The development of a Relative Risk Method model based on the risk management of aquatic ecosystems influenced by construction activities

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Thesis submitted for the degree *Philosophiae Doctor* in Environmental Sciences at the Potchefstroom Campus of the North-West University

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October 2017

DECLARATION

I declare that this thesis which I submitted for the degree of doctor of Philosophy (PhD) in the Department of Environmental Sciences and Management at the Potchefstroom Campus of the North-West University is original and has not been submitted by me for a degree at any institution. All assistance that I received has been fully acknowledged.

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Kaajial Durgapersad Date

SUMMARY

The water quality of Wilge River sub-catchment in the upper Olifants water management area is under threat from a number of land-use activities. The aims of this study is to use existing environmental monitoring and biomonitoring tools that are routinely applied in environmental assessments in South Africa and to interpret the results in a uniform risk-based format to allow for informed decision-making relating to the potential risk of impacts of construction activities on the aquatic resources. Water quality, toxicity, macroinvertebrates, fish and wetland status were evaluated for the period 2006-2014 for the Wilge River sub-catchment B20F area. The relationship between water quality, toxicity and the biological responses were evaluated using relevant multivariate (principal component and redundancy) statistical analyses and piper analyses.

Water quality results showed that the combination of land-use impacts has affected the water quality in the Wilge River sub-catchment B20F area. The main sets of stressors therefore, is acidic water containing heavy and trace metal ions and sulphate that is attributable to abandoned mining and nutrient concentrations originating from agricultural and livestock runoff, and from untreated or poorly treated sewage. The careful management and mitigation of these pollutant sources are essential to ensure compliance to the Wilge River IWRMP RWQOs. The four-tiered toxicity assessments were found to be applicable and appropriate for measuring the change in toxicity hazards due to a range of land-uses and produced additional information when considering the relative health of a water resource under stress. The hazard categories of the sampling sites were found to have a predominantly moderate hazard to toxicity. Thus implying that the cumulative effects of the impacts, i.e. agriculture, livestock farming, mining, the construction site and the quarry are contributing to the increasing toxicity in the catchment.

Comprehensive macroinvertebrate studies show that considerable variations occurred with regard to the families found between the various surveys and between each of the sampling sites. A dominance of families found had a preference for low to very low water quality, probably due to the changes in land-use. Macroinvertebrate assemblage within the Wilge River sub-catchment B20F area show that it is in a poor state of health and it is therefore imperative to maintain the ecological integrity of the Wilge River.

The fish assessments showed that the highest species diversity was found at sampling reaches 4 and 5, with all fish species on the Fish Reference Frequency of Occurrence (FFROC) list being found. Anthropogenic factors such as, impeding structures i.e. dams, bridges and roads have affected fish migration in sampling reaches 1, 2 and 3 showing lower species diversity and higher fish species absences in these reaches. Fish assemblage structures were shown not to have altered due to changes in land-use as the ecological categories remained similar from assessments carried out from 2006 to 2014.

WET-health assessments of the wetlands, indicated that the wetlands conditions were found to have deteriorated from March 2010 to December 2011 in the wetland complexes assessed, and can be attributed to the changing land-use. Improvements to the wetland ecological status from August 2012 to December 2014 can be due to a decrease in construction activities and an increase in wetland rehabilitation efforts implemented.

The Bayesian Network-Relative Risk Method was applied as a tool to perform a regional-scale, multiple-stressor ecological risk assessment in the Wilge River sub-catchment area in the Upper Olifants River catchment. The results of this study demonstrated that the bayesian network can be used to calculate risk for multiple stressors, and that they are a powerful tool for informing future management strategies for aquatic ecosystem management in the Wilge River sub-catchment. The evidence based outcomes can facilitate informed environmental management decision-making. The careful management and mitigation of pollutant sources are essential to ensure compliance to the Wilge River Integrated Water Resource Management Plan Resource Water Quality Objectives.

Keywords: relative risk assessment, water quality, macroinvertebrates, fish integrity, toxicity, wetlands, Olifants catchment, Wilge River sub-catchment, construction, land-use

ACKNOWLEDGMENTS:

I would like to acknowledge and thank with the sincerest appreciation, the following individuals and institutions:

My mentor Professor Victor Wepener, for your leadership, support and advice in the study. Thank you, for the time we have worked together.

My co-supervisor, Dr Gordon O'Brien, for your advice. Thank you for the opportunity to work with you.

Dave Lucas for the guidance, support and motivation throughout my studies. Thank you for your contribution to this study.

Dr Wynand Malherbe for his guidance and advice in the fish chapter.

Gerhard Brits and Michelle Bester Geographic Information Systems specialists, Eskom Academy of Learning. Eskom College, for their invaluable help with the ESI-GIS Maps.

Siphiwe Mahlangu, Environmental manager at Kusile power station for quick responses to data and information requests. It is very much appreciated.

Reshen Naidoo and Prof Walter Schmitz for their part in arranging for me to work from Witwatersrand University. I really appreciate everything you did, and it directly impacted on me meeting my deadlines.

Prof Louis Jestin, Eskom Power Plant Engineering Institute Programme (EPPEI) for allowing me the resources to complete this study.

My parents Ugrasen and Renuka Sukhraj, who instilled in me the virtues of perseverance and commitment and encouraged me to strive for excellence. Thank you for your example, guidance and support.

And finally, to my daughters Kirthi and Adithi for their unlimited love and support.

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CHAPTER 1: INTRODUCTION AND PROBLEM FORMULATION

1.1 Introduction

Water is essential for the survival and sustainability of all life on earth. It is one of the key and most indispensable of all our natural resources. It is fundamental to life, the quality of life, environment, food production, hygiene, industry, and power generation. Yet, water demand and pollution from human activities are continuously increasing (OECD, 2012). Water can be the limiting factor when it comes to economic growth and social development, especially in South Africa where it is a relatively scarce resource that is distributed unevenly geographically, through time and socio-politically. Environmental issues pertaining to water are recognised as one of the major environmental concerns for the coming decades (UNESCO, 2006). Prosperity for South Africa depends upon sound management and utilisation of our many natural and other resources, with water playing a pivotal role (DWA, 2004).

The continued deterioration in the ecological state of South Africa's surface aquatic ecosystems is causing an inevitable decline in the provision of key ecosystem services upon which the social and economic development of the country depends (Driver *et al*., 2005; MEA, 2005; Ashton, 2007). Water is a scarce commodity in South Africa (Tyson, 1987), which is disproportionately distributed, primarily in the limited river networks and few natural lakes (Davies and Day, 1998; Ashton, 2007). At present only approximately 30% of South Africa's main rivers are still intact and sustainable while a staggering 47% have been modified to varying degrees and 23% have been irreversibly transformed (Nel *et al.*, 2004). With restricted water resources, increased water pollution due to increased urbanisation, agricultural and industrial activities, inappropriate management and control of water resources and quality have exacerbated the already alarming situation (Oberholster *et al.*, 2008). In an attempt to address and establish integrated management plans for surface aquatic ecosystems in South Africa, all stakeholders need to become more closely engaged in the social and institutional decision making processes (Ashton, 2007). Along with the Department of Water and Sanitation (DWS), the custodian of South Africa's water resources, other stakeholders of these aquatic ecosystems include higher education institutions are required to contribute towards the establishment integrated management plans.

South Africa's Reconstruction and Development Programme are already based on the fundamental concept that people who are affected by decisions should take part in making these decisions (DWAF, 2004). As such, the management framework for freshwater aquatic ecosystems in South Africa allows for the participation of stakeholders of aquatic ecosystems (DWAF, 2004). Through the DWS, the National Water Resource Strategy describes how the water resources of South Africa should be protected, used, developed, conserved, managed and controlled in accordance with the requirements of the law in South Africa (DWAF, 2004). Within this strategy, the approach adopted to manage the equitable balance between the use and protection of surface aquatic ecosystems falls within the framework of the Integrated Water Resource Management Plan (IWRMP). The IWRMP is a process which promotes the co-ordinated development and management of water, land and related resources, in order to maximise the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystem components (Global Water Partnership, 1999). In addition, the IWRMP is a process, and an implementation strategy, which aims to facilitate equitable access to, and sustainable use of, water resources by all stakeholders at catchment, regional, national, and international levels, while maintaining the characteristics and integrity of water resources at the catchment scale within established limits (DWAF, 2004). These concepts are formalised into the constitution of South Africa in the form of the National Water Act, (Act No 36 of 1998), which details a progressive approach to water resource management in South Africa.

It has widely been accepted that the overall goal of environmental management should be environmental, social and economic sustainable development. Social and economic sustainable development is essential to improve continuously the quality of life of the world's population. Environmental sustainability ensures that this is achieved without causing environmental deterioration in either this or future generations. Over the years, different environmental management approaches have been developed, usually for specific purposes within environmental management, and new approaches are regularly published in the literature. All of these approaches are continually being developed further as practitioners seek ways of addressing broader aspects of sustainable development.

Risk assessment can be defined as the process of assigning magnitudes and probabilities to the adverse effects of anthropogenic activities or natural catastrophes

(Suter, 1993). These effects are termed hazards. The existence of a hazard and the related uncertainty of the hazards effects, result in the formulation of risk. Risk is the probability or likelihood of a prescribed undesired effect occurring and impacting an environment (Suter, 1993). The Regional-Scale Risk Assessment that makes use of the Relative Risk Model (RRM), developed by Landis and Wiegers (1997), is implemented on a large spatial scale and facilitates the consideration of multiple sources of multiple stressors affecting multiple endpoints, including the ecosystem dynamics and characteristics of the landscape that may affect the risk estimate. Following the initial development, the RRM has been refined into the working method which has been tried and tested in numerous Environmental Risk Assessments (ERA) around the world (Chen and Landis, 2005; Colnar and Landis, 2007; Landis and Thomas, 2009; Apitz, 2011). With the opportunity to test the RRM approach through so many case studies the approach has been criticized (Cook *et al*., 1999, Cormier *et al*., 2000), validated and refined into the working methods presented by Landis (2005) and Colnar and Landis (2007).

1.2 Problem formulation

The water quality of the surface water in the Olifants Water Management Area (WMA) is under threat as a result of industry, mining and agriculture (De Villiers and Mkwelo, 2009). The upper Olifants River catchment is the most important source of coal in South Africa, and acid mine drainage (AMD) originating primarily from old, abandoned mines has been identified as one of the major long-term water quality impacts in the catchment (Hobbs *et al.*, 2008). The quality of the water resource is also under threat from a number of sources, besides the coal mining industry i.e., urban development and poorly performing municipal wastewater treatment plants. There has been a steady deterioration in the water quality of the major dams since the 1970s (DWA, 2009). The coal mining, previously concentrated in the Klip River catchment is expanding to the Wilge River sub-catchment, as the coal reserves in the Middleburg and Witbank Dam catchments are insufficient to meet demands (DWAF, 2004). Currently the coal mining activities in the Wilge River catchments are low and the water quality is good in this catchment (DWA, 2009). The Wilge River sub-catchment B20F study area currently has the following land-use categories: mostly agricultural – commericial irrigated and fertilised land; coal mining i.e. closed New Largo colliery; livestock farming – livestock watering, combination of free range cattle and impounded cattle, chicken and pigs;

Kendal power station; Kusile construction site; quarrying and infrastructure, i.e. railway tracks and tar or gravel roads. The Regional-Scale Risk Assessment that makes use of the RRM, developed by Landis and Wiegers (1997), can be customized to address the threats of multiple sources of multiple stressors to local habitat and endpoints. In South Africa, the proposed National Water Resource Strategy (NWRS) will give practical effect to the management of water as a scarce resource within the framework of the National Water Act, (Act No 36 of 1998). In order to comply with the NWRS, the IWRMP is in the process of being developed and implemented for the Upper Olifants River catchment. This project will make a valuable contribution towards the development of the proposed IWRMP's through providing and environmental assessment framework based on a risk-based set of protocols.

1.2.1 Ten procedural steps of the Relative Risk Method framework

The RRM framework adapted by O'Brien and Wepener (2012) for the Regional-Scale Risk Assessment for the management of the aquatic ecosystems of South Africa was applied in this study. The process involved **10 steps** which include:

STEP 1: List the important management goals for the region;

Because the Kusile construction site was used as a case study, specific management goals of Kusile (i.e. Water Use Licenses (WUL)/Environmental Management Plans) were used.

STEP 2: Generate a map on which the potential sources and habitats relevant to the established management goals are indicated;

Geographic Information System (GIS) and Google maps were used to generate suitable study area maps of the Wilge River sub-catchment of the Upper Olifants River catchment.

STEP 3: Demarcate the map into regions based on a combination of the management goals, sources and habitats;

Land-use, varous other layers and site visits were used to identify potential sources and stressors. These sources of information were used to demarcate the study area into risk regions.

STEP 4: Construct a conceptual model that links the source stressors to receptors and to the assessment endpoints;

Kusile specialist studies, environmental specialists and managers were consulted on the Kusile construction site and were used to construct source-stressor-habitat-endpoint relationships.

The next four steps (STEP 5-8) of the RRM (5. Decide on a ranking scheme to calculate the relative risk to the assessment endpoints; 6. Calculate the relative risks; 7. Evaluate uncertainty and sensitivity analysis of the relative rankings; 8. Generate testable hypotheses for future field and laboratory investigations to reduce uncertainties and to confirm the risk rankings) will require field- and laboratory based testing

Steps 5-8 are based on the generation of data which were used to populate and validate the conceptual model. This validation was carried out to provide ranking schemes for further risk calculation.

Water Quality: Chemistry parameters carried out by South African National Accreditation System (SANAS) accredited laboratories, acquired from the specialist studies were collated. Water quality samples were taken monthly for the period 2006-2014 from all available sites for chemistry analyses. Water samples were collected, and analysed for a number of water quality variables, including pH, sulphate (SO_4), aluminum (Al), total dissolved salts (TDS), chloride (Cl), electrical conductivity (EC), sodium (Na), total hardness (Thard), total alkalinity (Talk), calcium (Ca), magnesium (Mg), potassium (K), nitrate (NO_3), ammonia (NH_3), and suspended solids (SS). All samples were stored in a cooler box and taken to the Eskom laboratory for analysis. The water quality was assessed according to the requirements for Kusile WUL and interim Resource Water Quality Objectives (RWQO) determined for the Wilge River sub-catchment.

Toxicology: The Direct Estimation of Ecological Effect Potential (DEEEP) analysis as required by the Kusile WUL were undertaken. Water samples were taken in 2 litre plastic bottles and transported in cooler boxes to the Golder Associates Laboratory. Four trophic levels of biota i.e., vertebrates (*Poecilia reticulata*), invertebrates (*Daphnia magna*), bacteria (*Vibrio fischeri*) and primary producers – algae (*Pseudokirchneriella subcapitata*) were exposed to the samples according to standard procedures under laboratory conditions and thereafter a risk/hazard category was determined by application of the latest DEEEP. This is a battery of tests that can measure toxicity of complex mixtures based on a set of parameters resulting from the outcome of effects, even if all constituents are not known. Consequently a hazard class was determined based on the resulting parameters of the battery of tests (DWA, 2003).

Biological Monitoring: South African Scoring System Version 5 (SASS5) for macro-invertebrates (Dickens and Graham, 2002); Fish Response Assessment Index (FRAI) for fish (Kleynhans, 2007) and wetland health i.e. WET-health (Macfarlane *et al.,* 2009) assessments were collated for the period 2006 to 2014. The macro-invertebrates were sampled in each biotope (stones, vegetation and gravel/sand/mud) group, identified and their relative abundance noted on the SASS5 datasheet. SASS5 scores were determined. A habitat assessment was also completed on site. Fish were assessed at sampling sites using an electro-shocker. The shocker was used to send electronic waves through the water. Waders and gloves were worn by the assessor for protection. The fish that were shocked were caught in a fish net. They were identified, assessed, counted and returned to the water. No fish were harmed in this assessment. WET-health assessment is carried out in the field by completing and noting a comprehensive list of wetland characteristics. The data were analysed and the assessment completed. The relationship between water quality, toxicity and the biological responses were evaluated using relevant multivariate (principal component and redundancy) statistical analyses and piper analyses.

Risk calculation, uncertainty and sensitivity analyses: Bayesian modeling were used to integrate the ranking schemes developed for the conceptual model and to provide a risk score for each risk region (Landis, 2005; Colnar and Landis, 2007). Monte Carlo permutation testing were applied for sensitivity and uncertainty analyses.

STEP 9: Test the hypotheses that were generated in STEP 8;

Hypotheses were generated through the selection of various management scenarios. The RRM model were then utilised to generate risk scores for the different scenarios, to the aquatic environment.

STEP 10: Communicate the results in a fashion that effectively portrays the relative risk and uncertainty in the response to the management goals.

Results based on the different scenarios were communicated by means of recommendations to an environmental management plan that addresses the risk posed to aquatic ecosystems by construction activities undertaken by Kusile power station.

The main aim of this study is to use existing environmental monitoring and biomonitoring tools that are routinely applied in environmental assessments in South Africa and to interpret the results in a uniform risk-based format to allow for informed decision-making relating to the potential risk of impacts of the Kusile construction activities on the local aquatic resources.

1.3 Study hypothesis

The hypotheses established for the study state:

Hypothesis 1: Land-use activities do not change the water quality and there is no influence of these water quality parameters on the toxicity to aquatic organisms.

Hypothesis 2: Land-use activities do not change the integrity of macroinvertebrate community structures.

Hypothesis 3: Land-use activities do not change the integrity of fish community structures.

Hypothesis 4: Land-use activities do not change the wetland integrity.

Hypothesis 5: The RRM is effective in achieving management goals.

1.3.1 Study aims

The aims of the study were established as follows:

- Development of a RRM framework based on activities related to construction activities of Kusile power station;
- Using the Kusile construction site as a case study, assess the impacts on local aquatic environments in terms of water quality and aquatic ecosystem health;

- Integrate data from existing aquatic assessment tools into an RRM protocol;
- Integrate the RRM-based outputs to refine the existing Environmental Management Plan of the Kusile construction site, based on data generated;
- Recommend environmental guidelines that would be implemented for construction activities at Kusile sites.

1.4 Structure of the thesis

This thesis is divided into eight chapters. References cited in each chapter are listed after each chapter.

Chapter 1: Introduction and Problem Formulation

This chapter provides the background to the problem, why and how the problem is addressed. The hypotheses, aims and objectives of the thesis are provided as well as an outline of the thesis.

Chapter 2: Water Quality Analysis of the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter provides an introduction (background and overview) on the water quality status of the Wilge River. Water quality data was evaluated using relevant statistical and multivariate principal component analyses. Piper analysis was undertaken as well. Spatial and temporal patterns of the water quality status of the Wilge River were discussed and interpreted in terms of the Integrated Water Resources Management Plan in South Africa.

Chapter 3: Survey of Macroinvertebrate Communities as an indicator of ecological health in the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter provides an introduction (background and overview) on the biological status of the Wilge River in terms of the macro-invertebrate community assemblages. SASS5, methodology, statistical analyses, diversity indices and RDA plots undertaken are described. Spatial and temporal patterns of the biological water quality status of the

Wilge River in terms of macroinvertebrate communities were discussed and interpreted in terms of the driving environmental variables responsible for the structures.

Chapter 4: The application of a direct toxicity assessment approach to assess the effect of construction activities on the ecological health of the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter provides an introduction (background and overview) on the toxicological status of the instream conditions of the Wilge River. Collation of toxicity data; DEEEP methodology and hazard classification undertaken are described. Spatial and temporal patterns of the toxicological status of the instream conditions of the Wilge River were discussed and interpreted.

Chapter 5: Use of fish as an indicator of ecological health in the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter provides an introduction (background and overview) on the biological status of the Wilge River in terms of the fish community assemblages. Collation of fish biological monitoring data, calculation of FRAI scores, statistical analyses – RDA plots were undertaken and described. Spatial and temporal patterns of the biological water quality status of the Wilge River in terms of fish communities were discussed and interpreted in terms of the driving environmental variables responsible for these structures.

Chapter 6: Use of Wetlands as indicators of ecological health in the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter provides an introduction (background and overview) on the biological status of the Wilge River in terms of wetlands. Wetland health i.e. WET-health (Macfarlane *et al.,* 2009) assessments, ecological importance and sensitivity and ecosystem services were collated for the period 2006 to 2014. WET-health assessment was carried out in the field by completing and noting a comprehensive list of wetland characteristics. Ecological importance and sensitivity and ecosystem services were evaluated from 2006-2014. The data were analysed and the assessment completed. Spatial and temporal patterns of the wetland ecological status of the Wilge River were discussed and interpreted in terms of the driving environmental variables responsible for these structures. The integrated wetland health assessment were discussed in relation to the ecological status of the different indicator groups.

Chapter 7: Using the Relative Risk model for a Regional-scale Ecological Risk Assessment of the Wilge River sub-catchment B20F area in the Upper Olifants River Catchment.

This chapter outlines the ten procedural steps undertaken in the development of the RRM framework. The important management goals for the Wilge River sub-catchment are listed. A map was generated on which the potential sources and habitats relevant to the established management goals were identified. A map demarcating the risk regions based on a combination of the management goals, sources and habitats was created. A conceptual model that links the source stressors to receptors and to the assessment endpoints was constructed. A ranking scheme to calculate the relative risk to the assessment endpoints was determined and relative risks calculated. The uncertainty and sensitivity analysis of the relative rankings were evaluated. Testable hypotheses for future field and laboratory investigations to reduce uncertainties and to confirm the risk rankings were generated and tested based on increased urban development and increased mining scenarios.

Chapter 8: Conclusions and Recommendations

This chapter provides a summary of the complete study and draws some conclusions. It also provides recommendations for environmental guidelines that would be incorporated into environmental management plans.

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CHAPTER 2: WATER QUALITY ANALYSIS OF WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

2.1 Introduction

The sustainable management of our aquatic resources is imperative for the survival and development of human communities. Rising world populations and consumption are increasing human demand for domestic, industrial, and agricultural water. Population numbers along with other global stressors will have a direct and significant impact on water quality and quantity. The legacy of pollution following the excessive use and abuse of surface aquatic ecosystems through for example, an increase in the salinisation of systems results in deleterious and often irreversible costs to the economic, social and ecological value of ecosystems. In particular the salinisation of aquatic ecosystems has resulted in many systems becoming totally unusable and practically void of biological diversities resembling their natural states (Williams, 2001). In many nations including South Africa salinisation is one of the most important factors contributing to the degradation of water quality of surface waters (e.g. Goetsch and Palmer, 1997; Davies and Day, 1998; Williams, 2001; Kefford *et al.*, 2004). In South Africa, the accumulation of dissolved inorganic salts or salinisation of ecosystem occurs as a result of either natural events such as the natural weathering of geological formations and more commonly by anthropogenic activities including agricultural activities, industrial activities and mining activities (Davies and Day, 1998). The main stressors of concern are usually toxic heavy and/or trace metal contamination, as well as nutrient enrichment (CSIR, 2010).

The water quality of the surface water in the Olifants Water Management Area (WMA) is under threat as a result of industrial, mining and agriculture (De Villiers and Mkwelo, 2009). The upper Olifants River catchment is the most important source of coal in South Africa, and acid mine drainage (AMD) originating primarily from old, abandoned mines has been identified as one of the major long-term water quality impacts in the catchment (Hobbs *et al.*, 2008). The mining is currently located in the Witbank and Middelburg Dam catchments as well as the Spookspruit and Klipspruit catchments. The water quality of the water resource is also under threat from a number of sources, besides the coal mining industry i.e., urban development and poorly performing municipal wastewater treatment plants. There has been a steady deterioration in the water quality of the major dams since the 1970s (DWA, 2009).

The salinity related variables in the Wilge River meet the Integrated Water Resource Management Plan (IWRMP) Resource Water Quality Objectives (RWQO), although the water quality is being threatened by increased mining activities in the catchment.· The trophic state of the Wilge River is also mesotrophic, with the 50 percentile phosphate concentration of about 0.02 mg/L (DWA, 2009). The coal mining, previously concentrated in the Klip River catchment is expanding to the Wilge River catchment, as the coal reserves in the Middleburg and Witbank Dam catchments are insufficient to meet demands (DWAF, 2004a). Currently the coal mining activities in the Wilge River catchments are minimal and the water quality is good in this catchment (DWA, 2009).

A new dry cooled Kusile power station is being constructed north of the existing Kendal power station in the Wilge River sub-catchment B20F area. The construction site is situated approximately 35 km west of Emalahleni (formerly known as Witbank), in the Mpumalanga Province. The coal mines provide coal for power generation in the local market and for export through the Richards Bay Coal Terminal (DWA, 2009). The activities regarding pollution of surface and groundwater resources by the power stations are managed by means of licensing procedures. The atmospheric deposition of emissions from the power stations has been cited as a source of salinity both in the Olifants and the Upper Vaal WMAs (DWAF, 2004b). Construction at Kusile power station has commenced in April 2008 and is planned to be completed in 2018. Large construction activities have been found to be responsible for, amongst other impacts, the salinisation of aquatic ecosystems (DWAF, 2004b). The station will consist of six units each rated at approximately 800 MW installed capacity giving a total of 4800 MW. As such it will be the fourth largest coal-fired power station in the world, once finished.

The need for aquatic resource conservation is increasing as a result of a decline in water quality of aquatic ecosystems due to increased pollution (Ashton *et al.,* 2008). If allowed to deteriorate the quality of water can adversely affect not only the aquatic ecosystem of the specific water resource, but the quality of the groundwater as well (CSIR, 2010). The aim of this chapter is to determine the spatial and temporal changes in selected water quality parameters, in the Wilge River sub-catchment B20F area over a period of 7 years (2008-2014) in order to determine possible impact of land-use activities. These data will be applied in the Relative Risk Model (RRM) to aid in the future management and conservation of the water resource.

2.2 Materials and methods

2.2.1 Study area

The study area, the quaternary drainage region B20F, is situated within the Wilge River catchment in the Olifants WMA4. The Olifants WMA is located within three provinces Gauteng, Mpumalanga and the Limpopo and covers an area of approximately 54 550 km^2 (DWA, 2011a). The Olifants River originates in Mpumalanga flowing northwards before curving in an easterly direction through the Kruger National Park and into Mozambique. In the National Water Resources Strategy (NWRS) (DWAF, 2004a), the Olifants WMA has been divided into 4 sub-areas: the Upper Olifants, Middle Olifants, Steelpoort and Lower Olifants sub-areas (Figure 2.1).

These four sub-areas of the Olifants WMA consistutes the following:

- Upper Olifants Catchment constitutes the catchment of the Olifants River down to Loskop Dam (B1 and B2);
- Middle Olifants Catchment comprises the catchment of the Olifants River downstream from the Loskop Dam to the confluence with the Steelpoort River (B3 and B5);
- Steelpoort Catchment corresponds to drainage region of the Steelpoort River (B4);
- Lower Olifants Catchment represents the catchment of the Olifants River between the Steelpoort confluence and the Mozambique border (B6 and B7).

The Upper Olifants River catchment consists of the Klip River (B1) and the Wilge River (B2) sub-catchments. The Upper Olifants River catchment is 12 264 km^2. Ogies town is located in the divide between the Klip and Wilge River sub-catchments. A number of the surrounding towns have been developed to accommodate power station and mining personnel (DWAF, 2004b).

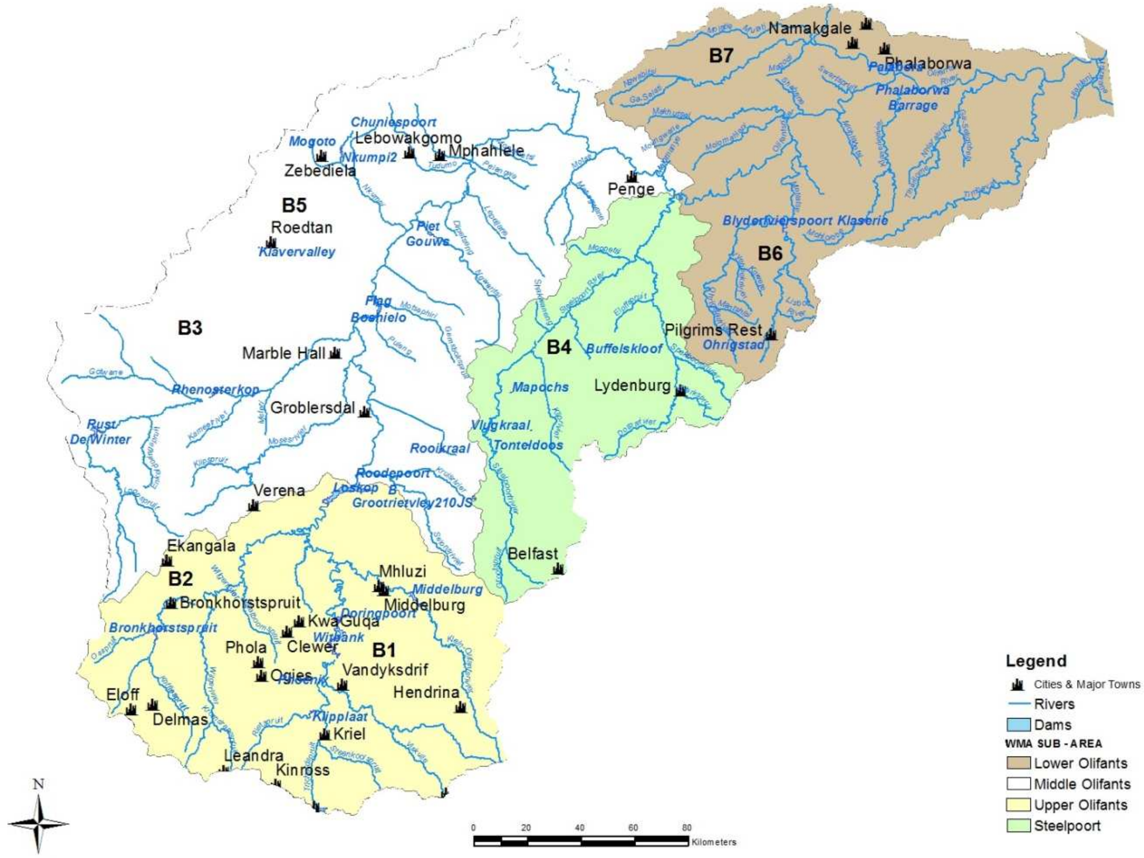


Figure 2.1: The Olifants River Water Management Area.

The Wilge River sub-catchment is 4 357 km^2 and the land cover is mostly rural in nature with the main activity being agriculture with the main towns of Bronkhorstspruit and Delmas (Figure 2.1 and 2.2).

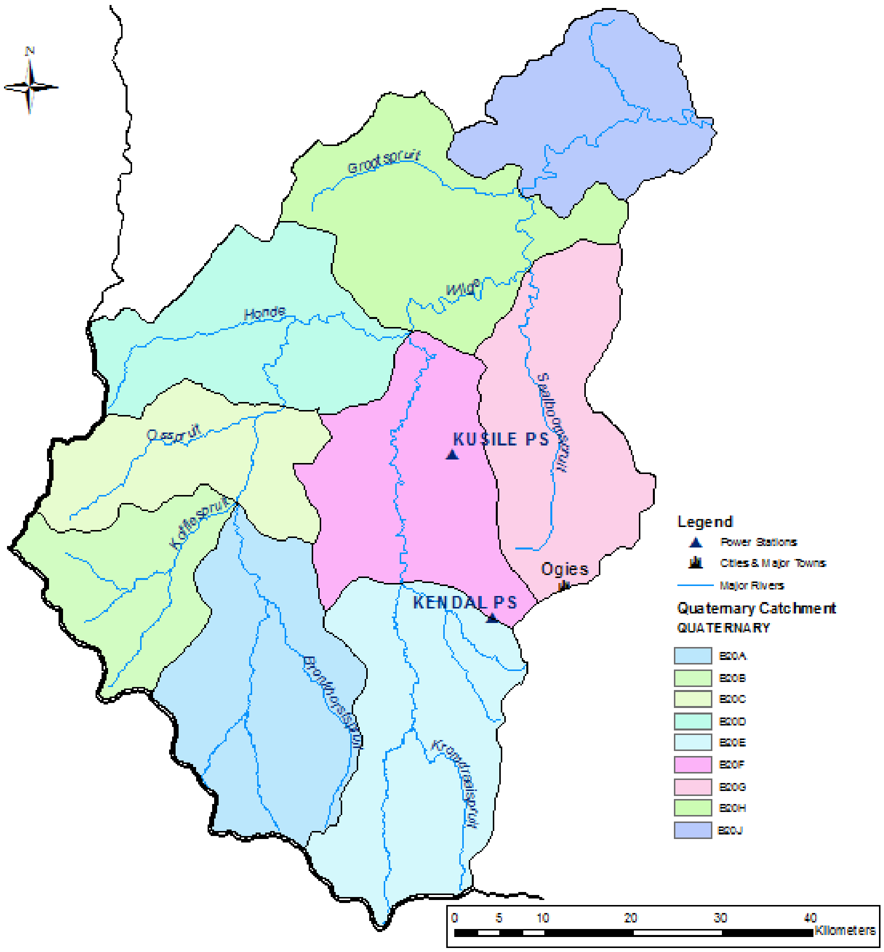


Figure 2.2: Wilge River B2 sub-catchment area in the Upper Olifants River Catchment.

This study area, quaternary drainage region B20F, is situated within the Wilge River (B2) sub-catchment in the Olifants WMA4 (Figure 2.2 and 2.3). The main river in the study area is the Wilge River with the Klipfonteinspruit and several unnamed tributaries joining the Wilge River. The B20F area has the following land-use categories: mostly agricultural – commercial irrigated, fertilised land and livestock farming – livestock watering, combination of free range cattle and impounded cattle, chicken and pigs; coal mining i.e. closed New Largo colliery; Kendal power station; Kusile construction site; quarrying and infrastructure, i.e. railway tracks and tar or gravel roads.

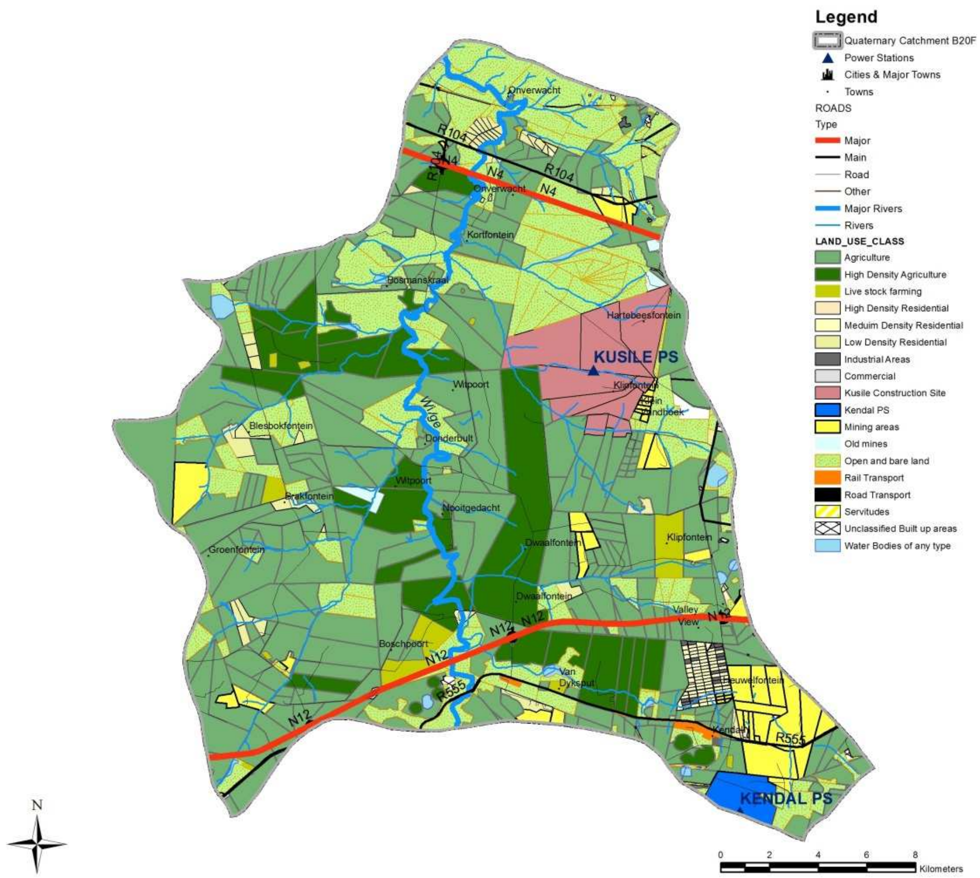


Figure 2.3: Land-use in Wilge River sub-catchment B20F Area.

The crops grown in the relatively small Grouwsberg irrigation district in the lower Wilge district include maize, potatoes, wheat and vegetables. The source of water for the irrigation is from river, farm dams and groundwater. Groundwater is abstracted extensively from the Delmas dolomites in the upper Bronkhorstspruit catchment. The irrigation water use has grown since 1995. The highest growth was in the Wilge River and the Bronkhorstspruit catchments. The growth in the groundwater abstraction from the Bronkhorstspruit catchment is particularly large. Much of the growth in the irrigation areas since 1995 is likely to be unlawful as very few abstraction licences have been issued by the Department (DWA, 2009).

According to Mucina and Rutherford (2006), the study area falls within the Grassland Biome, Mesic Highveld Grassland Bioregion (Figure 2.4). At a finer level, the study area is classed as Eastern Highveld Grassland and Rand Highveld Grassland, with patches of Eastern Temperate Freshwater Wetlands vegetation occurring and being associated with the larger pans of the area. Rand Highveld Grassland and Eastern Temperate

Freshwater Wetlands are listed as *Vulnerable* on the National List of Threatened Ecosystems (GN 1002 of 2011) for Mpumalanga Province, while Eastern Highveld Grassland is listed as *Vulnerable* on a national scale.

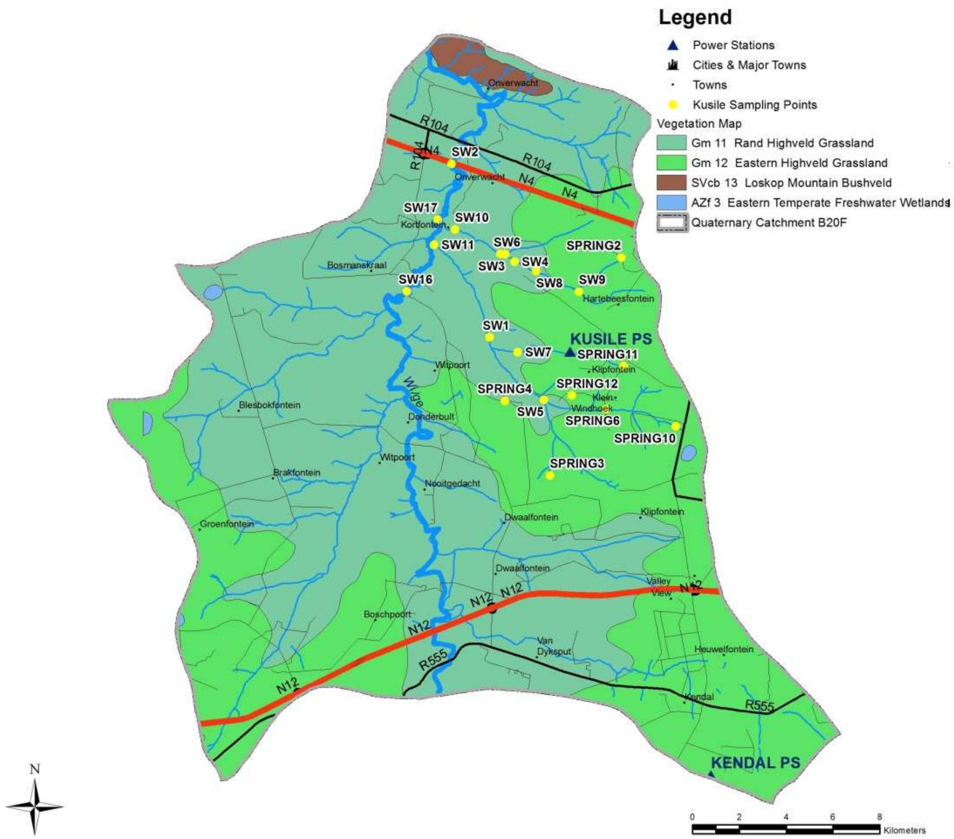


Figure 2.4: Vegetation Types in Wilge River sub-catchment B20F Area.

This area is generally flat to gently undulating. The Wilge sub-catchment B20F area is underlain by Pretoria shale and quartzite capped on the high ground by Dwyka tillite and shale (Figure 2.5). The entire sequence has been intruded by a dolerite sill. There are no significant anomalies associated with apparent conductive zones interpreted as highly weathered and/or fractured zones (Kok *et al.*, 2013). Four land types were found occurring on site. The deepest, most well-drained soils occur within the south west of the study area, while shallower soils with more impeded drainage, and thus more conducive to wetland formation, occur within the eastern half of the study area. Extensive shallow, rocky soils also occur on site, specifically in the north and north-west of the site.

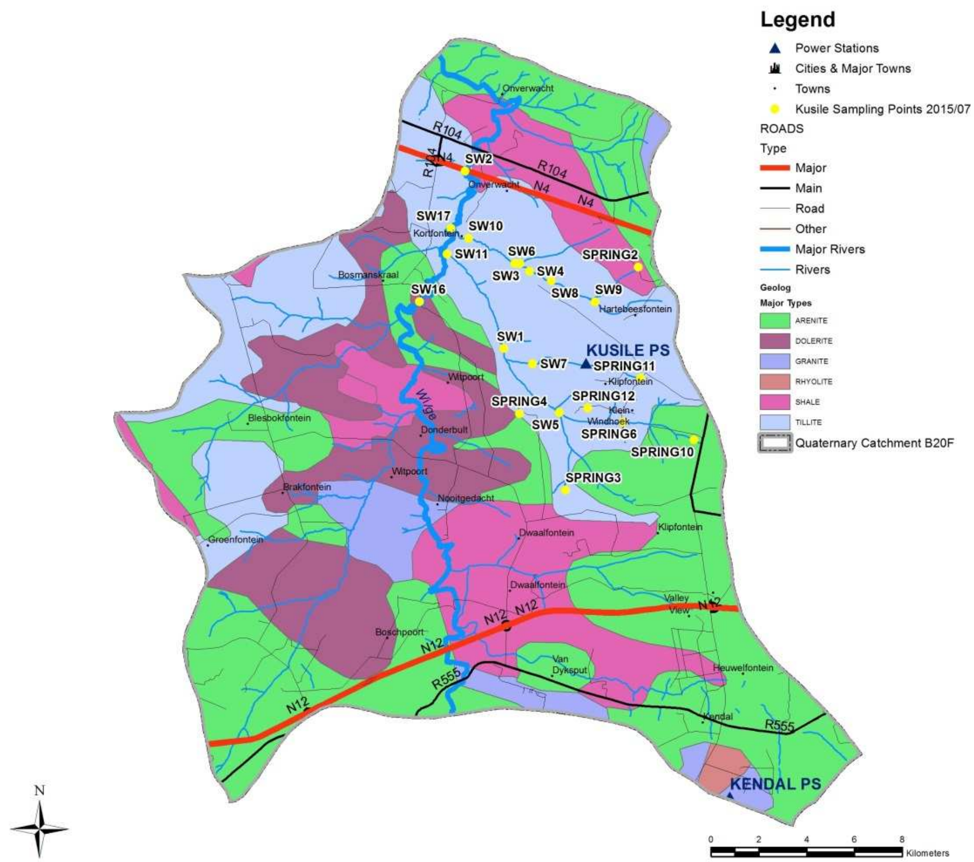


Figure 2.5: Geology and Mines in Wilge sub-catchment B20F Area.

2.2.2 Site selection

Water quality sampling sites in the Wilge River sub-catchment B20F area are shown in Figure 2.6. An assessment of the spatial and temporal changes of water quality in the B20F study area was undertaken by analysing historical water quality data obtained from the routine water quality monitoring carried out by Kusile power station. Kusile power station has 24 sampling points on the tributaries entering and leaving its construction site. Of these, 20 were chosen in this study (Table 2.1).

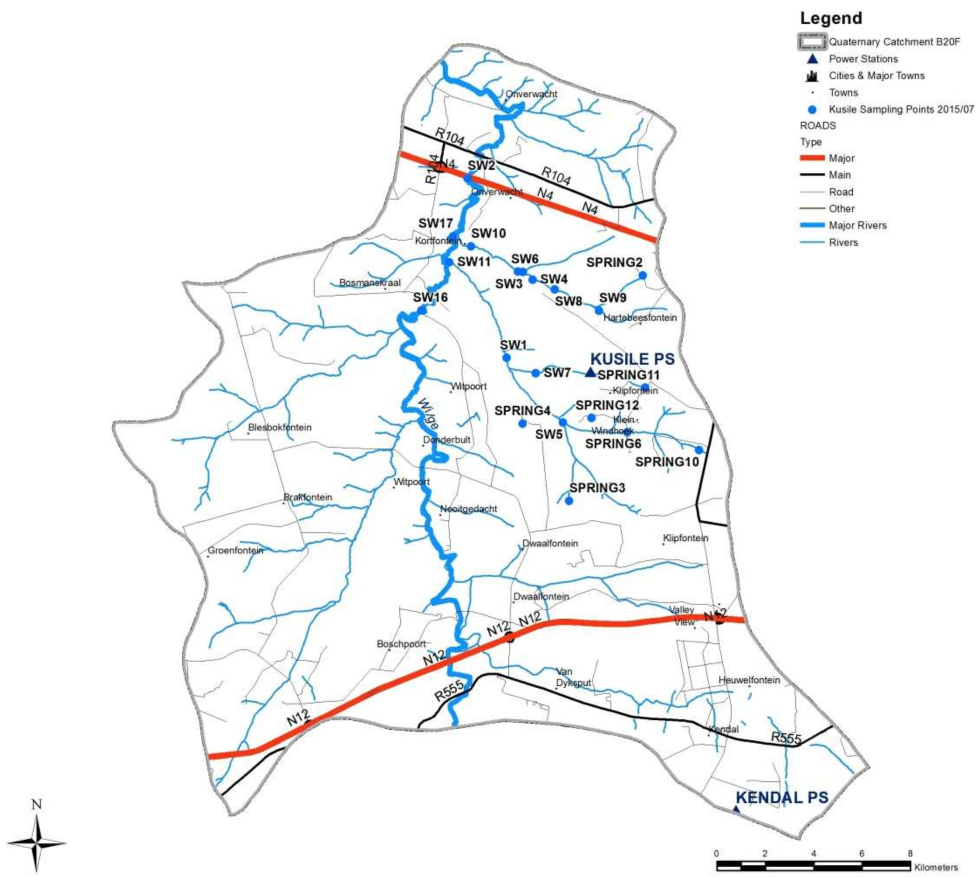


Figure 2.6: Water Quality Sampling Sites in Wilge River sub-catchment B20F Area.

The Kusile construction footprint is drained via two predominantly south east to north west flowing surface water drainage features. For the purposes of this study, the catchment areas have been divided into the following three areas as per Kok *et al.,* (2013): C11 catchment, Klipfonteinspruit catchment and the Wilge River catchment.

- The C11 catchment drains the northern portions of the Kusile construction footprint and consists the following sample sites Spring2, SW9, SW8, SW4, SW3, SW6 and SW10;
- The Klipfonteinspruit catchment drains the central and southern portion of the Kusile construction footprint and consists of the following sample sites Spring10, Spring6, Spring12, SW5, Spring3, Spring4, Spring11, SW7, SW1 and SW11;
- The Wilge River catchment into which the C11 and Klipfonteinspruit catchments discharge consists of the following sample sites SW16, SW17 and SW2.

Table 2.1: Water quality sampling points and GPS co-ordinates.

Catchment	Site	Latitude	Longitude	Description
C11	Spring2	-25.889300	28.933720	Located near the quarry and drains into the unnamed northern tributary
	SW9	-25.907380	28.927020	Located upstream on the unnamed northern tributary of the Wilge River
	SW8	-25.894600	28.900940	Located on the unnamed tributary of the Klipfonteinspruit, between sites SW9 and SW4
	SW4	-25.890750	28.890270	Located downstream on the unnamed northern tributary of the Wilge River
	SW3	-25.888100	28.889150	Drains the tributary into the unnamed northern tributary of the Wilge River just before site SW6
	SW6	-25.887970	28.887230	Located on the unnamed northern tributary of the Wilge River, downstream of site SW4
	SW10	-25.878530	28.869820	Located on unnamed northern tributary just before confluence with Wilge River
Klipfonteinspruit	Spring3	-25.973220	28.906320	Located on the tributary that drains into the Holfonteinspruit
	Spring4	-25.944490	28.888930	Drains into the Klipfonteinspruit, downstream of site SW5
	Spring6	-25.947600	28.927970	Located on Klipfonteinspruit before the confluence with the Holfonteinspruit
	Spring10	-25.954280	28.954620	Located on Klipfonteinspruit downstream of the closed New Largo colliery
	Spring11	-25.931100	28.934600	Located upstream of Kusile construction site
	Spring12	-25.942360	28.914660	Pan located next to construction site draining into the Klipfonteinspruit
	SW1	-25.914240	28.880640	This site is located on the Klipfonteinspruit, downstream of the unnamed tributary
	SW5	-25.938870	28.894710	Located at the confluence of the Klipfonteinspruit and the Holfonteinspruit
	SW7	-25.926361	28.894417	Located on the unnamed tributary of the Klipfonteinspruit
	SW11	-25.884390	28.861700	Located on Klipfonteinspruit just before confluence with Wilge River
Wilge	SW2	-25.864310	28.868930	Located in the Wilge River most downstream site of the B20F catchment
	SW16	-25.902190	28.851420	Located on Wilge River before confluence with Klipfonteinspruit
	SW17	-25.874760	28.863130	Located on the Wilge River just before the unnamed northern tributary of the Wilge River

The water quality in the Wilge River is considered good as indicated by the data collected at the Department of Water Affairs (DWA) sampling point B2H014Q01 just downstream of the Kusile sampling site SW2, for the last 10 years (Figure 2.7). Except for ammonia, which has been set the strictest guideline objective for aquatic ecosystem

(DWAF, 1996). The 90 percentile ammonia values were found to be within the interim RWQOs. The main impactors on the Wilge River upstream of sampling point B2H014Q01 are agriculture and some mining activities (DWA, 2014).

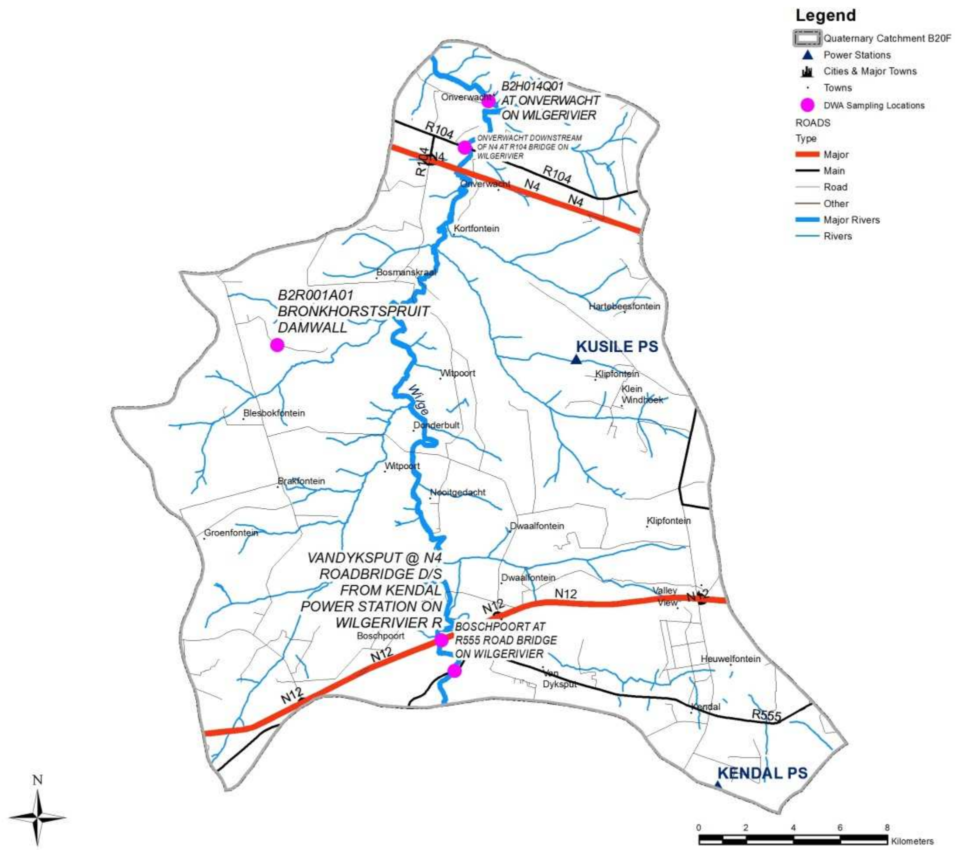


Figure 2.7: DWA Sampling Points in Wilge River sub-catchment B20F Area.

2.2.3 Water chemistry sampling and analysis

The samples were taken and analysed on a monthly basis from October 2008 till December 2014 for this study. Three different laboratories were used for the various water quality constituents determined for period 2008-2014. During June 2008-May 2012 the water samples were analysed by the UIS Analytical Services (Pty) Ltd laboratory. During June 2012-December 2013 the water samples were analysed by the SANAS Accredited Pelindaba Analytical Laboratory (PAL) at NECSA (SANAS Accreditation number: T0168). And during February 2014-December 2014 the water samples were analysed by Regen waters laboratories. All analyses were kept consistent in order to be comparable.

The surface sampling was conducted in accordance with the Minimum Requirements for Water Monitoring at Waste Management Facilities (DWAF, 1998). Two 1 litre plastic bottles were rinsed and filled with water at the site. Samples from the springs and surface water monitoring sites were collected by immersing the sample container in the water in order to collect a grab sample. Sample containers were prepared by the laboratory. Samples used for analyses of trace elements, i.e. Inductively Coupled Plasma-Mass Spectrometry (ICP-MS) were acidified with nitric acid (HNO_3). Samples were stored in cool containers after collection and transported to the laboratories within 24 hours after collection. The samples were accompanied by a chain of custody form, which lists all samples submitted, date and time sampled and constituents to be analysed. In addition to the collection of water samples field measurements were also conducted. Field measurements were taken for pH, electrical conductivity (EC), dissolved oxygen (DO), turbidity and temperature at all accessed sampling sites using a handheld meter. Calibration of the meter was conducted daily prior to sampling.

The samples were analysed for a number of water quality variables. The following analytical methods were used:

- Determination of pH, EC (mS/m), DO (% saturation) and turbidity measured in Nephelometric Turbidity Unit (NTU) with a handheld meter in water samples. In cases (For period December 2008- July 2012 and 2014) where DO was measured in mg/L, a conversion to % saturation was determined by assuming an average temperature of 15 ℃ and consequently using saturation level of 10 (USEPA, 2015);
- The spectrophotometric analysis and calculation in determination of ammonia in water samples;
- Suspended solids (SS) by filtration, measurement and calculation;
- Anions (NO_3^-, SO_4^{2-} , PO_4^{3-}) by ion chromatography;
- The determination of sodium and potassium by atomic absorption spectrophotometery (AAS);
- The determination of calcium and magnesium by Indictively Coupled Plasma Optical Emission Spectrometry (ICP-OES);

- The determination of the remaining elements i.e. aluminium (Al), iron (Fe), manganese (Mn), zinc (Zn), bromine (Br), cadmium (Cd), lead (Pb) and mercury (Hg) with ICP-MS analysis.

2.2.4 Statistics

Standard descriptive statistics were carried out on Microsoft Office Professional Plus 2010, Microsoft Excel Version: 14.0.7015.1000, 32-bit. In cases where concentrations were found to be below the detection limit (BD) of the instrument, average values were used in the statistical analysis. Box and whisker graphs were created using GraphPad Prism version 5.00 for Windows, GraphPad Software, San Diego California USA, www.graphpad.com. Graphs depict minimum, maximum, mean, 25th percentile and 75th percentile values.

Principle Component Analysis (PCA) was used to explore the relationship between water chemistry variables and spatial distribution of sampling sites. Analyses were conducted using the CANOCO for Windows package, version 4.5 (Ter Braak and Smilauer, 2002). A piper diagram was used to illustrate distinct water quality classifications (Piper, 1944; Appelo and Postma, 1993), which were then described according to their cationic and anionic distributions. These diagrams plot cations and anions as separate ternary plots to show the clustering of data points, which indicate whether or not samples have similar chemical compositions in terms of their major ions. The piper diagram was constructed using Aquachem Water Quality Analysis Software Version 5.1, Piper hydrogeochemical diagram by Carlos E. and Molano C.

2.3 Results and discussion

An assessment of the spatial and temporal changes of salinity levels in the Wilge River sub-catchment B20F study area was undertaken by analysing historical water quality data obtained from the routine water quality monitoring carried out by Kusile power station. The descriptive statistics of the water quality parameters are summarised and presented in Table 2.2. All data presented in Table 2.2 were used in the box and whisker graphs, PCA and piper diagrams (Appendix A, Figures 1-17; Figures 2.8-2.19).

Table 2.2: Descriptive statistics of the water quality parameters measured at all the sites in the study area between 2008 to 2014.

Parameters	pH	Conductivity	Turbidity	DO	Suspended Solids	Ammonia	Nitrate	Phosphate	Sulphate	Aluminium	Iron	Manganese	Zinc	Bromine	Cadmium	Lead	Mercury
Units		(mS/m)	(NTU)	(% Sat)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(µg/L)	(µg/L)	(µg/L)	(µg/L)	(µg/L)
Mean	7.29	21.29	93.84	64.00	55.85	0.22	1.18	0.35	47.07	0.69	1.67	0.17	13.99	72.98	0.45	1.44	0.56
Standard Error	0.02	0.50	6.91	1.28	5.25	0.02	0.08	0.01	2.48	0.05	0.18	0.01	1.47	2.07	0.03	0.14	0.03
Median	7.48	15.80	14.50	59.40	16.40	0.10	0.37	0.40	17.37	0.24	0.70	0.09	3.90	89.50	0.50	0.50	0.50
Mode	7.70	7.00	0.00	60.00	150.00	0.10	0.15	0.40	0.15	0.05	0.03	0.05	0.50	100.00	0.50	0.50	0.50
Standard Deviation	0.75	15.66	204.13	37.04	154.47	0.42	2.64	0.38	78.08	1.40	5.57	0.32	32.70	43.68	0.47	2.96	0.51
Sample Variance	0.56	245.26	41670.44	1372.03	23861.39	0.18	6.95	0.14	6095.91	1.96	31.08	0.10	1069.12	1907.95	0.22	8.77	0.26
Kurtosis	3.20	7.12	18	280	353	72	32	90	20	46	380	137	67	8.92	171	94	80
Skewness	-1.52	1.99	3.69	12.97	15.87	7.18	5.19	7.85	3.75	5.78	17.20	9.12	7.20	1.49	11.88	8.07	8.44
Range	5.53	133.80	2050.00	878.50	3678.00	5.39	24.39	5.33	690.00	16.00	137.00	5.90	373.56	357.60	7.17	42.00	5.80
Minimum	3.77	BD	BD	0.50	BD	0.01	0.01	0.05	BD	0.01	BD	BD	0.44	2.40	0.02	BD	0.07
Maximum	9.30	134	2050	879	3678	5.39	24	5.38	690	16	137	5.90	374	360.00	7.19	42	5.87
Sum	7258	21179	81921	53700	48310	85	1143	244	46698	604	1651	130	6927	32551	112	627	133
Count	995	995	873	839	865	384	967	699	992	876	991	743	495	446	250	434	238
Confidence Level(95%)	0.05	0.97	13.56	2.51	10.31	0.04	0.17	0.03	4.86	0.09	0.35	0.02	2.89	4.06	0.06	0.28	0.07

DO – Dissolved Oxygen

BD – Below Detection limit of the Instrument

The National Water Act, (Act No 36 of 1998) and its regulations prohibit the application of the General and Special Discharge Limits in the Wilge River and Olifants River catchments. Therefore, during 2009, a study was undertaken by the Department of Water Affairs & Forestry to develop an IWRMP for the Upper and Middle Olifants River catchment. As part of this study the catchment was divided into management units (MU). Interim RWQOs were set for each of the MUs.

The implementation of the RWQOs has the following objectives and implications:

- To ensure that the water users receive water of a suitable quality, while still allowing the catchment activities and potential economic growth to be realised;
- To ensure adequate protection of aquatic and river ecologies;
- The RWQOs have determined the assimilative capacity available in the system and therefore dictate the extent of the water quality management and pollution control interventions required.

The Wilge River sub-catchment B20F area falls within MU 22 management unit. The IWRMP for the Upper and Middle Olifants River catchment, recommends the following Resource Water Quality Objectives for the Wilge River catchment (Table 2.3).

Table 2.3: Integrated Water Resource Management Plan Interim Resource Water Quality Objectives determined for the Wilge River sub-catchment B20F area.

Water quality Variables	Units	IWRMP Units for Wilge sub-catchment B20F area
PHYSICAL		
Electrical Conductivity	mS/m	40 (PRWQ)
Dissolved Oxygen	% Sat	70 (AER)
pH		6.5-8.4 (IMS)
Suspended solids	mg/L	-
Turbidity	NTU	-
CHEMICAL, INORGANIC		
Alkalinity	mg $CaCO_3$/L	120 (PS)
Boron	mg/L	0.5 (IMS)
Calcium	mg/L	25 (PS)
Chloride	mg/L	20 (PS)
Fluoride	mg/L	0.5 (PS)
Magnesium	mg/L	20 (PS)
Potassium	mg/L	10 (PS)
Sodium	mg/L	20 (PS)
Sodium Adsorption Ratio (SAR)	meql0.5	1.0(PS)
Sulphate	mg/L	60 (PS)
Total Dissolved Solids	mg/L	280 (PS)
CHEMICAL, ORGANIC		
Dissolved Organic Carbon	mg/L	10 (DI)
METALS, DISSOLVED		
Iron	mg/L	1.0 (DI)
Manganese	mg/L	0.18 (AER)
Aluminium	mg/L	0.02 (AER)
Chromium VI	mg/L	0.05 (DF)
PLANT NUTRIENTS		
Ammonia*	mg/L as N	0.007 (AER)
Nitrate	mg/L as N	6 (DF)
Phosphate	mg/L as P	0.05 (AER)
Total Phosphorus	mg/L as P	0.25 (AER)
Total Inorganic Nitrogen	mg/L as N	2.5 (AER)
MICROBIOLOGICAL		
Escherichia coli (E.coli)	# per 100mℓ	130 (RFC)
Chlorophyll a	mg/L	0.02 (RIC)

Where: PS – present water quality status, ie. The 95 percentile concentration determined over the period 1997 to 2006 is used; AER – Aquatic Ecological Reserve as determined in the 2001 study; AET – Aquatic ecoltoxicological test results; ITWQR – Irrigation TWQR used for salt sensitive crops; IMS – Irrigation requirement used for moderately salt sensitive crops;SW – Stock watering; DI – Domestic informal water use; DF – Domestic formal use; RIC – Recreation intermediate contact; RFC – Recreation full contact; PRWQ – Current RWQO, based on previous studies; IND – industrial * - Free unionised NH_3

All the parameter results were compared to the Wilge River IWRMP RWQOs. For parameters analysed where the IWRMP RWQOs were not given, the Target Water Quality Objectives (TWQO) of the South African water quality guidelines for Aquatic Ecosystems (DWAF, 1996), were consulted (Table 2.4). The DWA TWQO are depicted for the following parameters: cadmium, lead, mercury and zinc. The average hardness in the study site based on the data available is ~80 mg/L, medium hardness (60-119 mg/L) so the TWQO for Cadmium and Lead with medium hardness (60-119) was used.

Table 2.4: South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996).

Water quality Variables	Units	Target Water Quality Objective
Cadmium (60-119 mg/L -hard water)	µg/L	0.25
Lead (60-119 mg/L -hard water)	µg/L	0.5
Mercury	µg/L	0.04
Zinc	µg/L	2

2.3.1 Box and whisker spatial change

Results of the spatial water quality changes within the Wilge River sub-catchment B20F area are presented in the following box and whisker graphs in Figures 2.8-2.10. Variation in yearly data over different sample sites to show temporal changes are also depicted in box and whisker graphs. Refer to Appendix A for the temporal changes in parameters.

The pH of natural waters is determined by both geological and atmospheric influences, as well as by biological activities. Most fresh waters are usually relatively well buffered and more or less neutral, with a pH range from 6 to 8, and most are slightly alkaline due to the presence of bicarbonates of the alkali and alkaline earth metals (DWAF, 1996). The pH values, range from the lowest value of 3.77 at Spring2 to the highest of 9.3 at SW3 (Figure 2.8). Slightly lower pH and overall largest variations in pH readings were found at Spring4 and Spring10.

The pH measured in the study area is similar to the Wilge River IWRMP RWQOs range of 6.5-8.4 that was set out for the Wilge River sub-catchment. The slightly lower pH and overall biggest variations in pH readings, found at Spring4 and Spring10 can be attributed to a two defunct mines present upstream of these sampling points. The water quality downstream of the upper Olifants River catchment is impacted by the discharge of acid water along the Kromdraaispruit. This is evident in the drop in the alkalinity at these sampling sites with the presence of increasing sulphate concentration (DWA, 2009; Dabrowski *et al.,* 2014). The recorded pH is approaching the acidic range during periods of low flow. Acidic conditions can also mobilise metals from the sediments (Calmano *et al.*, 1993). The Wilge River has a crucial mediating effect downstream of its confluence with the Olifants River, improving and restoring water quality and ecological health of the Olifants River up to the Loskop Reservoir. Therefore, it is vital that mining operations, in future, are planned and operated in such a manner that they have minimum effect on water quality and environmental flows in the Wilge River sub-catchment. It is likely that the Loskop Reservoir acts like a sink for metal-enriched sediments washed down the Olifants River during high-flow periods and further research is required to determine the fate of these metals in the reservoir (Dabrowski *et al.*, 2014).

The EC is the measure of the ability of water to conduct an electrical current as a result of the presence of ions such as carbonate, bicarbonate, chloride, sulphate, nitrate, sodium, potassium, calcium and magnesium in the water (DWAF, 1996). Wastewater, industrial and domestic effluents often contain high amounts of dissolved salts. High salt concentrations in effluents can increase the salinity, which may result in adverse ecological effects on the aquatic biota (Fried, 1991). For this reason, EC can serve as a useful indicator of water quality. The EC concentrations were found to exceed the Wilge River IWRMP RWQOs of 40 mS/m at the following sites: Spring6 and SW5 (Figure 2.8). Exceedances were observed at Spring4 (70 mS/m) in April 2010, Spring6 (91 mS/m) in February 2013, Spring10 (133 mS/m) in December 2012, SW1 (65 mS/m) in October 2010, SW5 (72 mS/m) in October 2010, SW6 (46 mS/m) in April 2010 and SW16 (47 mS/m) in August 2014. The high EC can be attributed to a defunct mines present upstream of Spring6 and SW5.

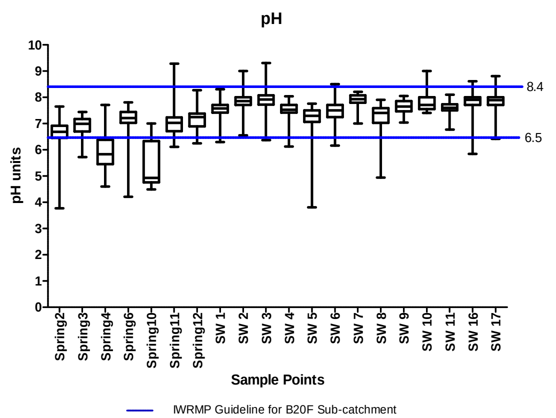
Suspended solids are the measure of the amount of material suspended in water. This includes a wide range of sizes of material, from colloids to large organic and inorganic particulates. The concentration of suspended solids increases with the discharge of sediment washed into rivers due to rainfall and resuspension of deposited sediment. As flow decreases, suspended solids settle out. Increases in suspended solids may also result from anthropogenic sources, including (a) discharge of domestic waste, (b) discharge of industrial effluents (i.e.pulp/paper mill, chin-clay, and brick and pottery industries), (c) discharge from mining operations and, (d) physical changes from the road, bridge and dam construction (DWAF, 1996). Suspended solids were found to be high at sample sites Spring12, SW1, SW4, SW7, and SW11 (Figure 2.8). An EC concentration of 3678 mg/L was observed at SW5 in April 2014. As most sample sites are downstream of Kusile's construction site increased EC can be partly attributed to the construction site and to a certain extent the quarry located in the vicinity of Spring2.

Turbidity refers to the "optical property that causes light to be scattered and absorbed rather than be transmitted in a straight line" (Davies and Day, 1998). Turbidity impacts on the clarity of the water and is due to the amount of suspended solids found in the water column. Sample sites with high turbidity readings were SW1, SW2, SW4, SW6, SW7, SW8, SW9, SW10, SW11 and SW17 (Figure 2.8). Exceedances were observed at SW1 (1210 NTU) in November 2011, SW2 (658 NTU) in November 2011, SW4 (978 NTU) in December 2010, SW6 (673 NTU) in April 2012, SW7 (834 NTU) in October 2012, SW8 (754 NTU) in January 2013, SW9 (983 NTU) in November 2013, SW10 (621 NTU) in September 2012, SW11 (2050 NTU) in September 2012 and SW17 (788 NTU) in October 2013. The highest turbidity readings were recorded at SW11 in 2012. Activities that may contribute to increased turbidity in the study area are the construction site, quarry, and livestock grazing. Buffer zones along and around all water resources should be identified and demarcated before activities are undertaken to allow for decreased turbidity impact to the water resources. Planning of site operation should ensure that all vegetation to not be stripped completely before construction commences in order to reduce turbidity. Drainage channels should be repaired with non-erodible soil (stony material with little soil) to prevent erosion problems from occurring at future construction sites and reduce the impacts of turbidity.

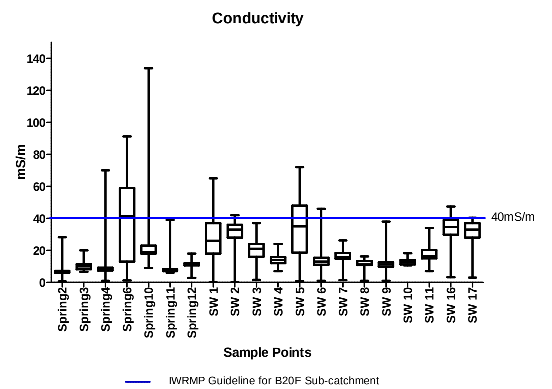
The maintenance of adequate DO is critical for the survival and functioning of the aquatic biota because it is required for the respiration of all aerobic organisms. Therefore, DO concentration provides a useful measure of the health of an ecosystem (DWAF, 1996). The median guideline for DO for the protection of aquatic biota is >5 mg/L (Kempster *et al.*, 1980). Majority of the DO values were found to be between 4.5-8 mg/L and are therefore not expected to have a limiting effect on aquatic biota. The Wilge River IWRMP RWQOs for DO is 70% saturation. Majority of the DO values were found to range between 40-80% saturation, in the study area (Figure 2.8). A 261% saturation reading of DO at sample site Spring2, in July 2011 may be an outlier. Sample sites with low DO concentrations were found at Spring12, Spring10, Spring3 and SW6.

Lowest DO values of 0.05% saturation were found at both Spring12 and SW6 in January 2012 and only at Spring12 in June 2011.

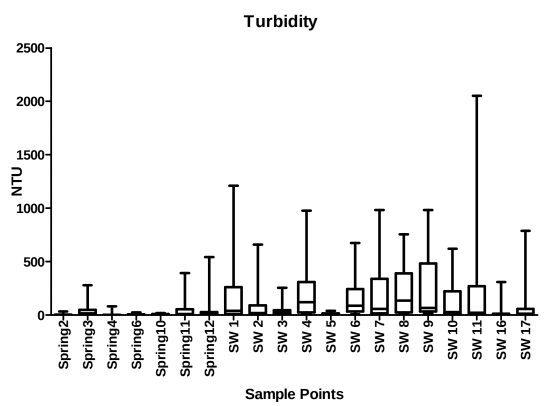
Ammonia is a pungent colourless gaseous compound of nitrogen and hydrogen that is very soluble in water and can easily be condensed into a liquid by cold and pressure. Ammonia reacts with NOx to form ammonium nitrate. It is a product of microbiological decay of plant and animal protein. Presence of ammonia in surface waters usually indicates domestic or agricultural pollution, for example, animal waste runoff. The discharge of effluent streams containing animal and human excrement, agricultural fertilisers and organic industrial wastes are the major sources of ammonia which enter the aquatic systems (DWAF, 1996). Majority of ammonia concentrations were found to be <0.5 mg/L of N (Figure 2.8). The Wilge River RQWO for ammonia is 0.007 mg/L of N. Exceedances were observed at Spring2 (5.39 mg/L) in February 2012, Spring6 (1.84 mg/L) in December 2014, and SW4 (1.82 mg/L) and SW8 (2.31 mg/L) in September 2014. An increase in ammonia was found in February 2012 (5.39 mg/L) and April 2012 (3.59 mg/L) at Spring 2 which resulted in the boxplot showing high variation. High ammonia readings were found at sample sites SW1 (0.70 mg/L), SW11 (0.69 mg/L) and SW17 (0.68 mg/L) in October 2012. Due to the sporadic increases in ammonia the spikes could be due to sewage spills from septic tanks, pit latrines or stockwatering areas.



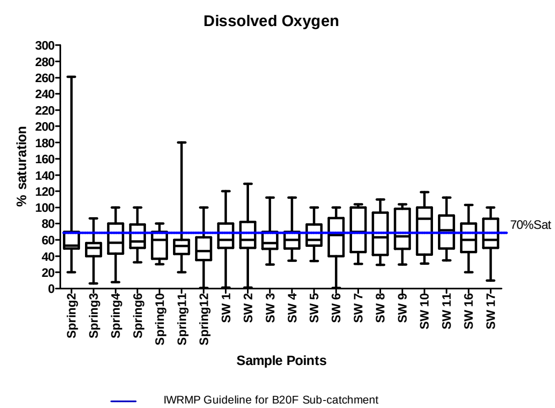
pH



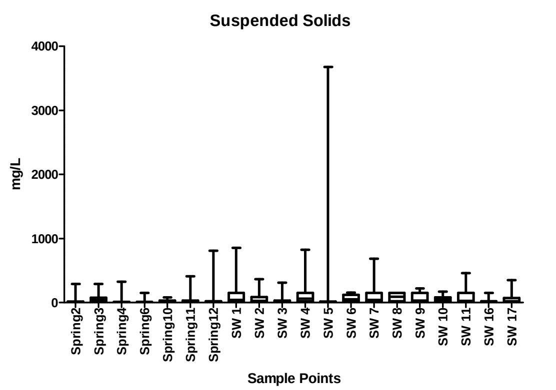
Electrical Conductivity



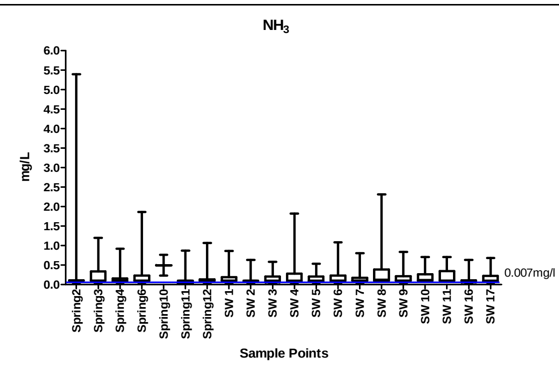
Turbidity



Dissolved Oxygen



Suspended Solids



Ammonia

Figure 2.8: Graphical representation of the Box and Whisker Graphs for pH, electrical conductivity, turbidity, dissolved oxygen, suspended solids and ammonia (2008-2014). Box plots represent the mean and upper and lower quartiles while the whiskers represent the 5th and 95th percentiles. The line represents the Integrated Water Resource Management Plan Resource Water Quality Objectives for each parameter when available.

A nitrate is any compound containing the nitrate group (such as a salt or ester of nitric acid) that can exist in the atmosphere or in water and that can have harmful effects on humans and animals at high concentrations. It is the most completely oxidised state of nitrogen found in water. High nitrate levels can occur naturally, but may indicate biological wastes in the water, run-off from heavily fertilised fields or industrial waste eflluents. The presence of nitrate in water bodies arises mostly from the use of fertilisers in agriculture and any nitrate not taken up by crops is likely to be dissolved in rainwater and either percolates down into groundwater or runs off into streams (DWAF, 1996). The RQWO for nitrate is 6 mg/L. Nitrate concentrations were found to exceed the Wilge River IWRMP RWQOs and show high variation, ranging from 2-18 mg/L at Spring10 (Figure 2.9). Exceedances were found at Spring2 (24.4 mg/L) in April 2012; Spring4 (16.6 mg/L) in October 2009; Spring10 (20.5 mg/L) in July 2011 and SW16 (11.7 mg/L) in July 2013.

Phosphate is a salt of phosphoric acid. Phosphates enter the water supply from domestic and industrial effluents, atmospheric precipitation and urban runoff. In particular, they drain from agricultural land, on which fertilisers have been applied. Phosphates are necessary for biological growth of aquatic plants, but too much phosphate causes excessive growth of aquatic plants and eutrophication of lakes (DWAF, 1996). The Wilge River IWRMP RWQOs for phosphate is 0.05 mg/L. Sample sites Spring4, SW6, SW16 and SW17 showed higher phosphate concentrations, with more readings exceeding the Wilge River IWRMP RWQOs (Figure 2.9). Phosphate concentrations were observed to display exceedances at Spring2 (5.38 mg/L), Spring12 (4.99 mg/L), SW1 (1.42 mg/L) and SW6 (3.35 mg/L) in February 2012. Sample sites SW6, SW16 and SW17 are located in the vicinity of the livestock farm.

Eutrophication resulting from nutrient (N-P) enrichment is globally considered to be one of the most serious threats to freshwater ecosystem services such as water quality and biodiversity. The principal anthropogenic point sources of inorganic nitrogen and phosphorus in aquatic ecosystems are: municipal sewage effluents and overflows of storm and sanitary sewers, wastewater from livestock farming, industrial wastewater effluents, and runoff from waste disposal sites, working mines and unsewered industrial sites. The main anthropogenic diffuse sources are: agricultural activities (use of manure and nitrogenous fertilizers, cultivation of N2-fixing crops), runoff from nitrogen saturated and burned forests and grasslands, urban runoff from unsewered, sewered and failed septic systems, runoff from construction sites and abandoned mines, polluted ground waters, anthropogenic atmospheric deposition loads (such as from fossil fuel combustion) and biomass burning. Globally, the dominant source of the increase, by a factor of about 4, in nutrient levels are widespread agricultural intensification and increased discharge of domestic wastes (De Villiers and Thiart, 2007). Nutrient concentration at Spring4 is indicative of potential eutrophic conditions.

Sulphate is essentially a salt of sulphuric acid. Sulphate is a natural forming mineral. When naturally occurring, they are often the result of the breakdown of leaves that fall into a stream, of water passing through rock or soil containing gypsum and other common minerals, or of atmospheric deposition. Point sources include sewage treatment plants and industrial discharges such as tanneries, pulp mills, and textile mills. Runoff from fertilised agricultural lands also contributes sulphates to water bodies (Davidson, 2000). The Wilge River IWRMP RWQOs for sulphate for the Wilge River sub-catchment is 60 mg/L. Sulphate concentrations found to exceed the Wilge River IWRMP RWQOs most often were at Spring6, Spring10, SW1, SW5, SW16 and SW17 (Figure 2.9). The highest sulphate reading of 690 mg/L was found at Spring10 in October 2010. Exceedances were observed at Spring4 (255 mg/L) in April 2010, Spring6 (516 mg/L) in February 2013, Spring10 (690 mg/L) in October 2010, Spring11 (128 mg/L) in April 2011, SW3 (153 mg/L) in September 2012 and SW6 (168 mg/L) in April 2010. Spatially, SW17, SW1 and SW5 all occur downstream of Spring6 and Spring10. This indicates that the sulphates entering the Klipfonteinspruit at Spring10, is being carried downstream, via Spring6, SW5, SW1, and SW17 into the Wilge River.

An increase in sulphate concentrations from upstream to downstream, as found in studies in the entire upper Olifants River catchment (Dabrowski and De Klerk, 2013) as well indicates that mining activities in the Wilge River sub-catchment have a progressively greater impact on water quality with increasing distance downstream. SW16 is located on the Wilge River upstream of the confluence between Klipfonteinspruit and the Wilge River, indicating that there is already high levels of suphate present in the Wilge River due to upstream impacts, prior to the confluence with the Klipfonteinspruit. At the Loskop Dam dissolved sulphate concentrations have increased more than 7-fold since the 1970s evidently due to increasing levels of pollution within the upper Olifants River catchment (De Villiers and Mkwelo, 2009).

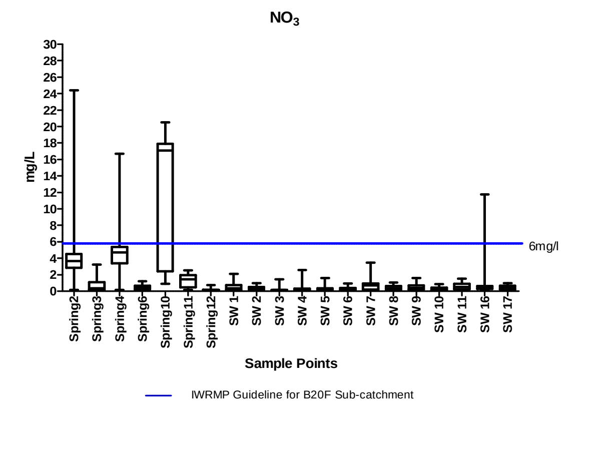
Aluminium is the third most abundant element in the earth's crust and constitutes 7.3% by mass. In nature, however, it only exists in very stable combinations with other materials (particularly as silicates and oxides). Aluminium is one of the principal particulates emitted from the combustion of coal, and aluminium fluoride is emitted from aluminium smelters. Industries using aluminium in their processes or in their products include the following: the paper industry, the metal construction industry, and the textile industry (DWAF, 1996). The Wilge River IWRMP RWQOs for aluminium is 0.02 mg/L. Aluminium concentrations were found to be higher than the Wilge River IWRMP RWQOs at Spring10, Spring11, Spring12, SW1, SW2, SW3, SW4, SW6, SW7, SW8, SW9, SW10, SW11 and SW17 (Figure 2.9). Exceedances were observed at Spring6 (13 mg/L) in February 2013, Spring12 (6.34 mg/L) in December 2009, SW1 (4.48 mg/L) in November 2013, SW2 (5.69 mg/L) in December 2009, SW3 (16 mg/L) in January 2012, SW4 (15 mg/L) in December 2009, and SW6 (7.92 mg/L) in January 2012. Aluminium contamination across the region in plants is partially caused by acidification of the soil by mining practises, which leach aluminium from the soil into the water, creating an uptake pathway for plants (Kok *et al.*, 2013).

Iron is a metal often found in waters. It is particularly a problem in ground water supplies, where the water is acidic and has passed through some iron bearing rock i.e. sulphide ores, igneous, sedimentary and metamorphic rock. The dissolved iron usually takes the form of ferric sulphate, which at pH values above 3.0 may become hydrolysed and form iron hydroxide. It is usually the occurrence of iron hydroxide rather than the iron itself that kills fish. Iron is also released into the environment by human activities, mainly from the burning of coke and coal, acid mine drainage, mineral processing, sewage, landfill leachates and the corrosion of iron and steel. Various industries that also use iron in their processes, or in their products, include: the chlor alkali industry, the household chemical industry, the fungicide industry and the petro-chemical industry (DWAF, 1996). Sample sites that showed iron values higher than the Wilge River IWRMP RWQOs of 1 mg/L were at sample sites, SW4, SW6, SW7, SW8, SW9 and SW10 (Figure 2.9). Eceedances were observed at Spring3 (137 mg/L) and SW11 (74 mg/L) in November 2014.

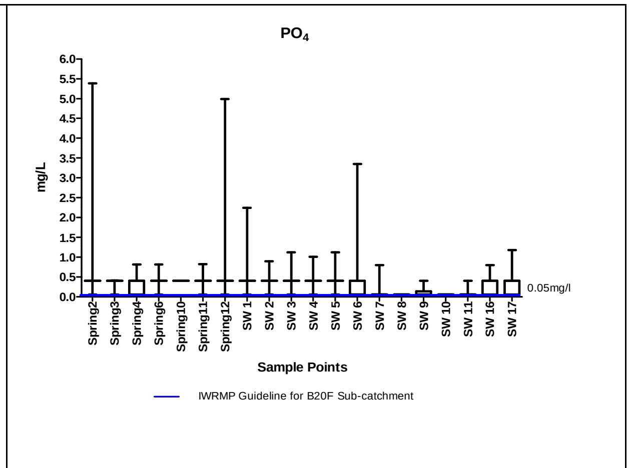
Manganese is a gray-white metal, resembling iron. It is a hard metal and is very brittle, fusible with difficulty, but easily oxidised. Manganese is rather electropositive and combines with some non-metals when heated. Soil sediments and metamorphic and sedimentary rocks are significant natural sources of manganese. Industrial discharges also account for elevated concentrations of manganese on receiving waters. Various industries use manganese, its alloys, and manganese compounds in their processes, or in their products of which include the steel industry (in the manufacture of dry cell batteries), the fertiliser industry (manganese is used as a micro-nutrient fertiliser additive), and the chemical industry in paints, dyes, glass, ceramics, matches and fireworks) (DWAF, 1996). Sample sites that showed manganese readings higher than the Wilge River IWRMP RWQOs of 0.18 mg/L were Spring6, Spring10, Spring12, SW1, SW5, SW7, SW8, SW9 and SW16 (Figure 2.9). Exceedances were found at Spring2 (0.9 mg/L) in April 2012, Spring3 (5.9 mg/L) in November 2014, Spring6 (0.91 mg/L) in September 2012, Spring10 (1.77 mg/L) in October 2010, Spring11 (1.4 mg/L) in April 2012, SW1 (0.71 mg/L) in September 2013, SW3 (1.51 mg/L) in September 2014, SW4 (2.38 mg/L) in July 2014, SW5 (1.58 mg/L) in October 2013, SW8 (0.72 mg/L) in February 2013 and SW9 (0.5 mg/L) in November 2012.

The cause of elevated metals (Al, Fe and Mn) and suspended solids at the numerous sample sites may be caused by the mineralisation and dissolution of geological formations and casing material. When comparisons are made to historical baseline data and hydrocensus data, results indicated that these metals were elevated since the pre-construction as well as outside of the site boundaries of groundwater (Kok *et al.*, 2013).

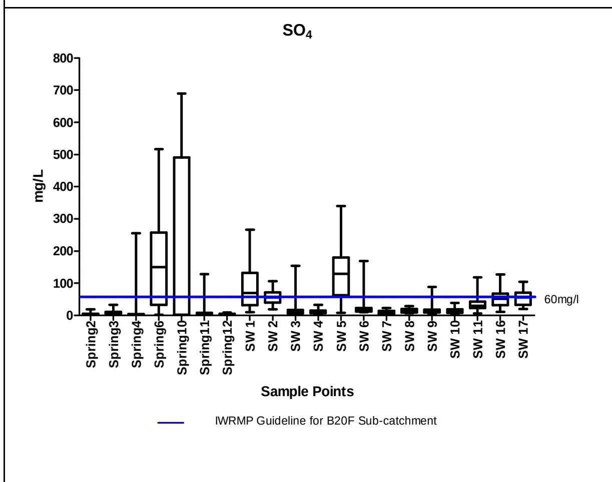
The quality of the mine water varies depending on the local geology. Associated with the low pH waters are iron, aluminium, and manganese. Mine water is generally high in dissolved solids with sulphate the dominant or indicator anion and calcium and magnesium the cations (Oberholster *et al*., 2011). Elevated concentrations of sulphate and manganese were found at Spring10. These, in addition to the low pH values, at Spring10, are indicative of mining impact. Spring10 is downstream of the closed New Largo Colliery. Mining to the east of Kusile site is impacting on the groundwater quality and it is shown from historical data that decant from New Largo mine takes place within the Wilge River sub-catchment B20F area and will affect water quality negatively over time as the pollution plume migrates towards the east (Kok *et al.*, 2013).



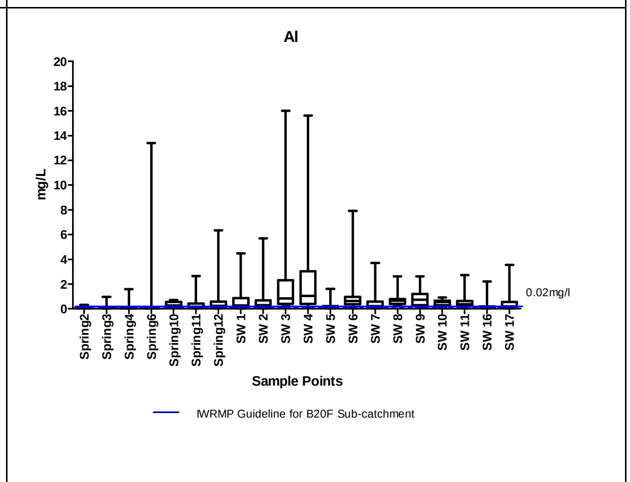
Nitrate



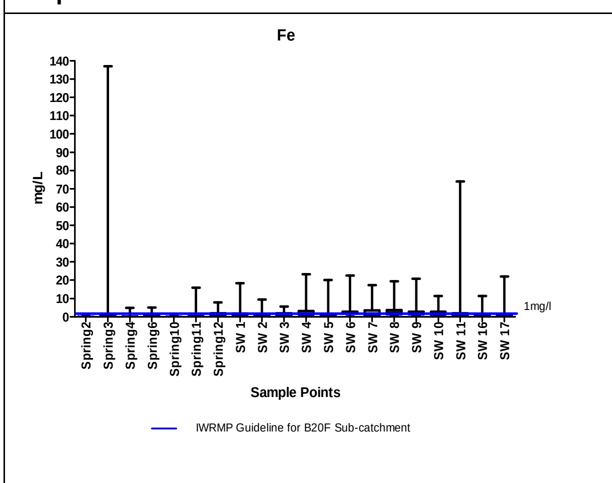
Phosphate



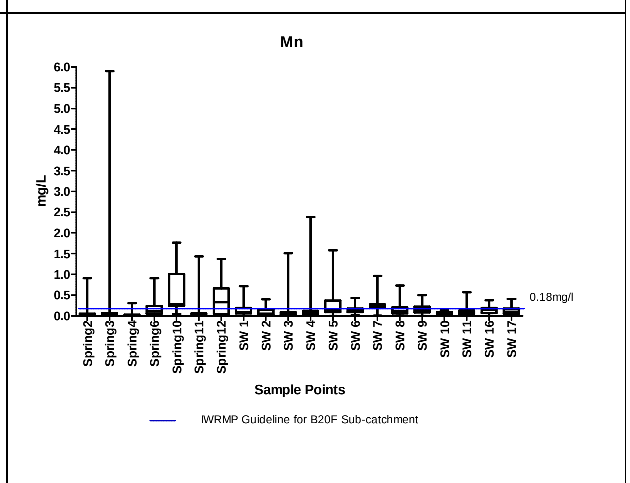
Sulphate



Aluminium



Iron



Manganese

Figure 2.9: Graphical representation of the Box and Whisker Graphs for nitrate, phosphate, sulphate, aluminium, iron and manganese (2008-2014). Box plots represent the mean and upper and lower quartiles while the whiskers represent the 5th and 95th percentiles. The line represents the Integrated Water Resource Management Plan Resource Water Quality Objectives for each parameter when available.

Zinc occurs in rocks and ores and is readily refined into a pure stable metal. It can enter aquatic ecosystems through both natural processes such as weathering and erosion, and through industrial activity. Soluble zinc salts (i.e., zinc chloride and zinc sulphate) or insoluble precipitates of zinc salts (i.e., zinc carbonate, zinc oxide and zinc sulphide) occur readily in industrial wastes. The carbonate, hydroxide and oxide forms of zinc are relatively resistant to corrosion and are used extensively in metal galvanising, dye manufacture and processing, pigments (paints and cosmetics), pharmaceuticals, fertilizer and insecticide industries (DWAF, 1996). Zinc concentrations were found to exceed the DWA TWQO throughout the study area (Figure 2.10). Exceedances were observed at Spring6 (351 µg/L) in February 2013, SW11 (310 µg/L) in April 2014 and SW17 (374 µg/L) in April 2012. The DWA TWQO for zinc is 2 µg/L.

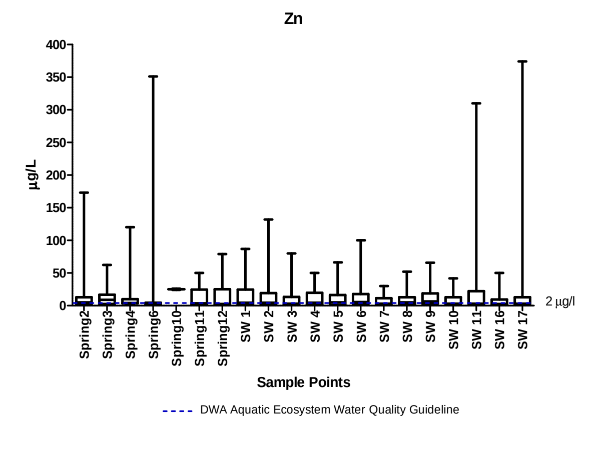
The bromine concentrations in natural ecosystems are usually very small. The three major manmade releases of Br and Br- into the environment are from mining, from emmisions of 1,2 dibromoethane, a scavenger in leaded fuel, and from the use of fertilizers and pesticides in agriculture. Bromine is released into the environment as a waste product of K mining (Flury and Papritz, 1993). There is no DWA TWQO for bromine. The highest bromine concentrations were found at sample sites Spring3 and Spring12 (Figure 2.10). Exceedances were observed at Spring3 (360 µg/L) in March 2014, (320 µg/L) in December 2014 and (230 µg/L) in February 2014. At Spring12 (350 µg/L) in December 2014, SW2 (230 µg/L) in April 2014, SW7 (147 µg/L) in September 2013, SW16 (138 µg/L) in April 2013 and SW17 (145 µg/L) in April 2013.

Cadmium is a metal element which is highly toxic to marine and freshwater aquatic life. Cadmium is present in the earth's crust at an average concentration of 0.2 mg/kg, usually in association with zinc, lead and copper sulphide ore bodies. Due to its abundance, large quantities of cadmium enter the global environment annually as a result of natural weathering processes. Cadmium is found at trace concentrations in fresh waters and mostly a result of industrial activity. The main sources of cadmium in the environment are due to: emissions to air and water from mining, metal (zinc, lead and copper) smelters, and industries involved in manufacturing alloys, paints, batteries and plastics, agricultural use of sludges, fertilizers and pesticides containing cadmium, burning of fossil fuels (very limited effect), and the deterioration of galvanized materials and cadmium-plated containers (DWAF, 1996). The DWA TWQO for cadmium (60-119 mg $CaCO_3$/L) is 0.25 µg/L. Cadmium concentrations were found to exceed the DWA

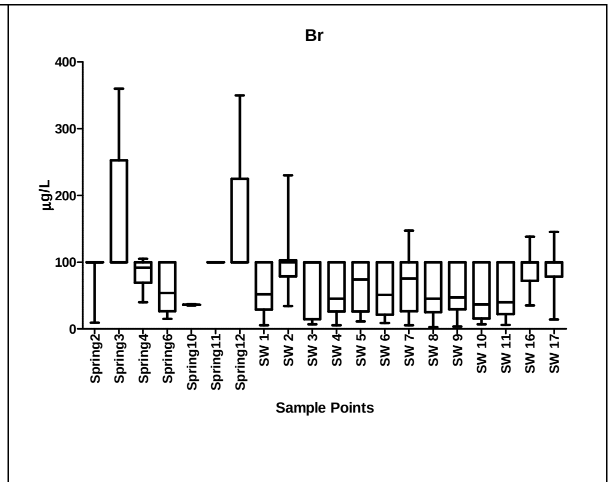
TWQO at Spring4, Spring12, SW1, SW5, SW6, SW8, SW9, SW11 and SW16 (Figure 2.10). A potential outlier was observed at SW6 (7.19 µg/L) in December 2014.

Lead is defined by the USEPA, (2015) as potentially hazardous to most forms of life, and is considered toxic and relatively accessible to aquatic organisms. Lead is principally released into the aquatic environment through the weathering of sulphide ores, especially galena. Since metallic lead and common lead minerals such as sulphides, sulphates, oxides, carbonates and hydroxides are almost insoluble, levels of dissolved lead (acetate and chloride salts) in aquatic ecosystems are generally low. Most of the lead entering aquatic ecosystems are associated with suspended sediments, while lead in the dissolved phase is usually complexed by organic ligands. The photolysis of lead compounds is an important process in the removal of lead from the atmosphere. The products of this photo-degradation are lead oxides and halides, which enter the aquatic ecosystems via direct deposition or surface runoff. The major sources of lead in the aquatic environment are anthropogenic and these include: precipitation, fallout of lead dust and street runoff (associated with lead emissions from gasoline-powered motor vehicles); industrial and municipal wastewater discharge; mining, milling and smelting of lead and metals associated with lead, e.g. zinc, copper, silver, arsenic and antimony; and combustion of fossil fuels (DWAF, 1996). The DWA TWQO for lead (60-119 mg $CaCO_3$/L) is 0.5 µg/L. Lead concentratons that were found to exceed the DWA TWQO at Spring12, SW1, SW2, SW3, SW4, SW6, SW7, SW8, SW9, SW10, SW11 and SW17 (Figure 2.10). Exceedances were observed at Spring2 (42 µg/L) in April 2012, Spring4 (12.3 µg/L) in February 2014, SW1 (27 µg/L) in November 2013, SW9 (13 µg/L) in November 2013 and SW11 (11 µg/L) in September 2012.

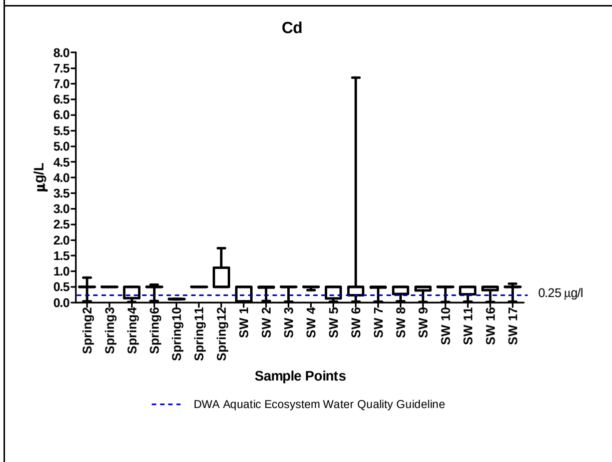
Mercury is a heavy metal that is of quite rare geological occurrence, and its concentration in the environment is normally very low. Mercury and mercury-organic complexes are of concern in the natural aquatic environment because of their extreme toxicity to aquatic organisms and the potential to bio-accumulate in the food chain. Intake of mercury can occur via air, food and water. Mercury may occur at high concentrations in water bodies subject to industrial pollution, or in the vicinity of industrial activities utilising or discharging mercury or compounds thereof. Important industries that use mercury in their processes, or in their products, include: the chlor-alkali industry, the paint industry, the fungicide industry, the paper and pulp industry, medical and dental industries, and the electrical equipment industry. Mercury has a strong affinity for sediments and suspended solids. Under anaerobic conditions, bacteria readily transform inorganic mercury into methyl mercury. Dissolved mercury salts are also easily absorbed by aquatic organisms and can be bio-accumulated. Methyl mercury, the most common form of mercury found in aquatic organisms, is lipid soluble (readily passes through plant and animal membranes) and is stored within the bodies of organisms. In aquatic animals, bio-accumulated mercury is stored in fatty tissues, whilst in aquatic plants, mercury is usually stored in roots and stems (DWAF, 1996). The DWA TWQO for mercury is 0.04 µg/L. Mercury concentrations were found to exceed the DWA TWQO throughout the study area (Figure 2.10). Exceedances were observed at SW2 (5.2 µg/L) in March 2012, SW7 (5.8 µg/L) in November 2014, SW8 (2.7 µg/L) in June 2012, and SW9 (1.8 µg/L), in June 2012.



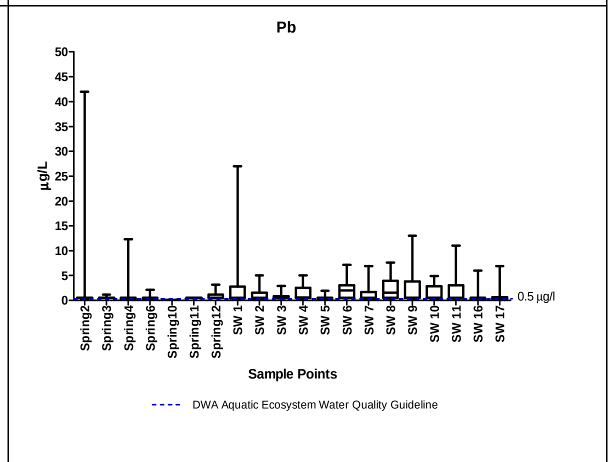
Zinc



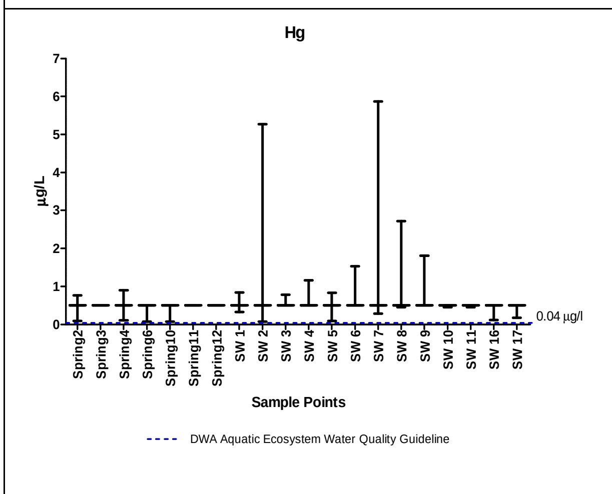
Bromine



Cadmium



Lead



Mercury

Figure 2.10: Graphical representation of the Box and Whisker Graphs for zinc, bromine, cadmium, lead and mercury (2008-2014). Box plots represent the mean and upper and lower quartiles while the whiskers represent the 5th and 95th percentiles. The line represents the DWA Target Water Quality Resource Water Quality Objectives for each parameter when available.

2.3.2 Box and whisker annual change

The pH in the Wilge River sub-catchment is mostly stable with high variation in pH occurring in 2008-2009 at sample sites Spring10, Spring3, SW3 and SW4 (Appendix A). During 2012 high variation in pH occurred at sample sites SW4, Spring2, SW3, SW1, SW5, and SW2. The highest EC concentrations were found at Spring6 in 2008-2009 and in Spring10 in 2010-2011. The highest turbidity readings were found at sample sites SW11, SW1, SW9, SW7, SW4, SW17, SW4 and SW8, occurred during 2012-2013. Dissolved oxygen was found to decrease at most sample sites from 2013 to 2014. The highest variation in DO readings was found at most sample sites in 2012. The highest suspended solids concentrations were found at sample sites SW4 and SW1. Both sampling points showed an increase in suspended solids in 2009 and a slow deterioration in suspended solids concentration till 2014.

The highest ammonia concentrations were found at sample sites Spring2 in 2012. Sample sites Spring2, SW9, SW5, SW3, SW11, SW16 and SW17 showed high variation in 2012. Highest nitrate concentrations were found at sample sites Spring2 in 2011-2012 and Spring10 in 2009-2011. Majority of the nitrate readings ranged from 7-17 mg/L from 2009-2011 at Spring10 and 2-20 mg/L at Spring2 in 2012 only. Highest phosphate concentrations were found at sample sites Spring2 and SW6 in 2012 and Spring12 in 2010. Sample sites Spring2, Spring6, SW1, SW2, SW3, SW4, SW5, SW6, SW16 and SW17 showed an increase in phosphate 2012. Highest sulphate concentrations were found at sample sites Spring10 from 2010-2011 and at Spring6 from 2008-2009. High sulphate concentrations (ranging from <0.15-500 mg/L) were found at sample sites SW1 and SW5 in 2010. Highest aluminium concentrations were found at sample sites SW4 and SW3. Sample site SW4 showed a decrease in aluminium concentrations from 2009 to 2014. Sample site SW3 showed high aluminium concentrations from 2008-2012. Aluminium concentrations decreased from 2013-2014 at SW3 and SW4. Highest iron concentrations were found at sample sites Spring3, SW4, SW8 and SW9. Sample site SW4 showed highest iron concentration in 2008 and a slow deterioration in iron concentration till 2013. Sample sites Spring3, SW4, SW8 and SW9 all showed an increase in iron concentration in 2014. Highest manganese concentrations were found at sample site SW4 and SW5. Sample site SW4 showed the highest variation in manganese concentrations in 2014. Sample site SW5 showed an increase in manganese concentrations from 2008 to 2013. Manganese concentrations were showed to decrease in 2014 at sample site SW5.

Highest zinc concentrations were found at Spring2 in 2012. Highest variations in zinc concentrations were found at SW11, SW9, SW2, SW1, Spring2 in 2014. SW5 and Spring6 showed high variation of data in 2013. Highest bromine concentrations were found at sample site Spring3 and Spring12 in 2014. Bromine concentrations were found to increase from 2012 to 2013 at sample sites Spring4, Spring6, SW1, SW2, SW4, SW5, SW6, SW7, SW8, SW9, SW10, SW11, SW16 and SW17. The highest cadmium concentrations were found at sample site Spring12 in 2014. The highest lead concentrations were found at sample site SW11 from 2013-2014. High mercury concentrations were found at sample sites SW2, SW8 and SW9 in 2012 and, at sample site SW7 in 2014.

2.3.3 Principle component analyses

PCA table was developed based on mean parameter readings for each site for the entire sampling period 2008-2014 (Figure 2.11). Refer to Appendix B, Tables 1-8 for mean and standard deviation used in the development of the PCA plots. The first and second axes explain 54.2% and 19.6% of the variation respectively. On the first axis turbidity, suspended solids, iron, aluminium and pH showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to turbidity, suspended solids and pH were found at sample sites SW6, SW9, SW4, SW8, SW11, and SW3. These sample sites also showed an association to mercury concentrations. A significant negative loading was found to these sample sites with the following parameters, phosphate, nitrate and ammonia concentrations. On the second axis, EC and sulphate showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to EC and sulphate was found at sample site SW5. A significant negative loading was found to these sample sites with bromine concentrations.

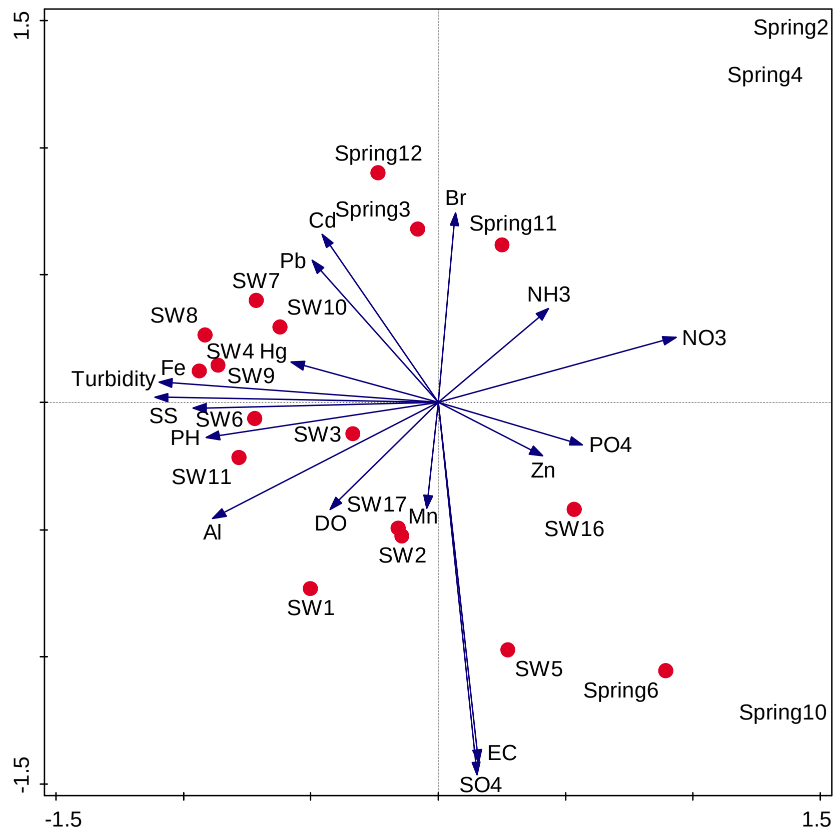


Figure 2.11: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2008-2014. The bi-plot represents 78.3% of the total variation in the data.

A PCA table was developed based on mean parameter readings for each site yearly. (i.e. 2008, 2009, 2010, 2011, 2012, 2013 and 2014 individually) showing temporal changes in data (Figure 2.12-2.18). Refer to Appendix B for mean and standard deviation tables used in the development of the yearly PCA plots. For the PCA plot for 2008 (Figure 2.12), the first and second axes explain 54.6% and 33.6% of the variation respectively. On the first axis aluminium and suspended solids are shown to be co-linear and depicted a strong relationship to each other. A significant positive loading to aluminium and suspended solids were found at sample sites SW4 and SW3. These sample sites also showed an association to manganese and iron concentrations although the managese contribution to the component loading is low. On the second axis, a strong co-linear relationship was found between EC and sulphate concentrations. A significant positive loading to nitrate concentrations were found at sample sites

Spring11 and Spring2. A significant negative loading was found to these sample sites with pH.

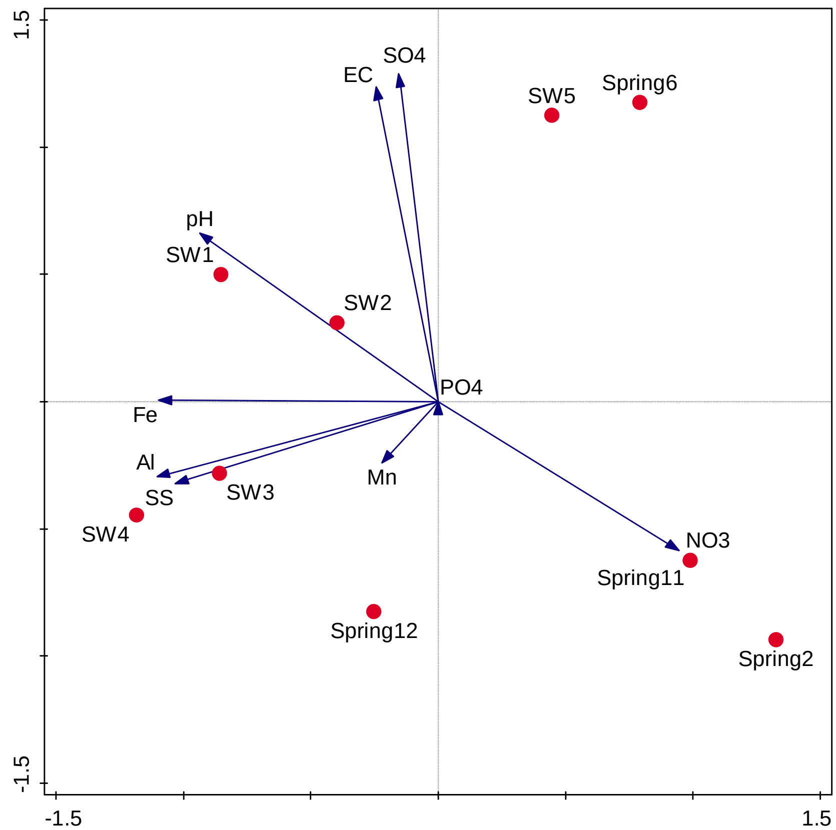


Figure 2.12: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2008. The bi-plot represents 88.21% of the total variation in the data.

For the PCA plot for 2009 (Figure 2.13), the first and second axes explain 57.8% and 19.7% of the variation respectively. On the first axis aluminium, suspended solids, iron and turbidity are shown to be co-linear and depicted a strong relationship to each other. A significant positive loading to these parameters were found at sample sites SW4 and SW3. These sample sites also showed an association to manganese concentrations. On the second axis, a strong co-linear relationship was found between EC and sulphate concnetrations. A significant positive loading to nitrate concentrations were found at sample sites Spring11, Spring10 and Spring3. A significant negative loading was found to these sample sites with pH.

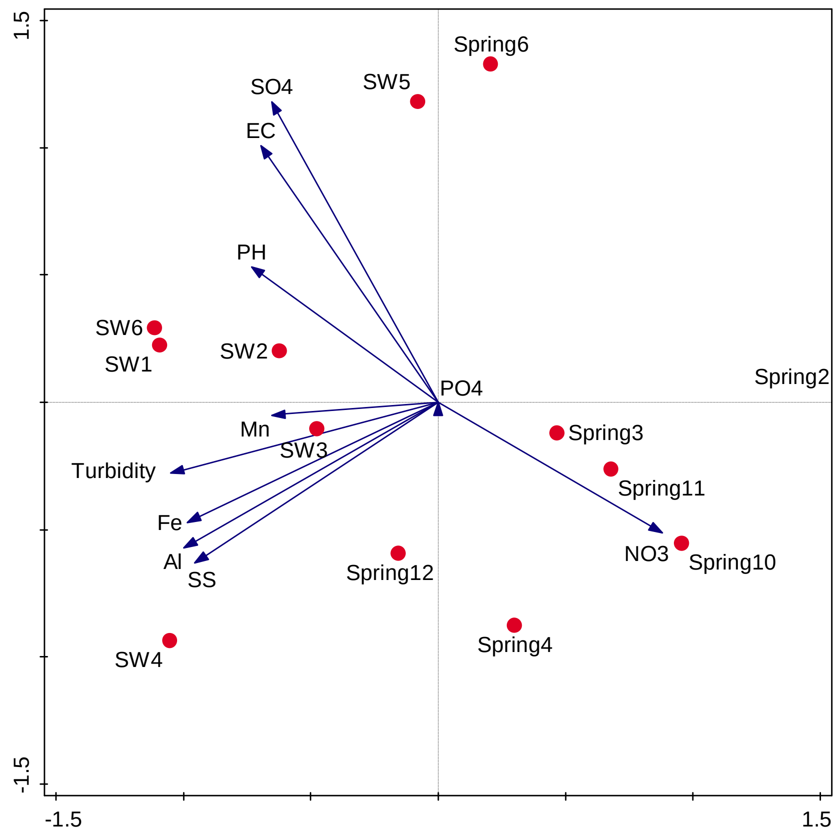


Figure 2.13: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2009. The bi-plot represents 77.51% of the total variation in the data.

For the PCA plot for 2010 (Figure 2.14), the first and second axes explain 44.3% and 29.6% of the variation respectively. On the first axis aluminium, iron and turbidity are shown to be co-linear and depicted a strong relationship to each other. A significant positive loading to these parameters were found at sample sites SW3, SW9, SW2, Spring12 and SW4. These sample sites also showed an association to SS, pH and phosphate concentrations. A significant negative loading was found to these sample sites with nitrate concentrations. On the second axis a strong co-linear relationship was found between EC and sulphate concnetrations. A significant positive loading to these parameters were found at sample sites SW5 and SW6.

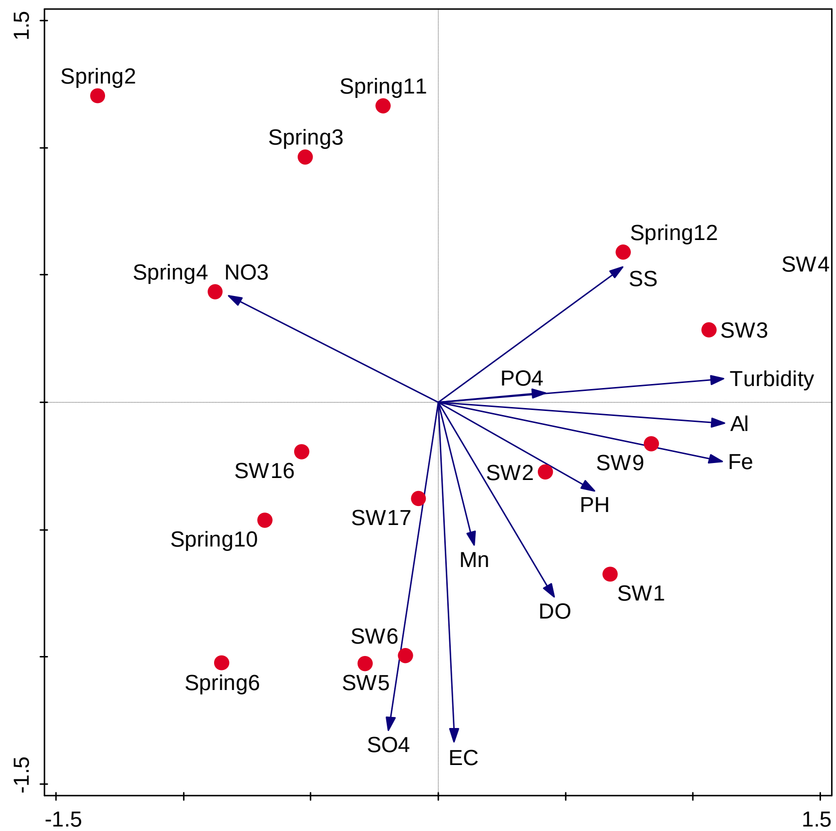


Figure 2.14: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2010. The bi-plot represents 73.84 % of the total variation in the data.

For the PCA plot for 2011 (Figure 2.15), the first and second axes explain 55.6% and 18.5% of the variation respectively. On the first axis turbidity, suspended solids, aluminium and pH showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to these parameters were found at SW6 and SW4. A significant negative loading was found with these parameters with nitrate concentrations. On the second axis, EC and sulphate showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to EC and sulphate were found at sample sites SW5, SW16, SW11 and Spring6. A slight negative loading was found to these sample sites with DO concentrations.

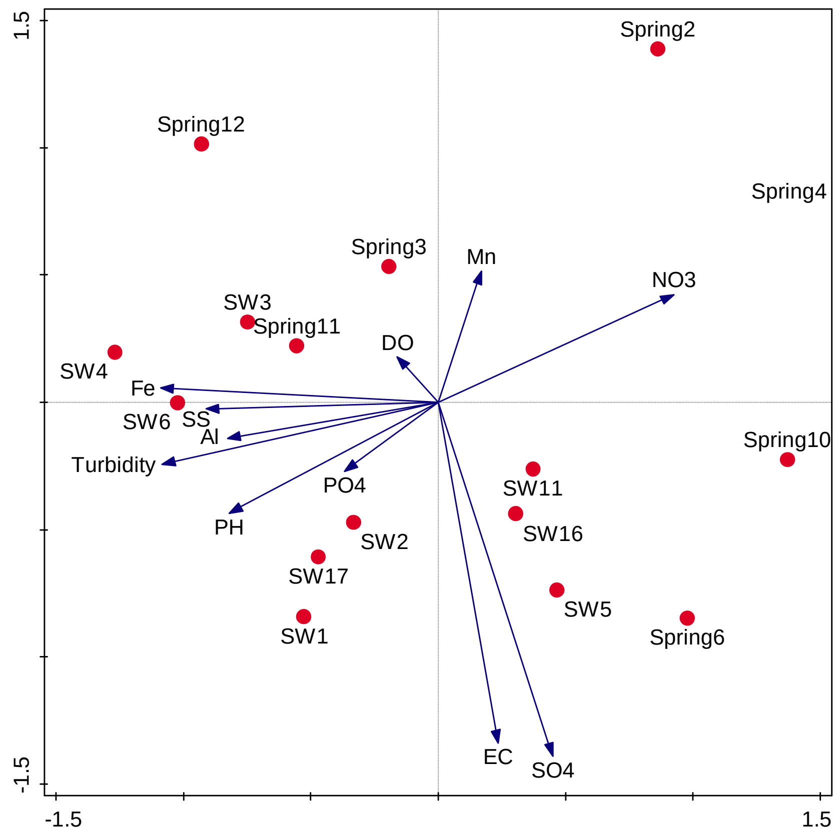


Figure 2.15: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2011. The bi-plot represents 74.09% of the total variation in the data.

For the PCA plot for 2012 (Figure 2.16), the first and second axes explain 44.2% and 19.3% of the variation respectively. On the first axis iron, aluminium, turbidity and suspended solids showed a significant positive loading at sample sites SW4, SW6, SW9, SW3, SW7, SW8, SW1, SW10, SW11, SW2 and SW17. A significant negative loading was found to these sample sites with nitrate, manganese and cadmium concentrations. On the second axis, EC, sulphate and ammonia showed to be co-linear and depicted a strong relationship to each other. A significant negative loading was found at sample sites Spring12, Spring3 and Spring11 with phosphate and zinc concentrations.

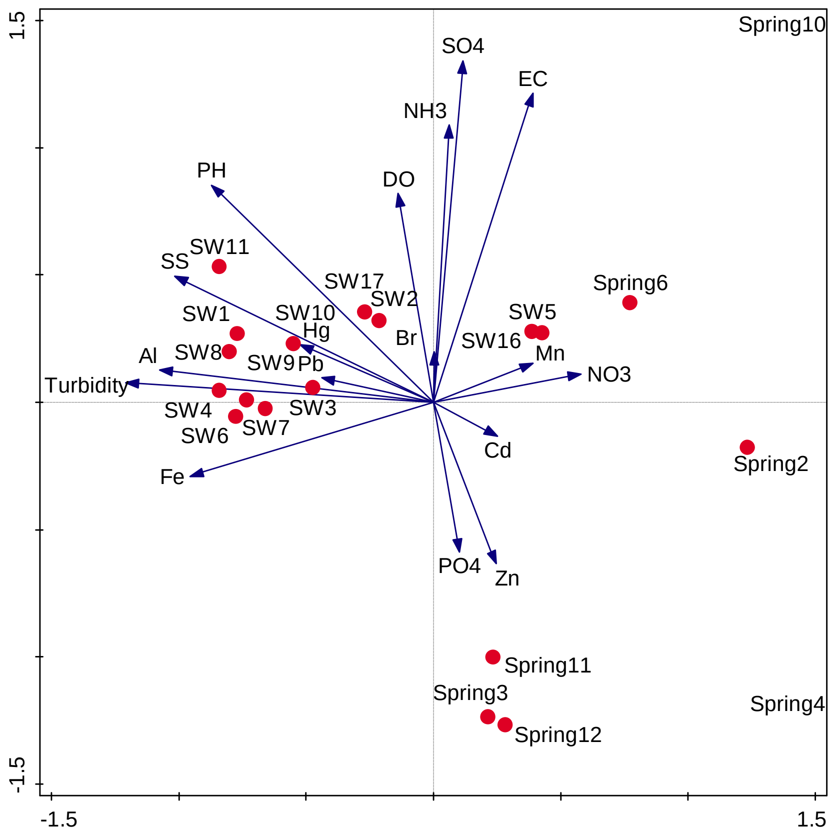


Figure 2.16: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2012. The bi-plot represents 63.48% of the total variation in the data.

For the PCA plot for 2013 (Figure 2.17), the first and second axes explain 57.9% and 25.2% of the variation respectively. On the first axis turbidity and lead showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to these parameters were found at sample sites SW4, SW7, SW9, SW6, SW8, SW11, SW10 and SW17. These parameters showed a strong association to suspended solids, DO and pH. On the second axis sulphate, manganese showed to be co-linear and depicted a strong relationship to each other. These parameters showed a strong association to zinc and EC. A significant positive loading to sulphate were found at sample sites SW5 and Spring6.

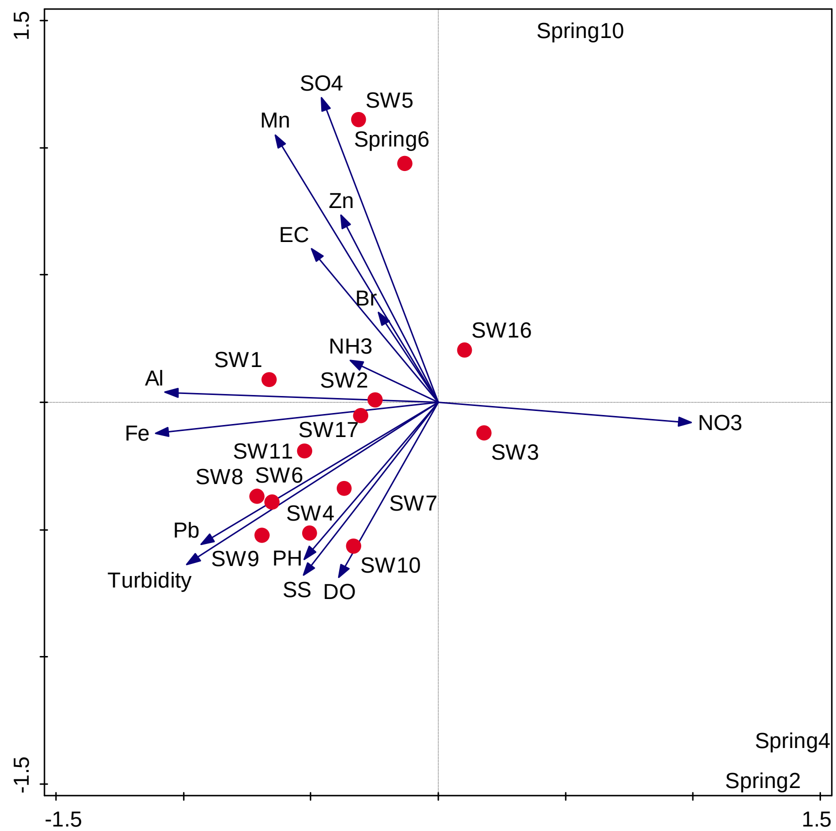


Figure 2.17: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2013. The bi-plot represents 83.11% of the total variation in the data.

For the PCA plot for 2014 (Figure 2.18), the first and second axes explain 56.4% and 16.2% of the variation respectively. On the first axis turbidity, suspended solids and aluminium showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to these parameters were found at sample sites SW8, SW10, Spring3, SW11 and SW9. These sample sites also showed an association to phosphate, cadmium, bromine, zinc and mercury concentrations. On the second axis EC, sulphate and manganese showed to be co-linear and depicted a strong relationship to each other. A significant positive loading to EC and sulphate were found at sample sites SW2, SW3, SW4 and SW5. A significant negative loading was found to these sample sites with nitrate and lead concentrations.

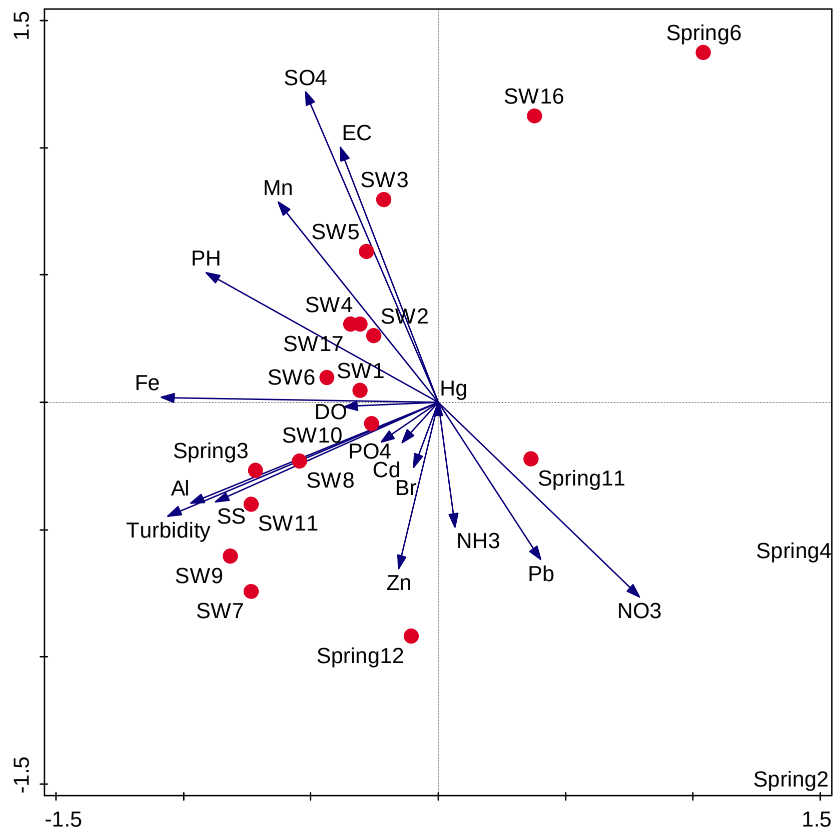


Figure 2.18: Principal component analysis biplot showing the influence of selected water quality parameters at sites in the Wilge River sub-catchment B20F area for the sampling period 2014. The bi-plot represents 72.66% of the total variation in the data.

A strong association to turbidity, iron and aluminium was noted from 2008 to 2009 at sample sites SW4, SW3 and Spring12 (Appendix A, Figures 1-17). In 2010, this was maintained, as well as increases in sulphate and EC were noted especially at sample sites SW5, SW6 and SW17. In 2011 turbidity, aluminium, iron and suspended solids were accompanied by an association to pH and phosphate. An additional sample site associated with these parameters are SW6. In 2012 turbidity, iron, aluminium and suspended solids were accompanied by an association to mercury and pH. More sample sites showed a positive loading to these parameters. In addition to SW3, SW4 and SW6 sample sites SW9, SW7, SW8, SW2, SW1, SW10, SW11 and SW17 showed a strong association with these parameters. In 2013 sample sites SW4, SW7, SW9, SW6, SW8, SW11, SW10 and SW1 showed a strong association to turbidity, iron, aluminium, lead, suspended solids, iron and pH. In 2014 sample sites SW8, SW11, SW6, SW9, SW10, SW7, Spring3 and Spring12 showed a strong association to turbidity, iron, aluminium and suspended solids. There has been a progressive increase in the number of sites in the Wilge River sub-catchment showing a change in turbidity and suspended solids from 2008 till 2014.

Piper diagrams (Figure 2.19), were drawn to illustrate the chemical water composition, in terms of major cations and anions. Most sample sites show HCO_3^- dominance with no cation dominance. The size of the circles reflect the TDS, the greater the circle the larger the TDS values. The TDS concentrations for sample sites Spring10, Spring6, SW5 and SW2 are closer to the SO_4+Cl/Ca+Mg apex and is indicative of polluted water. Spring12, SW3, SW4 and Spring11 indicate high sodium bicarbonate/ chloride waters. Sample sites SW1, SW5, Spring 10 and Spring6 are Ca-SO_4 dominant, which suggests the influence of coal mining in these sample sites.

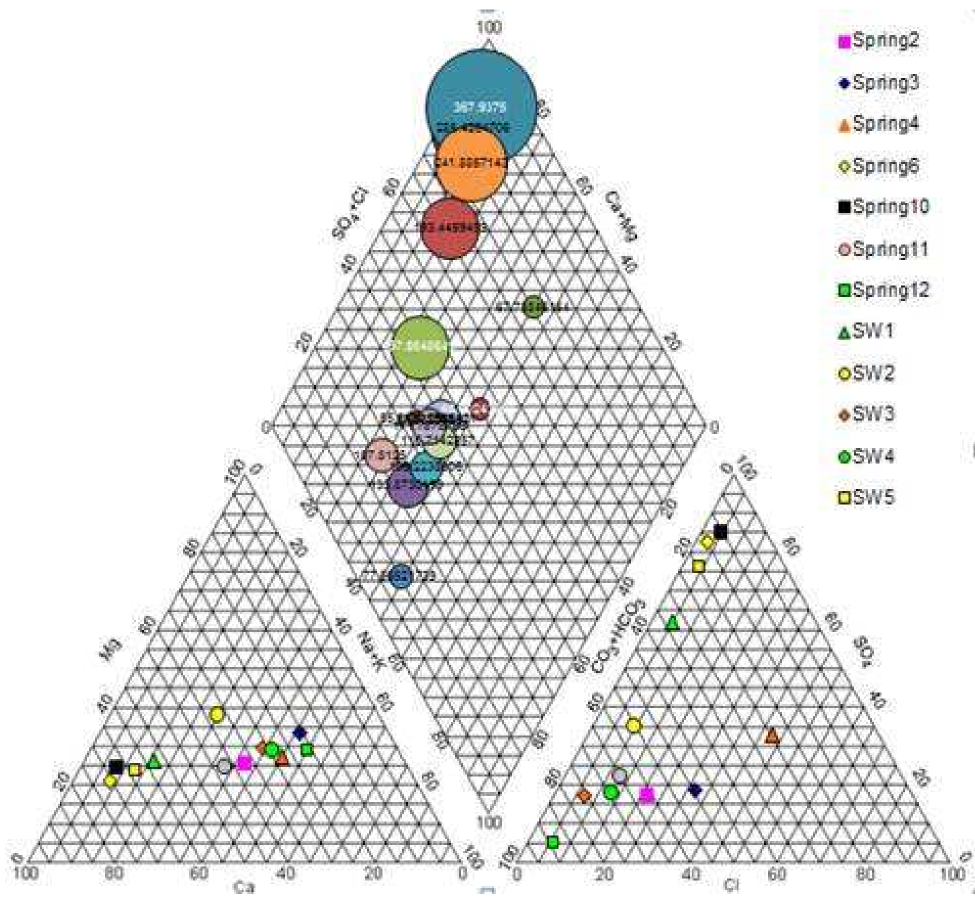


Figure 2.19: Piper diagrams showing the seasonal variation of the mean values for selected water quality parameters taken during January 2008 and December 2014 in the Wilge River sub-catchment B20F area.

The pollutants entering the Wilge River catchment from mining (acidic water, heavy metals and sulphates); untreated or poorly treated sewage and chemical toilets (high nutrient) and agriculture (high nutrient) have had a significant adverse impact on the water quality of the Olifants River system. This can cause general acidification of the system, the input or mobilisation of heavy metal ions plus sulphates and other contaminants via acid mine drainage, effluents containing a variety of potential pollutants, excessive nutrient inputs (phosphorus and nitrogen) from agricultural activities and sewage effluent from intensive agriculture leading to possible widespread eutrophication.

2.4 Conclusions

The Olifants River, into which the Wilge River drains, is one of the main river systems in South Africa and has often been described as one of the most polluted rivers in Southern Africa (Heath *et al.*, 2010). The main land-uses in the Wilge River sub-catchment B20F study area are extensive agricultural activities, pivot irrigation, livestock farming (i.e. cattle, pig and chicken farming), rural development, infrastructure (i.e. railways and tar and/or gravel roads), industrial activities (i.e. construction footprint), coal mining and quarrying. The combination of these anthropogenic stressors and general decline in the operation and management of waste water treatment has resulted in increased pollutants in the catchment (Figures 2.8-2.10). The pollutants generated and ecological processes affected by these activities include a general acidification of the system and the input or mobilisation of heavy metal ions, conductivity (of which sulphate is the major constituent) and other contaminants via acid mine drainage, excessive nutrient inputs (phosphorus and nitrogen) from agricultural activities and sewage effluent and livestock farming (e.g., feedlots).

Acid mine drainage from abandoned coal mines has serious effects on the aquatic ecosystem, resulting in toxic effects at several localised sites (Figure 2.19). Considering the current extent of coal mining in the upper Olifants River catchment, effective management and planning of mines is essential to minimise the effects of this problem in future.

Continued mining and power generation activities in the Olifants River catchment have led to a general acidification of the water system and the input and mobilisation of heavy metals (Dabrowski *et al*., 2014; Hobbs et al., 2008). Elevated levels of heavy metals in an aquatic system pose a significant threat to aquatic life an ultimately to human health, reaching considerable concentrations in plants and biota through bioaccumulation. As the Wilge River currently discharges the Olifants River and significantly improves its water quality, future mining and development activities activities in the Wilge River sub-catchment should be carefully planned and operated so as to ensure sufficient flows of acceptable quality to prevent further deterioration of water quality in the Olifants River and downstream reservoirs (Dabrowski *et al*., 2014).

From the combined results of the risk assessment (Genthe *et al.*, 2013) it was concluded that the combination of untreated sewage, mining and industrial practises severely compromise the water quality of the Olifants River, and have the potential to seriously impact on human health, especially in lower income communities that make direct use of untreated river water.

South Africa's National Water Act, (Act No 36 of 1998) is widely regarded as the most advanced water legislation framework in the world. Several technical approaches have been developed to facilitate the proper implementation of the National Water Act, (Act No 36 of 1998), namely Reserve Determination (DWA, 2011b) and the National Classification System (DWA, 2013). It is imperative that DWA together with local and provincial authorities and water users enforce the water management guidelines, maximum allowable thresholds and resource water quality objectives that have been recommended by these tools. Effective management of the water quality in the upper Olifants River catchment requires a truly collaborative approach between government, water resource managers, businesses and communities, with government playing the leading role in directing efforts and evaluating their efficiency.

The combination of land-use impacts has affected the water quality in the Wilge River sub-catchment B20F area. The two main sets of stressors therefore, are acidic water containing heavy and trace metal ions and sulphate that is attributable to abandoned mining and nutrient concentrations originating from agricultural and livestock runoff, and from untreated or poorly treated sewage. The careful management and mitigation of these pollutant sources are essential to ensure compliance to the Wilge River IWRMP RWQOs thus, achieving long-term sustainability of acceptable water quality and ecosystem health in the upper Olifants River catchment.

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CHAPTER 3: THE APPLICATION OF A DIRECT TOXICITY ASSESSMENT APPROACH TO ASSESS THE EFFECT OF CONSTRUCTION ACTIVITIES ON THE ECOLOGICAL HEALTH OF THE WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

3.1 Introduction

The scarcity of water is closely linked with water pollution because pollution renders water unfit for various purposes (Muller *et al*., 2009). Several land-use activities contribute to the deterioration of water quality and the loss of beneficial use from polluted water. Although pollutants enter the aquatic ecosystems as mixtures of chemicals the pollution control has mainly been through controlling concentrations of individual chemical components of these mixtures (Muller and Palmer, 2004). Most techniques used in detecting specific chemicals in complex wastewater discharges of surface waters have toxicological, environmental and analytical considerations and pose significant difficulties (Jooste and Herbst, 2004).

Toxicity testing of complex effluents has been shown worldwide to be an effective management option in preventing deteriorating water quality in aquatic ecosystem (Grothe *et al.,* 1996). Laboratory toxicity tests are used to manage environmental resources such as water quality and are considered to the first step in a tiered approach in establishing guidelines for setting maximum acceptable concentrations for specific pollutants (Kimball and Levin 1985, Chapman 1995, Muller and Palmer, 2004). However, no single toxicity test has proven to be suitable to assess all adverse ecological effects because individual organisms differ in susceptibility to different chemicals (Rand *et al.,* 1995; Chapman, 2000 and DWAF, 2003). Consequently, several different bioassays at different levels of biological complexity and trophic levels need to be used simultaneously to adequately assess if a potential hazard is posed (Jergentz *et al*., 2004).

In using this approach, a variety of test organisms are exposed to the complex effluent and the effect on the organisms is quantified. Results of these tests are used to estimate the effluent concentration health status, which will provide adequate aquatic ecosystem protection. The effluent discharge license can then be specified in terms of a toxicity test end-point rather than relying only on chemical concentrations of the complex effluent.

International toxicity approaches are used in water quality monitoring programmes in the United States of America (USA), Canada, European Union (EU), United Kingdom (UK), Netherlands, Australia and New Zealand. These include the whole effluent toxicity (WET), Chemical Specific Approach (chemical-specific water quality based limits that include an acute and chronic value), and Biological Bioassessment approach (incorporates chemical, physical and biological data). The Biological Test Method (reference method for determining acute lethality of effluents to rainbow trout – EPS1/RM/13, 2000), Biological Test Method (reference method for determining acute lethality of effluents to *Daphnia magna* – EPS1/RM/14, 2000), Biological Test Method (reference method for determining acute lethality of sediment to marine and estuarine amphipods – EPS1/RM/35, 1998), Biological Test Method (reference method for determining toxicity of sediment using luminescent bacteria in a solid-phase test– EPS1/RM/42, 2002) are applied in Canada. The acute toxicity for fish, acute toxicity for daphnia, algal growth inhibition and bacterial inihibition toxicity tests are applied in the EU (Chapman *et al*., 2011). The direct toxicity assessment (DTA) is adopted in the United Kingdom (UK), and the Total Effluent Milieu hygiene (TEM) or whole effluent environmental risk approach is used in the Netherlands (DWAF, 2003). These approaches are based on similar fundamental concepts by using an array of acute and chronic toxicity test endpoints and, in certain instances such as TEM, include in addition other 'indirect' hazard parameters such as oxyen depletion potential, bioaccumulation and mutagenicity to ascertain ecological effects of pollutants.

The Australian and New Zealand Environment and Conservation Council (ANZECC) has developed a set of guidelines - Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC, 2000) for managing of water quality in Australia and New Zealand. The ANZECC guidelines provide an authoritative reference for water quality management in New Zealand and Australia, particularly for toxic contaminants. Guideline values for many toxicants are listed in the guideline documents and are derived from standarised toxicity tests (Hart, 2001). In New Zealand WET testing protocols have been used since the 1980s and at present there are standardized protocols for freshwater invertebrate *Ceriodaphnia dubia* (water flea), *Paracalliope fluviatilis* (amphipod), *Gobiomorphus cotidianus* (freshwater fish), *Dunaliella tertiolecta* (marine algae) and *Rhombosolea plebeia* (marine fish), *Selenastrum capricornutum* (freshwater algae) and *Fellaster zelandiae* (marine invertebrate) (Hall and Golding, 1998) are being undertaken.

The availability of standardized toxicity test protocols for WET testing and the application of these tests in the assessment of complex industrial wastes have led to routine toxicity testing in USA, Germany, Canada and New Zealand for compliance and regulatory monitoring. Toxicity limits are set by the regulatory authority and have been included internationally (eg. USA and Canada) as conditions of effluent discharge licenses (Chapman *et al*., 2011). Toxicity limits are used in the same manner that chemical limits are used and should be viewed as equivalent to emission standards for chemical and physical parameters.

In South Africa, the Department of Water and Sanitation (DWS) has implemented the Direct Estimation of Ecological Effect Potential (DEEEP), which includes the international approaches (using TEM as foundation) and has become an integral first-tier tool used in the ecological hazard assessment of complex waste discharges (DWAF, 2003, Ansara-Ross *et al*., 2009). The DEEEP is a multifaceted approach that uses representative organisms from different trophic levels of the food chain (fish, invertebrates, algae and bacteria) and endpoints (both lethal and sublethal) to reflect the overall impact of toxicants, with provision to define acceptable ecological hazards providing protection to aquatic ecosystems (Jooste and Herbst, 2004, Liu and Dutka, 1999).

This suite of hazard assessment methods were selected for inclusion in the National Toxicity Monitoring Programme (NTMP) for surface waters (DWAF, 2005). Through the use of various aquatic toxicity test methods the NTMP aims to report on both the potential for toxic effects to selected test organisms and on potentially toxic substances in South African inland surface water resources (DWAF, 2006). The NTMP was designed in anticipation of the DWAF's Resource Classification System (Murray *et al.*, 2004). It will play a support role and provide supplementary information to various national monitoring programmes currently being implemented by the DWS that will focus on determining resource quality objectives for South African water resources (Chapman *et al*., 2011). The main objective of the NTMP is to monitor South African water resources in terms of (a) the toxic effects to selected organisms and (b) selected potentially toxic substances. The four rapid, simple and inexpensive internationally standardized toxicity bioassays that form part of the DEEEP and NTMP were adapted and incorporated into this study. These include: the vertebrates (*Poecilia reticulata*), invertebrates (*Daphnia magna, Daphnia pulex*), bacteria (*Vibrio fischeri*) and primary producers (*Pseudokirchneriella subcapitata*).

The extent of ecological hazard for each individual series of toxicity test parameters may vary. The need for approaches in aggregating the individual parameters within the hazard assessment has been identified (DWAF, 2003). Jooste and Herbst (2004) recognised that criteria for ecological hazard assessments need to be validated locally before finally adopting them. By assessing each parameter according to a certain criterion that determines a specific rating, an overall ecological hazard can be predicted, i.e. no, slight, moderate, high and very high hazards. The currently applied assessment methods are sufficiently flexible to allow preliminary criteria to be used as a first-tiered approach, to be refined and validated over time with further biological assessments and chemical analyses (DWAF, 2003). With the implementation of the NTMP a relatively simple classification system based on the occurrence or absence of toxicity was proposed (DWAF, 2005).

Continued mining, industrial and power generation activities in the Olifants River catchment have led to a general acidification of the water system and the input and mobilisation of heavy metals (Oberholster *et al*., 2010; Hobbs *et al*., 2008). Elevated levels of heavy metals in an aquatic system pose a significant threat to aquatic life an ultimately to human health, reaching considerable concentrations in plants and biota through bioaccumulation. Heavy metals are of particular significance in ecotoxicology because of their high degree of persistence, and all have the potential to be toxic to living organisms (Dyer and Belanger, 1999) above threshold concentrations.

This chapter evaluates the potential hazardous effects of land-use impacts in the nearby receiving waters of the Olifants River WMA, Wilge River sub-catchment B20F area using a combination of toxicological bioassays that allows for a comprehensive and comparative assessment of spatial and temporal variability of effects. Toxicity tests using four trophic levels of biota i.e., bacteria (*V. fischeri*), unicellular algal (*P. subcapitata*), water flea (*D. magna, D. pulex*) and fish (*P. reticulata)* were undertaken.

3.2 Materials and methods

3.2.1 Site description

Eight toxicity assessment sites Spring12, Spring6, T3 (2.5 km downstream of Spring3 just after the confluence of the Holfonteinspruit and unnamed tributary), SW7, SW11, SW6, SW2, SW17 were selected (Figure 3.1 and Table 3.1). Toxicity analyses were undertaken in March and May 2014. Additional toxicity analyses were undertaken at sampling sites Spring 6 and T3 in November 2012, July 2013, June 2014 and November 2014; and only at Spring6 in January 2014. Toxicity analyses and hazard classification of these tributaries will give a good indication of the potential impacts, which could affect the biotic integrity of the downstream Wilge River site.

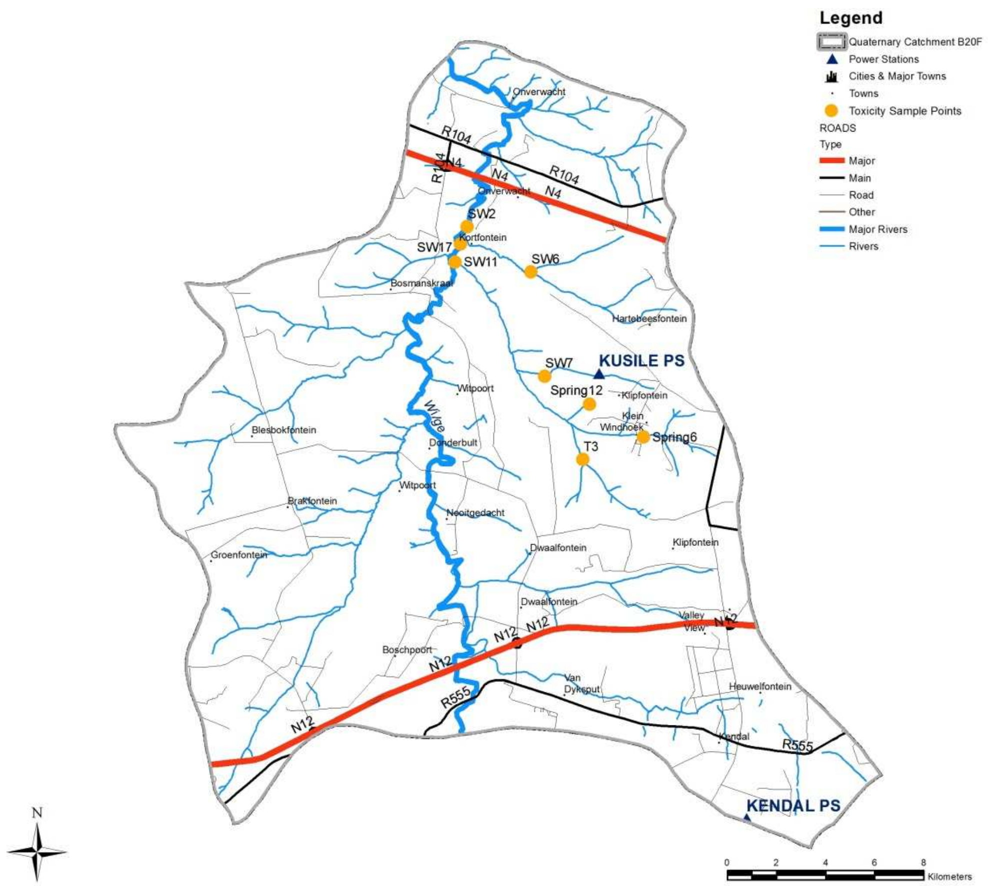

Figure 3.1: Sites at which water samples were collected for Direct Estimation of Ecological Effect Potential (DEEEP) analyses.

Table 3.1: Description of Toxicity sampling sites.

Site	Latitude	Longitude	Site Description
Spring12	-25.94236	28.91466	Pan located next to construction site draining into the Klipfonteinspruit
Spring6	-25.94760	28.92797	Site on Klipfonteinspruit upstream of proposed ash dump facility
T3	-25.95689	28.90842	Site 2.5km downstream of Spring 3 after the confluence of the Holfonteinspruit and unnamed tributary
SW7	-25.92578	28.89394	Site on Unnamed tributary of the Klipfonteinspruit
SW11	-25.88439	28.86170	Site on Klipfonteinspruit before confluence with Wilge River
SW6	-25.88797	28.88723	On northern tributary of the Wilge River
SW2	-25.85330	28.86847	Downstream site on Wilge River
SW17	-25.87476	28.86313	Site on Wilge Rvier

3.2.2 Methodology

3.2.2.1 Toxicity Assesments

Two 2 litre plastic bottles were rinsed and filled with water at the site. Samples from the surface water monitoring sites were collected by immersing the sample container in the water and collecting a grab sample. Samples were stored in cool containers after collection and transported to the laboratories within 24 hours after collection. The samples were accompanied by a chain of custody form, which lists all samples submitted, date and time sampled and constituents to be analysed. Toxicity tests using four trophic levels of biota i.e., bacteria *V. fischeri*, unicellular algal (*P. subcapitata*), water flea (*D. magna, D. pulex*) and fish (*P. reticulata)* were conducted within 24 hours of sampling. Water quality parameters (pH, conductivity and dissolved oxygen) were measured at the starting time of the *Daphnia* testing.

Toxicity testing is applied by exposing biota under laboratory conditions to water sources (pollution control dams and effluent sources) in order to determine the potential risk of such waters to the biota of the receiving water bodies. Acute (and short-chronic) toxicity testing is applied by exposing biota to water sources in order to determine the potential risk of such waters to the biota/biological integrity of the receiving water bodies. A risk category is determined based on the percentage of mortalities (or inhibition-stimulation) of the exposed biota by application of the latest DEEEP methodology. Consequently a hazard class was determined based on the resulting parameters of the battery of tests (DWAF, 2003). It is important to note that the hazard classification is based on the standardised battery of selected test biota and therefore represents the risk/hazard towards similar biota in the receiving aquatic environment. The toxicity hazard is therefore in terms of the aquatic biotic integrity and does in no way represent toxicology towards humans or other mammals. All tests were conducted in environmental controlled rooms using the following internationally standardised methods:

The *V. fischeri* bioluminescent test was conducted using the standard method: EN ISO 11348-3, (European Standard, 1998). There was no deviation from the standard method. The test sample volume of 500 µl was used with exposure periods: 15 and 30 minutes. Three replicates were undertaken in the November 2012, July 2013, January 2014, June 2014 and November 2014 assays. Two replicates were undertaken in the March 2014 and May 2014 assays. Growth inhibition was measured on a Luminoscan TL, Hygiene Monitoring System. Test endpoint for the screening test was determined by comparing the percentage growth inhibition or stimulation of luminescence relative to the control.

The *P. subcapitata* growth inhibition test was conducted using the standard method OECD Guideline 201, (1984). There was no deviation from the standard method. Tests using, *P. subcapitata*, Printz (CCAP 278/4 Cambridge, UK) were carried out in environmentally controlled rooms (21-25℃). Tests were undertaken with an exposure period of 72 h; test sample volume of 25 ml and test chamber type of 10 cm long cell. Three replicates were undertaken in the November 2012, July 2013, January 2014, June 2014 and November 2014 assays. Two replicates were undertaken in the March 2014 and May 2014 assays. Optical density (OD) using a Jenway 6300 spectrophotometer was used to establish growth inhibition or growth stimulation relative to a control. Percentage growth inhibition or stimulation was measured respectively as a reduction or increase in growth rate relative to a control carried out under identical conditions. Test endpoint for the screening test was determined by comparing the percentage growth inhibition or stimulation relative to the control.

The *D. magna, D. pulex* acute toxicity test was conducted using the standard method USEPA, (1993). *D. magna* was used in the November 2012, July 2013, January 2014, June 2014 and November 2014 assays. *D. pulex* was used in the March 2014 and May 2014 assays. There was no deviation from the standard method. The test species, *D. magna, D. pulex* of less than 24 hours old were used and the assay was carried out in environmentally controlled rooms at a temperature of 21±2 ℃. Five test organisms per beaker with 4 replicate beakers per sample were used. Test sample volume of 25 ml was used with an exposure period: 24 and 48 h. Test endpoint for the screening test was determined by calculating percentage mortality.

The *P. reticulata* acute toxicity test was conducted using the standard method: USEPA, (1996). There was no deviation from the standard method. The test species, *P. reticulata* of less than 21 days old were carried out in environmentally controlled rooms at temperature of 21±2 ℃. Six test organisms per beaker with 2 replicate beakers per sample were used in the November 2012, July 2013, January 2014, June 2014 and November 2014 assays. Ten test organisms per beaker with 2 replicate beakers per sample were used in the March 2014 and May 2014 assays. A sample volume of 200 ml was used with an exposure period of 96 h. Test endpoint for the screening test was determined by calculating percentage mortality.

Test endpoint for the definitive test is determined by calculating the EC20 and EC50 values for all toxicity tests. Statistical analyses were undertaken on an EXCEL spreadsheet.

3.2.2.2 Hazard classification

DWA has recommended the DEEEP protocols and hazard classification. The risk category equates to the level of acute or chronic risk posed by the selected potential pollution sources on the receiving rivers/streams. A 10% effect for daphnia and guppies, while a 20% effect for algae and bacteria (*Vibrio*) is the safe concentration that should not have any level of toxic effect on the receiving environment, (DWAF, 2003). A 50% effect is regarded as an acute toxicity for all of the tests (daphnia, guppies, algae and bacteria)

The hazard criterion described by Ansara-Ross *et al*., (2009) was adapted for this study. The scoring system comprises five ranking classes that range from 'not acutely hazardous or toxic' to extremely acutely hazardous or toxic'. Once the effect for each test series was determined, the sample was ranked in one of the five classes on the basis of highest toxic response shown by at least one of the tests applied. The effect results of each test series were then given a weight hazard score (WHS) as indicated in Table 3.2.

Table 3.2: Criteria for ecological hazard assessment for discharges/receiving water proposed for the Direct Estimation of Ecological Effect Potential (DEEEP) method.

Parameter	Effect endpoint	Effect values	Hazard description	WHS* per
Vibrio fischeri (bacteria)	Percentage mortality at 100% sample	-10-(+)10%	No inhibition or stimulation	0
		-20-(+)20%	Negligible stimulation or inhibition	0
		>(+)20%	Moderate to high stimulation	1
		(-)20-(-)30%	Slight inhibition	1
		(-)30-(-)40%	Moderate inhibition	2
		<(-)40%	High inhibition	3
Pseudokirchneriella subcapitata (unicellular algae)	Percentage mortality at 100% sample	-10-(+)10%	No inhibition or stimulation	0
		-20-(+)20%	Negligible stimulation or inhibition	0
		>(+)20%	Moderate to high stimulation	1
		(-)20-(-)30%	Slight inhibition	1
		(-)30-(-)40%	Moderate inhibition	2
		<(-)40%	High inhibition	3
Daphnia magna, Daphnia pulex (water flea)	Percentage mortality at 100% sample	<10%	No toxic effect.	0
		10%	Negligibly acutely toxic	0
		20%	Slightly acutely toxic	1
		30%	Moderately acutely toxic	2
		>30%	Highly acutely toxic	3
Poecilia reticulata (fish)	Percentage mortality at 100% sample	<10%	No toxic effect.	0
		10%	Negligibly acutely toxic	0
		20%	Slightly acutely toxic	1
		30%	Moderately acutely toxic	2
		>30%	Highly acutely toxic	3

*Weight hazard score

A cumulative WHS for all tests was then calculated for each sample by adding the individual WHS. A hazard category was then assigned to the cumulative WHS for each sample as indicated in Table 3.3. As each sample is weighted according to its level of toxicity (no hazard to extreme hazard) observed for each test performed, if none of the tests detected any toxicity, the sample would have a weight of 0%, while extreme hazards detected by all tests will result in a weight of 100%. The weighting system therefore provides a measure to compare relative toxicity on a scale between 0 and 100, and toxicity hazards can therefore be compared between samples that fall within the same class. This hazard category can then be assessed in terms of ecological and management viewpoint.

Table 3.3: Hazard assessment categories for the various toxicity endpoints.

Hazard category	Hazard description	Result	Cumulative hazard score
A	No hazard due to toxicity	None of the tests shows a toxic effect.	0
B	Slight hazard due to toxicity	The cumulative hazard score of one of the toxicity tests was 1	1
C	Moderate hazard due to toxicity	The cumulative hazard score of one of the toxicity tests was between 2 and 5	2-5
D	High hazard due to toxicity	The cumulative hazard score of one of the toxicity tests was between 6 and 10	6-10
E/F	Extreme hazard due to toxicity	The cumulative hazard score of one of the toxicity tests was greater than 10	>10

3.2.3 Water quality parameters

In situ water quality analyses i.e. pH, electrical conductivity (EC), dissolved oxygen (DO), and temperature (Temp) were undertaken at each sampling site. Compact handheld instrument EXTECH II was used to record water quality parameters pH, EC, DO and Temp were measured *in situ* at all sites during March 2014 and May 2014 surveys. Water quality parameters pH, EC, DO and Temp were measured at sampling sites Spring 6 and T3 in the November 2012, July 2013, January 2014, June 2014 and November 2014 surveys by Eskom's YSI 556 Multiparameter system (MPS). DO was measured in mg/L, and converted to % saturation using, USEPA, (2015). All meters were calibrated to ensure accuracy of the results.

Physical, chemical water quality samples were taken at sampling sites Spring 6 and T3 in November 2012, July 2013, June 2014 and November 2014. Physical, chemical water quality samples were taken at the sampling sites Spring 6 only in January 2014. Two 1 litre plastic bottles were rinsed and filled with water at the site. The first 1 litre bottle was stored in the cooler box and the second sample was preserved with nitric acid (HNO_3) for heavy metals analyses before being stored in the cooler box. The samples were analysed for a number of water quality variables including, alkalinity (Alk), aluminum (Al), ammonia (NH_3), cadmium (Cd), chloride (Cl), chemical oxygen demand (COD), conductivity (EC), dissolved organic carbon (DOC), iron (Fe), fluoride (F), Potassium (K), magnesium (Mg), manganese (Mn), sodium (Na), nickel (Ni), nitrate (NO_3), pH, ortho-phosphate (PO_4), strontium (Sr), sulphate (SO_4), total dissolved solids (TDS), total organic carbon (TOC), total hardness (TH), total suspended solids (TSS) and turbidity (NTU). All samples were taken to the South African National Accreditation System (SANAS) Accredited Eskom Analytical chemistry laboratory for analysis using the same methods as previously described in Chapter 2.

The Wilge River sub-catchment B20F area falls within MU 22 management unit. The IWRMP for the Upper and Middle Olifants River catchment, recommends the following Resource Water Quality Objectives for the Wilge River sub-catchment (DWA, 2009). (Chapter 2, Table 2.3). All the parameter results were compared to the IWRMP RWQOs. For parameters analysed where the IWRMP RWQOs were not given, the Target Water Quality Objectives (TWQO) of the South African water quality guidelines for aquatic ecosystems (DWAF, 1996). Chapter 2, Table 2.4, or the Australian and New Zealand Guidelines (ANZECC, 2000) were consulted (Table 3.4).

Table 3.4: Australian and New Zealand Water Quality Guidelines (ANZECC, 2000).

Water quality	Units	Target Water Quality Objective
Arsenic	µg/L	1
Cyanide	µg/L	4
Copper	µg/L	1
Nickel	µg/L	8
Selenium	µg/L	5

3.3 Results and discussion

Water quality parameters (pH, EC, DO and Temp) were analysed for each sampling site on which the bioassays were carried out (Table 3.5). The pH readings were within the Intergrated Water Resource Management Plan (IWRMP), resource water quality objectives (RWQO) range of 6.5-8.4 for all sampling sites except, for SW11 in March 2014 with a pH reading of 6.13 being recorded; and at Spring12, T3, SW6 and SW11 in May 2014 with pH of 6.39, 6.35, 6.31 and 6.38 being recorded respectively.

Table 3.5: *In situ* water quality analyses for toxicity sampling points.

Parameter	Date	Spring12	Spring6	T3	SW7	SW11	SW6	SW2	SW17
pH	Nov-12		7.52	7.53					
EC (mS/m)			12.08	5.43					
DO (%Sat)			82.14	82.36					
pH	Jul-13		8.10	8.40					
EC (mS/m)			8.70	7.42					
DO (%Sat)			92.39	94.49					
pH	Jan-14		7.10						
EC (mS/m)			22.90						
DO (%Sat)			88.20						
pH	Mar-14	6.94	6.99	7.12	6.96	6.13	6.49	6.63	6.64
EC (mS/m)		14.33	14.45	10.06	16.74	22.00	25.60	14.45	26.70
DO (%Sat)		63.07	72.33	57.11	57.33	15.10	65.82	76.30	74.86
pH	May-14	6.39	6.51	6.35	6.31	6.36	6.57	6.55	6.74
EC (mS/m)		12.08	7.46	14.19	14.33	14.34	14.45	15.59	10.26
DO (%Sat)		96.97	16.52	83.93	101.46	96.18	100.90	104.27	99.44
pH	Jun-14		7.60	7.50					
EC (mS/m)			11.00	8.80					
DO (%Sat)			79.38	82.69					
pH	Nov-14		7.40	7.20					
EC (mS/m)			4.40	5.40					
DO (%Sat)			81.59	80.49					

All EC values were within the IWRMP RWQO limit of 40 mS/m. Dissolved oxygen percentage saturation (DO%) was found to be within the IWRMP limit for all sampling sites except Spring12, T3, SW7, SW11 and SW6 with a DO% of 63%, 57%, 57%, 15% and 65% being recorded respectively, and at Spring6 in May 2014 with a DO% of 16 being recorded. The water sample from sampling site SW11 and Spring6 were aerated before the toxicity biosassays were undertaken.

The comprehensive chemistry analyses were carried out on the sampling sites Spring6 and T3 in the November 2012, July 2013, January 2014, June 2014 and November 2014 surveys (Table 3.6). Exceedance of the Wilge River IWRMP RWQOs for iron and aluminium at T3 was found in all surveys. Spring6 showed exceedance of the Wilge River IWRMP RWQOs for iron, aluminium and manganese in most sampling events. T3 showed exceedance of the Wilge River IWRMP RWQOs for ammonia in July 2013 survey only. Spring6 and T3 showed an exceedance of the Wilge River IWRMP RWQOs for nickel in the November 2014 survey only. Spring6 showed exceedances of the Wilge River IWRMP RWQOs for calcium, manganese and sulphate in the January

2014 survey. Spring6 also showed a decrease in pH in the January 2014 survey indicating a possible effluent discharge at this site prior to the survey.

Table 3.6: Chemistry analyses at Spring6 and T3 for November 2012-2014.

Date	Nov-12		Jul-13		Jan-14	Jun-14		Nov-14	
Sample Name	Spring6	T3	Spring6	T3	Spring6	Spring6	T3	Spring6	T3
Alkalinity Total (mg/L $CaCO_3$)	ND	ND	16.54	18.51	9.49	8.42	17.84	19	31.7
Aluminium as Al (mg/L)	0.09	0.09	0.04	0.64	0.02	0.02	0.22	0.06	0.31
Ammonia as N (mg/L)	<0.005	<0.005	<0.005	0.1	<0.005	<0.005	<0.005	<0.005	<0.005
Calcium as Ca (mg/L)	18	5.1	12	5.8	28	11	4	7.2	5.7
Chloride as Cl (mg/L)	1.41	0.8	2.49	2.66	1.93	3.59	5.47	1.76	6.83
Chemical Oxygen Demand (mg/L)	10	11	<1	2	1	3	5	<1	12
Conductivity (mS/m)	15	7.3	10.3	9.1	18.4	11.9	10.9	8.46	10.4
High Level DOC Elementar (mg/L)	4.44	4.27	1.7	3.1	3.14	1.4	2.74	4.01	8.43
Iron as Fe (mg/L)	1.4	2.2	0.32	1.8	0.6	0.24	2.1	1.3	3
Flouride as F (mg/L)	0.21	0.22	0.17	0.22	0.18	0.06	0.14	0.15	0.46
Potassium as K (mg/L)	1.4	0.91	1.5	1.4	2.1	1.3	1.1	1.2	1.7
Magnesium as Mg (mg/L)	3	1.8	1.7	2.4	5.3	1.9	1.8	2	3.4
Manganese as Mn (mg/L)	0.19	0.18	0.04	0.1	1.4	0.06	0.02	0.23	0.09
Sodium as Na (mg/L)	4.7	5.4	3.8	7.3	3.8	4.7	7.9	4.4	8.6
Nickel as Ni (mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	0.01	0.02
Nitrate as N (mg/L)	0.12	<0.02	0.37	<0.02	0.07	0.42	0.07	0.09	0.05
pH @ 25 °C	6.9	6.78	6.85	6.94	6.39	7	7.06	7.14	7.37
Ortho Phosphate as PO4 (mg/L)	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090	<0.090
Sulphate (mg/L)	33.08	3.9	17.1	13.71	85.37	34.71	17	19.98	12.94
TDS (mg/L)	95.9	55.3	83.6	65.1	163.8	76.3	75.3	61.4	68.6
TSS (mg/L)	<10.00	<10.00	<10.00	79.35	<10.00	ND	ND	10.53	26.19
Turbidity (NTU)	2.07	5.67	3.29	77.5	1.22	1.05	29.6	5.84	30

Wilge Integrated Water Resource Management Plan Resource Water Quality Objectives (DWAF, 2009)
South African water quality guidelines for Aquatic Ecosystems TWQO (DWAF, 1996)
Australian and New Zealand Guidelines (Australian and New Zealand Environmental and Conservation Council, 2000)
Exceedances of the respective quality guidelines

ND – Analysis not done by the laboratory

Percentage inhibition or stimulation for *V. fischeri* toxicity bioassays are presented in Figure 3.2. Growth inhibition or stimulation greater than 20% indicated toxicity or excessive nutrients respectively. Growth inhibition was indicated at all sampling sites in March 2014 with the highest growth inhibition of 71% noted at SW11. Growth inhibition was indicated at all sampling sites in May 2014 except SW17. The highest growth inhibition of 29% was noted at Spring6 in May 2014. Growth stimulation of 24% was noted at only one sampling site, T3 in November 2014.

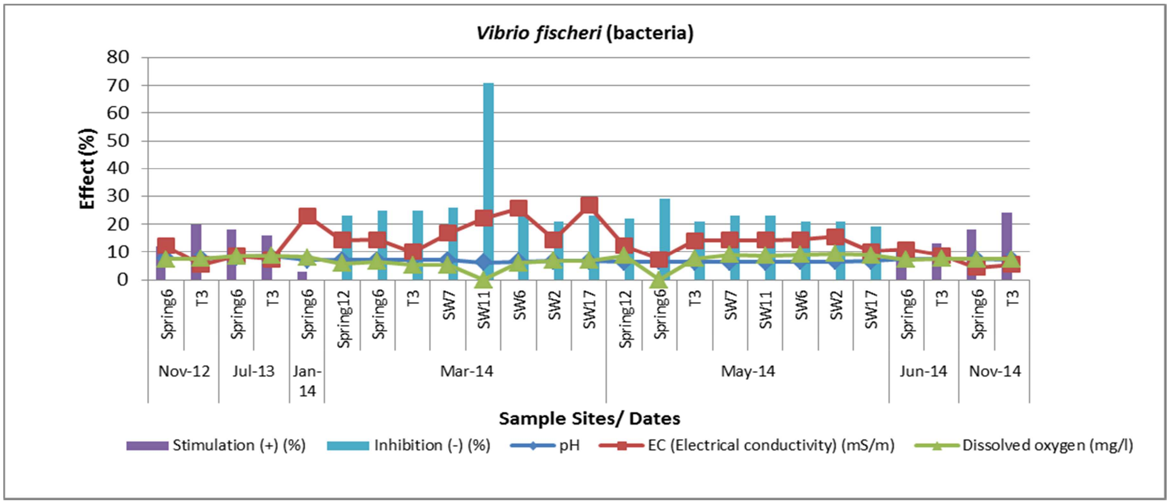

Figure 3.2: Effect data for *Vibrio fischeri* growth or inhibition assay.

Percentage inhibition or stimulation for *S. capricornutum* toxicity bioassays are presented in Figure 3.3. Growth inhibition or stimulation greater than 20% indicated toxicity or excessive nutrients respectively. Growth inhibition of 50% was noted at sampling site SW11 in March 2014 and 47% at sampling site SW6 in May 2014 respectively. Growth stimulation was noted at all sites in March 2014 except SW11. Growth stimulation was noted at sampling sites Spring12, T3, SW7 and SW11 in May 2014

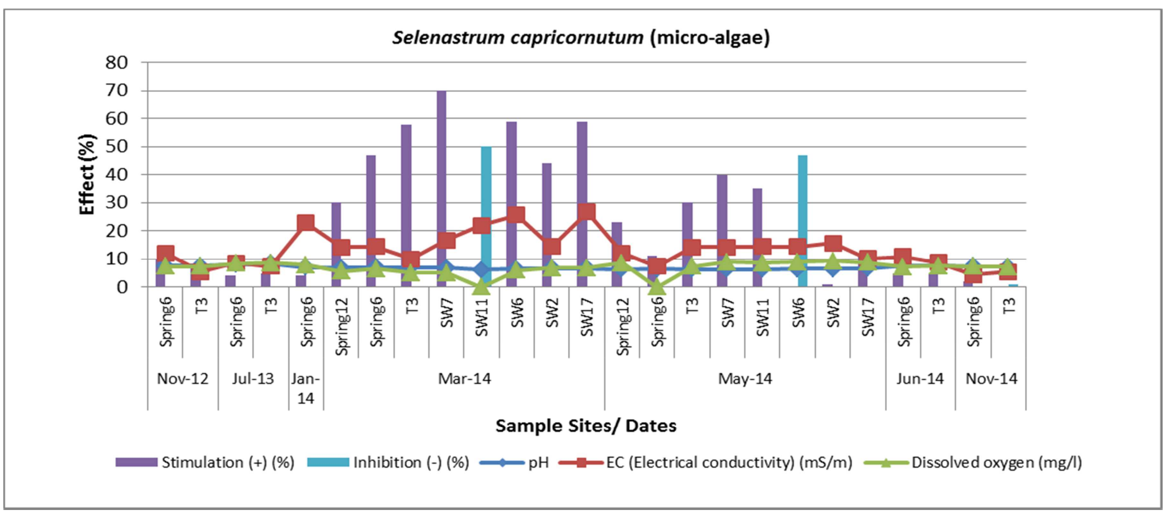

Figure 3.3: Effect data for *Selenastrum capricornutum* growth or inhibition assay.

The percentage mortality for each site for *D. magna, D. pulex* after 48 hours exposure is shown in Figure 3.4. Mortalities greater than 10% indicated toxicity. *Daphnia* mortalities greater than 10% were recorded for all sites in March 2014 except T3. T3 indicated a mortality of 10%. Sampling site SW11 indicated the highest mortality of 100%. *Daphnia* mortalities greater than 10% were recorded for at sampling sites SW2 and SW17 in May 2014 and sampling sites Spring6 and T3 in June 2014.

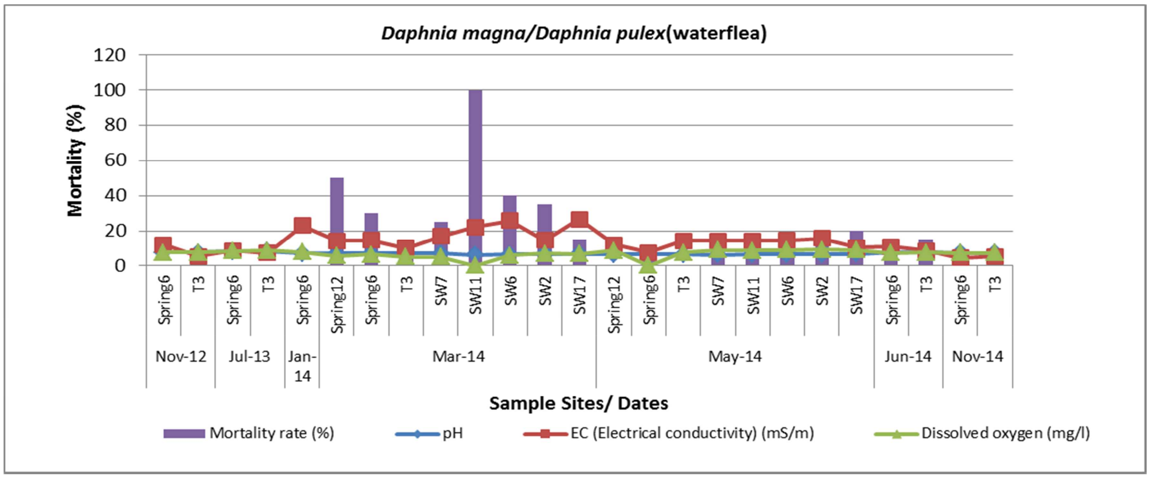

Figure 3.4: Effect data for *Daphnia magna/ Daphnia pulex* lethality assay.

The percentage mortality for each site for *P. reticulata* after 96 hours exposure is shown in Figure 3.5. Mortalities greater than 10% indicated toxicity. *P. reticulate* mortalities of 10% were recorded at Spring6 and T3 in June 2014.

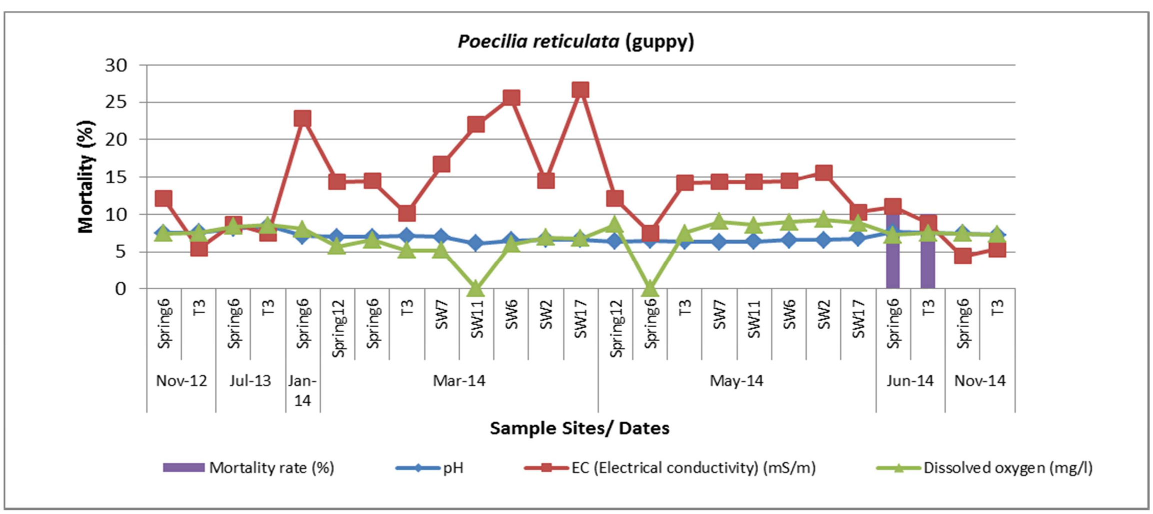

Figure 3.5: Effect data for *Poecilia reticulata* lethality assay.

By applying weight hazard scores for each sampling site the large array of data can be simplified and an overall ecological hazard class per site can be calculated (Table 3.7 and Figure 3.6). Hazard assessment categories were proposed to standardise the output of the different toxicity assessments.

Table 3.7: Effect class categories for the toxicity bioassays with associated weighted hazard scores (WHS) for sampling sites in the Wilge River sub-catchment B20F area.

Date	Site	*V. fisheri*		*S. capricornatum*		*D.magna/pulex*		*P.reticulata*		Hazard Status	
		% Effect	WHS	% Effect	WHS	% Effect	WHS	% Effect	WHS	Cumulative WHS	Hazard Category
Nov-12	Spring6	12	0	10	0	0	0	0	0	0	A
	T3	20	0	8	0	0	0	0	0	0	A
Jul-13	Spring6	18	0	4	0	0	0	0	0	0	A
	T3	16	0	5	0	0	0	0	0	0	A
Jan-14	Spring6	3	0	4	0	0	0	0	0	0	A
Mar-14	Spring12	-23	1	30	1	50	3	0	0	5	C
	Spring6	-25	1	47	1	30	2	0	0	4	C
	T3	-25	1	58	1	10	0	0	0	2	C
	SW7	-26	1	70	1	25	2	0	0	4	C
	SW11	-71	3	-50	3	100	3	0	0	9	D
	SW6	-23	1	59	1	40	3	0	0	5	C
	SW2	-21	1	44	1	35	3	0	0	5	C
	SW17	-23	1	59	1	15	1	0	0	3	C
May-14	Spring12	-22	1	23	1	0	0	0	0	2	C
	Spring6	-29	1	11	0	5	0	0	0	1	B
	T3	-21	1	30	1	0	0	0	0	2	C
	SW7	-23	1	40	1	10	0	0	0	2	C
	SW11	-23	1	35	1	5	0	0	0	2	C
	SW6	-21	1	-47	3	5	0	0	0	4	C
	SW2	-21	1	1	0	20	1	0	0	2	C
	SW17	-19	0	12	0	20	1	0	0	1	B
Jun-14	Spring6	10	0	5	0	15	1	10	0	1	B
	T3	13	0	5	0	15	1	10	0	1	B
Nov-14	Spring6	18	0	2	0	0	0	0	0	0	A
	T3	24	1	-1	0	0	0	0	0	1	B

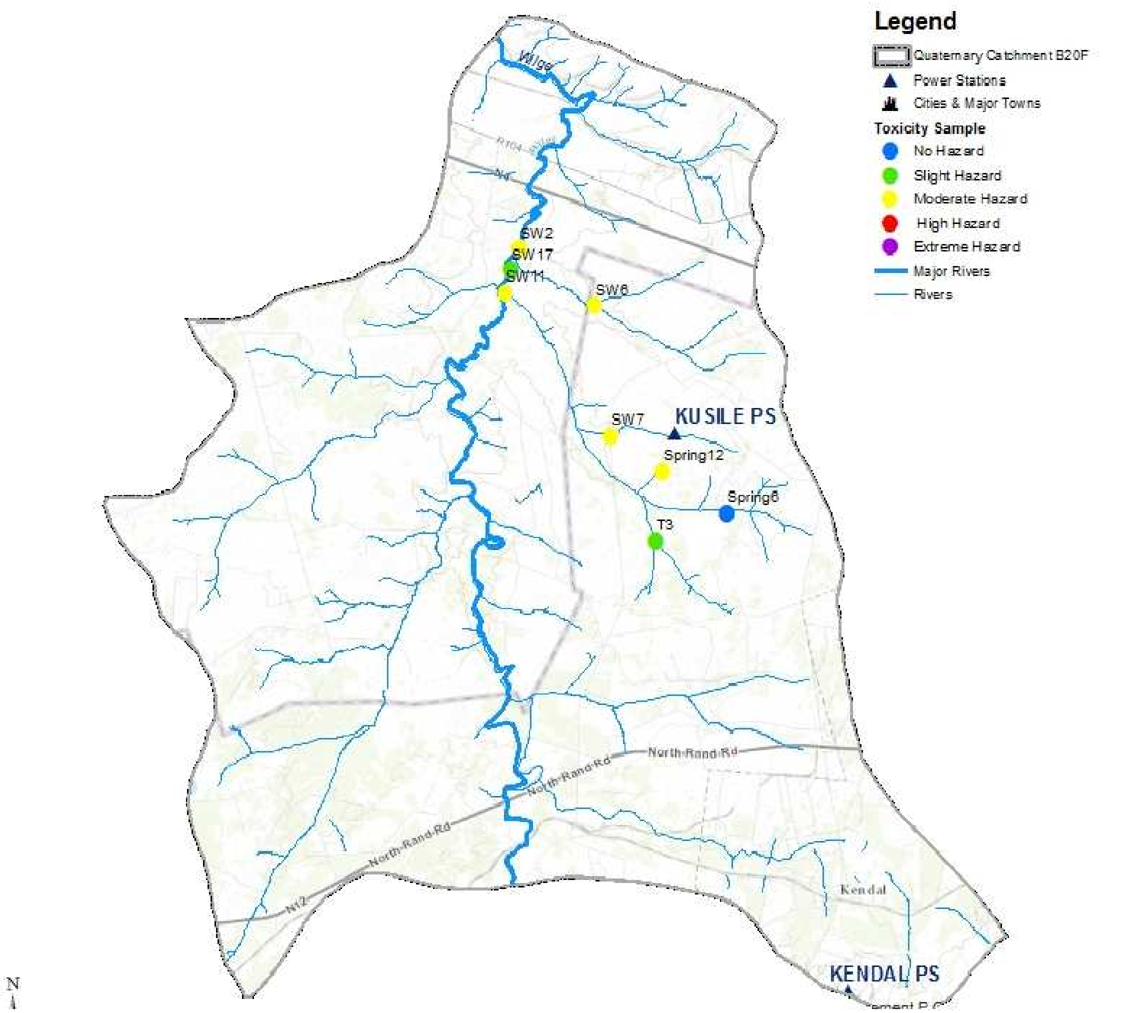

Figure 3.6: Hazard categories for last sampling survey for all toxicity sampling sites completed in 2014. Last sampling survey was May 2014 for sampling sites Spring12, SW7, SW11, SW6, SW2 and SW17; and November 2014 for sampling sites Spring6 and T3.

Sampling site Spring6 indicated unimpacted and in hazard category A in November 2012, July 2013 and January 2014. However in the March 2014 survey, Spring6 received a cumulative WHS score of 4 thereby falling within the C hazard category, showing a high potential to elicit harmful impact on the aquatic ecosystem. Spring6 is downstream of the closed New Largo colliery and agricultural and livestock farming impacts. Spring6 showed exceedances of the Wilge River IWRMP RWQOs for iron, aluminium, manganese, nickel, calcium and sulphate in the surveys undertaken from November 2012 to November 2014. Spring6 also showed a decrease in pH in the January 2014 survey indicating mining activities and impact at the sampling site. Spring6 improved to a hazard category of B in May 2014 and June 2014. In November

2014 Spring6 had recovered to its original hazard category of A indicating a potential impact to the water resources at Spring6 during the period January 2014 (after the survey) to March 2014.

Sampling site T3 was also found to be unimpacted and fell into hazard category A in November 2013 and July 2013. T3 received a cumulative WHS of 2 in the March 2014 and May 2014 surveys and fell into the hazard catergory C. T3 is found on the confluence of the Holfonteinspruit and an unnamed tributary. T3 showed exceedance of the Wilge River IWRMP RWQOs for iron, aluminium, ammonia and nickel in the surveys undertaken from November 2012 to November 2014. It is impacted mainly by agriculture. The hazard category of T3 improved to B in June 2014 and November 2014.

Spring12 indicated a cumulative WHS of 5 in March 2014 and improved to a WHS of 2 in the May 2014 survey however it remained in the hazard category of C. Spring12 is a pan draining into the Klipfonteinspruit, downstream of Spring6. It is impacted on by agriculture and construction. SW7 indicated a cumulative WHS of 4 in March 2014 and improves to a WHS of 2 in the May 2014 survey however it remained in the hazard category of C. SW7 is impacted on by agriculture and construction. SW11 indicated a cumulative WHS of 9 in March 2014 and improved to a WHS of 2 in the May 2014 survey. SW11 exhibited the highest WHS of all the surveys undertaken in the study. SW11 improved from a hazard category of D to C. SW11 is impacted on by agriculture, livestock farming, mining and construction.

SW6 indicated a cumulative WHS of 5 in March 2014 and improved to a WHS of 4 in the May 2014 survey however it remained in the hazard category of C. SW6 is located on the northern tributary of the Wilge River and is impacted on by agriculture, construction and a quarry. SW2 indicated a cumulative WHS of 5 in March 2014 and improved to a WHS of 2 in the May 2014 survey however it remained in the hazard category of C. SW2 is located on the Wilge River and is the last downstream site of the study on the Wilge River sub-catchment B20F area. SW17 indicated a cumulative WHS of 3 in March 2014 and improved to a WHS of 1 in the May 2014 survey. The hazard category of sampling site SW17 improved from a C to a D. SW17 is located on the Wilge River and is impacted on by agriculture, livestock farming and construction.

The potential hazardous effects of land-use impacts in the Olifants River WMA, Wilge River sub-catchment B20F area were evaluated using the, vertebrates (*P. reticulata*), invertebrates (*D. magna, D.pulex*), bacteria (*V. fischeri*) and primary producers (*P. subcapitata*) assays. Increases in percentage inhibition for *V. fischeri* showed an association to *D. magna/ D.pulex* mortality rates. Increases in percentage stimulation for *V.fischeri* showed an association to *P. reticulata* mortality rates, thus showing the advantage of the four-tiered toxicity assesment.

The hazard categories of the sampling sites were mostly found to be C and have a moderate hazard to toxicity, with their cumulative hazard of the toxicity test being between 2-5. Thus implying that the cumulative effects of the impacts, i.e. agriculture, livestock farming, mining, the construction site and the quarry are contributing to the increasing toxicity in the catchment. Sampling site SW11 fell into the D category in the March survey, which indicated high hazard due to toxicity. SW11 is located downstream of the Klipfonteinspruit just before the confluence with the Wilge River. Sampling site SW11 was found to have a cumulative WHS of 9 with the highest WHS calculated due to 71% inhibiton for the *V. fischeri* assay, 50% inhibition for the *P. subcapitata* assay and 100% mortality in the *Daphnia* assay. SW11 is impacted on by agriculture, livestock farming, mining and construction.

The Crocodile River system toxicity analyses, as shown by Ansara-Ross *et al*., (2009) had a much higher toxicity in comparison to the Wilge River. The Crocodile River system has intensive agricultural and urban discharges Intensive agricultural sites showed the highest effects to all tested biota. Receiving water at urban sites associated with increased nutrients and lowest pesticide usage showed few adverse effects, while the relatively unimpacted site indicated no hazard to any organism, and only a slight stimulation to algal growth (Ansara-Ross *et al*., 2009). The cumulative effect of agriculture, livestock farming, mining, the construction site and the quarry impact the Wilge River sub-catchment B20F area and are contributing to the increasing toxicity in the catchment.

3.4 Conclusions

The study showed the usefulness of combining a series of toxicity tests and the role that this can potentially play in assessing the potential ecological impacts in rivers that are affected by many different land-uses. The results showed that the use of toxicity tests produced additional information when considering the relative health of a water resource under stress. Due to the many land-uses present in the catchment, when a site has an unacceptable toxicity level and is assessed as having a high hazard score, it would be valuable to undertake comprehensive chemistry analysis at these specific sites temporally in order to assess changes in the toxicity hazard score to help identify the cause and either deterioration or improvement in conditions. The four-tiered toxicity assessments were applicable and appropriate for measuring the change in toxicity hazards due to a range of land-uses. It provides a means of protecting the ecological integrity of aquatic ecosystems by providing the environmental concerned individuals to effectively monitor and manage the water resources in the Wilge River sub-catchment B20F area.

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CHAPTER 4: SURVEY OF MACROINVERTEBRATE COMMUNITIES AS AN INDICATOR OF ECOLOGICAL HEALTH IN THE WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

4.1 Introduction

Macroinvertebrates are ubiquitous, have a high diversity, range of sensitivities, and well established sampling methodologies (Fonesca and Esteves, 1999). These organisms are therefore used extensively to assess pollution in freshwater environments (Jones *et al*., 2010; Jones *et al*., 2011). More specifically, changes in the macroinvertebrate community structure are widely used in pollution assessment studies (Bollmohr and Schulz, 2009). Community composition is influenced by an array of anthropogenic and non-anthropogenic factors, and direct causal relationships are typically not evident (Long and Chapman 1985; Gaston and Edds 1994). Research indicates that family level identification provides sufficient taxonomic resolution to detect community responses to human disturbance (Warwick 1988a; 1988b; Bouwman and Bailey 1997; Heino, 2008). Macroinvertebrates may also demonstrate the effects of past and present pollution incidents in terms of the way species have established themselves (Jefferies and Mills, 1990).

Advantages of macroinvertebrate sampling include the ability to identify taxa at various levels of taxonomic classification, which allows for cheap, straightforward and rapid on-site identification (Newson, 2005). Benthic macroinvertebrates are used to provide a general characterization of the health of a stream ecosystem, because they are relatively easy and inexpensive to collect, particularly if qualitative sampling is undertaken; they are largely non-mobile and thus representative of the location being sampled, which enables effective spatial analyses of pollutant or disturbance effects to be undertaken and they have a rapid lifecycle often based on seasons and their largely sedentary habits (Dallas *et al*., 2010). Currently, in Southern Africa, the South African Scoring System (SASS) has proven to be a fairly accurate biomonitoring tool in evaluating water quality in streams and rivers, and is now in its 5th revised form namely SASS5 (Dickens and Graham, 2002). Hill (2005) showed that integration of biological indicators (e.g. aquatic invertebrates) with chemical (e.g. metals) and physical (e.g. sediment) indicators ultimately provides information on the ecological state of a river

The Wilge River sub-catchment, with existing land-uses, namely, agriculture (main feature in the area), mining and power generation activities, is under pressure from nutrients and sulphate inputs as well (De Villiers and Mkwelo, 2009; DWAF, 2004). The in-stream and riparian habitats in this ecoregion illustrates a fair to unacceptable state, with the general condition reflecting a poor status. Biological communities further reflect a fair to unacceptable health (DWAF, 2001). South African National Biodiversity Institute (SANBI) data collections show that the aquatic species richness reflects a fair to tolerable health (Powrie LW. 2015), (Figure 4.1).

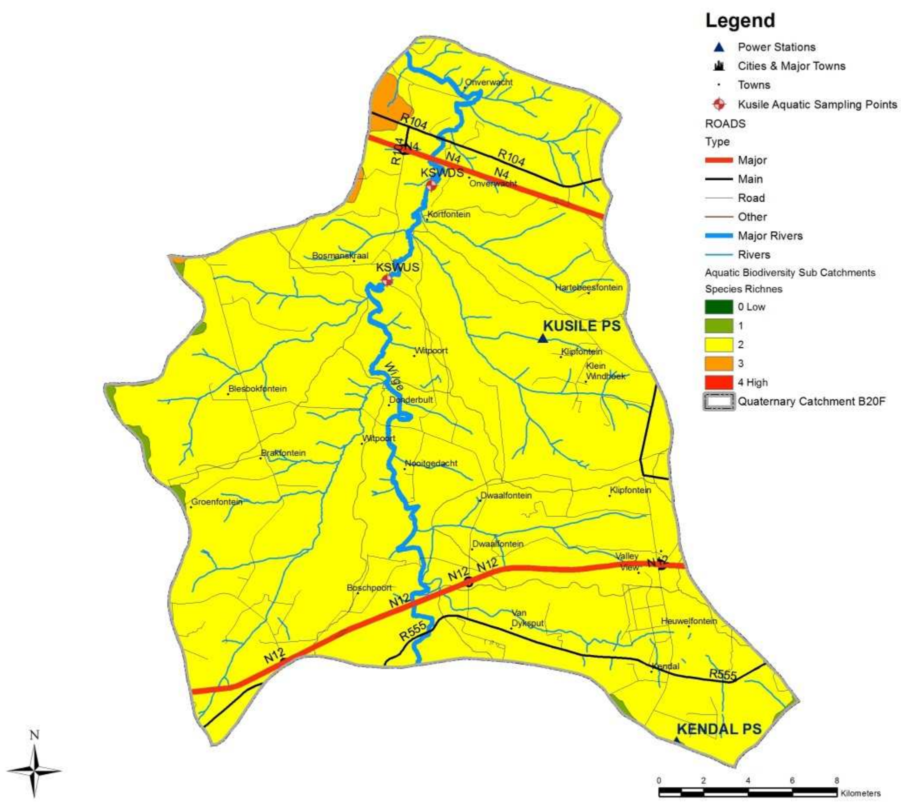


Figure 4.1: Aquatic biodiversity showing degress of species richness in Wilge River sub-catchment B20F area (Powrie LW. 2015).

However, CSIR (2012), showed that the ecological status at Loskop dam, located downstream from the confluence with the Wilge River, was recorded as 'good to fair', despite problems with mine effluent draining into the Wilge River tributary, and frequent fish deaths in the Loskop Dam. Nonetheless, there is still a concern that the rivers in this study area already contain high sediment (turbidity) and nutrient loads due to the land-use in the area (Farrell *et al.*, 2015). Over-grazing and highly erodible soils is causing severe erosion, resulting in high levels of suspended solids being transported into the Wilge River. Increases in sedimentation erosion may result in a loss of habitat diversity and quality thus contributing to biological community and integrity alteration. Macroinvertebrate assemblages are often used to determine biotic integrity or ecological health of river ecosystems (Oberholster *et al*., 2005; Malherbe *et al.,* 2010). Farrell *et al.*, (2015) showed that seasonal changes, associated flow conditions, *in situ* water quality and habitat availability play a crucial role in the complexity of aquatic macroinvertebrate communities along the Wilge River.

The aim of this chapter was to investigate the macroinvertebrates communities within the Wilge River sub-catchment B20F area to identify driving variables that influence these communities both spatially and temporally, in order to determine whether the macroinvertebrate communities can be used as indicators of water quality and ecosystem health.

4.2 Materials and methods

The study area for this project was located within the Olifants River Water Management Area (WMA4), within the quaternary drainage region B20F in the Wilge River catchment. The sample sites are located on the Wilge River, the unnamed tributary of the Klipfonteinspruit River and on an unnamed tributary of the northern tributary of the Wilge River and the Klipfonteinspruit River (Table 4.1 and Figure 4.2). These sample sites are impacted on by extensive agricultural activities (i.e. cattle, pig and chicken farming), pivot irrigation, rural development, infrastructure (i.e. railways and tar and/or gravel roads), industrial activities (i.e. construction footprint and existing industrial complexes) and mining and quarries (from tributaries entering the Wilge River) (Farrell *et al*., 2015).

Table 4.1: Biomonitoring sampling points and GPS co-ordinates.

Site	Latitude	Longitude	River	Description
SW5	-25.93887	28.89471	Klipfonteinspruit	This site is located at the confluence of the Klipfonteinspruit and the Holfonteinspruit.
SW7	-25.926361	28.894417	Unnamed tributary of the Klipfonteinspruit	This site is located on the unnamed tributary of the Klipfonteinspruit.
SW1	-25.91424	28.88064	Klipfonteinspruit	This site is located on the Klipfonteinspruit, downstream of the unnamed tributary
SW9	-25.90738	28.92702	Unnamed northern tributary of the Wilge River	This site has been located as an upstream point for the northern tributary.
SW4	-25.89075	28.89027	Unnamed northern tributary of the Wilge River	This site has been located as a downstream point for the northern tributary.
S2	-26.04485	28.86745	Wilge River	This site is located in the upper Wilge River Catchment, where the R555 crosses the Wilge River.
S3	-25.96092	28.85101	Wilge River	This site is located on the Wilge River upstream of the Klipspruit confluence.
S4	-25.95442	28.83999	Klipspruit	This site is located on the Klipspruit River, west of the Wilge River.
SW16	-25.90219	28.85142	Wilge River	This site is located in the Wilge River upstream of the Topigs farm.
SW17	-25.87476	28.86313	Wilge River	This site is located in the Wilge River downstream of the Topigs farm.
SW2	-25.86431	28.86893	Wilge River	This site is located in the Wilge River downstream of the B20F catchment

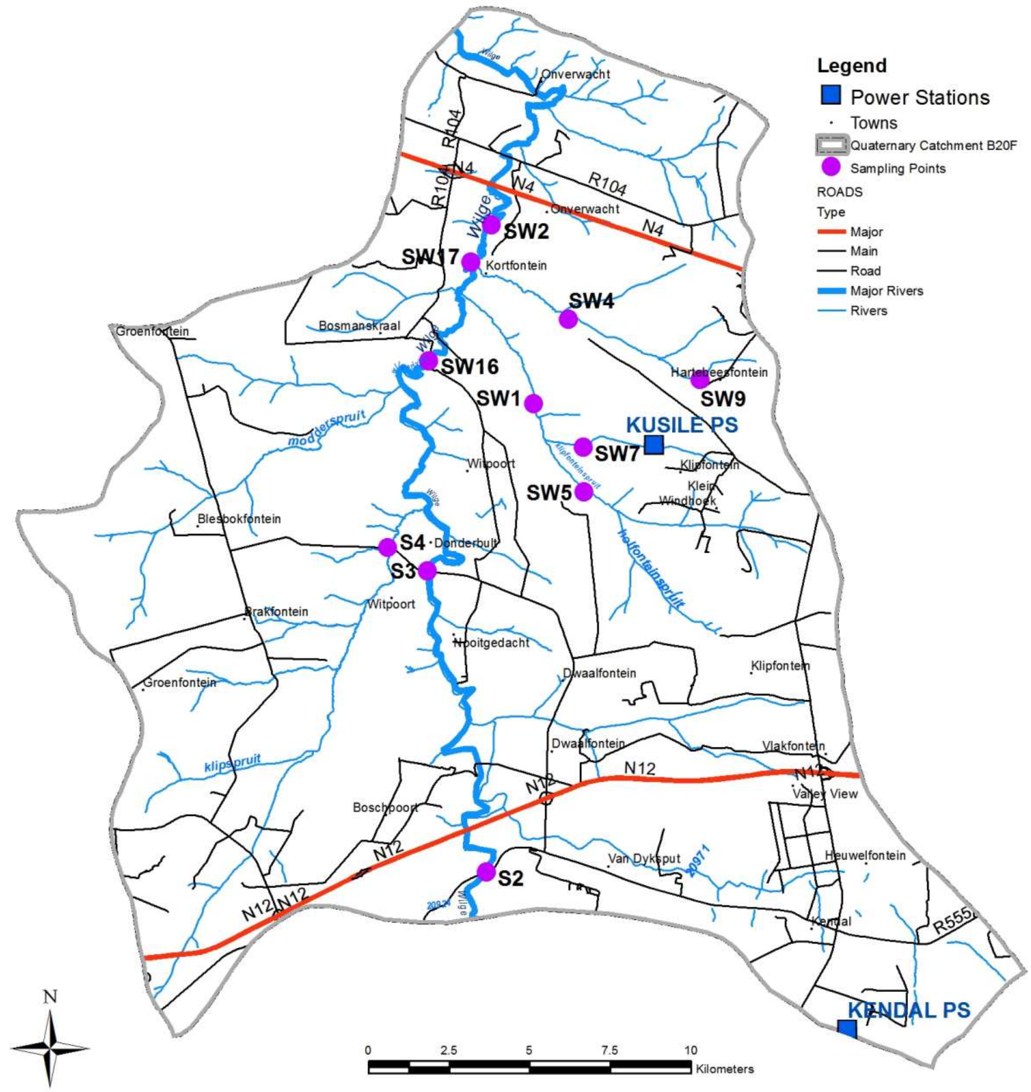


Figure 4.2: Map of Kusile power station study area, indicating streams and selected monitoring sites.

Twenty four surveys were carried out at the sampling sites between March 2010 to December 2014 (Table 4.2). For 18 surveys, *in situ* water quality analyses (pH, electrical conductivity (EC), dissolved oxygen (DO); oxygen saturation (DO%), total dissolved solids (TDS) and temperature (TEMP) were undertaken at all 11 sampling sites. In cases where DO was measured in mg/L, a conversion to % saturation was calculated by using the Saturation Level Table from US EPA, (2015). The (SASS5) analyses (presence and absence only) and Integrated Habitat Assessment System (IHAS) analyses were undertaken as well. Aquatic assessments were undertaken in March 2010, June 2010, September 2010, December 2010, March 2011, June 2011, September 2011, November 2011, August 2012, December 2012, February 2013, May 2013, August 2013, November 2013, March 2014, May 2014, August 2014 and December 2014. For the remaining 6 surveys at sampling sites, SW16 and SW2, *in situ* water quality analyses (pH, EC, DO, TDS and TEMP); comprehensive chemistry (Refer to 4.2.2) analyses (taken for laboratory analyses); SASS5 (actual macroinvertebrate counts) and IHAS were undertaken in May 2012, November 2012, July 2013, January 2014, June 2014 and November 2014.

The data were compared to the aquatic assessments done for the EIA in September 2006 at sampling sites SW5, SW1, S2, S3 and S4. For the EIA, *in situ* water quality analyses, SASS5 analyses (presence and absence only) and IHAS analyses were undertaken.

Table 4.2: List of survey dates and sampling sites.

	EIA	18 Surveys	6 Surveys
Sample Points	SW5, SW1, S2, S3 and S4	SW5, SW7, SW1, SW9, SW4, S2,S3,S4,SW16,SW17 and SW2	SW16 and SW2
Parameters	In situ chemistry analyses, SASS5 analyses (presence and absence only) and, IHAS analyses were undertaken.	In situ chemistry analyses, SASS5 analyses (presence and absence only) and, IHAS analyses were undertaken.	In situ chemistry analyses, comprehensive chemistry analyses, SASS5 (actual macroinvertebrate counts) and, IHAS analyses were undertaken.
Survey Dates	Aquatic assessments were undertaken in September 2006.	Aquatic assessments were undertaken in March 2010, June 2010, September 2010, December 2010, March 2011, June 2011, September 2011, November 2011, August 2012, December 2012, February 2013, May 2013, August 2013, November 2013, March 2014, May 2014, August 2014, and December 2014.	Aquatic assessments were undertaken in May 2012, November 2012, July 2013, January 2014, June 2014 and November 2014.

4.2.1 *In situ* water quality

Water quality parameters pH, EC, DO and/or DO%, TDS and Temp were measured *in situ* at all surveys during 2006-2014 using a portable handheld instruments. Table 4.3 presents the different water quality equipment used to measure *in situ* variables between 2012 and 2014. All meters were calibrated to ensure accuracy of the results.

Table 4.3: Instruments used to measure *in situ* water quality parameters.

Parameters	EIA	18 Surveys		6 Surveys	
	2006	2010-2013	2014	May 2012	Nov 2012-2014
Conductivity	By calculation	Eutech EC tester II Dual range	EXTEXH II handheld	Hanna Instruments HI98129)	YSI 556 Multiparameter system (MPS)
pH	pH scan	Eutech pH tester	EXTEXH II handheld	Hanna Instruments HI98129)	YSI 556 Multiparameter system (MPS)
DO%	By calculation	Eutech CyberScan DO110	By calculation	Hanna Instruments HI9143	YSI 556 Multiparameter system (MPS)
TDS	TDS scan	By calculation	By calculation	By calculation	YSI 556 Multiparameter system(MPS)
Temperature	Alcohol Thermometer	Eutech CyberScan DO110	EXTEXH II handheld	Hanna Instruments HI9143	YSI 556 Multiparameter system (MPS)

All data were converted to similar units in order to be comparable. All the parameter results were compared to the Wilge River IWRMP RWQOs. Refer to Chapter 2, Table 2.3. As there are no temperature guidelines in the IWRMP, temperature readings were compared to the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) of 5-30 ℃.

4.2.2 Chemistry Analyses

Samples for omprehensive chemistry analyses were taken at sample sites SW16 and SW2 only during the May 2012, November 2012, July 2013, January 2014, June 2014 and November 2014 surveys. Two 1 litre plastic bottles were rinsed and filled with water at the site. The first 1 litre bottle was stored in the cooler box and the second sample was preserved with nitric acid (HNO_3) for heavy metals analyses before being stored in the cooler box. The samples were analysed for a number of water quality variables, including, aluminum (Al), ammonia (NH_3), antimony (Sb), arsenic (As), cadmium (Cd), calcium (Ca), chemical oxygen demand (COD), chloride (Cl), cyanide (CN), cobalt (Co), total chromium (Cr), EC, dissolved organic carbon (DOC), iron (Fe), fluoride (F), potassium (K), magnesium (Mg), manganese (Mn), sodium (Na), nickel (Ni), nitrate (NO_3), lead (Pb), pH, ortho-phosphate (PO_4), strontium (Sr), sulphate (SO_4), TDS, total organic carbon (TOC), turbidity, vanadium (V) and zinc (Zn). All samples were taken to Eskom Analytical chemistry laboratory for analyses.

The following analytical methods were used for the analyses of water quality variables:

- Determination of pH with a Metrohm 862 Compact Titrosampler; EC (mS/m) with a Metrohm 912 Conductometer and turbidity (NTU) with a Hach 2100AN Turbidometer in water samples;
- TDS by calculation;
- TOC by Elemental analyser;
- The determination of dissolved organic carbon (DOC) was undertaken with an APOLLO 9000;
- The spectrophotometric analysis and calculation in determination of NH_4^+ and COD in water samples;
- Anions (NO_3^-, SO_4^{2-},PO_4^{3-}, Cl^- and F^-) by ion chromatography;
- As, Se and Hg with an Atomic Absorption Spectrometer;
- The determination of the remaining elements in liquid samples, with ICP-MS analysis.

The Wilge River sub-catchment B20F area falls within MU 22 management unit. The IWRMP for the Upper and Middle Olifants River catchment, recommends the following Resource Water Quality Objectives for the Wilge River sub-catchment. All the parameter results were compared to the IWRMP RWQOs. For parameters analysed where the IWRMP RWQOs were not given, the Target Water Quality Objectives (TWQO) of the South African water quality guidelines for aquatic ecosystems (DWAF, 1996), (Chapter 2, Table 2.3 and 2.4), and the Australian and New Zealand Guidelines (ANZECC, 2000) were consulted (Chapter 3, Table 3.4).

4.2.3 Aquatic invertebrate assessment: South African Scoring System 5.

Benthic macroinvertebrate communities of the selected sites were investigated according to the SASS5 approach (Dickens and Graham, 2002). This method is based on the British Biological Monitoring Working Party (BMWP) method and has been adapted for South African conditions by Dr. F. M. Chutter (Thirion *et al.,* 1995). The SASS method is a rapid, simple and cost effective method, which has progressed through four different upgrades/versions. The current upgrade is Version 5, which is specifically designed to comply with international accreditation protocols.

4.2.3.1 Sample Collection

An invertebrate net (30 x 30 cm square with 1 mm mesh netting) was used for the collection of the organisms. The available biotopes at each site were identified on arrival. Each of the biotopes was sampled by different methods explained below.

Stone (S) Biotopes

Stones in current (SIC) or any solid object: Refers to a biotope where movable stones of at least cobble size (3 cm diameter) to approximately 20 cm in diameter, within the fast and slow flowing sections of the river are present. Kick-sampling is used to collect organisms in this biotope. This is done by putting the net on the bottom of the river, just downstream of the stones to be kicked, in a position where the current will carry the dislodged organisms into the net. The stones are then kicked over and against each other to dislodge the invertebrates (kick-sampling) for ± 2 minutes.

Stones out of current (SOOC): Refers to a biotope where the river is still, such as behind a sandbank or ridge of stones or in backwaters. Collection is again done by the method of kick-sampling, but in this case the net is swept across the area sampled to catch the dislodged biota. Approximately 1 m^2 is sampled in this way.

Bedrock or other solid substrate: Refers to bedrock which includes stones greater than 30cm, which are generally immovable, including large sheets of rock, waterfalls and chutes. The surfaces are scraped with a boot or hand. The net is swept across the area sampled to catch the dislodged biota for ± 1 minute.

Vegetation (VG) Biotopes

Marginal vegetation (MV): This is the overhanging grasses, bushes, twigs and reeds growing on the edge of the stream, often emergent, both in current (MvegIC) and out of current (MvegOOC). Sampling is done by holding the net perpendicular to the vegetation (half in and half out of the water) and sweeping back and forth in the vegetation (± 2 m of vegetation).

Submerged vegetation (AQV): This vegetation is totally submerged and includes Filamentous algae and the roots of floating aquatics such as water hyacinth. Sampled by pushing the net (under the water) against and amongst the vegetation in an area of approximately one square meter.

Gravel, Sand and Mud (GSM) biotopes

Sand: This includes sandbanks within the river, small patches of sand in hollows at the side of the river or sand between the stones at the side of the river. This biotope is sampled by stirring the substrate by shuffling or scraping of the feet, which is done for half a minute, whilst the net is continuously swept over the disturbed area.

Gravel: Gravel typically consists of smaller stones (2-3 mm up to 3 cm). Sampling is similar to that of sand.

Mud: It consists of very fine particles, usually as dark-collared sediment. Mud usually settles to the bottom in still or slow flowing areas of the river. Sampling is similar to that of sand.

Hand picking and visual observation

Before and after disturbing the site, approximately 1 minute of “hand-picking” for specimens that may have been missed by the sampling procedures was carried out.

4.2.3.2 Sample preparation

The organisms sampled in each biotope group were identified, enumerated and noted on the SASS5 datasheet. The SASS5 score and Average score per Taxon (ASPT) were determined, based on macroinvertebrate diversity. The ASPT scores provide an indication of the average tolerance/ intolerance of the aquatic macroinvertebrate community at each site. Modelled reference conditions for the Highveld Ecoregion were obtained from Dallas (2007) (Table 4.4). In certain cases when the class could not have been determined with both SASS5 and ASPT scores (i.e. one score out of range) then the SASS5 score alone was used to determine the class.

Table 4.4: Modelled reference conditions for the Highveld Ecoregion (11) based on South African Scoring System 5 (SASS5) and Average Score Per Taxon (ASPT) values (Dallas, 2007).

SASS	ASPT	Class	Description
>124	>5.6	A	Unimpaired. High diversity of taxa with numerous
83-124	4.8-5.6	B	Slightly impaired. High diversity of taxa, but with fewer
60-82	4.6-4.8	C	Moderately impaired. Moderate diversity of taxa
52-59	4.2-4.6	D	Considerablely impaired. Mostly tolerant taxa present
30-51	Variable	E	Severely impaired. Only tolerant taxa present
<30	Variable	F	Critically impaired. A few tolerant taxa present

4.2.4 Habitat Assessment

An evaluation of habitat quality and availability to biota is critical to any assessment of ecological integrity and should be conducted at each site at the time of biological sampling. On site habitat assessments were conducted by using existing habitat evaluation indices (McMillan, 1998). The Integrated Habitat Assessment System (IHAS) was developed specifically for use with the SASS5 index and rapid biological assessment protocols in South Africa. The index considers sampling habitat and stream characteristics. The sampling habitat is broken down into categories, these being stone-in-current, vegetation and other habitat/general. All of these add up to a possible 100 points (or percentage). It is presently thought that a total IHAS score over 65% represents good habitat conditions, a score over 55% indicates adequate/fair habitat conditions and anything below 55% is poor (MacMillan, 1998) (Table 4.5).

Table 4.5: Integrated Habitat Assessment System Scoring Guidelines (Version 2).

IHAS score	Description
>65%	Good
55-65%	Adequate/Fair
<55%	Poor

4.2.5 Soil erosion and sediment deposition

The visual assessment approach adopted by Golder Associates Africa Project (2012) is also included in this study. During the visual assessment each sampling site was evaluated for signs of recent soil erosion and sediment deposition. The scoring system is represented in the Table 4.6.

Table 4.6: Scoring system for assessing current erosion and sediment deposition.

Parameter	Score	Interpretation
Soil Erosion	0	None
	1	Slight
	2	Moderate
	3	Large, active
	4	Severe
Soil deposition	0	None
	1	Slight covering, muddy water
	2	Moderate
	3	Large plants extend through sediment layer
	4	Severe, forming a fan

4.2.6 Statistical analyses

Standard Excel column graphs for IHAS, SASS and ASPT were carried out on Microsoft Office Professional Plus 2010, Microsoft Excel Version: 14.0.7015.1000, 32-bit. The specific water quality requirements (Thirion, 2007) for all taxa were used. Redundancy Analysis (RDA) was used to explore the relationship between water chemistry variables and spatial distribution of macroinvertebrates. Actual species counts, species presence and absence and water quality parameters measured in water were included in the analysis. Analysis was conducted using the CANOCO for Windows package, version 4.5 (Ter Braak and Smilauer, 2002).

4.3 Results and discussion

4.3.1 *In situ* water quality and presence and absence macroinvertebrate data (2006-2014) for all 25 surveys

In situ water quality

Water quality is one of the most important factors which influence an aquatic ecosystem's integrity, as the distribution of aquatic freshwater organisms is controlled mainly by water quality characteristics, including dissolved oxygen, acidity and nutrient content (Dallas and Day, 1993). Conductivity, pH, EC, DO and/or DO%, TDS and TEMP were measured *in situ* at all sites for 25 surveys (including the EIA) and were compared to the Wilge River IWRMP RWQOs guideline and the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996). These guidelines are used to assess the present condition of the river systems and the extent of degradations.

Data from September 2006 to December 2014 illustrate that the pH values have fluctuated both spatially and temporally and in most instances have been within the Wilge River IWRMP RWQOs (Figure 4.3). The pH values recorded have been mostly alkaline. The pH readings were found to exceed the Wilge River IWRMP RWQOs upper limit at all the sampling sites. This was observed at SW5 in the September 2011 survey, SW7 in September and November 2011 surveys, SW4 in the September 2010 and December 2010 surveys; at S3 in the March 2010 and June 2011 surveys; at S4 in the December 2010 and June 2011 surveys; at SW16 in the March 2010 and June 2011 surveys; at SW17 in the March 2010 December 2010 and June 2011 surveys; at SW2 in the March 2010, December 2010 and June 2011 surveys. Sampling site S2 exceeded the Wilge River IWRMP RWQO of 8.4 on numerous occasions in the sampling period.

The pH readings were found to fall below the Wilge River IWRMP RWQOs lower limit at sampling sites, SW5, SW4, SW16 and SW2. This was observed once at SW5 in the May 2013 survey with a pH reading of 5.9; SW4 in the February 2013 survey with a pH reading of 6.2; at SW16 in May 2012, August 2012 and January 2014 surveys with pH readings of 5.9; 5.9 and 5.8 respectively. The pH reading fell below the Wilge River IWRMP RWQOs lower limit at sampling site SW2 in May 2012, November 2013 and January 2014 surveys with pH readings of 6.3, 6.1 and 4.23 respectively.

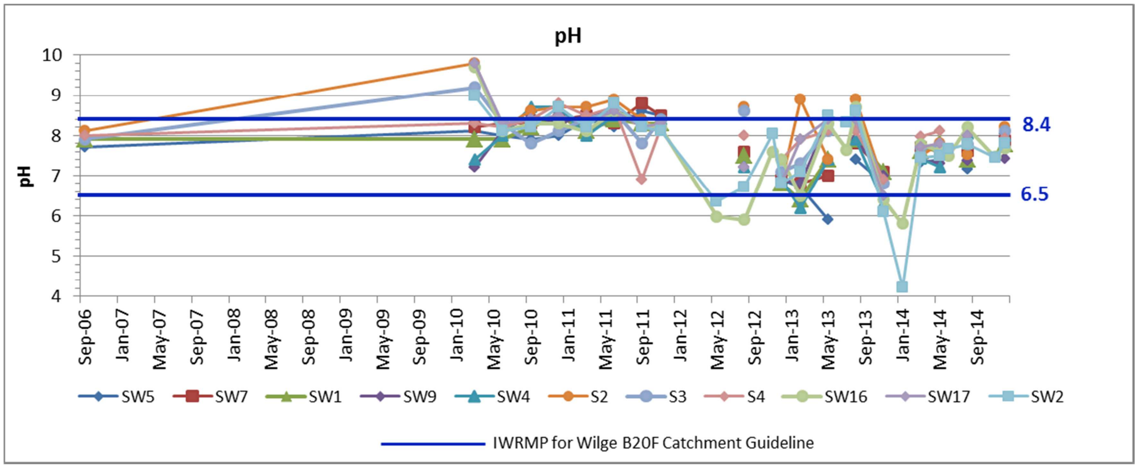


Figure 4.3: Line graph showing pH levels recorded at the eleven sampling sites during 2006-2014. Blue lines indicate the Integrated Water Resources Management Plan Guideline for pH.

The exceeded and low pH values may have resulted in a range of physiological stresses on the aquatic biota at the time of surveys. However, the pH subsequently recovered and returned back to its general alkaline trend along the Wilge River following the surveys.

Data from September 2006 to December 2014 illustrate that the EC values have fluctuated both spatially and temporally in most instances at sampling sites, SW7, SW9, SW4 and S4 have stayed below the Wilge River IWRMP RWQOs (Figure 4.4). The EC at sampling sites S2 and S3 exceeded the Wilge River IWRMP RWQOs most instances. The highest EC reading was recorded at the most upstream site of the Wilge River, sampling site S2 in the May 2013 survey. The EC readings subsequently decreased in a downstream direction along the Wilge River.

The highest EC reading was recoded at the most upstream site of the Wilge River, sampling site S2. The EC readings subsequently decreased in a downstream direction along the Wilge River. As the EC readings were found to be generally higher within the Wilge River, this may likely be due to an input of salts along the river, possibly by agricultural activities.

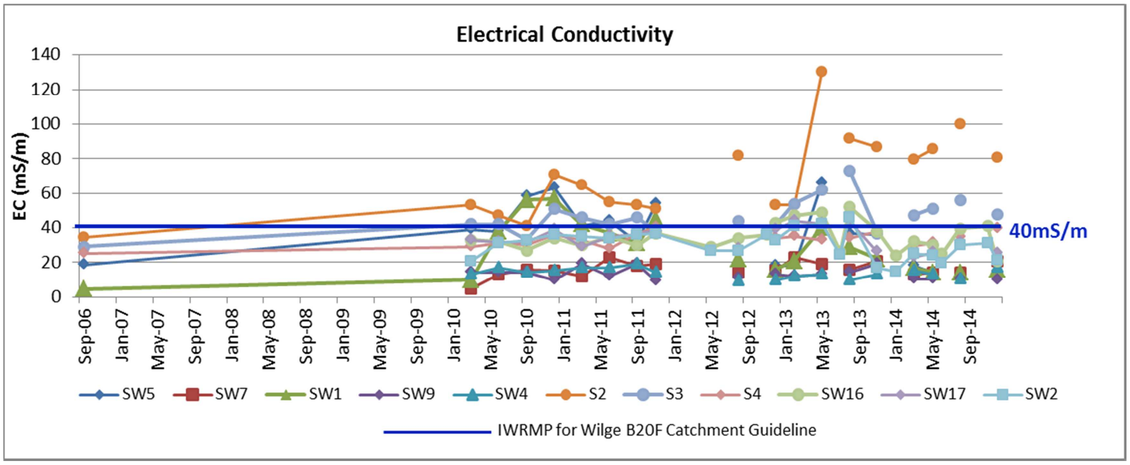


Figure 4.4: Line graph showing electrical conductivity levels recorded at the eleven sampling sites during 2006-2014. The blue line indicates the Integrated Water Resources Management Plan Guideline for electrical conductivity.

Percentage oxygen saturation is the amount of oxygen (O_2) in a litre of water relative to the total amount of oxygen that the water can hold at that temperature. DO levels fluctuate seasonally and diurnally over a 24 hour period and vary with water temperature and altitude. The DO% of 80-120% is needed to protect aquatic biota through most life stages and a DO% below 40% would be lethal (DWAF, 1996). The Wilge River IWRMP RWQOs for DO% is 70% to which most of the sampling site complied. DO% was observed to be below 40% often at SW9, SW2 and SW16 (Figure 4.5). DO% levels fell below 40% at sampling sites SW4, S3, S4, SW16, SW17 and SW2 during the March 2010 survey. The lowest reading of 8.3 was recorded at SW2 in March 2010. Since the March 2010 survey, the DO% had subsequently recovered and in most instances exceeded the Wilge River IWRMP RWQOs limit. The drop might been due to an instrument error.

Eutrophication is associated with nutrient enrichment which may be a contributing factor to the low DO concentrations throughout the catchment. It may further be associated with a combination of gradients and habitat (Davies and Day, 1998). All the sampling sites are located at shallow gradients, coupled with limited rocky habitats, which functions as an aeration mechanism, thus oxygenating the water. Furthermore, the amount of oxygen dissolved in the water is influenced by the aeration rate from the atmosphere, temperature, air pressure and salinity, as well as from the comparative rates of respiration and photosynthesis (Davies and Day, 1998). The low DO% readings observed may have a limiting effect on aquatic biota.

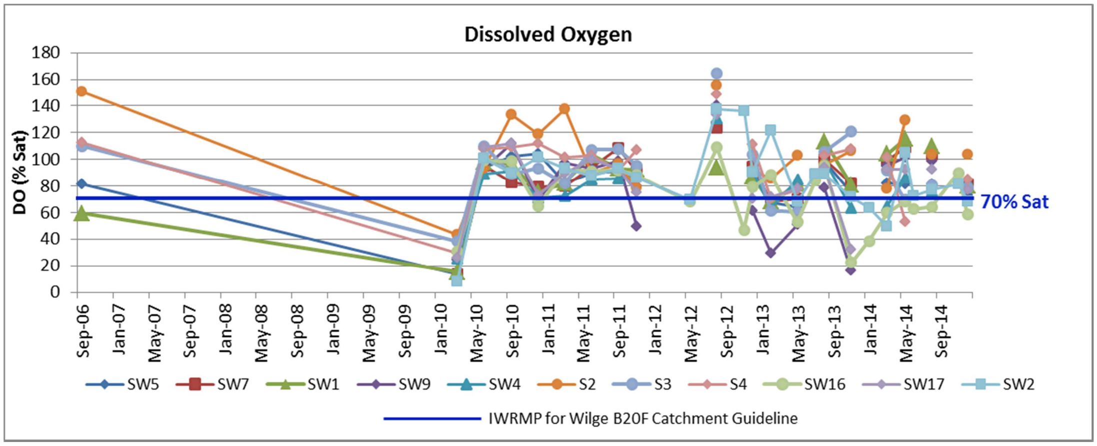


Figure 4.5: Line graph showing dissolved oxygen levels recorded at the eleven sampling sites during 2006-2014. The blue line indicates the Integrated Water Resources Management Plan Guideline for dissolved oxygen.

The temperatures of inland waters range from 5-30 °C, as per the South African Water Quality Guidelines: Aquatic Ecosystems (DWAF, 1996). Historical data illustrates that the temperature during September 2006 to December 2014 at all sampling sites during all surveys were observed to be within these guidelines (Figure 4.6).

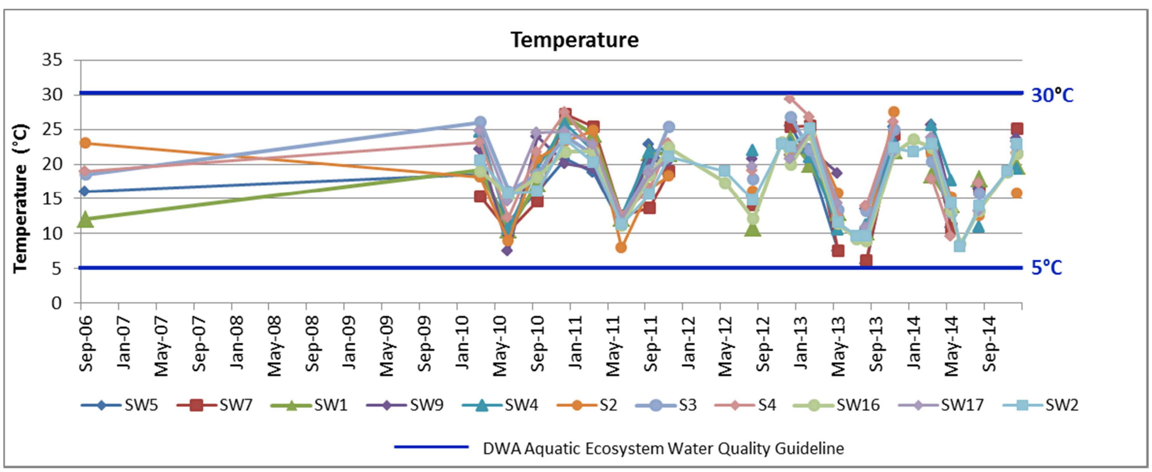


Figure 4.6: Line graph showing temperature levels recorded at the eleven sampling sites during 2006-2014. Blue lines indicate the South African Water Quality Guideline: Aquatic Ecosystem (1996) for temperature.

Water temperatures play an important role in aquatic ecosystems by affecting the rates of chemical reactions and therefore also the metabolic rates of organisms (DWAF, 1996). Temperature varies with seasons and affects the rate of development, reproductive periods and emergence time of aquatic macroinvertebrates (Thirion, 2007). Increasing temperature is known to cause slight decrease in pH (Gleick, 1993). The temperatures were considered normal for these systems and clearly reflected seasonal variation over time. Therefore, temperatures were not expected to have a limiting effect on aquatic biota.

Macroinvertebrate Community

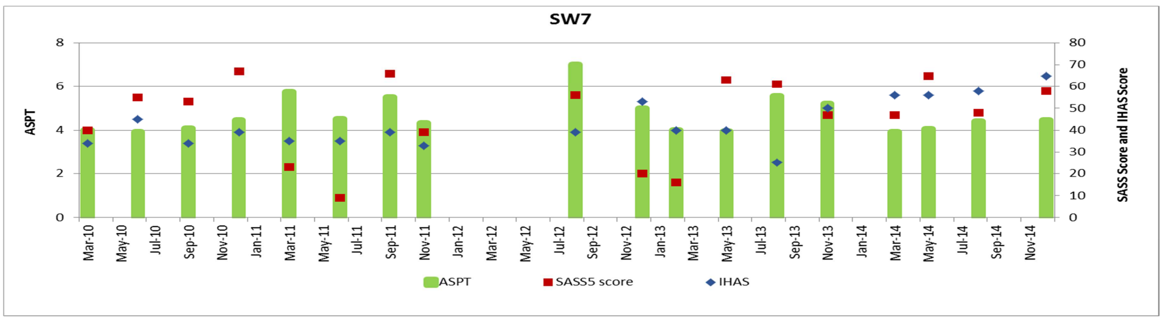
The SASS5, ASPT, IHAS scores and percentage taxa per survey showing specific water quality requirements (Chutter, 1998; Thirion, 2007) for all sample sites during 25 surveys (including the EIA) are shown in Figures 4.7-4.17. SASS5 scores range from 8 at sample site SW7 in June 2011 survey to 165 at sample site S3 in November 2013 survey. IHAS scores ranged from 14 at sampling site S3 in the August 2012 survey to 86 at S3 in the December 2012 survey. ASPT scores ranged from 2.59 at sample site SW5 in February 2013 to 7.73 at sample site S3 in March 2010. Refer to Appendix B for the SASS, IHAS, ASPT and number of taxa data from September 2006 to December 2014.

Overall the data indicated that the SASS5, IHAS and ASPT scores were variable both spatially and temporally. The SASS5 and ASPT scores at SW5 fluctuated throughout the study period (Figure 4.7(A)). Habitat conditions at SW5 were found to be poor and not exceeding 56% throughout the study period. The highest SASS5 scores of 82 and 80 for the March 2014 and May 2014 surveys respectively had low ASPT scores, indicating the presence for higher numbers of macroinvertebrates but ones that have very low water quality preferences as seen in Figure 4.7(B). ASPT score was found to be 4.7 for the initial survey in September 2006, improve to 5.7 in the June 2010 survey and to 6.0 in the June 2011 survey. The lowest ASPT score for SW5 was 2.59 in the February 2013 survey. Macroinvertebrates found at SW5 with high requirements for water quality preferences were Baetidae (>2 species) and Hydropsychidae (>2 species). SW5, a downstream point on the Klipfonteinspruit, is impacted upon by associated infrastructure i.e. bridge construction which results in ongoing deposition of sediment in the river channel at this point. As Hydropsychidae prefer cobbles as their preferred habitat (De Moor and Scott, 2003; Thirion, 2007) the change of biotope may limit their presence and result in lower SASS5 and ASPT scores.

Figure 4.7: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW5 from September 2006 to December 2014.

The SASS5 and ASPT scores at SW7 fluctuated throughout the study period (Figure 4.8(A)). Habitat conditions at SW7 were found to be between poor to fair throughout the study period. The highest SASS5 scores of 67 and 66 were found during the December 2010 and September 2011 surveys respectively. Macroinvertebrates that have high water quality preferences were not found at sampling site SW7 until the August 2012 survey (Figure 4.8(B)). ASPT score was found to be 7.0 for the August 2012 survey with no significant change in habitat conditions indicating an improvement in water quality at SW7 in August 2012. Macroinvertebrates found at SW7 with high water quality preferences were Baetidae (>2 species), Heptagenidae and Hydropsychidae (>2 species). SW7 is situated downstream of a newly constructed road. Construction activities and lack of bank rehabilitation may result in ongoing deposition of sediment in the river channel at this point. As Heptagenidae and Hydropsychidae prefer cobbles as their preferred habitat (Barber-James and Lugo-Oritz, 2003; De Moor and Scott, 2003; Yabe and Nakatsugawa, 2004; Thirion, 2007) the change in biotope may limit their presence and result in lower SASS5 and ASPT scores.

A



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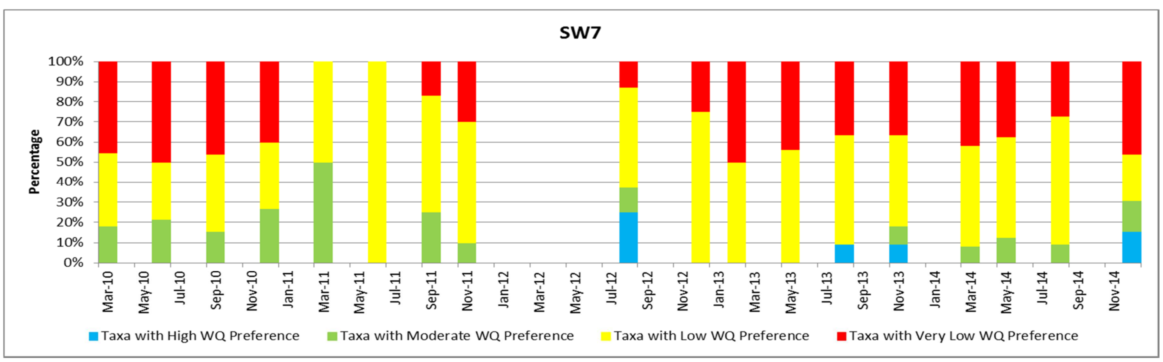


Figure 4.8: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW7 from March 2010 to December 2014.

The SASS5, ASPT and IHAS scores at SW1 showed an improvement during the study period (Figure 4.9(A)). Habitat conditions at SW1 were found to be between poor to fair throughout the study period. Habitat conditions improved from poor to fair in December 2012 and thereafter improved to good in March 2014. The improvement of the stones and vegetation biotopes, which provides habitat for most macroinvertebrates, resulted in the subsequent increase in the SASS5 scores. The highest SASS5 scores of 102 and 120 were found at the May 2014 and August 2014 surveys respectively. Macroinvertebrates that have high water quality preferences were not found at sampling site SW1 until the June 2011 survey (Figure 4.9(B)). There has been an increase in ASPT scores and a collaborating increase in the presence of macroinvertebrates that have high water quality preferences at SW1 from June 2011 onwards. Macroinvertebrates found at SW1 with high water quality preferences were Baetidae (>2 species), Heptagenidae and Hydropsychidae (>2 species). As Heptagenidae and Hydropsychidae prefer cobbles as their preferred habitat (Barber-James and Lugo-Oritz, 2003; De Moor and Scott, 2003; Thirion, 2007) the change of biotope may limit their presence and result in lower SASS5 and ASPT scores.

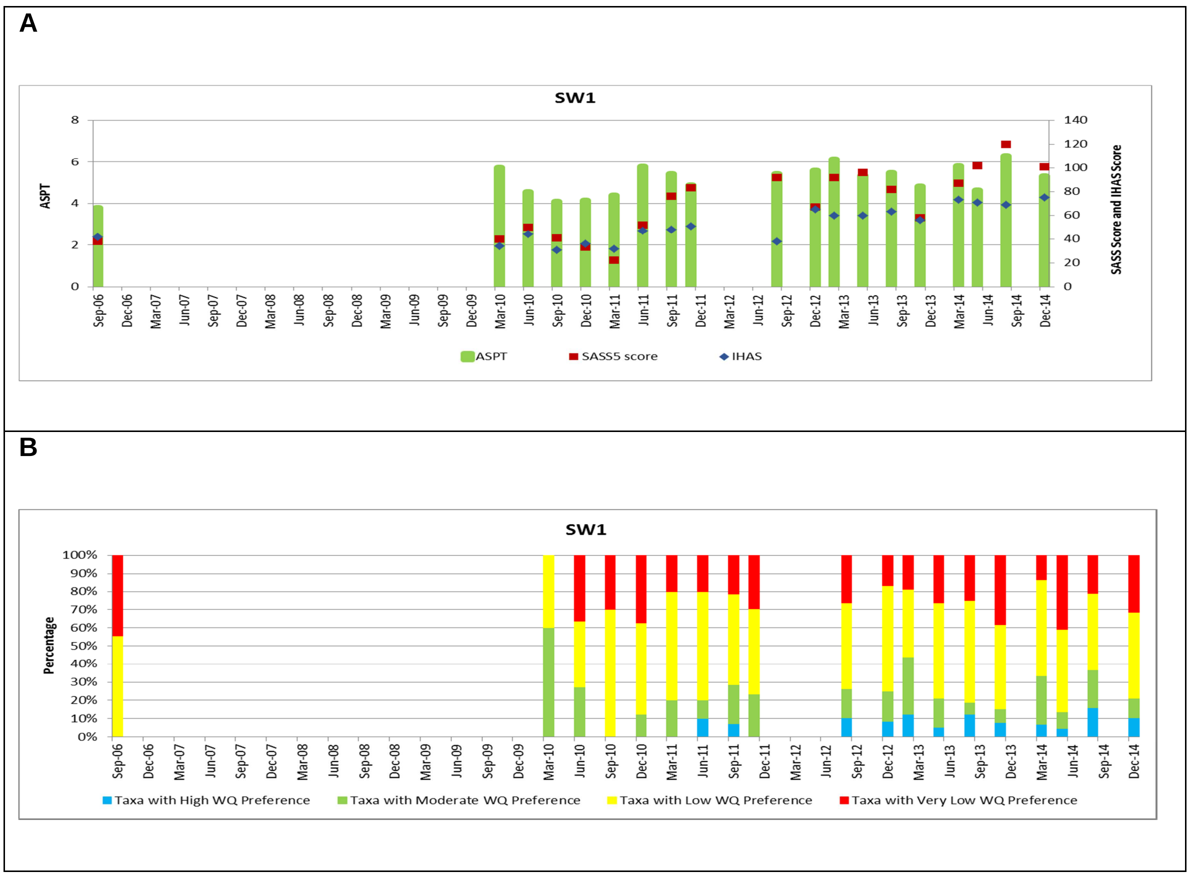


Figure 4.9: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW1 from September 2006 to December 2014.

The SASS5, ASPT and IHAS scores at SW9 showed a combined deterioration from June 2011 to November 2013 (Figure 4.10(A)). Subsequent improvements in scores were found from March 2014. As habitat conditions deteriorated at SW9 from good (65) in March 2010 to the lowest score of 32 in May 2013, SASS5 and ASPT scores showed a resulting decrease. The decrease in habitat conditions show an increase in macroinvertebrates that prefer moderate to very low water quality conditions, indicating a deterioration in water quality at SW9 as well. Macroinvertebrates that have high water quality preferences were not found at sampling site SW9 until the August 2014 survey once habitat conditions improved (Figure 4.10(B)). Macroinvertebrates found at SW9 with high water quality preferences were Baetidae (>2 species) only.

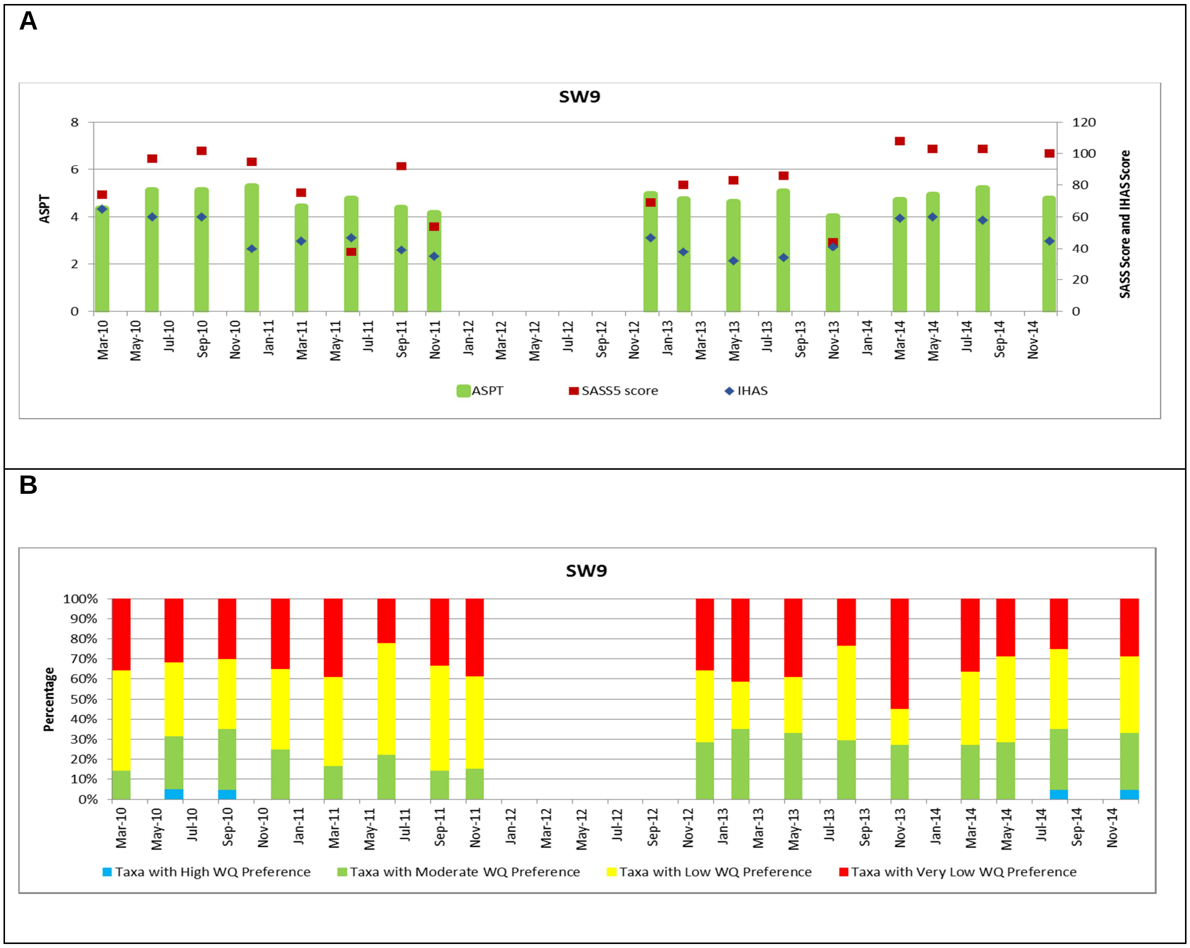


Figure 4.10: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW9 from March 2010 to December 2014.

The SASS5, ASPT and IHAS scores at SW4 fluctuated throughout the study period (Figure 4.11(A)). Habitat conditions at SW4 was found to be between poor to fair throughout the study period except for the June 2010 survey when the highest IHAS score of 67 was achieved. Stones and vegetation biotopes which provide habitat for most macroinvertebrates were limited at this site, and the presence of mostly reduced flow conditions may be contributing to the low SASS5 scores. The highest SASS5 scores of 118 and 113 were found at the June 2010 and February 2013 surveys respectively. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.11(B)). However, an increase in the percentage of macroinvertebrates that prefer very low water quality was found in the latest survey. Macroinvertebrates found at SW4 with high water quality preferences were Amphipoda, Baetidae (>2 species) and Heptagenidae. As Heptagenidae prefer cobbles as their preferred habitat (Barber-James and Lugo-Oritz, 2003; De Moor and Scott, 2003; Thirion, 2007) the lack of stones biotope limits their presence and resulted in lower SASS5 and ASPT scores.

Figure 4.11: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW4 from March 2010 to December 2014.

IHAS scores at S2 deteriorated from November 2011 to November 2013 (Figure 4.12(A)). The decrease in habitat conditions resulted in a decrease in SASS5 scores in August 2012 and December 2012. However, even with the decrease in IHAS scores, SASS5 scores increased from December 2012 to December 2014 indicating improved water quality. Biotope availability was maintained throughout the study period and the decrease in IHAS scores were from a change in physical stream conditions i.e. erosion or disturbances to the river. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.12(B)), except for the August 2012 survey indicating a possible incident affecting the physical stream conditions after the September 2011 survey at S2.

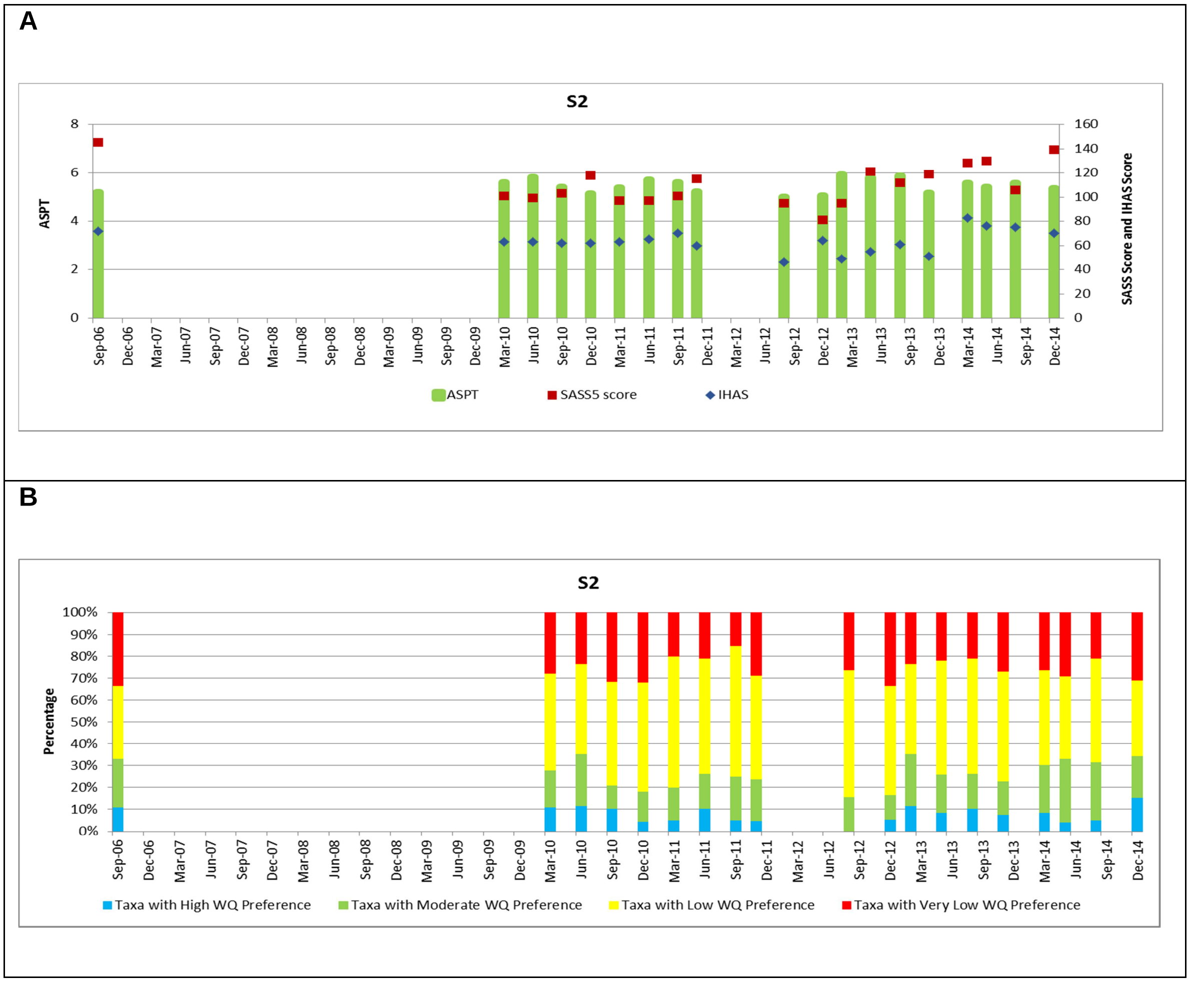


Figure 4.12: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site S2 from September 2006 to December 2014.

IHAS scores at S3 deteriorated from November 2011 to August 2012 (Figure 4.13(A)). The decrease in habitat conditions resulted in a decrease in SASS5 scores in August 2012 and December 2012. However even with the decrease in IHAS scores, SASS5 scores increased from February 2013 to December 2013 indicating improved water quality. Biotope availability was maintained throughout the study period and the decrease in IHAS scores were from a change in physical stream conditions i.e. erosion or disturbances to the river. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.13(B)), except for the November 2011 survey. Sampling site S3 is downstream of S2, indicating that a possible incident occurred after the September 2011 survey to affect the physical stream conditions at S2 and that it had an effect at sample site S3

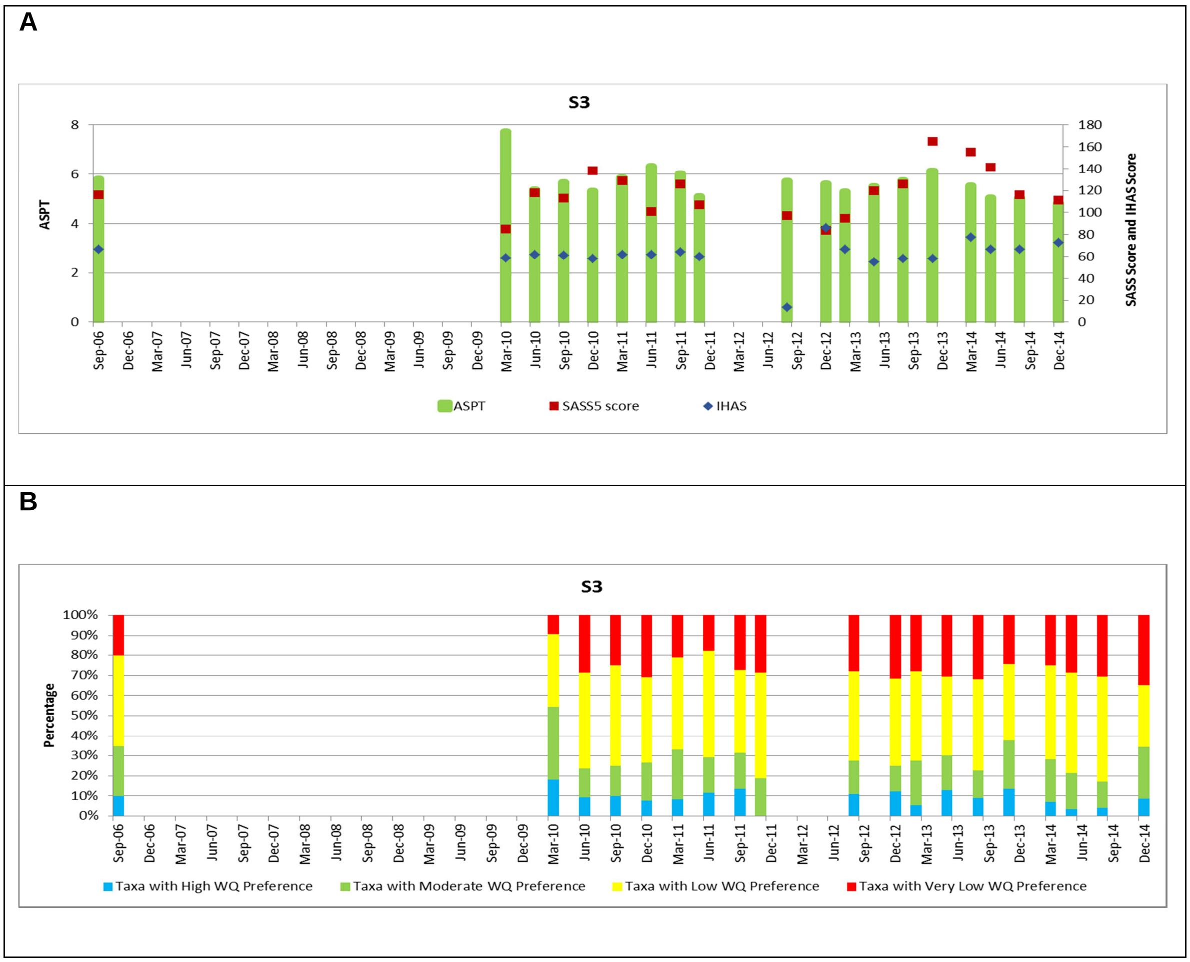
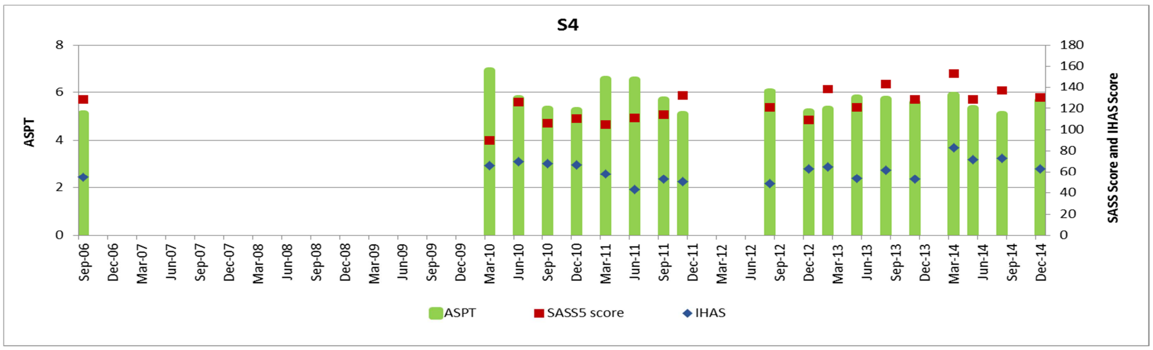


Figure 4.13: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site S3 from September 2006 to December 2014.

Habitat conditions at S4 were found to be between good to fair except during June 2011 to November 2013 (Figure 4.14(A)). Habitat conditions were found to decrease both in terms of biotope availability and physical stream conditions. However even with the decrease in IHAS scores for this period, the SASS5 scores were maintained indicating good water quality. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.14(B)), except for the November 2011 survey, indicating a possible incident affecting the water quality at S4 before the November 2011 survey. Sampling site S4 is located on the Klipspruit River.

A



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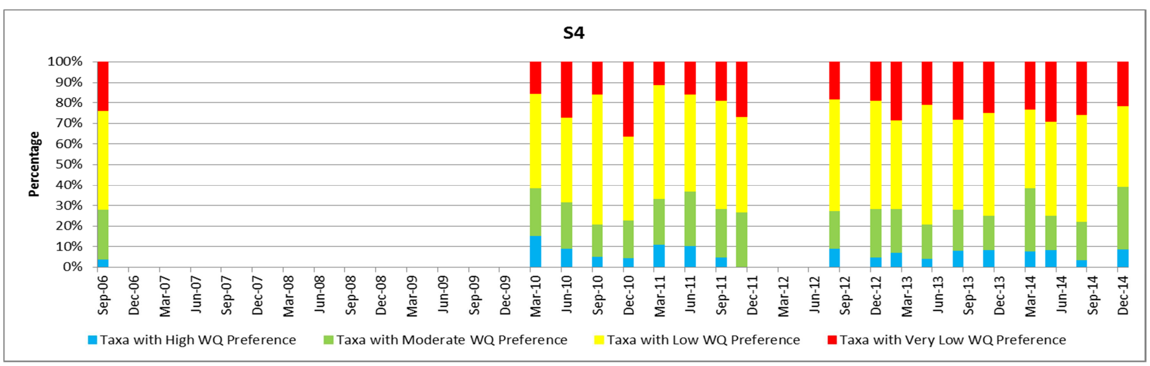

Figure 4.14: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site S4 from September 2006 to December 2014.

The SASS5, ASPT and IHAS scores at SW16 fluctuated throughout the study period (Figure 4.15(A)). Habitat conditions at SW16 was found to be between good to fair throughout the study period except for March 2011 and August 2012 surveys when the lowest IHAS score of 41 and 49 was attained respectively. Stones and vegetation biotopes which provides habitat for most macroinvertebrates were limited at this site for these two surveys, and the presence of mostly reduced flow conditions may be contributing to the low SASS5 scores. The highest SASS5 scores of 155 and 139 were found at the December 2010 and March 2010 surveys respectively. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.15(B)), however, were lacking in the December 2010, August 2012, December 2012, November 2013, January 2014 and November 2014 surveys. Macroinvertebrates found at SW16 with high water quality preferences were Baetidae (>2 species), Heptagenidae and Hydropsychidae (>2 species). As Heptagenidae and Hydropsychidae prefer cobbles as their preferred habitat (Barber-James and Lugo-Oritz, 2003; De Moor and Scott, 2003; Thirion, 2007) the lack of stones biotope limits their presence and resulted in lower SASS5 and ASPT scores March 2011 and August 2012 surveys.

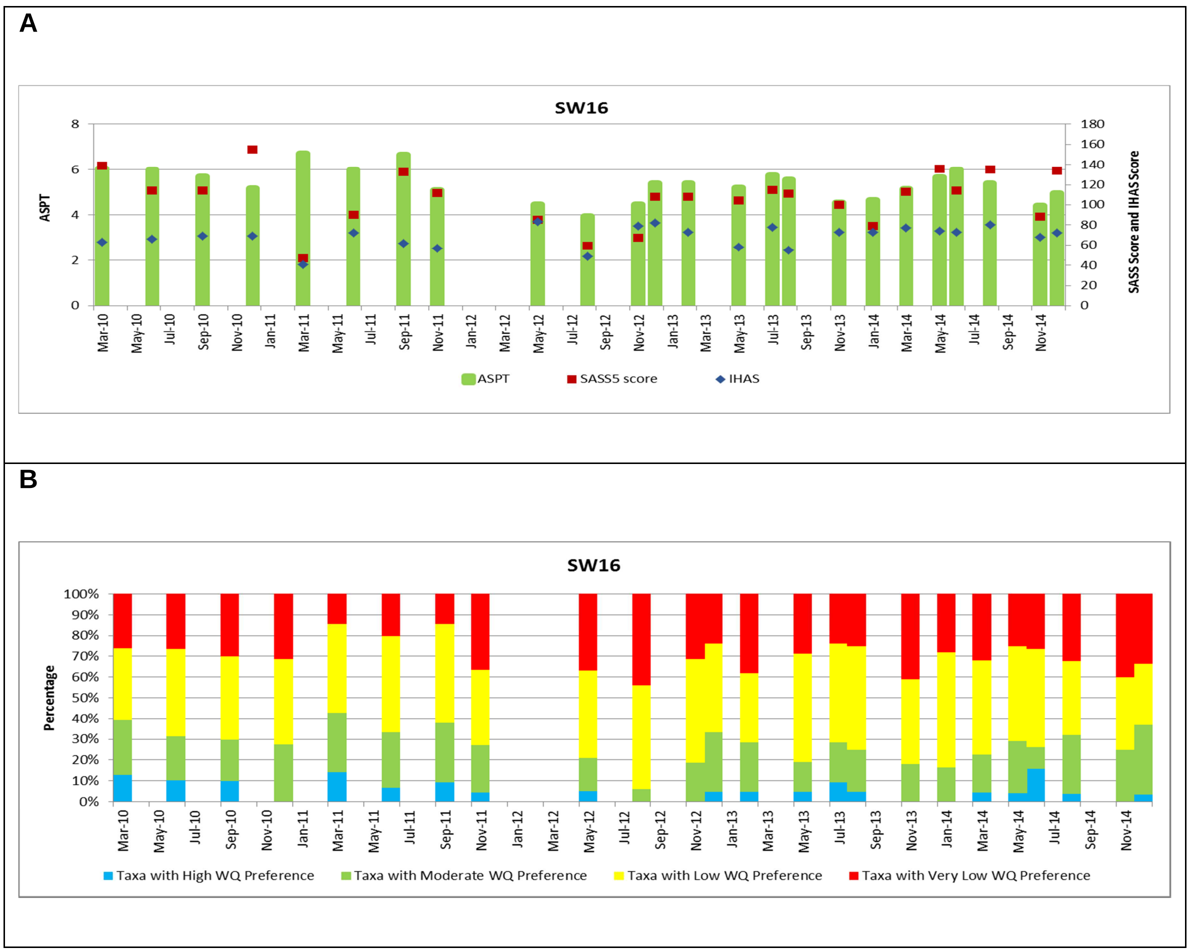

Figure 4.15: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW16 from March 2010 to December 2014.

The SASS5, ASPT and IHAS scores at SW17 fluctuated throughout the study period (Figure 4.16(A)). Habitat conditions at SW17 were found to be mostly poor from March 2010 to November 2013, only showing a slight improvement in habitat conditions from

March 2014. The steep, eroded and incised banks with a deeply eroded channel and lack of flow conditions resulted in overall low SASS5 and ASPT scores in both wet and dry conditions. The highest SASS5 scores of 158 and 137 were found at the December 2014 and August 2014 surveys respectively. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.16(B)). Macroinvertebrates found at SW17 with high water quality preferences were Baetidae (>2 species), Heptagenidae and Hydropsychidae (>2 species). The poor habitat conditions may have limited their presence and resulted in lower SASS5 and ASPT scores in the March 2011, November 2012, February 2013, August 2013 and November 2013 surveys.

Figure 4.16: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW17 from March 2010 to December 2014.

The SASS5, ASPT and IHAS scores at SW2 fluctuated throughout the study period (Figure 4.17(A)). Habitat conditions at SW2 were found to be mostly fair to good, only showing poor habitat conditions in the March 2010, September 2010, March 2011 and August 2012 surveys. The highest SASS5 scores of 158 and 146 were found at the December 2014 and May 2013 surveys respectively. Macroinvertebrates that have high water quality preferences were found throughout the sampling period (Figure 4.17(B)), however, were lacking in the March 2010, March 2011 and August 2012 surveys. Macroinvertebrates found at SW2 with high water quality preferences were Baetidae (>2 species), Heptagenidae, Perlidae and Hydropsychidae (>2 species). The poor habitat conditions may have limited their presence and resulted in lower SASS5 and ASPT scores.

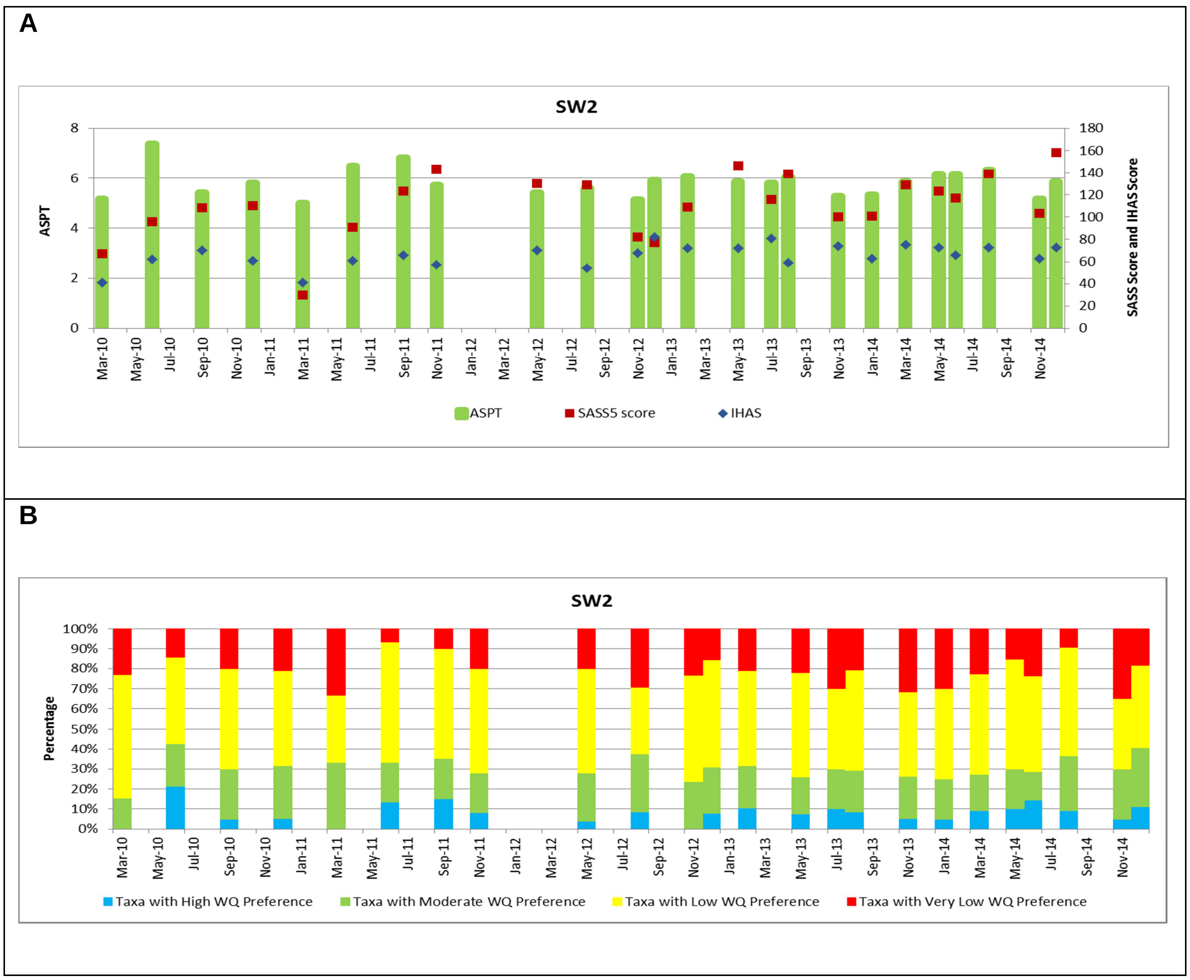

Figure 4.17: SASS, ASPT, IHAS scores (A) and percentage taxa per survey showing specific water quality requirements (Thirion, 2007) (B) for sampling site SW2 from March 2010 to December 2014.

The decrease in biotope availability by the lack of stones biotope at sample sites SW5, SW16 and decrease in stones and vegetation habitat at sample sites SW7, SW1, SW4, SW17 and SW2 has resulted in low SASS5 scores at these sites. Most macroinvertebrates with high requirements for water quality have a specific habitat preference to cobbles (Palmer, 2000; Yabe and Nakatsugawa, 2004; Thirion, 2007). The presence of macroinvertebrates with high requirements for water quality preferences are thus limited when the stones biotope in unavailable and resulted in lower SASS5 and ASPT scores. Sample site SW9 showed a decrease in habitat conditions and water quality from June 2011 to November 2013. The decrease in habitat conditions show an increase in macroinvertebrates that prefer moderate to very low water quality conditions, indicating a deterioration in water quality at SW9 as well. Macroinvertebrates that have high water quality preferences were not found at sampling site SW9 until the August 2014 survey once habitat conditions improved.

Sample sites S2 and S3 on the Wilge River and S4 on the Klipspruit River all showed a decrease in habitat conditions after the September 2011 survey. However, even with the decrease in IHAS scores, SASS5 scores were maintained. Biotope availability was maintained throughout the study period and the decrease in IHAS scores were from a change in physical stream conditions i.e. erosion or disturbances to the river. Macroinvertebrates that have high water quality preferences were found throughout the sampling period, indicating no change in water quality. SASS and ASPT scores were modelled on reference conditions for the Highveld Ecoregion, obtained from Dallas (2007) and are displayed on Table 4.7. In cases where the SASS5 and ASPT scores fell into different categories, the SASS category was given preference. IHAS scores were determined as per MacMillan (1998) are displayed in Table 4.8.

All the sampling sites found on the Wilge River, except SW17 were found to be in the A or B category indicating unimpaired to slightly impaired biological integrity, with high diversity of taxa with sensitive taxa present. Sampling site SW5 was found to be in a critically impaired condition in March 2010, December 2010, June 2011, December 2012, February 2013 and August 2013. SW7 was found to be in a critically impaired condition in March 2011, June 2011, December 2012 and February 2013. Sampling sites SW1 and SW17 was found to be critically impaired in March 2011. Biotic integrity was found to mostly decrease during the low flow survey when compared to the high flow conditions. This is likely due to the seasonality and reduced flow.

Table 4.7: Class: Modelled reference conditions for the Highveld Ecoregion (11) based on South African Scoring System 5 (SASS5) and Average Score Per Taxon (ASPT) values on Table 4.4 (Dallas, 2007).

	Sep-06	Mar-10	Jun-10	Sep-10	Dec-10	Mar-11	Jun-11	Sep-11	Nov-11	May-12	Aug-12	Nov-12	Dec-12	Feb-13	May-13	Jul-13	Aug-13	Nov-13	Jan-14	Mar-14	May-14	Jun-14	Aug-14	Nov-14	Dec-14
SW5	E	F	D	E	F	E	F	D	E		E		F	F	E		F	E		C	C		E	E	D
SW7		E	D	D	C	F	F	C	E		D		F	F	C		C	E		E	C		E		D
SW1	E	E	E	E	E	F	D	C	B		B		C	B	B		C	D		B	B		B		B
SW9		C	B	B	B	C	E	B	D				C	C	B		B	E		B	B		B		B
SW4		C	B	C	C	B	C	C	B		E		C	B	B		C	D		D	D		E		E
S2	A	B	B	B	B	B	B	B	B		B		C	B	B		B	B		A	A		B		A
S3	B	B	B	B	A	A	B	A	B		B		B	B	B		A	A		A	A		B		B
S4	A	B	A	B	B	B	B	B	A		B		B	A	B		A	A		A	A		A		A
SW16		A	B	B	A	E	B	A	B	B	D	C	B	B	B	B	B	B	C	B	A	B	A	B	A
SW17				B	B	F	D	C	B		B		E	C	C		E	D		A	B		A		A
SW2		C	B	B	B	E	B	B	A	A	A	C	C	B	A	B	A	B	B	A	B	B	A	B	A

Table 4.8: Integrated Habitat Assessment System (IHAS) Scores Habitat modelled on IHAS Scoring guidelines (Version 2) on Table 4.5 (MacMillan, 1998).

	Sep-06	Mar-10	Jun-10	Sep-10	Dec-10	Mar-11	Jun-11	Sep-11	Nov-11	May-12	Aug-12	Nov-12	Dec-12	Feb-13	May-13	Jul-13	Aug-13	Nov-13	Jan-14	Mar-14	May-14	Jun-14	Aug-14	Nov-14	Dec-14
SW5	52	45	56	37	42	41	40	46	41		25		42	29	32		40	29		51	50		39		38
SW7		34	45	34	39	35	35	39	33		39		53	40	40		25	50		56	56		58		65
SW1	42	34	44	31	36	32	47	48	51		38		65	60	60		63	56		73	71		69		75
SW9		65	60	60	40	45	47	39	35				47	38	32		34	41		59	60		58		45
SW4		46	67	60	50	45	62	45	50		27		64	62	35		40	60		42	32		41		44
S2	72	63	63	62	62	63	65	70	60		46		64	49	55		61	51		83	76		75		70
S3	67	59	62	61	58	62	62	64	60		14		86	67	55		58	58		78	67		67		73
S4	55	66	70	68	67	58	43	53	51		49		63	65	54		62	53		83	72		73		63
SW16		63	66	69	69	41	72	62	57	83	49	79	82	73	58	78	55	73	73	77	74	73	80	68	72
SW17				45	56	46	40	44	49		30		51	45	21		36	38		75	74		75		77
SW2		41	62	70	61	41	61	66	57	70	54	68	82	72	72	81	59	74	63	75	73	66	73	63	73

Sediment is an important component of an aquatic ecosystem in that it provides habitat, feeding and spawning areas for aquatic fauna such as fish and benthic macroinvertebrates. Changes in soil erosion and sedimentation deposition are noted in Table 4.9 and 4.10 respectively.

Table 4.9: Visual assessment of change in soil erosion at monitoring sites as adopted by Golder Associates Africa Project (2012).

	Dec-12	Feb-13	Mar-14	May-14	Aug-14	Nov-14
SW5	1	2	3	3	3	2
SW7	4	4	4	4	4	4
SW1	3	3	3	3	3	3
SW9	2	2	2	2	2	2
SW4	1	3	4	4	4	4
S2	0	0	2	2	2	2
S3	2	0	2	2	2	2
S4	0	3	2	2	3	2
SW16	0	0	2	2	2	2
SW17	0	0	2	2	2	2
SW2	2	2	2	2	3	3

Table 4.10: Visual assessment of change in sedimentation deposition at monitoring sites as adopted by Golder Associates Africa Project (2012).

	Dec-12	Feb-13	Mar-14	May-14	Aug-14	Nov-14
SW5	3	3	2	2	2	4
SW7	0	0	1	1	1	2
SW1	0	0	2	2	3	3
SW9	0	0	2	2	2	4
SW4	1	1	2	2	4	4
S2	0	0	2	2	2	2
S3	0	0	2	2	2	2
S4	0	1	2	2	2	4
SW16	0	0	2	2	2	2
SW17	0	0	2	2	2	2
SW2	0	0	2	2	2	2

Soil erosion scores were found to either remain the same or increase at the sampling sites from December 2012 to November 2014. Sampling sites S2, SW16 and SW17 on the Wilge River moved from a non-eroded state to a moderately eroded state. Sampling site S3 on the Wilge River remained in a moderately eroded state. Sampling site SW2 on the Wilge River moved from a moderately eroded state in December 2012 to a largely active eroded state in November 2014. Sampling site S4 on the Klipspruit River moved from a non-eroded state to a moderately eroded state. Sampling site SW7 on the unnamed tributary of the Klipfonteinspruit remained in a severely eroded state from

December 2012 to November 2014. SW7 is situated downstream of a newly constructed road. Construction activities and lack of bank rehabilitation may result in ongoing deposition of sediment in the river channel at this point. Sampling site SW1 on the Klipfonteinspruit remained in a largely actively eroded state from December 2012 to November 2014 while sampling site SW5 on the Klipfonteinspruit was found to be steadily eroding from December 2012 to August 2014 with improvement noted in November 2014. SW5, a downstream point on the Klipfonteinspruit, is impacted upon by associated infrastructure i.e. bridge construction which results in ongoing deposition of sediment in the river channel at this point. Sampling site SW9 on the unnamed tributary of the Wilge River remained in a moderately eroded state, while sampling site SW4 on the unnamed tributary of the Wilge River changed from a slightly eroded state to a severely eroded state from December 2012 to November 2014.

There was no soil deposition in December 2012 at any of the sampling sites except SW5 and SW4. All sample sites indicated an increase in soil deposition with sampling sites, SW5, SW4, SW9 and S4 showing severe soil deposition. Sampling site SW1 showed largely active soil deposition occurring with large plants extending through the sediment layer in November 2014. The rest of the sampling sites all had moderate soil deposition by November 2014.

Excess sediments in itself can cause deterioration of water quality and impact on river ecology but it can also be a means of attachment of pollutants such as nutrients or metals by adsorption and transport of pollutants. The poor land management practices and changes in land-use have caused erosion, siltation of rivers, unstable river banks and beds and with this mobilisation of sediment associated with pollutants. The sediments often act as a ‘sink’ for pollutants, which can be released back into the water column during disturbance, whether natural or anthropogenic (CSIR, 2012). Better land management practices are suggested at sample sites SW5. SW4, SW7, SW1, SW9 and S4 that indicate soil erosion and sedimentation deposition occurring.

Constrained RDA triplot diagrams (Figures 4.18 – 4.23) illustrate measured water quality variables, sampling sites and the presence or absence of macroinvertebrate families for year 2006, 2010, 2011, 2012, 2013 and 2014. The surveys were separated into high flow and low flow periods. Surveys undertaken in March 2010, December 2010, March 2011, November 2012, December 2012, February 2013, November 2013, January 2014, March 2014, November 2014 and December 2014 were included in the high flow category. Surveys undertaken in September 2006, June 2009, June 2010, September 2010, June 2011, September 2011, May 2012, August 2012, May 2013, July 2013, August 2013, May 2014, June 2014 and August 2014 were included in the low flow category.

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the September 2006 survey (Figure 4.18), explains a total of 73.60% of the observed variation with the first axis explaining 55.19% and the second axis 18.41% of the variation. Species selection of 80-100% was selected to show the 11 best fitting families. The triplot indicated that environmental factors EC, DO, Temp, pH and explanatory factors macroinvertebrate families Ancylidae, Ceratopognidae, Hydropsychidae, Leptophlebiidae, Lestidae, Pleidae, Potamonautidae, Simuliidae, Turbellaria, Oligochaeta and Coenagrionidae were determining factors for the distribution of the sample sites under consideration. Although IHAS score data was included in the RDA dataset, it was not selected as a determining factor in the distribution of the data. A co-linear relationship with pH and macroinvertebrate families Ancylidae, Ceratopognidae, Hydropsychidae, Leptophlebiidae, Lestidae, Pleidae, Potamonautidae, Simuliidae and Turbellaria was indicated on axis 1 and an association was found with sample site S4 in low flow periods. Coenagrionidae indicated a significant negative loading to these variables. EC, DO and Temp indicated a co-linear relationship with each other and an association to sample site S2. Macroinvertebrate family Oligochaeta, explained the variation in the distributive trends on axis 2 with an association to SW5.

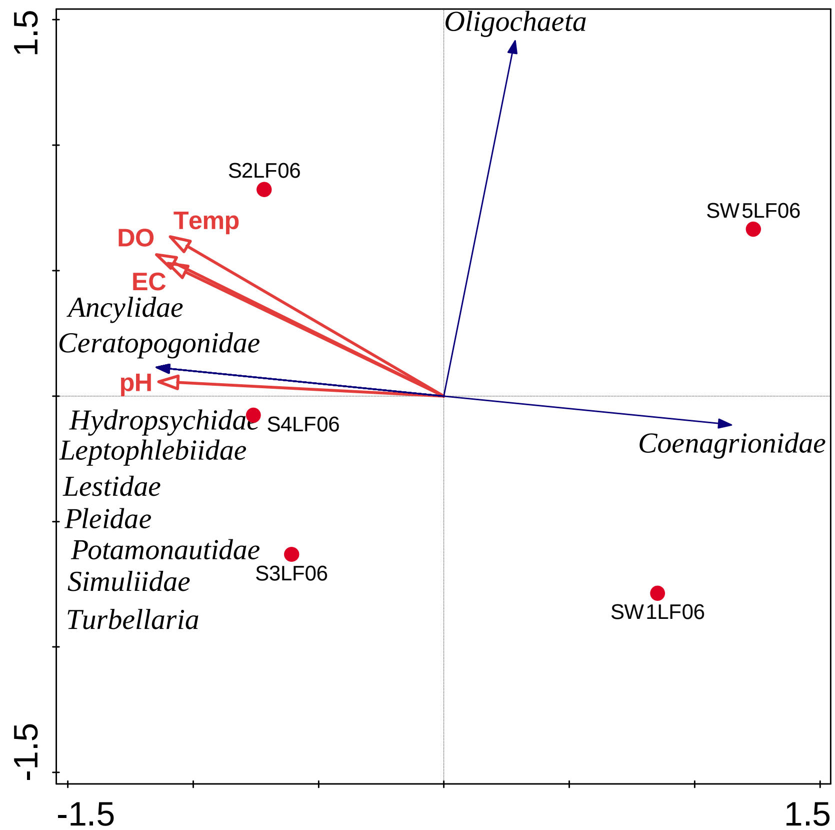

Figure 4.18: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at five sampling sites (SW5, SW1, S2, S3 and S4), during the September 2006 survey showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Species selection of 80-100% was selected to show the 11 best fitting families. Explained variation on Axis 1 is 55.19% and on Axis 2 is 18.41% (Total variation explained is 73.60%).

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the 2010 surveys (Figure 4.19), explains a total of 68.23% of the observed variation with the first axis explaining 44.49% and the second axis 23.74% of the variation. Species selection of 30-100% was selected to show the 5 best fitting families. The distribution was explained between the IHAS score and macroinvertebrate families Leptophlebiidae, Turbellaria, Heptageniidae and Elmidae on axis 1. Sample sites associated with this relationship are mostly sample sites found on the Wilge River i.e. S2, S3, SW16, SW17, SW2 and the Klipspruit River, S4. Sample sites found not to be associated with these parameters are SW5, SW1, SW9, SW4 and SW7.

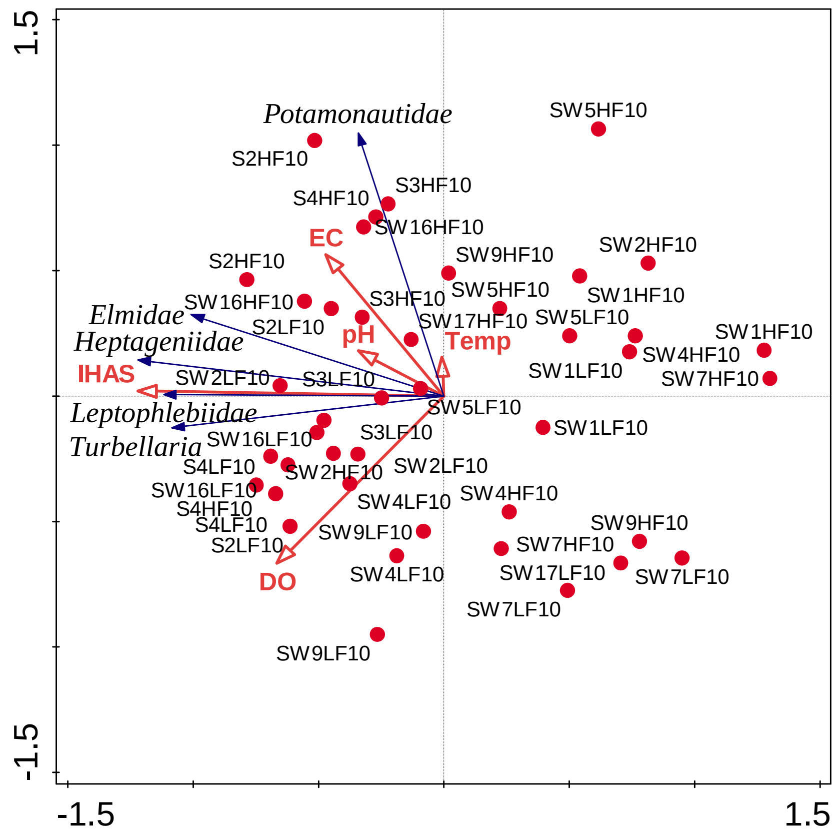

Figure 4.19: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at eleven sampling sites (SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2), for surveys March 2010-December 2010 showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp), Integrated Habitat Assessment Score (IHAS) and; taxa (blue arrows). Species selection of 30-100% was selected to show the 5 best fitting families. Explained fitted variation on Axis 1 is 44.49% and on Axis 2 is 23.74% (Total variation explained is 68.23%).

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the 2011 surveys (Figure 4.20), explains a total of 77.02% of the observed variation with the first axis explaining 60.03% and the second axis 16.99% of the variation. Species selection of 30-100% was selected to show the 7 best fitting families. The distribution was explained between IHAS score and macroinvertebrate families Elmidae, Leptophlebiidae and Tricorythidae on axis 1. Sample sites S2, S3, SW2 and SW16 were found to be associated to these parameters during low flow periods. A co-linear relationship was found between EC and macroinvertebrate families Ancylidae, Ceratopognidae and Corbiculidae. Sample sites S2, S3 and SW16 were found to be associated to these parameters during high flow periods. The distribution was explained by temperature on axis 2. Sample sites associated with temperature are SW1 and S4. Sample sites associated with the explanatory factors (environmental and macroinvertebrate families) are mostly sample sites found on the Wilge River i.e. S2, S3, SW16, SW2 and the Klipspruit River, S4. Sample sites found not to be associated with these parameters are SW5, SW1, SW9, SW4 SW17 and SW17. Sample site Sw17 is located on the Wilge River downstream of the confluence with the Klipfonteinspruit River which is impacted on by SW5, SW7 and SW1.

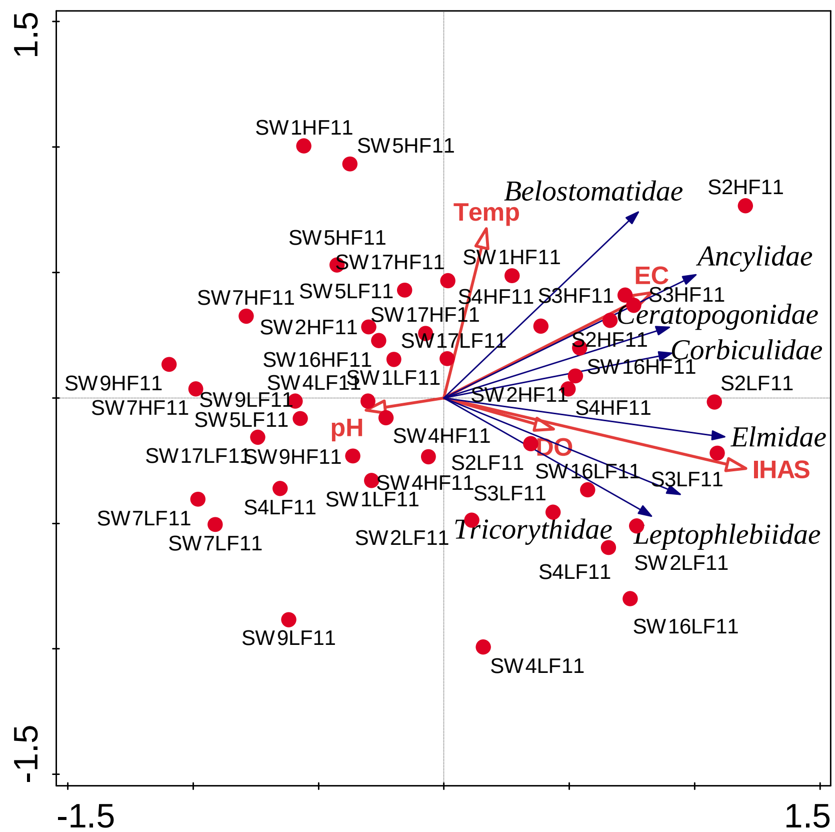

Figure 4.20: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at eleven sampling sites (SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2), for surveys March 2011-November 2011 showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp), Integrated Habitat Assessment Score (IHAS) and; taxa (blue arrows). Species selection of 30-100% was selected to show the 7 best fitting families. Explained fitted variation on Axis 1 is 60.03% and on Axis 2 is 16.99% (Total variation explained is 77.02%).

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the 2012 surveys (Figure 4.21), explains a total of 65.76% of the observed variation with the first axis explaining 35.69% and the second axis 30.07% of the variation. Species selection of 20-100% was selected to describe the 12 best fitting families. The distribution of data was explained by macroinvertebrate family Coenagrionidae on the first axis with an association to sample point SW16 during high flow periods. An association between IHAS, and macroinvertebrate families Simuliidae and Veliidae was found on the first axis. A negative loading to this relationship was noted at sample sites SW7, SW4 and SW5 indicating poor habitat conditions at these sites.

On the second axis a co-linear relationship was found to be between EC and macroinvertebrate families Turbellaria, Hydracarina, Hydropsychidae, Elmidae and Corbiculidae. Sampling sites associated with these parameters include SW2, S2 and S3. There was a negative loading to these parameters with macroinvertebrate family Dixidae. The sample sites associated is SW7, SW5 and SW4. Dixidae was only found to be present at SW4 in the August 2012 survey and at SW9 in the December 2012 survey. Dixidae has a preference to moderate water quality (Thirion, 2007). Notonectidae was found to be associated with sample site SW9. Notonectidae was found at sample sites SW1 and SW4 in the August 2012 survey and at sample site SW9, SW4, S4, SW17 and SW2 in the December 2012 surveys. Notonectidae has a preference to low water quality (Thirion, 2007). Sample sites associated with the explanatory factors (environmental only) are mostly sample sites found on the Wilge River i.e. S2, S3, SW16, SW2 and the Klipspruit River, S4. Sample sites found not to be associated with these parameters are SW5, SW1, SW9, SW4, SW7 and SW17.

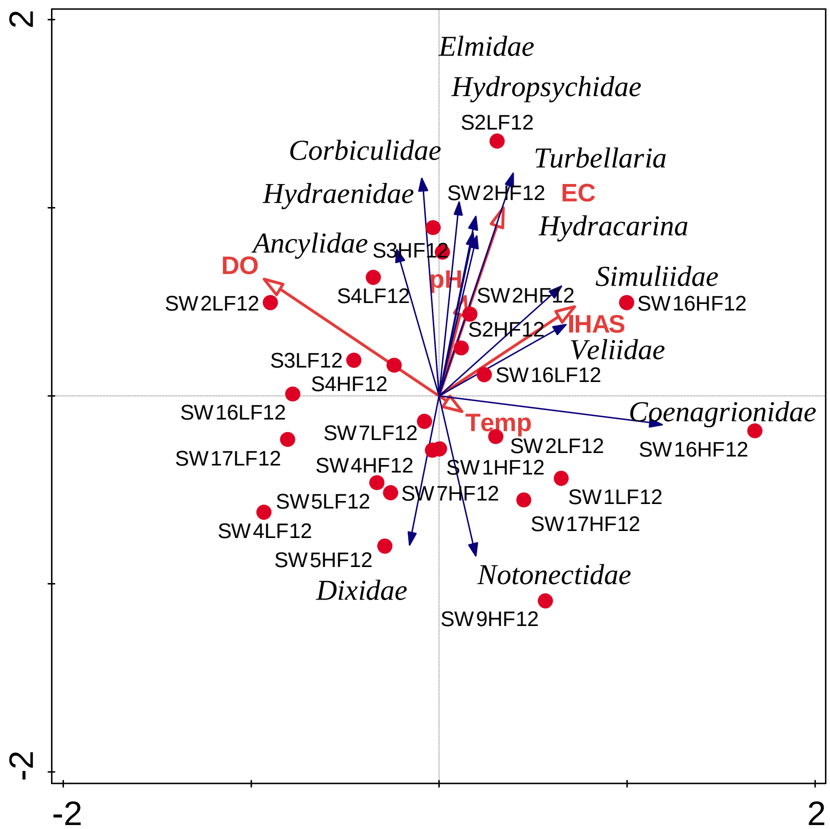

Figure 4.21: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at eleven sampling sites (SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2), for surveys May 2012-December 2012 showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp), Integrated Habitat Assessment Score (IHAS) and; taxa (blue arrows). Species selection of 20-100% was selected to show the 12 best fitting families. Explained fitted variation on Axis 1 is 35.69% and on Axis 2 is 30.07% (Total variation explained is 65.76%).

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the 2013 surveys (Figure 4.22), explains a total of 72.65% of the observed variation with the first axis explaining 55.17% and the second axis 17.48% of the variation. Species selection of 30-100% was selected to describe the 6 best fitting families. The distribution was explained by IHAS score and macronvertebrate families Hydropyschidae, Leptophlebiidae and Elmidae on axis 1. An association was found with sample sites S2, SW2, S3, S4 and SW16 during high flow periods. Sample sites that indicateded a negative loading to IHAS score are SW4, SW9 and SW7. A strong co-linear relationship was found between temperature and macroinvertebrate family

Potamonautidae on axis 2. Sampling sites associated with these parameters SW16, SW2, SW7, SW17 and S3 during high flow periods. Sample sites associated with the explanatory factors (environmental and macroinvertebrate families) are mostly sample sites found on the Wilge River i.e. S2, S3, SW16, SW2 and the Klipspruit River, S4. Sample sites found not to be associated with these parameters are SW5, SW1, SW9, SW4, SW7 and SW17.

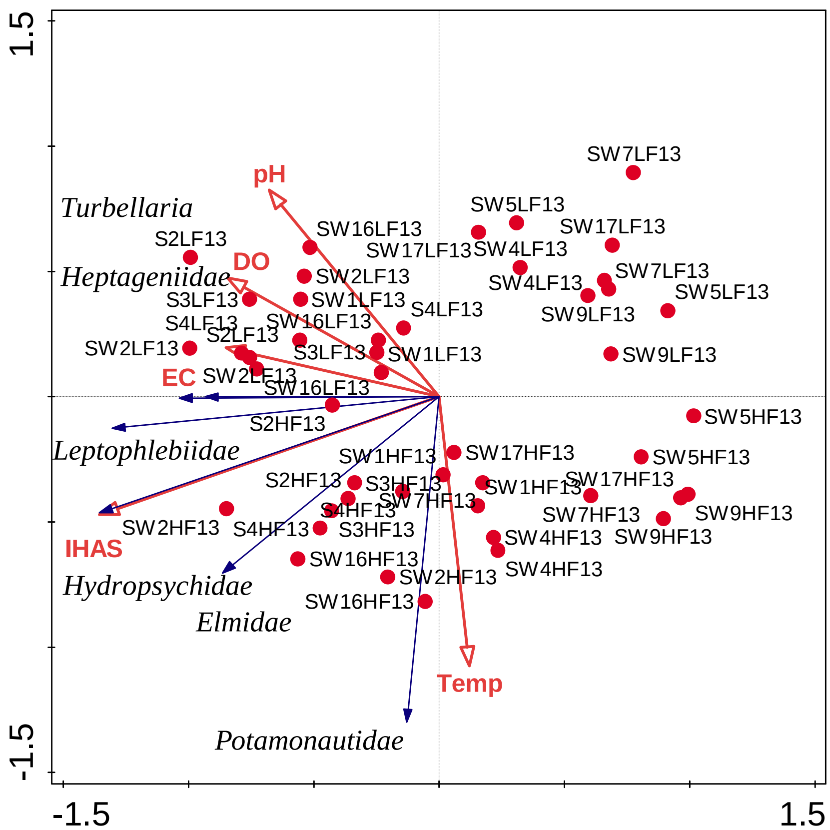

Figure 4.22: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at eleven sampling sites (SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2), for surveys February 2013–November 2013 showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) Integrated Habitat Assessment Score (IHAS) and; taxa (blue arrows). Species selection of 30-100% was selected to show the 6 best fitting families. Explained fitted variation on Axis 1 is 55.17% and on Axis 2 is 17.48% (Total variation explained is 72.65%).

Explanatory factors (environmental and macroinvertebrate families) in the RDA diagram illustrating the 2014 surveys (Figure 4.23), explains a total of 76.31% of the observed variation with the first axis explaining 58.61% and the second axis 17.70% of the variation. Species selection of 30-100% was selected to describe the 6 best fitting families. Sample sites were limited to the 35 best fitting sites. The distribution was explained between the IHAS score, and macroinvertebrate family Leptophlebidae, Corbiculidae, Hydropsychidae, Heptageniidae and Simulidae on axis 1. Sample sites associated with these parameters are SW2, S4, SW17 and SW16. Sample sites that indicaed a strong negative loading to these parameters are SW5, SW4, SW7 and SW9. EC indicated a strong co-linear relationship with macroinvertebrate family Elmidae. Sample sites associated with these parameters include SW17, SW1, S3 and S2. Sample sites that displayed a negative loading to these parameters are SW5, SW4, SW7 and SW9. Sample sites associated with the explanatory factors (environmental and macroinvertebrate families) are mostly sample sites found on the Wilge River i.e. S2, S3, SW16, SW17, SW2 and the Klipspruit River, S4. Sample sites found not to be associated with these parameters are SW5, SW1, SW9, SW4 and SW7. This implies an improvement at sample site SW17 in the 2014 surveys.

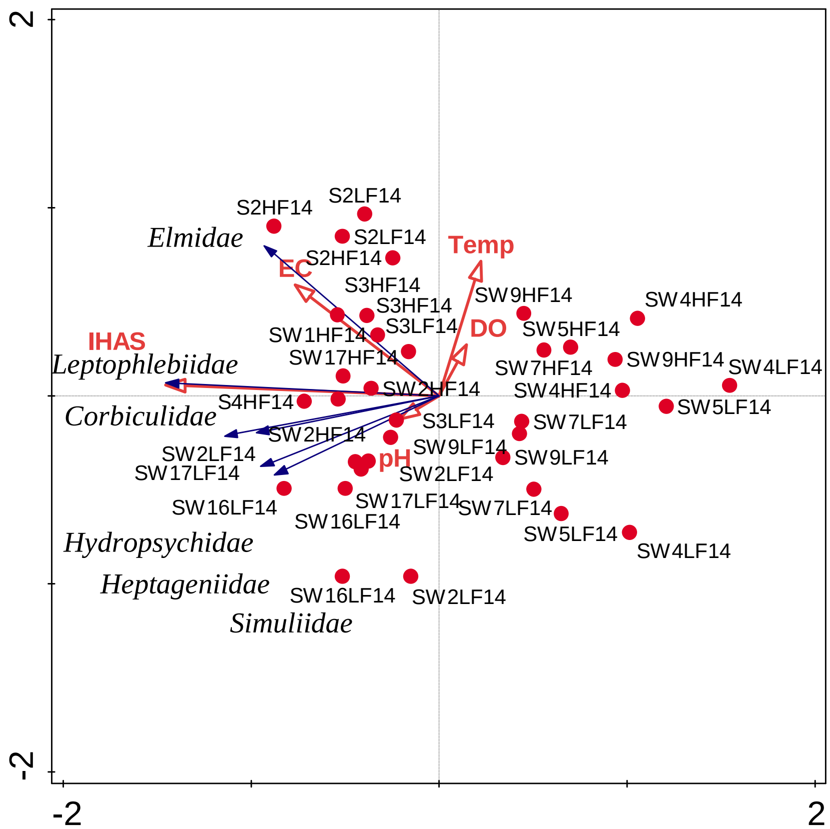

Figure 4.23: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area at eleven sampling sites (SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2), for surveys March 2014–December 2014 showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp), Integrated Habitat Assessment Score (IHAS) and; taxa (blue arrows). Species selection of 30-100% was selected to show the 6 best fitting families. Sample sites were limited to the 35 best fitting sites. Explained fitted variation on Axis 1 is 58.61% and on Axis 2 is 17.70% (Total variation explained 76.31%).

In 2006 environmental factors EC, DO, Temp and pH were driving factors for the distribution of the sample sites under consideration (S2, S3 and S4 only). Although IHAS score data was included in the RDA dataset, it was not selected as a determining factor in the distribution of the data. Sample sites associated with these parameters are S4 located on the Klipspruit River; and S2 and S3 located on the Wilge River. Sample sites located on the Klipfonteinspruit (SW5 and SW1), show no association with EC, DO, Temp and pH.

For 2010 to 2014 EC and IHAS are the strong driving factors for the distribution of the sample sites. There is a strong association with these parameters and the sample sites located on the Wilge River (S2, S3, SW16, SW17 and SW2) and Klipspruit River (S4). Sample sites located on the Klipfonteinspruit (SW5 and SW1), the unnamed tributary of the Klipfonteinspruit (SW7) and the unnamed tributary of the Wilge River (SW9 and SW4) show no association with EC and IHAS score.

Families that were found to be present most often during all the surveys and EIA include Chironomidae, Caenidae, Coenagrionidae, Dytiscidae, Gyrinidae, Leptophlebiidae, Corixidae, Simulidae and Baetidae 2spp. Most families found have a preference of low to very low water quality (Thirion, 2007), indicating that the water of the Wilge River sub-catchment B20F area is in a poor state of health. Families that were found to be present most often during the high flow periods include Baetidae, Chironomidae, Coenagrionidae, Hydropsychidae, Caenidae, Potamonautidae, and Dytiscidae. Coenagrionidae, Hydropsychidae, Potamonautidae prefer very fast to moderately fast velocities so it is expected (Thirion, 2007). Chironomidae, Caenidae, and Dytiscidae prefer slow to very slow velocities (Thirion, 2007) but were found in high flow conditions. Average velocity in the tributaries of the Wilge River sub-catchment B20F area during high flow conditions is 2.67 m/s (Maliba and Durgapersad, 2013). Families that were found to be present most often during the low flow periods include Baetidae, Chironomidae, Hydropsychidae, Caenidae, Dytiscidae, Coenagrionidae and Gyrinidae.

Chironomidae, Caenidae and Dytiscidae prefer slow to very slow velocities so it is expected (Thirion, 2007). Hydropsychidae, Coenagrionidae and Gyrinidae prefer very fast to moderately fast velocities (Thirion, 2007) but were found in low flow conditions. Average velocity in the tributaries of the Wilge River sub-catchment B20F area during high flow conditions is 1.16 m/s (Maliba and Durgapersad, 2013). Families that were completely absent during the low flow periods include Amphipoda, Heliodae, Palaemonidae and Platycnemidae. Amphipoda and Platycnemidae prefer very fast to moderately fast velocities so it is expected (Thirion, 2007). Heliodae and Palaemonidae were found rarely found in the 25 surveys occurring only once and twice respectively in the low flow conditions. Families that were completely absent during the high flow periods were Bulinae, Chlorocyphidae, Coelenterata, Haliplidae, Philopotamidae, Psychodidae and Syrphidae. Bulinae, Chlorocyphidae, Haliplidae, Psychodidae and Syrphidae prefer slow to very slow velocities so it is expected (Thirion, 2007).

Coelenterata and Philopotamidae were rarely found in the 25 surveys occurring only once and twice respectively in the high flow conditions.

Explanatory factors macroinvertebrate families Ancylidae, Ceratopognidae, Hydropsychidae, Leptophlebiidae, Lestidae, Pleidae, Potamonautidae, Simuliidae, Turbellaria, Oligochaeta and Coenagrionidae were found to be associated with sample sites S2, S3 and S4 in 2006. For 2010-2014 the macroinvertebrate family Elmidae was found to be associated with S2, S3, S4, SW16, SW17 and SW2. Leptophlebiidae was found to be associated with these sample sites in 2010, 2011, 2012, 2013 and 2014. Heptagenidae was found to the associated with these sample sites in 2010, 2013 and 2014. Elmidae, Leptophlebiidae and Heptagenidae have a preference for high to moderate water quality (Thirion, 2007).

4.3.2 Comprehensive chemistry and actual macroinvertebrate counts (2012-2014) for 6 surveys only

Comprehensive *Chemistry analyses (2012-2014)*

The comprehensive chemistry analyses show an exceedance of the Wilge River IWRMP RWQOs for ammonia at both sampling sites for all surveys (Table 4.11). Aluminium showed an exceedance of the Wilge River IWRMP RWQOs for all surveys except for SW2 in the May 2012 survey. Sample site SW2 just met the limit of 0.02 mg/L for the Wilge River IWRMP RWQOs. The DWA TWQO for mercury is 0.04 µg/L. The mercury analyser on which the mercury analyses were carried out had a detection limit of 1 µg/L, and therefore 0.5 µg/L was used as the average value in the graphs. The 0.5 µg/L used exceeds the DWA TWQO for mercury, however the mercury concentrations may be lower at the sampling sites. A mercury analyser with a lower detection limit is required. Calcium concentrations were found to exceed the Wilge River IWRMP RQWO at SW16 in the November 2012, June 2014 and November 2014 surveys. Conductivity, sodium and sulphate were found to exceed the Wilge River IWRMP RWQO at SW16 and SW2 in the July 2013 survey, indicating an incident prior to sampling. Conductivity, magnesium, manganese and sodium were found to exceed the Wilge River IWRMP RWQOs at SW16 in the November 2014 survey. Lead was found to exceed the DWA TWQO at SW16 and SW2 in the April 2012 survey. Ortho-phosphate was found to exceed the Wilge River IWRMP RWQOs at SW2 in the January and November 2014 surveys. Zinc was found to exceed the DWA TWQO at SW16 and SW2 in the January 2014 survey.

Table 4.11: Water quality variables measured at the time of sampling at SW16 and SW2 biomonitoring sites (from April 2012 to November 2014).

Sample Name and survey	SW16	SW2	SW16	SW2	SW16	SW2	SW16	SW2	SW16	SW2	SW16	SW2
	May-12		Nov-12		Jul-13		Jan-14		Jun-14		Nov-14	
Aluminium as Al (mg/L)	0.19	0.02	0.27	0.38	0.12	0.1	0.5	6.2	0.07	0.14	0.06	0.66
Ammonia as N (mg/L)	0.0025	0.1	0.0025	0.1	0.0025	0.0025	0.1	0.1	0.1	0.0025	0.1	0.2
Arsenic as As(ug/L)	0.51	0.78	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Calcium as Ca (mg/L)	21	23	26	24	25	25	18	16	27	22	28	22
Cadmium as Cd (ug/L)	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25
Chloride as Cl (mg/L)	11.47	11.06	10.68	14.57	12.62	11.69	8.4	4.48	10.78	8.76	19.88	14.96
Cyanide as CN (ug/L)	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5
Conductivity (mS/m)	31	28.9	36.6	36	46	42.3	20.4	13	39	30.9	41	31.3
Copper as Cu (mg/L)	2.5	2.5	20	20	20	10	10	20	10	10	10	10
Iron as Fe (mg/L)	0.5	0.36	0.63	0.87	0.16	0.16	0.74	4.6	0.42	0.57	0.36	1.3
Flouride as F (mg/L)	0.25	0.28	0.32	0.3	0.18	0.17	0.24	0.33	0.17	0.15	0.32	0.32
Mercury as Hg (ug/L)	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Potassium as K (mg/L)	2.3	2.3	3.6	4	2.2	2.3	2.7	3.7	2.6	2.2	3.6	3.5
Magnesium as Mg (mg/L)	16	17	22	20	19	18	14	9.2	19	14	22	17
Manganese as Mn (mg/L)	0.06	0.1	0.16	0.15	0.08	0.05	0.26	0.18	0.04	0.0025	0.29	0.15
Sodium as Na (mg/L)	15	18	16	19	28	24	13	10	20	16	22	16
Nickel as Ni (ug/L)	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	20	20
Nitrate as N (mg/L)	0.01	0.01	0.26	0.23	0.32	0.17	0.69	0.72	0.52	0.51	0.28	0.41
Lead as Pb(ug/L)	2	2	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
pH @ 25 °C	7.58	7.76	7.16	7.13	7.76	7.81	6.8	6.78	7.3	7.22	7.66	7.55
Ortho Phosphate as PO4 (mg/L)	0.045	0.045	0.045	0.045	0.045	0.045	0.045	0.2	0.045	0.045	0.045	0.2
Selenium as Se(ug/L)	1	1	1	3	1	1	4	1	1	1	1	1
Sulphate (mg/L)	37.15	30.88	28.47	43.54	73.08	67.44	20.75	17.04	85.77	63.31	55.89	40.22
Turbidity (NTU)	4.05	15.1	9.33	18.2	2.67	3.15	3.51	655	2.88	2.1	3.22	268
Zinc as Zn(ug/L)	0.25	0.25	0.25	0.25	2	2	0.25	8	0.25	0.25	0.25	0.25

Wilge Integrated Water Resource Management Plan Resource Water Quality Objectives (DWAF, 2009)
South African water quality guidelines for Aquatic Ecosystems TWQO (DWAF, 1996)
Australian and New Zealand Guidelines (Australian and New Zealand Environmental and Conservation Council, 2000)
Exceedances of the respective quality guidelines

Aluminium and ammonia were found to exceed the Wilge River IWRMP RWQOs almost throughout this study period. Other parameters that exceeded the Wilge River IWRMP RWQOs at 2 or more occasions in the study period were calcium, magnesium and manganese at the SW16 sample point and orthophosphate at the SW2 sampling sites. Indicating that calcium, magnesium and manganese are entering the catchment upstream of the SW16 sample point in the Wilge River. Also indicating that the orthophosphate concentrations are entering the Wilge River from either or both of the tributaries of the Wilge River between the SW16 and SW2 sampling sites. Conductivity, sodium and sulphate concentrations were found to exceed the Wilge River IWRMP RWQOs in the July 2013 survey, at the SW16 and SW2 sampling sites indicating an incident during this period. Sulphate concentrations were found to exceed the Wilge River IWRMP RWQOs in the June 2014 survey, once again at the SW16 and SW2 sampling sites. Zinc concentrations exceeded the DWA TWQO in the January 2014 survey at the SW2 sample point. Calcium, conductivity, magnesium, manganese and sodium exceeded the Wilge River IWRMP RWQOs in the November 2014 survey indicating an incident.

Macroinvertebrate Community (2012-2014)

During this study period a total of 43 taxa were identified and counted during the 6 surveys at the two sampling sites. Twenty five (25) of these taxa were each represented by less than 1% of the total number of specimens sampled, while the remaining 18 taxa were each represented by more than 1% of the total specimens collected (Table 4.12). The Baetidae were dominant during the study, followed by Simulidae, Chironomidae and Corixidae. Baetidae and Dytisicidae were found at all the sampling sites, during all the surveys.

Table 4.12: Taxa listed according to the number of specimens collected as occurring in less or more than 1% of the total number of specimens found during the study.

Taxa % occurrence <1%		Taxa % occurrence >1%	
Porifera	Leptoceridae	Turbellaria	Gyrinidae
Coelenterata	Elmidae	Oligochaeta	Ceratopogonidae
Hirudinea	Hydrophilidae	Atyidae	Chironomidae
Potamonautidae	Culcidae	Baetidae	Simulidae
Hydracarina	Tabanidae	Caenidae	Sphaeridae
Trcorythidae	Tipulidae	Heptageniidae	
Chlorocyphidae	Ancylidae	Leptophlebiidae	
Gomphidae	Lymnaeidae	Coenagrionidae	
Belostomatidae	Physidae	Corixidae	
Gerridae	Planobinae	Notonectidae	
Hydrometridae	Corbiculidae	Veliidae	
Nepidae	Unionidae	Hydropsychidae	
Pleidae		Dytisicidae	

The total number of specimens collected per site ranged from 88 at SW2 in the November 2012 survey to 622 at SW2 in the July 2013 survey. Of the total of 2698 specimens found over the entire study period during all the surveys, 1852 were found during the low flow periods (May 2012, July 2013 and June 2014 surveys) and 846 were found during the high flow periods (November 2012, January 2014 and November 2014 surveys). The dominant taxa found during the low flow periods were Baetidae, Chironomidae, Simulidae and Caenidae. The dominant taxa found during the high flow periods were Corixidae, Baetidae, Simulidae and Caenidae.

The total SASS5 scores increased largely from the upstream to the downstream site. This improvement in biotic integrity was despite a lower IHAS score at the downstream site. The water quality was better at the downstream Wilge River site, which resulted in better SASS5 scores (higher biotic integrity). This corresponds well with the water quality variables tested during this assessment. Specifically, the pH improved from well below target values at the upstream site to marginally below target values at the downstream site. This indicates a positive influence from the two tributaries entering the Wilge River between these two sites.

The sampling sites during all the surveys were dominated by aquatic macroinvertebrate taxa with a low/very low requirement and moderate requirement for unmodified water quality. Some taxa with moderate requirement for unmodified water quality were present. Only one macroinvertebrate Heptagenidae, which has a high requirement for unmodified water quality, was present in all surveys except November 2014. The SASS5 scores ranged from 67 at SW16 in the November 2014 survey to 130 at SW2 in the May 2012 survey. The ASPT scores ranged from 4.4 at SW16 in the November 2014 survey to 6.16 at SW2 in the June 2014 survey. All SASS5 and ASPT scores were from to be lower at the SW16 site compared to the SW2 site. IHAS scores (Table 4.8) showed higher habitat availability and suitability at the upstream site.

The diversity of habitat types, together with good flow conditions and water quality at SW16 and SW2, support a diverse and abundant aquatic macroinvertebrate community. A combination of habitat availability, good water quality and flow conditions/velocities at a particular site, as found by Farrel *et al*., (2015) were the primary variables that supported a diverse aquatic macroinvertebrate community. The overall ASPT values recorded in the study area were indicative of taxa that were tolerant to moderately tolerant of pollution levels. Most families found have a preference of low to very low water quality (Thirion, 2007), indicating that the water of the Wilge River sub-catchment B20F area is in a poor state of health.

RDA diagram, (Figure 4.24) shows measured water quality variables, sampling sites and actual macroinvertebrate counts of families on a constrained RDA triplot. Explanatory factors (environmental and macroinvertebrate families), explains a total of 49.09% of the observed variation with the first axis explaining 32.76% and the second axis 16.33% of the variation. Species selection of 50-100% was selected to show the 15 best fitting families.

The triplot shows that the variables sulphate, conductivity, sodium and total dissolved solids explained the variation in the distribution on axis 1. Macroinvertebrate families that showed a strong association with these parameters are Sphaeridae, Turbellaria, Baetidae, Chlorocyphidae, Unionidae, Ancylidae, Ceratopogonidae and Caenidae. Although IHAS score data was included in the RDA dataset, it was not selected as a determining factor in the distribution of the data. The parameters iron, nickel, phosphate and turbidity explained the variation in the distributive trends on axis 2. Macroinvertebrate families associated with these parameters are Notonectidae, Veliidae and Atyidae. Macroinvertebrate Hydropsychidae and Simuliidae showed a strong negative loading to these parameters. Sample point SW2 was found to be associated with these conditions on high flow conditions in 2014 and low flow conditions in 2012. Sample site SW16 was found to be associated with these conditions during high flow conditions in 2014.

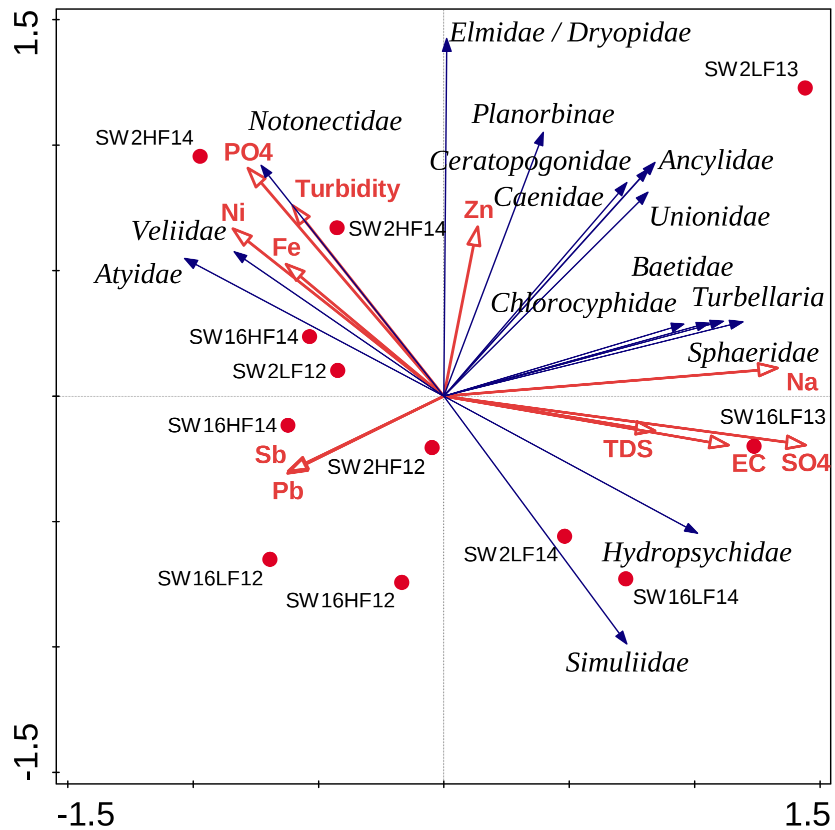

Figure 4.24: Constrained redundancy analysis for macroinvertebrates in Wilge River sub-catchment B20F area (2012-2014 Actual counts) for two sampling sites (SW16 and SW2), and six surveys (May 2012-November 2014) showing water chemistry factors (red arrows) electrical conductivity (EC), total dissolved solids (TDS), sulphate (SO_4), sodium (Na), zinc (Zn), turbidity (Turbidity), phosphate (PO_4), nickel Ni), iron (Fe), antimony (Sb), Lead (Pb) and; taxa (blue arrows). Species selection of 50-100% was selected to show the 15 best fitting families. Explained variation on Axis 1 is 32.76% and on Axis 2 is 16.33% (Total variation explained is 49.09%).

For surveys from 2012-2014 showing sample sites SW16 and SW2 only, the environmental factors explaining the distribution of the data are EC, sulphate and sodium. Macroinvertebrate families Sphaeridae and Turbellaria (which prefer very low water quality) and Chlorocyphidae (which prefers moderate water quality were found associated with these parameters. Environmental factors phosphate, nickel, turbidity and iron also explained the distribution of the sample sites showing a positive loading to these parameters during high flow conditions and a negative loading during low flow conditions in 2014. Macroinvertebrate families found associated during the high flow conditions are Atyidae and Veliidae (which prefer moderate water quality) and

Notonectidae (which prefers very low water quality). Macroinvertebrate families found associated during the low flow conditions are Hydropsychidae (which prefers high to moderate water quality) and Simuliidae (which prefers low water quality).

4.4 Conclusions

Biodiversity is frequently seen as an indicator of ecological health. From the *in situ* analyses of all the sample sites high pH occurrances were found at SW17, SW2 and S2 on the Wilge River. Low pH was found to occur at SW16 which is located also on the Wilge River but downstream of its confluence with the Klipfonteinspruit during the biomonitoring surveys. High conductivity was found at sample sites S2 and S3 on the Wilge River downstream of the confluence with the Klipfonteinspruit. Low DO concentrations were found at sample site SW9 which is found on the unnamed northern tributary of the Wilge River and at sample sites SW2 and SW16 on the Wilge River during the biomonitoring surveys. Soil erosion was found to increase at all sample sites from December 2012 to November 2014 except, S3, SW7, SW9 and SW1 where the soil erosion state remained the same. Sample sites SW7 and SW4 were in a severely eroded state in November 2014. Sedimentation deposition was found to increase at all sample sites from December 2012 to November 2014. Sample sites SW5, SW9, SW4 and S4 showed severe sedimentation deposition in November 2014.

From the biomonitoring analyses of all the sample sites ASPT scores were found to be highest at S3, S4 and SW2. Sample site S3 is located on the Wilge River, downstream of the confluence with the Klipspruit on which sample site S4 is found and downstream of the Klipfonteinspruit. Sample site SW2 is located on the Wilge River, upstream of all the sample sites. IHAS scores were found to be high at sample sites on the Wilge River S2, S3, SW16, SW7, SW2 and S4 on the Klipspruit River. IHAS scores were found to be lower at SW5, SW7 and SW4. Higher SASS5 scores were found at sample sites S4, S2 and S3, which is most probably due to the better habitat conditions found at these sites. SASS5 scores were found to be lower at SW5, SW7, SW1, SW17 and SW2. The IHAS scores are quite low at sample sites SW5 and SW7 and can be the reason attributed to the low SASS5 scores. Sample site SW1 has since January 2013 showed an increase in IHAS scores and subsequent increase in SASS5 scores as well. Sample site SW17 and SW2 both are located on the Wilge River and has relatively good IHAS scores. It is a concern that the SASS5 scores were found to be lower at these sites. SASS5 scores have shown improvement at SW17 and SW2 from March 2014.

The sampling sites during all the surveys were dominated by aquatic macroinvertebrate taxa with a low/very low requirement and moderate requirement for unmodified water quality. Some taxa with moderate requirement for unmodified water quality were present. Only one macroinvertebrate Heptagenidae, which has a high requirement for unmodified water quality, was present in all surveys except November 2014. The SASS5 scores ranged from 67 at SW16 in the November 2014 survey to 130 at SW2 in the May 2012 survey. The ASPT scores ranged from 4.4 at SW16 in the November 2014 survey to 6.16 at SW2 in the June 2014 survey. All SASS5 and ASPT scores were found to be lower at the SW16 site compared to the SW2 site. IHAS scores showed higher habitat availability and suitability at the upstream site.

The diversity of habitat types, together with good flow conditions and water quality at SW16 and SW2, support a diverse and abundant aquatic macroinvertebrate community. A combination of habitat availability, good water quality and flow conditions/velocities at a particular site, as found by Farrell *et al*., (2015) were the primary variables that supported a diverse aquatic macroinvertebrate community. The overall ASPT values recorded in the study area were indicative of taxa that were tolerant to moderately tolerant of pollution levels. Most families found have a preference of low to very low water quality (Gerber and Gabriel, 2002; Thirion, 2007), indicating that the water of the Wilge River sub-catchment B20F area is in a poor state of health.

In 2006 environmental factors EC, DO, Temp and pH were driving factors for the distribution of the sample sites under consideration (S2, S3, S4, SW5 and SW1 only). Sample sites associated with these parameters are S4 located on the Klipspruit River; and S2 and S3 located on the Wilge River. Sample sites located on the Klipfonteinspruit (SW5 and SW1), show no association with EC, DO, Temp and pH. For 2010 to 2014 EC and IHAS are the strong driving factors for the distribution of the sample sites. There is a strong association with these parameters and the sample sites located on the Wilge River (S2, S3, SW16, SW17 and SW2) and Klipspruit River (S4). Sample sites located on the Klipfonteinspruit (SW5 and SW1), the unnamed tributary of the Klipfonteinspruit (SW7) and the unnamed tributary of the Wilge River (SW9 and SW4) show no association with EC and IHAS score.

Explanatory factors macroinvertebrate families Ancylidae, Ceratopognidae, Hydropsychidae, Leptophlebiidae, Lestidae, Pleidae, Potamonautidae, Simuliidae, Turbellaria, Oligochaeta and Coenagrionidae were found to be associated with sample sites S2, S3 and S4 in 2006. For 2010-2014 macroinvertebrate family Elmidae was found to be associated with S2, S3, S4, SW16, SW17 and SW2. Leptophlebiidae was found to be associated with these sample sites in 2010, 2011, 2012, 2013 and 2014. Heptagenidae was found to the associated with these sample sites in 2010, 2013 and 2014. Elmidae, Leptophlebiidae and Heptagenidae have a preference for high to moderate water quality (Gerber and Gabriel, 2002; Thirion, 2007).

From this macroinvertebrate study it was evident that considerable variations occurred with regard to the families found between the various surveys and between each of the sampling sites. These variations probably resulted from a combination of environmental and physical conditions, which include the conditions of the habitat, habitat availability changing with flow, seasonal variation in macroinvertebrate communities and various anthropogenic influences such as flow modifications (impoundments), agriculture and mining. A dominance of families found has a preference for low to very low water quality, probably due to the changes in land-use. Ultimately, in order to manage and conserve the Wilge River, it is essential to understand the catchment and its impeding land-uses (Farrell *et al.,* 2015). Macroinvertebrate assemblage within the Wilge River sub-catchment B20F area show that it is in a poor state of health and it is therefore imperative to maintain the ecological integrity of the Wilge River and strive to improve it.

4.5 References

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CHAPTER 5: USE OF FISH AS AN INDICATOR OF ECOLOGICAL HEALTH IN THE WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

5.1 Introduction

The freshwater ichthyofauna of South Africa is currently threatened by several factors which include development, water withdrawal, agricultural pollution, domestic and industrial effluents (Ashton, 2007). In the multi-use environment of water resources biological monitoring has gained momentum in developing countries because it is a fast and cost-effective approach for assessing the effects of environmental stressors (Bere and Tundisi, 2010). Biological monitoring has increased due to growing concerns of water pollution, flow regulation, and effects of land-use changes on aquatic ecosystems among other challenges.

The analysis of fish has long been applied to assess the integrity and/or identify impacts affecting freshwater aquatic ecosystems (Barbour *et al*., 1999). Fish community attributes were used to assess ecological integrity of stressed ecosystems (Karr, 1981). Fish communities reflect overall ecological integrity by integrating different stressors over time and thereby providing a broad measure of their aggregate impact. The monitoring of fish communities therefore provides a reliable ecological measure of fluctuating environmental conditions. Although degraded water quality conditions continue to pose the greatest threat to fish health in this system (De Villiers, 2007), additional impacts such as habitat alteration, flow regime modifications, barriers for migration, disturbance to wildlife and/or the impact of non-endemic alien or introduced fishes may be affecting the fish communities (Wepener *et al*., 2011).

Fishes possess a suite of advantages which have made them to be widely used for measuring a wide range of human impact types on aquatic environments (Barbour *et al*., 1999). Thus, fish can be used as indicators for habitat degradation (Gorman and Karr, 1978; Raburu and Masese, 2012), metal pollution levels (Authman *et al*., 2015), deforestation (Roth *et al*., 1998), water quality (Kadye, 2008) and flow regulation (Bain *et al*., 1988). Fishes are migratory and they are considered sensitive organisms for continuum disruptions (Northcote, 1998). In Southern Africa, the Fish Assemblage Integrity Index (FAII) developed by Kleynhans (1999), and the Fish Response

Assessment Index (FRAI) also developed by Kleynhans (2007), has been used to determine the status of the fish assemblage in relation to human-induced factors. FAII was used to account for both anthropogenic and environmental factors. The index evaluates the impact of exotic species, which were a threat to fish in many Zimbabwean rivers (Minshull, 1993), as well as assessing the impact of human activities, including pollution and siltation. Although the FRAI uses essentially the same information as the FAII it does not follow the same procedure. The FAII was developed for application in the broad synoptic assessment required for the River Health Programme (RHP) (Dallas, 2005), and does not have a particularly strong cause-and-effect basis. The purpose of the FRAI, on the other hand, is to provide a habitat-based cause-and-effect in order to interpret the deviation of the fish assemblage from the reference condition.

Ibanez *et al.,* (2007) noted that it is important to identify distinct assemblages in river systems, so that each can be explicitly integrated into conservation plans. It is also important to understand how both natural biogeography and anthropogenic alterations influence assemblages (Matthews, 1998). Rigorous and integrated conservation approaches are needed to inform catchment management in the face of competing demands for water use across South Africa (Ashton, 2007). Single species conservation approaches may focus on a species of concern such as an endemic or endangered species. Focal species, which are particularly sensitive to environmental disturbance or limited in their habitat needs, and umbrella species, whose habitat needs overlap with many other species, may also be conservation targets (Abell, 2002). For instance, conservation management activities in South Africa often target focal species based on their regional endemism or specialised environmental requirements (Roux et al., 2008). Similarly, Rivers-Moore *et al*., (2007) noted that recent research priorities for freshwater conservation in the KwaZulu-Natal Province include developing a list of freshwater umbrella species, whose distribution ranges will need to be modelled. Rashleigh *et al.,* (2009) investigated patterns in freshwater fish assemblages in the Olifants River catchment and noted that species richness on its own may not be an optimal measure for conservation planning; and suggested that the densities of selected species that are less tolerant to flow alteration, or the ratios of such species to more tolerant species, may be better conservation measures.

Conservation efforts for the Olifants River must also consider fish assemblages and biodiversity within the broader context of the region (Roux *et al*., 2002). In an analysis of riverine conservation in the South African Lowveld, Roux *et al*., (2008) concluded that the optimal set of planning units for protecting regional biodiversity would not include the Olifants River, since neighbouring catchments possessed higher conservation value. Dudgeon *et al.,* (2006) showed that the long-term protection of freshwater biodiversity requires a mixture of strategies that include reserves for high value areas and species or habitat-centred plans that reconcile conservation with water resource use for human-modified ecosystems. An ecosystem approach to management can be used to reconcile these alternative services and ensure sustainability in the Olifants River (Jewitt, 2002). This will be necessary to maintain genetic diversity (Wishart and Davies, 2003), as well as the provisioning of goods and services essential to people in communities within the catchment (Skukuza Freshwater Group, 2006).

The comprehensive Ecological Reserve determination for the Olifants River WMA was undertaken by the DWAF in 2000. The key results of the study relevant to this study were the determination of the Present Ecological Status (PES), a proposed ecological management class (EMC), the environmental water requirements (EWR), ecological water quality requirements and the ecological importance and sensitivity. Most of the B20F study area was categorisied to be in a medium to poor state (DWA, 2009).

The Olifants River catchment experiences an extreme demand for natural resources, and associated land modification and pollution. Thus, river ecosystems in this area are generally in a fair to poor condition. The upper reaches of the Olifants River catchment are characterised mainly by mining, agricultural and construction activities (WRC, 2001). Mining-related disturbances are the main causes of impairment of river health. Over-grazing and highly erodible soils result in such severe erosion, which after heavy rains has a red-brown colour from the suspended sediments. There is also an extensive invasion by alien vegetation, and to a lesser extent, alien fauna (WRC, 2001).

The aim of this chapter was to analyse the fish communities in the Wilge River sub-catchment B20F area, in order to better understand the natural and anthropogenic influences on these fish communities and to identify driving variables that influence these communities both spatially and temporally, in order to determine whether the fish communities can be used as indicators of water quality and ecosystem health.

5.2 Materials and methods

The study area for this project was located within the Olifants River Water Management Area (WMA4), within the quaternary drainage region B20F in the Wilge River sub-catchment. The eleven sample sites were located on the Wilge River, the Klipfonteinspruit River, on the unnamed northern tributary of the Wilge River and on an unnamed tributary of the Klipfonteinspruit (Figure 5.1). These sample sites are impacted on by extensive agricultural activities (i.e. cattle, pig and chicken farming), pivot irrigation, rural development, infrastructure (i.e. railways and tar and/or gravel roads), industrial activities (i.e. construction footprint and existing industrial complexes) and mining and quarries (from tributaries entering the Wilge River) (Farrell *et al*., 2015).

Eighteen fish surveys were carried out at eleven sample sites, SW5, SW7, SW1, SW9, SW4, S2, S3, S4, SW16, SW17 and SW2 in the Wilge River sub-catchment B20F area from March 2010 to December 2014. Surveys were conducted in March 2010, June 2010, September 2010, December 2010, March 2011, June 2011, September 2011, November 2011, August 2012, December 2012, February 2013, May 2013, August 2013, November 2013, March 2014, May 2014, August 2014 and December 2014. This data was compared to the aquatic assessments done for the Environmental Impact Assessment (EIA) in September 2006 at sample sites SW5, SW1, S2, S3 and S4.

As fish are mobile and can move extensive distances within a system, the fish community at a specific site monitored in isolation will not be a true reflection of the fish community response to stressors. Sample sites in this study were therefore grouped into 5 reaches in order to be analysed more effectively (Table 5.1 and Figure 5.1).

Table 5.1: Reaches and associated sampling sites.

Reaches	Associated sites
1	SW4 and SW9
2	SW5, SW7 and SW1
3	S4
4	S2, S3 and SW16
5	SW17 and SW2

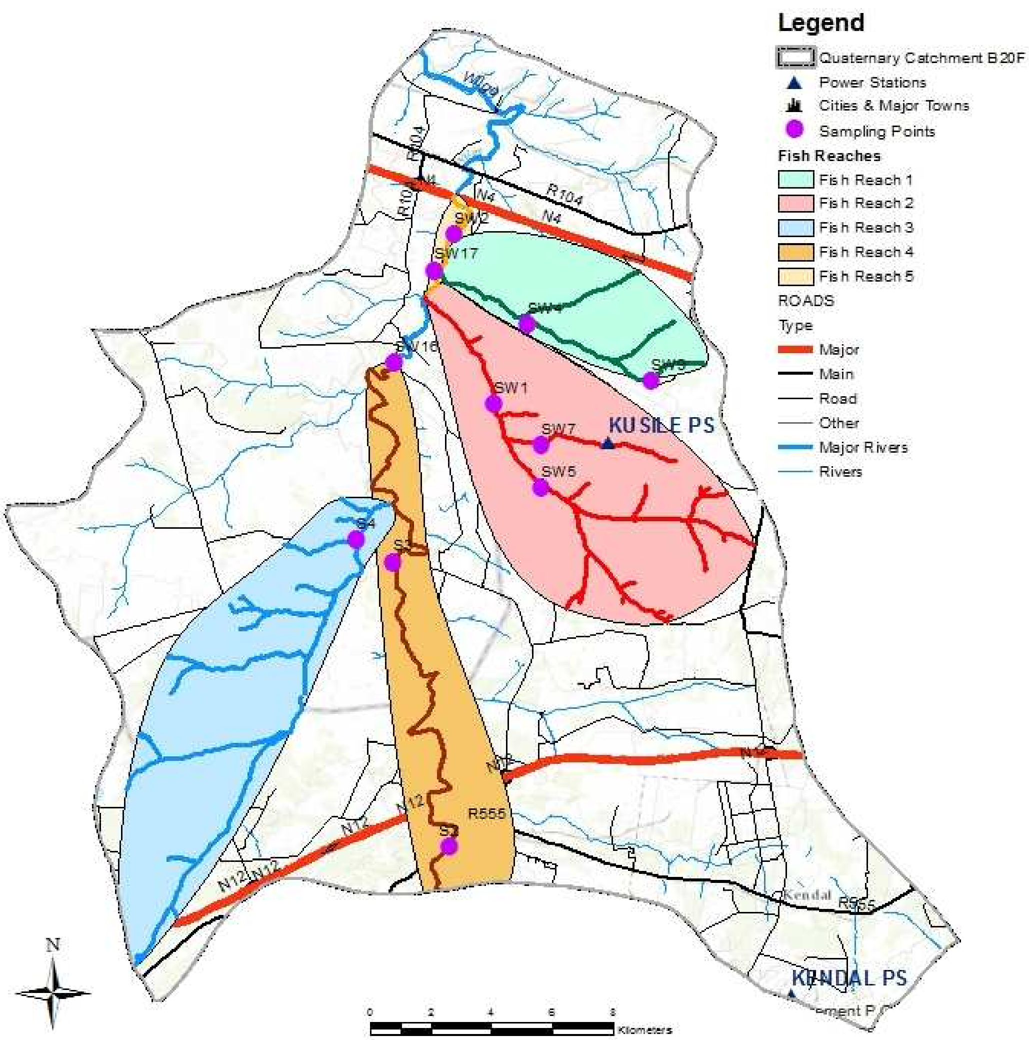

Figure 5.1: Map of Kusile power station study area, indicating streams and selected monitoring sites.

The fish assessment was completed using standard electrofishing techniques (Plafkin *et al*., 1989). Electrofishing was conducted using a SAMUS electrofisher Model number:

SAMUS725MP. This method relies on an immersed anode and cathode to temporarily stun fish in the water column; the stunned fish can then be scooped out of the water with a net for identification. The responses of fish to electricity are determined largely by the type of electrical current and its wave form. These responses include avoidance, electrotaxis (forced swimming), electrotetanus (muscle contraction), electronarcosis (muscle relaxation or stunning) and death (USGS, 2004). Electrofishing is regarded as the most effective single method for sampling fish communities in wadeable streams (Plafkin *et al*., 1989). All fish were identified in the field using the guide Freshwater Fishes of Southern Africa (Skelton, 2001) and released back into the river at the point of capture.

FRAI was used to assess the fish populations of the river sites in this chapter. FRAI is an assessment index that is based on the intolerance and preferences of a given fish assemblage and the response of the assemblage to changes in certain environmental variables. The intolerances and preferences of an assemblage is categorised into metrics which relate to the environmental requirements and preferences of each species. The FRAI index relies on the use of a reference fish species list as well as a fish frequency of occurrence within the system. This information is available in the Reference Fish Frequency of Occurrence (FFROC) (Kleynhans *et al.,* 2007). If this data is not available other sources of data need to be studied to populate the reference list. The reference list in this chapter was compiled using the Reference FFROC (Kleynhans *et al.*, 2007), previous biomonitoring reports for the Kusile power station construction project (Ferreira, 2014) and the River Health Programme (RHP) (DWAF, 2008; Dallas, 2005) (Table 5.2). The information collected from the RHP monitoring localities 4OF154, 40F148, B2 Wilge Spitz and 40F137 were used to determine the reference list. Although *Enteromius trimaculatus* was not found during the entire study period from September 2006 to December 2014, it was included in the FFROC list as is was found to be present by Kleynhans *et al*., (2007). *Enteromius trimaculatus* was sampled and positively identified during the River Health Programme (RHP) reference FFROC determination in South Africa. Two separate FFROC lists were used for the FRAI assessment i.e. one for the Wilge River and another for the associated tributaries. *Chiloglanis pretoriae* and *Labeobarbus cylindricus* were not found to be present in the tributaries, as they prefer fast-flowing permanent waters and were therefore left out of the tributary FFROC list.

Table 5.2: Fish Reference Frequency of Occurrence (FFROC) for B20F Wilge River sub-catchment B20F area.

Fish species	Wilge River FFROC	Tributary FFROC
Enteromius anoplus	5	5
Enteromius paludinosus	3	3
Enteromius trimaculatus	3	3
Chiloglanis pretoriae	3	
Clarias gariepinus	3	3
Labeo cylindricus	2	
Labeobarbus marequensis	3	3
Labeobarbus polylepis	5	5
Pseudocrenilabrus philander	4	4
Tilapia sparrmanii	3	3

Species that prefer fast-flowing permanent waters and were therefore left out of the tributary FFROC list.

The reference data obtained were entered into the FRAI 2.0 model, (Kleynhans, pers. com. 2016[1]) and an index value is then automatically calculated. This index value relates to the ecological categories indicated in Table 5.3. The final FRAI value can be adjusted through changes in each metric. This will occur if changes in habitat availability, cover, water quality, etc. took place at a particular site.

The FRAI index does make use of a ranking and weighting system (Kleynhans, 2007). The principle of following a ranking-weighting approach is so that not all driver or biological response metrics will have the same relative ecological significance in all types of rivers. That is, a particular metric may be seriously modified but it may be of relatively low significance in terms of the functioning and integrity of the river. In another river (or different section of the same river) this metric may, however, be of very high ecological importance. The ranking-weighting process is thus done separately from the rating and should not be influenced by it. The FRAI was applied to the 5 different reaches and an ecological category for each reach was determined.

[1] Dr Neels Kleynhans, DWS Resource Quality Information Services, July 2016

Table 5.3: Ecological Categories (EC) used in interpreting RHP data (DWAF, 2008).

Class	Ecological	Description
A	Natural	Unmodified state - un-impacted state, conditions natural.
B	Good	Largely natural - few modifications, mostly natural.
C	Fair	Moderately modified – Community modifications, some impairment of river health
D	Poor	Largely modified – Distinct impairment of river health, impacted state.
E	Seriously modified	Seriously modified – most community characteristics modified, seriously impacted state.
F	Critically modified	Critically modified - extremely low species diversity and abundance, unacceptable modified state.

Redundancy Analysis (RDA) was used to explore the relationship between water chemistry variables and spatial distribution of fish speices. Actual species counts, species presence and absence and water quality parameters were included in the analysis. Analysis was conducted using the CANOCO for Windows package, version 4.5 (Ter Braak and Smilauer, 2002).

5.3 Results

The expected species and habitat preferences are indicated in Table 5.4. Of the 13 species that are expected to occur within the study area, three species are exotic i.e. *Cyprinus carpio, Gambusia affinis* and *Micropterus salmoides*. The remainder of the 9 species are all expected to occur within the study area. The sampling technique used (electrofishing) limits sampling to wadeble areas. This generally has little impact on the sampling of fish communities, but in larger systems (such as the Wilge River) certain species preferring deep waters, will only be sampled in rare occasion. This includes species such as the redeye labeo (*L. cylindricus*) and the lowveld largescale yellowfish (*Labeobarbus marequensis*).

Table 5.4: Habitat preferences and details of observed fish species (Skelton, 2001; Kleynhans, 2003; Kleynhans, 2007).

Scientific Name	Common Name	Habitat Preference	Intolerance Rating
Enteromius anoplus	Chubby head barb	Occurs in a wide variety of habitats from small streams to large rivers and lakes.	2.6
Enteromius paludinosus	Straightfin barb	Hardy species preferring well-vegetated waters and marginal areas of large rivers and slow flowing streams.	1.8
Enterormius trimaculatus	Threespot barb	Hardy species commonly occurring in a wide variety of habitats, especially well vegetated areas.	2.2
Chiloglanis pretoriae	Shorspine suckermouth	Shallow rocky ridges, riffles and rapids of permanent rivers.	4.6
Clarias gariepinus	Sharptooth catfish	Occurs in numerous habitats but prefers slow flowing waters and thrive in dams.	1.2
*Cyprinus carpio**	Carp	Hardy species occurring in variety of habitats, prefers standing or deep slow flowing waters with soft sediments.	1.4
*Gambusia affinis**	Mosquitofish	Tolerant of wide temperature range preferring slow flowing water with plant cover.	2.0
Labeo cylindricus	Redeye labeo	Clear, running water in rocky habitats of small and large rivers.	3.1
Labeobarbus marequensis	Lowveld largescale Yellowfish	Requires flowing waters in perennial rivers. Requires riffle areas for spawning.	2.6
Labeobarbus polylepis	Bushveld smallscale Yellowfish	Cool waters, occurring in deep pools of flowing waters of permanent rivers and dams.	3.1
*Micropterus salmoides**	Largemouth bass	Clean standing or slow flowing rivers and adapted to thrive in farm dams.	2.2
Pseudocrenilabrus philander	Southern mouthbrooder	Wide variety of habitats from standing to flowing waters. Prefers well vegetated backwater areas.	1.3
Tilapia sparrmanii	Banded tilapia	Tolerant species with wide habitat preference but usually found in standing waters with emergent or submerged vegetation.	1.3

* refers to exotic species

(green)	1 – 2	Tolerant
(yellow)	>2 – 3	Moderately tolerant
(orange)	>3 – 4	Moderately intolerant
(red)	4 – 5	Intolerant

In order to assess the Red Data Book status of the expected fish assemblage, the International Union of Conservation of Nature and Natural Resources (IUCN) Red List of threatened species was consulted (IUCN, 2016). Of the ten endemic fish species expected to occur in the sampling area, one, *P. philander* is currently not listed on the IUCN Red List; the rest are currently listed as Least Concern on the IUCN Red List. Species in this category are considered to be widespread and abundant (IUCN, 2016).

Figures 5.2-5.6 show observed fish species from September 2006 till December 2014 for sampling reaches 1-5 respectively. Although fish assessments completed for the Environmental Impact Assessment (EIA) in September 2006 at all sampling sites SW5, SW1, S2, S3 and S4, fish were found and identified at S2, S3 and S4. No fish were found at SW5 and SW1 during the EIA in September 2006.

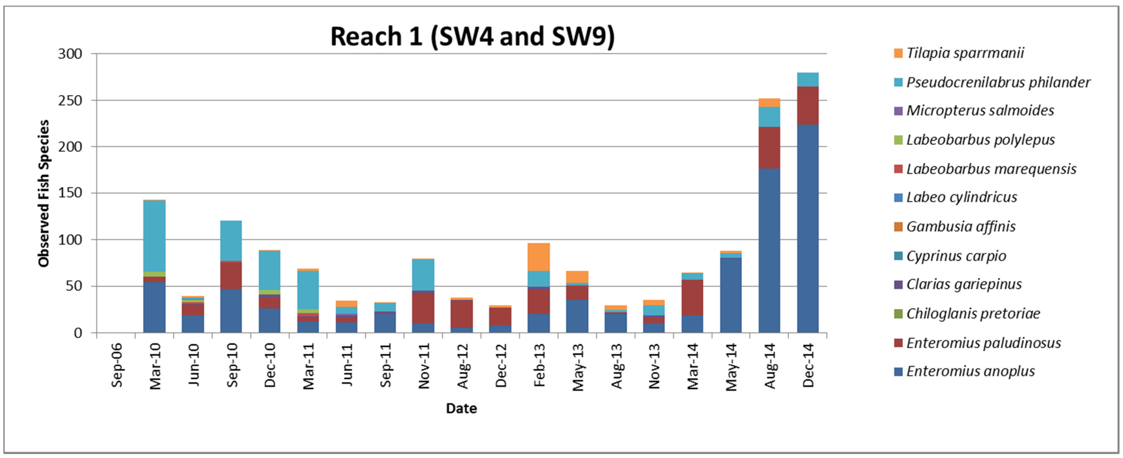

Figure 5.2: Observed fish species at sampling reach 1 from March 2010 till December 2014.

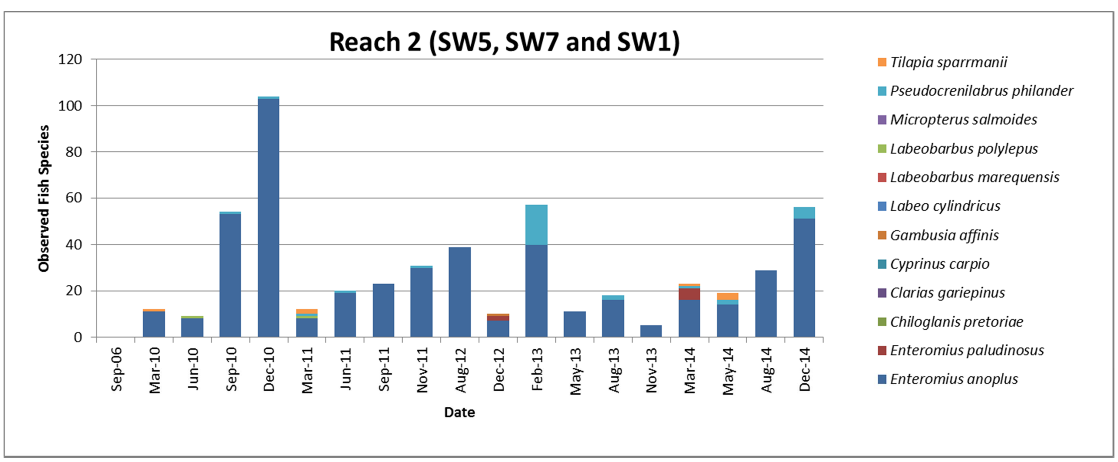

Figure 5.3: Observed fish species at sampling reach 2 from March 2010 till December 2014.

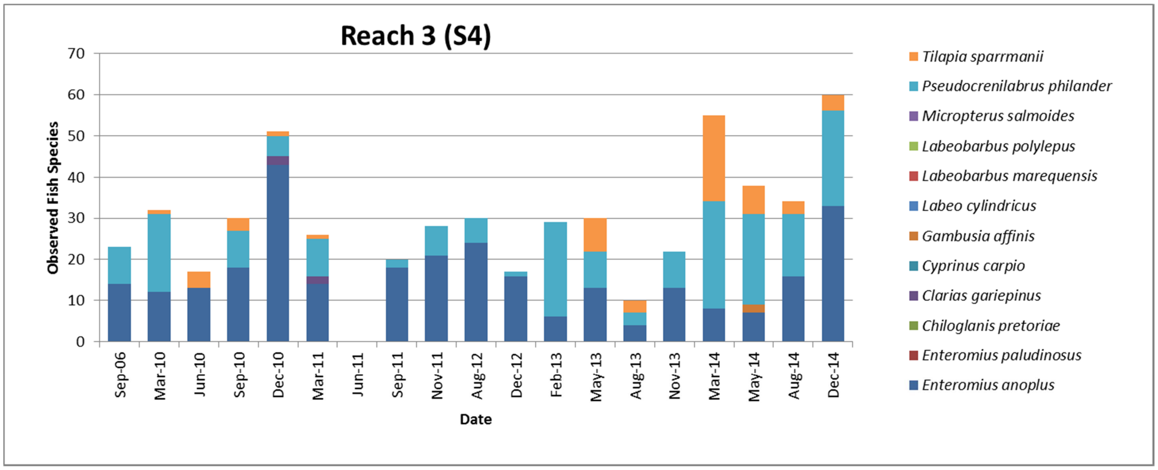

Figure 5.4: Observed fish species at sampling reach 3 from March 2010 till December 2014.

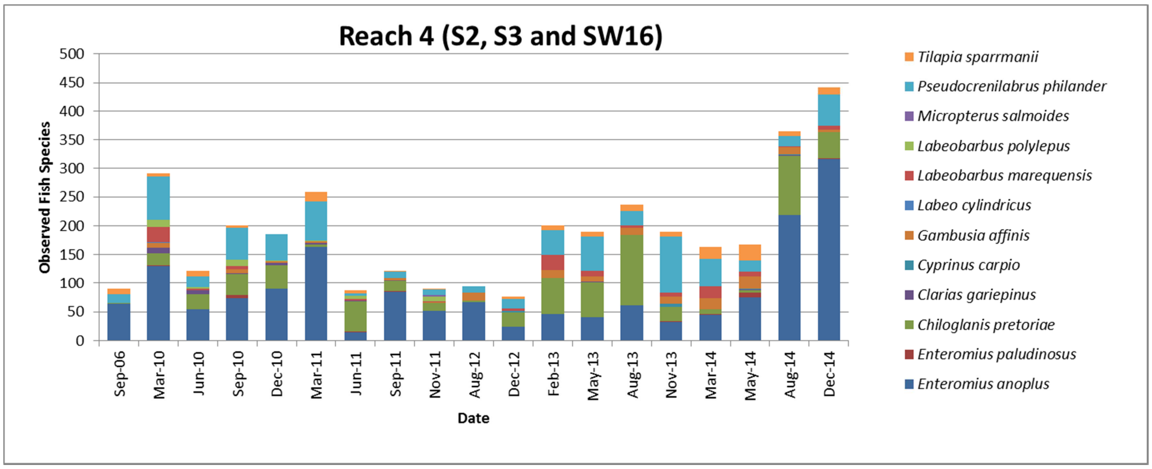

Figure 5.5: Observed fish species at sampling reach 4 from March 2010 till December 2014.

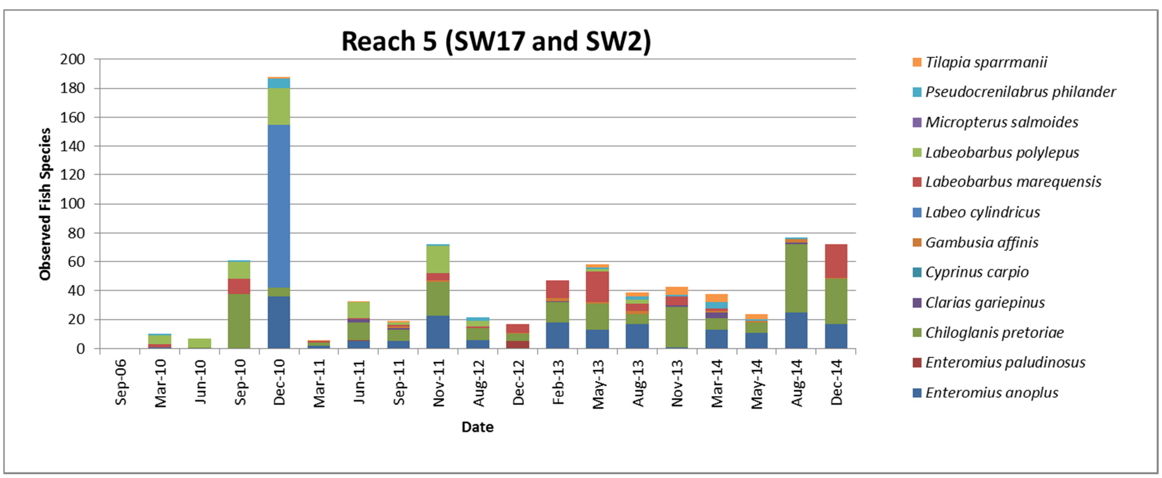

Figure 5.6: Observed fish species at sampling reach 5 from March 2010 till December 2014.

In the 19 (including the EIA) surveys carried out from September 2006 to December 2014, 12 fish species were found with a total number of 7073 fish identified during the study period. The chubby head barb (*E. anoplus*), southern mouthbrooder (*P. philander*), and the shorspine suckermouth (*C. pretoriae*) were the most abundant species observed, with a total number of 3420, 1284 and 939 fishes being counted, respectively. Fish species that were found in low abundance were *M. salmoides, C. carpio* and *C. gariepinus*. *Micropterus salmoides* and *C. carpio* are exotic species. *Clarias gariepinus* thrive in dams and prefer standing or slow flowing waters as habitat. *Cyprinus carpio* was found to occur only in the Wilge River at reach 4 (sampling sites S2, S3 and SW16). Exotic species *G. affinis* was found to occur in higher abundances at reaches 4 and 5 (sampling sites S2, S3 SW16, SW17 and SW2) in the Wilge River and the northern tributary of the Wilge River at SW4 only at reach 1. Two *G. affinis* specimens were found at reach 3 (sample site S4) in the Kipspruit River for the May 2014 survey. The highest species diversity was found at sampling reaches 4 and 5, with all fish species on the reference list (Table 5.4) being found. Sampling site SW7 in sampling reach 2 showed the lowest biodiversity with only *E. anoplus* being found for the duration of the study period. However further downstream of the sampling reach at SW1, *Enteromius anoplus, Enteromius paludinosus, G. affinis. P. philander and Tilapia sparrmanii* were found to be present. These are mostly hardy species with a wide habitat preference.

Fish assessments were carried out for the EIA in September 2006 at sample sites SW5, SW1, S2, S3 and S4 only. *Enteromius anoplus, C. pretoriae, T. sparrmanii and P. philander* were found at all sample sites on the Wilge River for the EIA. Even though sampling sites SW5 and SW1 were sampled for the EIA in September 2006, no fish were found during the fish assessment.

Interpretation of the FRAI scores is based on the ecological categories of DWAF (2008). Tables 5.5 shows the ecological categories which were determined for the sampling reaches 1-5 using the FRAI version 2 model (Table 5.5).

The fish community surveys in the Wilge River sub-catchment B20F area show the reaches in the tributaries to be in a seriously modified state and the reaches in the Wilge River to be in a moderately modified state. At Reach 1 ecological category was found to be moderately modified in March 2010 and this situation was maintained until deterioration occurred in June 2011 and subsequent surveys till present. A number of dams and roads have been constructed within the active channel of the tributaries. Movement of species in the northern tributary (represented by SW4) is still severely disrupted due to approximately eight impeding structures in an eight kilometre section of the water resource.

At Reaches 2 and 3 the ecological category was maintained between seriously and critically modified throughout the study period of September 2006 to December 2014 except for a slight improvement to a largely modified category for Reach 2 in March 2011 before deteriorating once again. The tributaries have been transformed through various land-use activities and several farm dams impede the movement of species into these areas.

Reach 4 and 5 is on the Wilge River. Reach 4 is in a mostly moderately modified ecological state. The reconstruction of a low level bridge at SW16 which was damaged during the high flow events is still of concern. Two small flumes appeared to be placed underneath this bridge. These flumes are not a sufficient route for migration and the bridge has also caused flooding of riparian vegetation and damming of the river upstream of the bridge. The change in ecological integrity downstream of the bridge is a possible indication that the bridge may cause long term alterations to the fish community in the sections above the bridge. The abundance and diversity was much lower at S3 when compared to SW16 and it is evident that the low-level bridge is limiting to upstream fish migration. The majority of the *C. pretoriae* and *L. marequensis* that were recorded were located at the downstream edge of the constructed road. Over time, the inadequately (as it did not take the fish migration into consideration) constructed road will affect biotic integrity up and downstream of the Wilge River. Reach 5 is in a seriously to critically modified ecological state. Fish assemblage structures were not altered due to changes in land-use as the ecological categories remained similar from assessments carried out from 2006 to 2014. Alternate driving variables such as habitat and flow changes may be attributed to the low ecological status of fish in the study area.

Table 5.5: Ecological Categories determined from Fish Response Assessment Index (FRAI) Scores.

Survey	Reach 1		Reach 2		Reach 3		Reach 4		Reach 5	
	SW9 SW4		SW7 SW5 SW1		S4		S2 S3 SW16		SW17 SW2	
Sep-06			0.0	F	20.4	E/F	50.6	D		
Mar-10	63.0	C	17.8	F	27.4	E	76.9	C	33.8	E
Jun-10	79.2	C/B	31.6	E	19.6	E/F	71.2	C	33.1	E
Sep-10	49.1	D	18.4	E/F	27.4	E	72.9	C	63.1	C
Dec-10	60.3	C/D	21.6	E/F	33.1	E	60.0	C/D	73.4	C
Mar-11	84.3	B	45.9	D	33.1	E	60.7	C/D	45.2	D
Jun-11	45.0	D	21.6	E/F	0.0	F	79.2	C/B	73.4	C
Sep-11	43.2	D	15.6	F	20.4	E/F	65.2	C	63.5	C
Nov-11	43.2	D	21.6	E/F	20.4	E/F	68.3	C	59.2	C/D
Aug-12	29.1	E	12.5	F	20.4	E/F	39.6	D/E	47.8	D
Dec-12	23.4	E	18.1	E/F	20.4	E/F	50.2	D	41.0	D/E
Feb-13	41.3	D/E	21.6	E/F	20.4	E/F	71.6	C	50.4	D
May-13	54.4	D	12.5	F	27.4	E	56.2	D	60.9	C/D
Aug-13	35.6	E	15.3	F	27.4	E	72.1	C	66.8	C
Nov-13	35.0	E	12.5	F	22.3	E	64.0	C	55.3	D
Mar-14	38.8	D/E	32.5	E	27.4	E	68.6	C	56.4	D
May-14	32.5	E	28.8	E	27.4	E	68.8	C	35.7	E
Aug-14	43.2	D	15.6	F	27.4	E	63.3	C	36.1	E
Dec-14	27.9	E	23.5	E	27.4	E	66.9	C	44.2	D

Constrained RDA (Figure 5.7) describes all the fish populations in the 19 surveys. There seems to be a strong association between electrical conductivity and *G. affinis* and sample site S2. There is a negative loading to this relationship by *E. paludinosus.* There is a strong correlation between temperature, pH and DO. Fish species associated with these parameters are *L. polyepus, L. cylindricus* and *C. gariepinus.* One sampling site found associated with these parameters is S2. Sampling sites found to be not associated to these parameters are SW7, SW9 and SW1

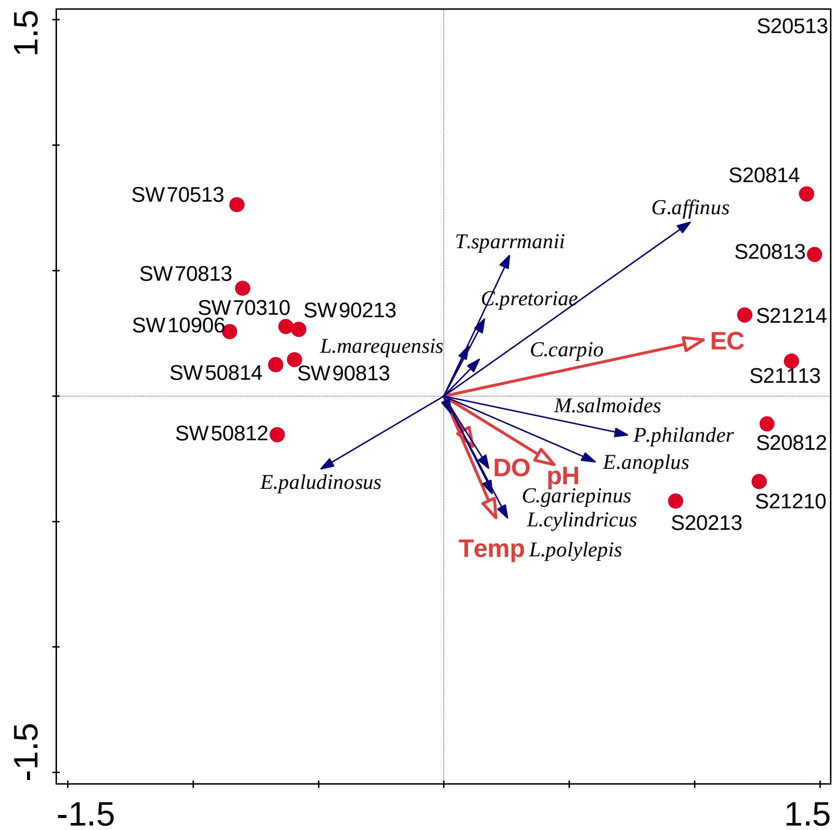

Figure 5.7: Constrained redundancy analysis for fish populations in Wilge River sub-catchment B20F area (2006-2014) for eleven sampling sites (SW4, SW9, SW5, SW7, SW1, S2, S3, S4, SW16, SW17 and SW2) and nineteen surveys (including EIA), showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 50-100% was selected to show the 16 best fitting sampling sites. Explained variation on Axis 1 is 76.67% and on Axis 2 is 12.54% (Total variation explained is 89.21%).

RDA diagrams that follow show measured water quality variables, sampling sites and fish species on a constrained RDA triplot for year 2010, 2011, 2012, 2013 and 2014 in Figures 5.8-5.12 respectively. In the 2010 surveys, a co-linear relationship was found between pH, *C. gariepinus*, electrical conductivity and *L. cylindricus* on axis 1 (Figure 5.8). Sampling sites associated with these parameters are S21210 and S30310. *Enteromius anoplus, L. polylepus, P. philander* and temperature showed a strong co-linear relationship on axis 1 and an association to sampling sites S41210 and SW90910. *Micropterus salmoides* showed a strong negative loading to this relationship and an association to these sampling sites SW50610, SW70710 and SW10610. On axis 2 *G. affinis* showed a strong negative loading to *E. paludinosus*.

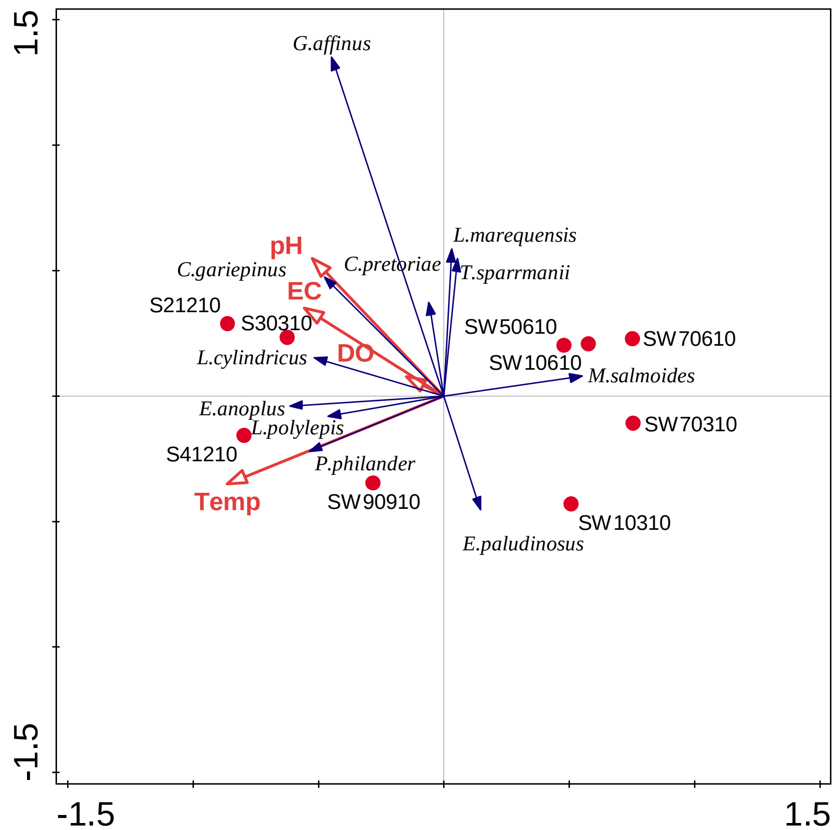

Figure 5.8: Constrained redundancy analysis for fish populations at all eleven sampling sites for 4 surveys (March 2010-December 2010) showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 20-100% was selected to show the 9 best fitting sampling sites. Explained variation on Axis 1 is 54.91% and on Axis 2 is 22.71% (Total variation explained is 77.62%).

A strong co-linear relationship was noted between *E. anoplus*, *G. affinis* and electrical conductivity on axis 1 in the 2011 surveys (Figure 5.9). A sampling site associated with this relationship is S20911. There was a negative loading to *E. paludinosus* and pH. Sampling site found to be assocated with these parameters is SW70911. On axis 2 there is a strong co-linear relationship noted between temperature, dissolved oxygen, *T. sparrmanii* and *P. philander*. Sampling site S4 (S40311 and S41111) were found to be associated with these parameters are during the March 2011 and November 2011 surveys. There is a negative loading to this relationship with *L. marequensis*.

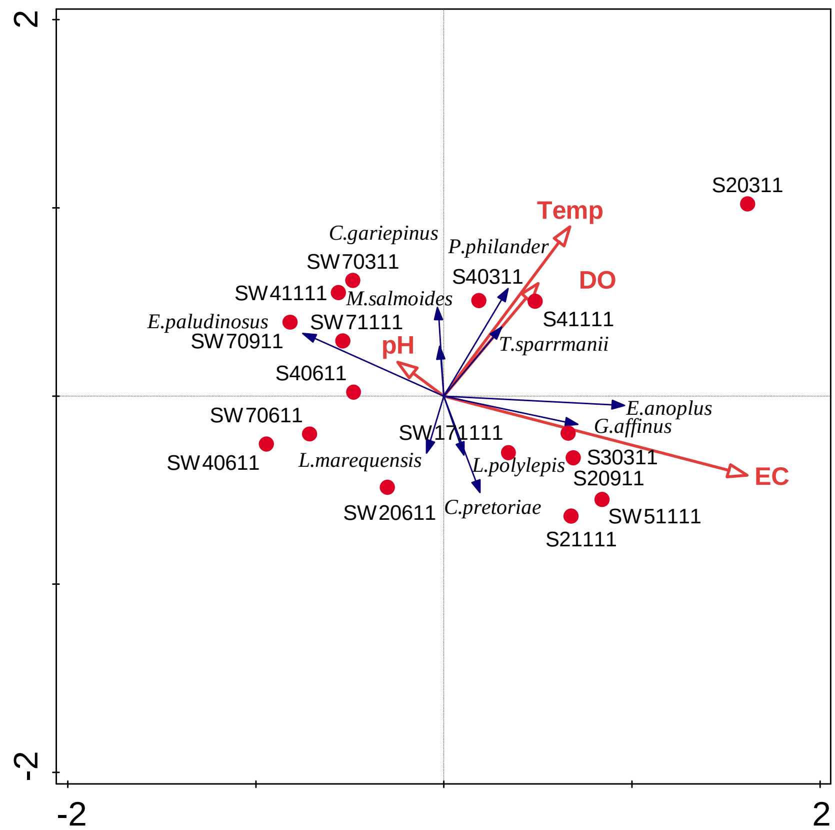

Figure 5.9: Constrained redundancy analysis for fish populations at all eleven sampling sites for 4 surveys (March 2011-November 2011) showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 20-100% was selected to show the 16 best fitting sampling sites. Explained variation on Axis 1 is 70.66% and on Axis 2 is 22.59% (Total variation explained is 93.25%).

A strong co-linear relationship was noted between electrical conductivity and *P. philander* on axis 1 for the 2012 surveys (Figure 5.10). A co-linear relationship was also noted between *E. anoplus,* dissolved oxygen and pH. Sampling sites associated with these parameters are S30812 and S40812. On axis 2 a strong relationship was found between *C. pretoriae, L. polylepis* and *L. marequensis*. A sampling site associated to these parameters was SW20812. On axis 2 a co-linear relationship was noted between temperature, *C. gariepinus, T. sparrmanii and E. paludinosus*. Sample sites associated with these parameters are SW71212, SW91212 and SW11212. There is a negative loading to this relationship with *G. affinis and C. carpio*

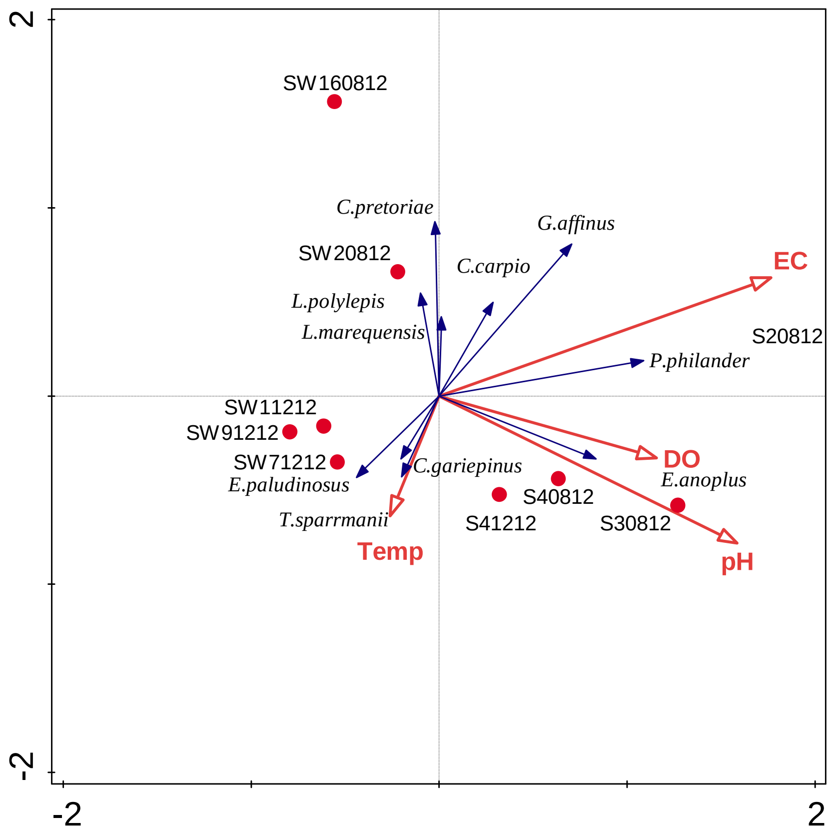

Figure 5.10: Constrained redundancy analysis for fish populations at all eleven sampling sites for 2 surveys (August 2012 and December 2012) showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 40-100% was selected to show the 9 best fitting sampling sites. Explained variation on Axis 1 is 72.13% and on Axis 2 is 21.36% (Total variation explained is 93.49%).

A strong co-linear relationship was noted between electrical conductivity, pH and *G. affinis* on axis 1 in the 2013 surveys (Figure 5.11). Sampling sites associated with these parameters are S20213, S20813 and S30813. There is a negative loading to this relationship by *E. paludinosus*. A strong co-linear relationship was found between temperature, *C. gariepinus and C. carpio* on axis 2. A sampling site found to be associated with these parameters was S41113. There was negative loading to this relationship from *T. sparrmanii, C. pretoriae and L. polylepus.*

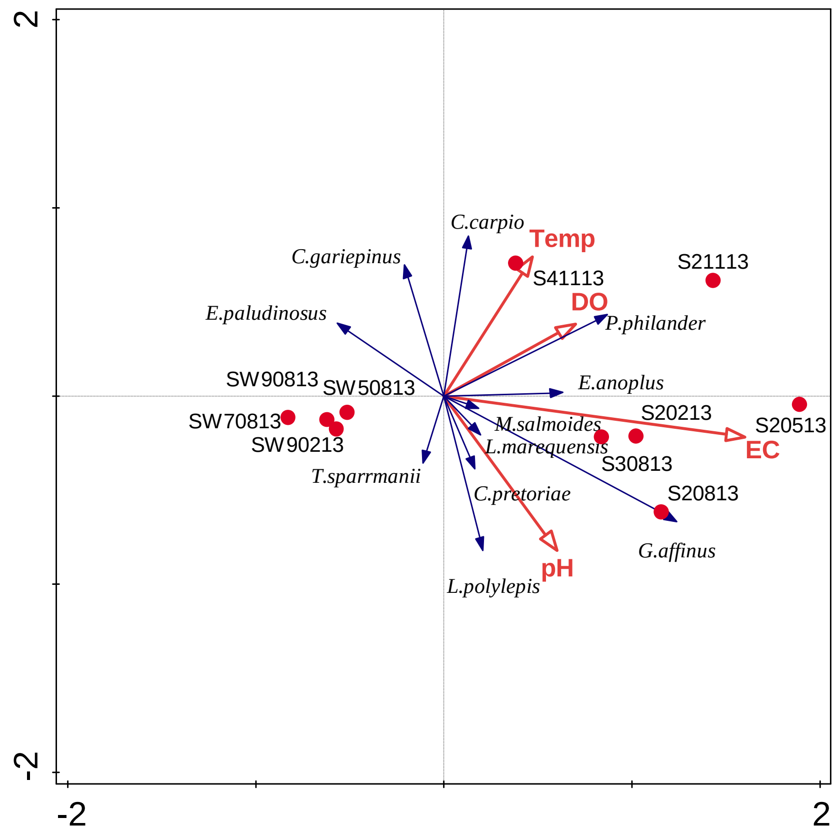

Figure 5.11: Constrained redundancy analysis for fish populations at all eleven sampling sites for 4 surveys (February 2013 and November 2013) showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 40-100% was selected to show the 10 best fitting sampling sites. Explained variation on Axis 1 is 80.03% and on Axis 2 is 13.09% (Total variation explained is 93.12%).

A strong co-linear relationship was found between electrical conductivity, *G. affinis, E. anoplus* and *T. sparrmanii* on axis 1 (Figure. 5.12). A co-linear relationship between *C. carpio* and pH was noted on axis 1. There was negative loading to this relationship from *E. paludinosus*. A co-linear relationship was found between *C. pretoriae* and *C. gariepinus.* Sampling sites found to be associated with these parameters are SW160314 and S40514. There is a negative loading to this relationship with dissolved oxygen. Sampling sites associated with dissolved oxygen are SW10814, SW10514, SW10314, SW70514, SW70814, SW70314 and SW50814.

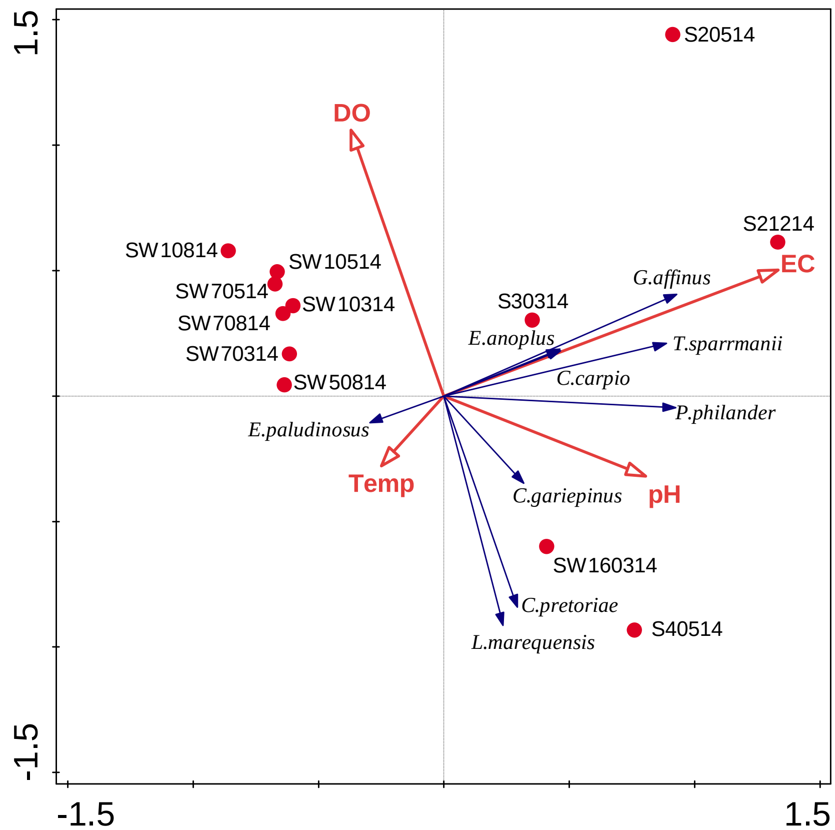

Figure 5.12: Constrained redundancy analysis for fish populations at all eleven sampling sites for 4 surveys (March 2014 and November 2014) showing abiotic factors (red arrows) electrical conductivity (EC), dissolved oxygen (DO), pH, water temperature (Temp) and; taxa (blue arrows). Sample selection of 40-100% was selected to show the 12 best fitting sampling sites. Explained variation on Axis 1 is 66.62% and on Axis 2 is 24.41% (Total variation explained is 91.03%).

5.4 Discussion

Rashleigh *et al*., (2009) showed that altitude and position in the catchment were the most important variables in describing fish assemblage structure in the Olifants River catchment. Altitude is a surrogate for many variables, including temperature, rainfall, soils, geology, and channel form. In this study fish species *C. pretoriae, C. carpio and L. cylindricus* were only sampled in the Wilge River at lower altitudes and not in the tributaries (Figures 5.2–5.6). The finding of higher species richness at lower altitude sites with high runoff is consistent with the finding of Kleynhans (1999) for the Crocodile River.

In addition to the natural influence of altitude, impact of human activity in the Olifants River catchment such as dry-land crops, dams, mining, construction activities and alien vegetation were the most significant pressures in this catchment. This finding is consistent with the assessment of the Water Research Commission (2001), which states that mining, predominantly for coal, and other industrial activities in this area are the main contributors to poor instream and riparian habitat conditions. Stream diversions occur as a result of agricultural and mining activities. In some parts, access roads, mostly related to mining and industrial activities, have resulted in severe disturbance of riparian habitats, and increased erosion of both land and riverbed. In some places the riverbeds are eroded down to the bedrock, leaving little suitable habitat for fish and aquatic invertebrates (WRC, 2001). Influences from dry-land crops include erosion and possible inputs of agrochemicals (DWAF, 2004; Ashton, 2007).

Fish populations at reaches 3, 4 and 5 i.e. sample sites S2 S3, S4, SW4 SW16, SW17, SW2 are affected by agricultural land-use (Figure 5.1, Table 5.5). Dams are widely recognised as significant influences on freshwater systems; major effects include altered timing and volume of flow, modification of habitat, and alteration of sediment, nutrient, and temperature regimes (Postel and Richter, 2003). Fish populations at sample sites SW4 and SW16 are affected by dams (Figures 5.4 and 5.5). Mining influences cause deterioration of water quality with rivers receiving pollutants from various diffuse and point sources. Fish assemblages, mostly with high abundances of omnivores and air breathers which are able to tolerate a variety of environmental conditions were shown to have a relatively low association with the water quality variables (Mangadze *et al*., 2015). Those that do not will be detrimentally affected by poor water quality. Fish populations at sampling sites SW5 and SW1 are affected by mining activities. Fish species *E. anoplus, E. paludinosus, T. sparrmanii and P. philander* were found to be mostly present at sampling sites SW5 and SW1 which is affected by mining activities (Figure 5.3). *Enteromius anoplus, E. paludinosus, T. sparrmanii* are all omnivores which are able to tolerate a variety of environmental conditions.

Construction activities create large areas of bare soil along the banks, which are thus more susceptible to erosion. Eroded soil causes increased turbidity and sedimentation in the rivers, smothering of interstitial habitats and fish gills. Fish populations at reach 1 (sample site SW9) and reach 2 (sample sites SW5, SW1 and SW7) are affected by construction activities (Figures 5.2 and 5.3).

Alien invasive vegetation can pose a threat to biodiversity, as they dominate natural habitats, by competing with natural species for space, water, sunlight, and other resources (WRC, 2001). They also reduce the structural diversity of the vegetation, and disrupt ecosystem dynamics, which can impact on the number and type of animal species that can be supported by the vegetation in that habitat. Alien vegetation can also destabilise riverbanks, because chemicals from their leaves and roots penetrate the soil and prevent other plant species from growing underneath them (WRC, 2001). This creates large areas of bare soil along the banks, which are thus more susceptible to erosion. Eroded soil causes increased turbidity and sedimentation in the rivers, smothering of interstitial habitats and fish gills, and increasing the likelihood and severity of flooding. There is also an extensive invasion by alien vegetation in the Olifants River catchment as noted by the WRC, (2001). The systematic removal of alien vegetation is being conducted by the national Working for Water Programme and in expected to considerably increase the long-term survival of rare and endemic species (Samways *et al*., 2011). For biodiversity conservation to be realised, it is essential that the presence of indigenous plants are maintained and encouraged.

Exotic fish species affect indigenous fish populations by competing for food and habitat. *Micropterus salmoides, C. carpio and G. affinis* are exotic fish species found to occur in the study area. *Micropterus salmoides* and *C. carpio* were found in low abundance. *Cyprinus carpio* was found only in the Wilge River at sample sites S2, S3 and SW16 (Figure 5.5). *Gambusia affinis* was found to be present in higher abundances at sample sites S2, S3 SW16, SW17 and SW2 in the Wilge River and the northern tributary of the Wilge River at SW4 only (Figures 5.2 and 5.6). Although current impact of exotics appear minimal, over time they need to be monitored as they may pose a problem to indigenous populations.

The desired ecological state for most of the upper region of the Olifants River is fair (WRC, 2001), which is the 2014 status for most of the monitoring sites found in the Reaches 4 and 5 on the Wilge River (Figures 5.5 and 5.6). The current (2014) status for most of the monitoring sites in Reach 1 on the northern tributary is poor and the monitoring sites of Reaches 2 and 3 were found to be seriously to critically modified (Figures 5.2, 5.3 and 5.4). Any restoration of instream habitat is unlikely to improve conditions unless there is a focus on the land-use practices that affect and sustain instream processes (Helfman, 2007). Because of the intense development pressure and natural resources demand in the upper Olifants River, it is unlikely that the system will be restored above the status of fair (Rashleigh *et al*., 2009). Therefore, the strategy for the Olifants River could follow the proposals of Seastedt *et al.,* (2008), who suggested that conservation efforts should focus less on restoring degraded ecosystems to their original state and more on sustaining current ecosystems so that they are resilient to further environmental change. Catchment management to support sustainable systems would encompass the active support of multiple stakeholders in order to create balance among economic, social, and ecosystem needs (Roux *et al*., 2006; Ashton, 2007).

The single identification of *E. trimaculatus* by Kleynhans *et al*., (2007) might have been an anomaly as it was not observed again the September 2006 EIA study or any of the eighteen subsequent surveys at any of the eleven sampling sites from March 2010 to December 2014. Consideration should be taken on its removal from the expected list for the Wilge River sub-catchment B20F area. *Enteromius paludinosus, C. gariepinus, L. polylepis, T. sparrmanii* and exotic species *C. carpio, G. affinis* and *M. salmoides* are tolerant species that prefer a wide range of habitats and were found throughout the study area. Fish species diversity within the Wilge River tributaries, i.e. sampling reaches 1 and 2 were found to be lower than within the sample sites of the Wilge River, i.e. sampling reaches 4 and 5. The dominance of *E. anoplus* and *P. philander* in the study area was expected as both species prefer a wide variety of habitats and are common in the Highveld area. The *L. polylepis* populations in the upper Olifants River catchment and in Gauteng rivers were deemed to be in a poor state and that population size and abundance continued to decline (Roux, 2008, O'Brien, 2009). The low fish diversity within the tributaries is primarily due to environmental drivers namely limited habitat availability and poor flow conditions. The Wilge River however, had a much higher diversity, with the exception of site SW17. Similar to the tributaries, this sample site has limited habitat availability, deep channel and no flow conditions. *Chiloglanis pretoriae, C. carpio and L. cylindricus* were only sampled in the Wilge River and not in the tributaries. *Labeobarbus cylindricus* prefers clear and running water in rocky habitats, which is not the case in the tributaries.

Chiloglanis pretoriae should be considered as a focal species as it prefers fast flowing riffle habitat and is sensitive to changes in both habitat and water quality. The absence of this species can be used as an indicator of pollution and habitat alterations (Skelton, 2001, Nel *et al*., 2007). The occurrence of this species below any of the construction activities is an indicator that current habitat and water quality conditions are still favourable for the occurrence of this species. The abundances and distribution of the species should continuously be monitored to determine whether the current increase in turbidity and sedimentation observed in some of the tributaries affects the occurrence of *C. pretoriae* in the lower reaches of the study area. *Chiloglanis pretoriae* does not migrate long distances and siltation of riffle habitats within the Wilge River could lead to a loss of this species in the study area.

Chiloglanis pretoriae was also suggested as a focal species by Roux *et al*., (2008) for the South African Lowveld region. It is anticipated that the *C. pretoriae* population in the Wilge River represents one of the last remaining populations of this species in the upper Olifants River catchment. However, *C. pretoriae* is unlikely to persevere if the elevated turbidity levels persist as its habitats will be directly impacted upon by sedimentation.

5.5 Conclusions

Current conservation efforts in South Africa include the identification of specific river segments for protection (Nel *et al.,* 2007). Rashleigh *et al*., (2009) identified supporting habitats for *C. pretoriae* as those with the higher percent of natural land-use and the water runoff above specified thresholds, so river segments with these characteristics should be conservation priorities. In order to sustain protection of specific river segments, it is also necessary to protect the ecological processes that shape fish assemblages within these segments. For example, we can retain longitudinal connectivity to maintain the process of seasonal migration (Freeman *et al.,* 2007). Connectivity also allows tributaries to serve as refugia for species (Nel *et al*., 2007). Moilanen *et al.,* (2008) showed that including connectivity in species distribution modelling had a major influence on the prioritisation of areas for conservation.

The highest species diversity was found at sampling reaches 4 and 5, with all fish species on the FFROC list being found. Impeding structures such as dams, bridges and roads are affecting fish migration in sampling reaches 1, 2 and 3 showing lower species diversity and higher fish species absences in these reaches. *Enteromius paludinosus* which is found abundantly in the sub-catchment is completely absent at sampling reach 3 for the study period. *Chiloglanis pretoriae* and *L. cylindricus* are completely absent at sampling reaches, 1, 2 and 3 for the study period. *L. marequensis* is completely absent at sampling reaches 2 and 3 for the study period. Many of the species (such as *L. marequensis* and *L. cylindricus*) are known to migrate tens of kilometres. The migration and movement of species into/from the tributaries may be influenced by these (dams, bridges and roads) structures. Certain taxa such as *L. marequensis* and *L. cylindricius* are also not expected to occur in the smaller tributaries due to a lack of habitat. Fish assemblage structures were shown not to have altered due to changes in land-use as the ecological categories remained similar from assessments carried out from 2006 to 2014 and may be attributed to changes in habitat and flow in this study area.

As species richness increased in association with dams, species richness may not be an optimal conservation target for this system. Rashleigh *et al*., (2009) suggested a series of indicators will be necessary to track and measure conservation success in the Olifants River catchment. Indicators such as the densities of selected species that are less tolerant to flow alteration, or the ratios of such species to more tolerant species, may be better conservation measures.

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CHAPTER 6: USE OF WETLANDS AS AN INDICATOR OF ECOLOGICAL HEALTH IN THE WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

6.1 Introduction

In basic terms a wetland would be defined as any part of a landscape where water accumulates long enough to influence the plants and animals in that area. Wetlands are officially defined in the National Water Act, (Act No 36 of 1998) as: "*land which is transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface or the land is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soil*".

The contribution of wetlands to the hydrological cycle is important in maintaining the health of the water resource in addition to providing other services (Emerton and Bos, 2004). Wetlands are among the world's most productive environments; cradles of biological diversity that provide the water and productivity upon which countless species of plants and animals depend for survival. Wetlands are indispensable for the ecosystem services they provide to humanity, ranging from: water purification and replenishment, biodiversity support fishing, grazing, and land for subsistence agriculture, provision of harvestable resources (reeds and other medicinal plants etc.), temporary storage of floodwaters and attenuation of flood peaks, erosion control (through sediment trapping and storage); socio-cultural significance, tourism and recreation, sustained stream flow and ground water recharge (Ramsar Convention Secretariat, 2004; Zedler and Kercher, 2005).

Worldwide, wetlands continue to be subjected to increasing human impacts which diminish their state of health or ecological condition (Junk, 2002). High levels of wetland degradation are reported for South Africa (Kotze *et al*., 1995). Wetlands have been exposed to a range of stress-causing alterations from activities such as dredging and filling operations, hydrologic modifications, pollutant runoff, eutrophication, impoundment, and fragmentation by roads and ditches (Klemas, 2011). These activities cause disruption to the ecological balance of animal and biotic reservoirs in wetlands (Ramsar Convention Secretariat, 2004). The spread of urbanisation and industrialisation has escalated wetland degradation in many parts of the world, in both developing and developed countries (Zedler and Kercher, 2005). Agricultural use and industrial production, pesticide residues, contamination of wetlands from chemicals outlets, change in natural habitats, over exploitation of natural resources, have caused potential risks to the wetland ecosystems (Kotze *et al.,* 2012). Development projects such as road construction, thermal power plants, transmission lines, oil and petrochemicals refineries and factories threaten the life of wetlands. In order to protect and manage wetlands in a sustainable way, it is necessary to reduce ecological risks that impact on the wetlands.

The South African Constitution (Act 108 of 1996) states that everyone has the right to an environment that is not harmful to their health or well-being and is protected for the benefit of present and future generations (Ch 2, para. 24). Thus, the conservation of wetlands is institutionally enshrined. Since the revision of the environmental policy and legislation that began after 1994, the legislative requirements for wetland management have become much more demanding - in particular the National Water Act, (Act No 36 of 1998) various parts of the National Environmental Management Act (1998), National Environmental Management Protected Areas Act (2003) and the National Environmental Management: Biodiversity Act (2004). The legal requirements are evolving so it is necessary to keep abreast of the requirements and adapt to changes as they occur. Due to the growing impact on wetlands, effective monitoring and management of wetlands are essential.

The main ecological value of the wetlands in the upper Olifants River catchment lies in the sheer abundance of wetland habitat in the region. This is probably the single most important ecological factor that has promoted the degradation of the wetlands (Palmer *et al*., 2002). If the wetlands of the study area were seen as a single system (65 000 ha), they may be treated differently in terms of a conservation strategy. Another factor that adds to the aggregate biodiversity value of the wetlands is their diversity of types, providing a high variety of wetland habitats. The fact that 17 hydro-geomorphic wetland types could be distinguished in the upper Olifants River catchment is of conservation significance alone (Palmer *et al*., 2002). Almost all of the wetlands in the study area (along the Wilge River, Klipfonteinspruit and the unnamed stream) have been denoted as Freshwater Ecosystem Priority Areas (Nel *et al.*, 2011). The main threats to the wetlands in the upper and middle Olifants River catchment are land-use changes due to mining, agriculture, dams, weirs, alien vegetation and potential forestry. Deteriorating water quality, in particular acid mine drainage, has also been identified as a threat to the wetlands (DWA, 2009). The assessment of the ecological condition of wetlands is needed for a range of purposes, including Environmental Impact Assessments (EIA), ecological reserve determinations and the planning and monitoring of wetland management and rehabilitation outcomes (Kotze *et al.,* 2012).

The comprehensive Ecological Reserve determination for the Olifants River WMA was undertaken by the DWAF in 2000. The key results of the study were the determination of the Present Ecological Status (PES), a proposed ecological management class (EMC), the environmental water requirements (EWR), ecological water quality requirements and the ecological importance and sensitivity (EIS). The majority of the Wilge River sub-catchment B20F study area is categorisied to be in a fair to medium state and should be managed as an EMC of C (DWA, 2009). A crude estimate made by Palmer *et al*., (2002) of the economic values of wetlands in the upper Olifants River catchment, suggest that, despite the wetlands having little or no indirect use value, their direct use value is not insubstantial. Their studies show that different types of wetlands are estimated to be worth R1 000 to R32 000 per ha.

The aim of this chapter was to investigate the wetlands within the Wilge River sub-catchment B20F area to identify driving variables that influence these communities both spatially and temporally, in order to determine whether the wetlands can be used as indicators of water quality and ecosystem health.

6.2 Materials and methods

6.2.1 Wetland Classification

The National Wetlands Inventory (Nel *et al.*, 2011) provides a series of maps detailing wetland classfication types throughout South Africa. Figure 6.1 shows the wetland types within the Wilge River sub-catchment B20F area.

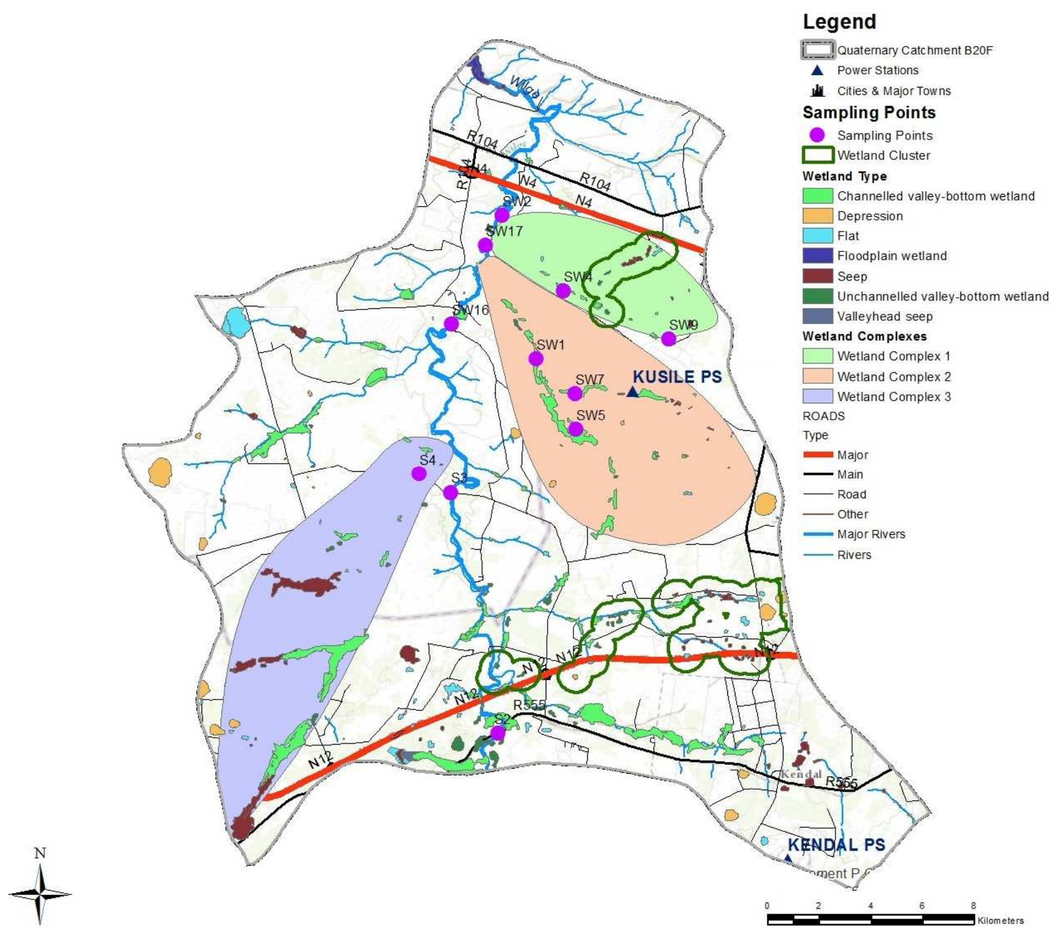

Figure 6.1: Wetlands in the Wilge River sub-catchment B20F area.

The wetland inventory (Nel *et al.*, 2011) identified 791 wetland ecosystem types in South Africa based on classification of surrounding vegetation (taken from Mucina and Rutherford, 2006) and hydro-geomorphic (HGM) wetland type; seven HGM wetland types are recognised and 133 wetland vegetation groups. Based on this classification, the following wetland vegetation types are indicated as occurring on the study area:

- Mesic Highveld Grassland Group 4_Depression;
- Mesic Highveld Grassland Group 4_Flat;
- Mesic Highveld Grassland Group 4_Floodplain;
- Mesic Highveld Grassland Group 4_Seep;
- Mesic Highveld Grassland Group 4_Valleyhead Seep;
- Mesic Highveld Grassland Group 4_Channelled valley-bottom wetland;
- Mesic Highveld Grassland Group 4_Unchannelled valley-bottom wetland.

As many of the sampling sites occur on the same system at different geographical points above and below the construction activities, the wetlands were thus not assessed in isolation, but as a large wetland complex. The different sites included in each of the wetland complexes are indicated in the Table 6.1.

Table 6.1: Associated sites and hydrogeomorphic types for Wetland Complexes 1, 2 and 3 according to (Nel *et al.*, 2011) and (Kassier, (2013).

Wetland Complex	Associated sites	Associated HGMs according to Nel *et al.*, (2011) study	Associated HGMs according to Kassier, (2013) study
1	SW4 and SW9	70% hillslope seeps, 10% unchannelled valley bottom wetlands, 5% channelled valley bottom wetlands and, 5% pans	85% hillslope seeps, 10% floodplain 5% unchannelled valley bottom wetlands,
2	SW5, SW7 and SW1	80% channelled valley bottom wetlands and 20% unchannelled valley bottom wetlands	95% hillslope seeps and 5% floodplain
3	S4	70% unchannelled valley bottom wetlands, 15% hillslope seeps, 10% channelled valley bottom wetlands and 5% pans	Not done

6.2.2 WET-Health Assessment

WET-Health is a tool designed to assess the health or integrity of a wetland (Macfarlane *et al*., 2009). Wetland health is defined as a measure of the deviation of wetland structure and function from the wetland's natural reference condition. This approach not only provides an indication of hydrological, geomorphological and vegetation health, but also highlights the key causes of wetland degradation. The WET-Health technique is therefore designed to both direct and monitor the effects of management interventions on wetland habitats. This tool should be very useful in planning and monitoring and evaluating the success of individual projects. The Wet-Health assessment allows for the assessment of each hydrogeomorphic (HGM) unit within a wetland to be assessed individually and provides an overall PES. There are many HGM units in each wetland complex (Table 6.1).

The PES for each of the wetland complex was determined using the updated methodology and ranking system by DWAF (2005) and DWAF (2007) for EIA in September 2006 and for assessments from March 2009 – August 2012 and, the WET-Health methodology ranking system by Macfarlane *et al*., (2009) for assessments from December 2012 – December 2014. For the EIA, high, moderate and low priority classes were determined from priority maps and interpreted according to the the DWAF (2005) ranking system (Table 6.2). For May 2014, August 2014 and December 2014 individual average hydrological, geomorphological and vegetation scores were determined based on the categories assigned in the assessments. These were allocated equal weighting to determine the PES.

Table 6.2: Impact scores and categories of Present Status used for describing the integrity of wetlands (Adapted from DWA (2005) and Macfarlane *et al*., (2009).

DWA(2005)		Macfarlane(2009)	Description	Health Category
Sep-06	Mar2009-Aug2012	Dec 2012-Dec2014		
	>4	0-0.9	Unmodified, natural	A
High	>3 and ≤4	1-1.9	Largely natural with few modifications. A slight change in ecosystem processes is decernible and a small loss of natural habitat and biota may have taken place.	B
	>2.5 and ≤3	2-3.9	Moderately modified. A moderate change in ecosystem processes and loss of natural habitats has taken place but the natural habitat remains predominantly intact.	C
Moderate	≤2.5 and >1.5	4-5.9	Largely modified. A large change in ecosystem processes and loss of natural habitat and biota has occurred.	D
Low	>0 and ≤1.5	6-7.9	The change in ecosystem processes and loss of natural habitat and biota is great but some remaining natural habitat features are still recognisable.	E
	0	8-10	Modifications have reached a critical level and the ecosystem processes have been modified completely with an almost complete loss od natural habitat and biota.	F

The PES was determined using the Wet-Health method which is based on the perceived changes in hydrology, vegetation and geomorphology in comparison to relative reference conditions.

A Level 1 assessment was undertaken. The Level 1 assessment is designed for use when many wetlands need to be assessed over a broad geographical area, whereas the Level 2 assessment is for a single wetland. The purpose of including the Wet-Health assessment was to quantify the extent of current land-use within the wetlands included in the study area. In most cases a particular land-use or impact is expressed as an overall percentage and this allows for the monitoring of changes (increases) in land-use or alterations over an extended period of time.

The WET-Health technique attempts to assess hydrological, geomorphological and vegetation health in three separate modules. Hydrology is defined in this context as the distribution and movement of water through a wetland and its soils, and focuses on changes in water inputs as a result of changes in catchment activities and characteristics that affect water supply and its timing, as well as on modifications within the wetland that alter the water distribution and retention patterns within the wetland. Geomorphology is defined in this context as the distribution and retention patterns of sediment within the wetland, and focuses on evaluating current geomorphic health through the presence of indicators of excessive sediment inputs and/or losses for clastic (minerogenic) and organic sediment (peat). Vegetation is defined in this context as the vegetation structural and compositional state, and evaluates changes in vegetation composition and structure as a consequence of current and historic onsite transformation and/or disturbance.

Each of these modules broadly follows a similar approach. Prior to assessment, the wetland is divided into hydrogeomorphic (HGM) units and their associated catchments. These are analysed separately for hydrological, geomorphological and vegetation health based on extent, intensity and magnitude of impact. This is translated into a health score. The approach is as follows: The extent of impact is measured as the proportion of a wetland and/or its catchment that is affected by an activity. Extent is expressed as a percentage. The intensity of impact is estimated by evaluating the degree of alteration that results from a given activity. The magnitude of impact for individual activities is the product of extent and intensity.

The magnitudes of individual activities in each HGM unit are combined in a structured and transparent way to calculate the overall impact of all activities that affect hydrological, geomorphological or vegetation health. Present State health categories, on an impact score scale of 1- 6 (or health category A-F), are as follows: natural, largely natural, moderately modified, largely modified, extensively modified, and critically modified.

6.2.3 Ecological importance and sensitivity

The Ecological Importance and Sensitivity (EIS) assessment was conducted according to the guidelines as described in the “Resource Directed Measures for Protection of Water Resources. Volume 4. Wetland Ecosystems” (DWAF, 1999). Here the DWAF defines “ecological importance” of a water resource as an expression of its importance to the maintenance of ecological diversity and function on local and wider scales. “Ecological sensitivity”, according to the DWAF (1999), refers to the system’s ability to resist disturbance and its capability to recover from disturbance once it has occurred. This was done in order to determine the ecological state of the wetlands and to provide an indication of the conservation value and sensitivity of the wetlands in the study area. In the method outlined by the DWAF a series primary and modifying determinants are used to determine the EIS of the wetland unit (Table 6.3).

Table 6.3: Score sheet for determining ecological importance and sensitivity (DWAF 1999).

Determinant
Primary Determinants
Rare and endangered species
Species/taxon richness
Diversity of Habitat types or features
Migration route/breeding and feeding site for wetland species
Sensitivity to changes in the natural hydrological regime
Sensitivity to water quality changes
Flood storage, energy dissipation and particulate/element removal
Modifying Determinants
Protected status
Ecological integrity

The series of determinants for EIS are assessed on a scale of 0 to 4, where 0 indicates no importance and 4 indicates very high importance. The median of the determinants is used to assign the Ecological Management Class (EMC) for the wetland complex (Table 6.4).

Table 6.4: Ecological importance and sensitivity categories. showing interpretation of median scores for biotic and habitat determinants (DWAF, 1999).

Ecological Importance and Sensitivity categories		Range of Median	Recommended Ecological Management Class
Very high	Wetlands that are considered ecologically important and sensitive on a national or even international level. The biodiversity of these wetlands is usually very sensitive to flow and habitat modifications. They play a major role in moderating the quantity and quality of water of major rivers.	>3 and ≤4	A
High	Wetlands that are considered to be ecologically important and sensitive. The biodiversity of these wetlands may be sensitive to flow and habitat modifications. They play a role in moderating the quantity and quality of water of major rivers.	>2 and ≤3	B
Moderate	Wetlands that are considered to be ecologically important and sensitive on a provincial or local scale. The biodiversity of these wetlands is not usually sensitive to flow and habitat modifications. They play a small role in moderating the quantity and quality of water of major rivers.	>1 and ≤2	C
Low/marginal	Wetlands that are not ecologically important and sensitive at any scale. The biodiversity of these wetlands is ubiquitous and not sensitive to flow and habitat modifications. They play an insignificant role in moderating the quantity and quality of water of major rivers.	>0 and ≤1	D

EIS was undertaken at Wetland complex 1 and 2 in March 2009. It was undertaken at Wetland complex 1, 2 and 3 in August 2012. In the December 2014 survey, a desktop study consisting of several annual Kusile wetland reports, topographical maps of the study area were used to undertake the EIS assessment.

6.2.4 Ecosystem services

Ecosystem services data was carried out in September 2006 and December 2014. The September 2006 assessment of the ecosystem services was conducted according to the guidelines as described by Kotze *et al*., (2005). A Level 2 assessment was undertaken which examines and rates the following services: flood attenuation; stream flow regulation; sediment trapping; phosphate trapping; nitrate removal; toxicant removal; erosion control; carbon storage; maintenance of biodiversity; water supply for human use; natural resources; cultivated foods; cultural significance; tourism and recreation; education and research. This method provides a scoring system for establishing wetland ecosystem services. It enables one to make relative comparisons of systems based on a logical framework that measures the likelihood that a wetland is able to perform certain functions. The characteristics were scored according to the following general levels of services provided (Table 6.5):

Table 6.5: Level of service ratings.

Score	Services Rating
0	Low
1	Moderately Low
2	Intermediate
3	Moderately High
4	High

The Ecosystem Services assessment in December 2014 was carried out using several annual Kusile wetland reports, topographical maps of the study areas and the WET-EcoServices Level 2 assessment, Kotze *et al*., (2009).

6.2.5 Soil erosion and sediment deposition

The visual assessment approach adopted by Golder Associates Africa Project (2012) is also included in this study as referenced in Chapter 3. During the visual assessment the wetlands at each sampling site was evaluated for signs of recent soil erosion and sediment deposition. The scoring system is represented in Chapter 4, Table 4.6.

6.3 Results and Discussion

The wetlands and water resources of the Wilge River sub-catchment B20F area are dominated by the Wilge River that drains from south to north. This area is generally flat to gently undulating which is underlain by Pretoria shale and quartzite capped on the high ground by Dwyka tillite and shale. The entire sequence has been intruded by a dolerite sill (Kok *et al.*, 2013). The deepest, most well-drained soils occur within the south west of the study area, while shallower soils with more impeded drainage, and thus more conducive to wetland formation, occur within eastern half of the study area. Extensive shallow, rocky soils also occur on site, specifically in the north and north-west of the site. Wetland Complex 1 is located in the northern tributary of the Wilge River. According to Nel *et al.*, 2011 wetland complex 1 has 70% hillslope seeps, 10% unchannelled valley bottomed wetland, 5% channelled valley bottomed wetlands and 5% pans. Comprehensive studies by Kassier (2013) show that wetland complex 1 has 85% hillslope seeps, 10% floodplain wetlands, and 5% unchannelled valley bottom wetlands

(Table 6.1). Wetland complex 1 was found to be in a largely modified ecological state (D) in September 2006. In March 2009 wetland complex 1 was found to be in a largely natural state with few modifications but with some loss of natural habitats (B) (Table 6.6). A slight deterioration to moderately modified state C was noted in July 2009 and then to D in March 2010 was noted. This was maintained till December 2011. The ecological state improved to C in August and December 2012 and then to B in May, August and November 2014

Wetland complex 2 is found on the Holfonteinspruit and the Klipfonfeinspruit water resources. According to Nel *et al.*, 2011 wetland complex 2 has 80% channelled valley bottomed wetlands and 20% unchannelled valley bottomed wetlands (Table 6.1). Comprehensive studies by Kassier (2013) show that wetland complex 2 has 95% hillslope wetlands and 5% floodplain wetlands. At its widest (the confluence with the Klipfonteinspruit), the floodplain is more than 600 m across. A number of smaller tributaries of the Wilge River drain across the study area roughly from east to west. Wetland complex 2 was found to be in a largely modified ecological state (D) in September 2006. In March 2009 and July 2009 wetland complex 2 was found to be in a largely natural state with few modifications but with some loss of natural habitats (B). A deterioration to largely modified state D was noted in March 2010. This was maintained till December 2011. The ecological state improved to B in August 2011 and deteriorated to D in December 2012 and then improved again to B in May, August and November 2014, inspite of part of the wetlands being lost due to Kusile power station construction which started in April 2008.

Wetland complex 3 is located in the south west of the Wilge River sub-catchment B20F area on the Klipspruit River. Wetland complex 3 according to Nel *et al.*, 2011 has 70% unchannelled valley bottom wetlands, 15% hillslope seeps, 10% channelled valley bottom wetlands and 5% pans (Table 6.1). Wetland complex 3 was found to be in a largely modified ecological state (D) in September 2006 till March 2011. In June 2011 and September 2011 the ecological state improved to C and then resumed to D in December 2011 again. The ecological state improved to A in August 2012 and to B in May, August and November 2014

6.3.1 Present Ecological Status WET-Health Assessment

Table 6.6: Present Ecological Status for Wilge River sub-catchment B20F area Wetland Complex 1, 2 and 3 from September 2006 to November 2014.

Wetland Complex	1	2	3
Sites	SW9 and SW4	SW5, SW7 and SW1	S4
Sep-06	D	D	D
Mar-09	B	B	
Jul-09	C	B	
Mar-10	D	D	D
Jun-10	D	D	D
Sep-10	D	D	D
Dec-10	D	D	D
Mar-11	D	D	D
Jun-11	D	D	C
Sep-11	D	D	C
Dec-11	D	D	D
Aug-12	C	B	A
Dec-12	C	D	
May-14	B	B	B
Aug-14	B	B	B
Nov-14	B	B	B

Wetland conditions were found to have deteriorated from March 2010 to December 2011 in the wetland complexes assessed, and can be attributed to the changing land-use i.e. increase in construction activities, mining activities and agricultural activities. Improvements to the wetland ecological status from August 2012 to December 2014 can be due to a decline in construction activities and an increase in wetland rehabilitation efforts implemented.

6.3.2 Ecological importance and sensitivity

Ecological Importance and Sensitivity is a concept introduced in the reserve methodology to evaluate a wetland in terms of:

- Ecological Importance;
- Hydrological Functions;
- Direct Human Benefits.

Most of the wetlands rated within the very high EIS category (A) are hillslope seepage wetlands that are still characterised by primary vegetation and are located within catchments consisting mostly of natural grasslands. The pan located to the south west of the power station is also rated as highly important and sensitivity due to the role it plays in supporting Red Data bird species especially. Wetland complex 1 and 2 were rated as moderate EIS category (C) in the March 2009 surveys (Table 6.7). The most significant and important wetland system and water resource within the study area is considered to be the Wilge River with its associated wetland systems. These wetlands make up 80% of the wetlands rated as being of high importance and sensitivity Wetland complex 1 was rated as low/marginal EIS category (D) in the August 2012 survey; these are mainly hillslope seepage wetlands that have been impacted by previous cultivation. Wetland complex 2 was rated as being high (EIS) category (A) in the August 2012 and December 2014 surveys. Wetland complex 3 was rated as being high (EIS) category (B) in the August 2012 and December 2014 surveys.

Table 6.7: Ecological Importance Sensitivity for Wilge River sub-catchment B20F area Wetland complex 1, 2 and 3 for March 2009, August 2012 and December 2014.

Date	Parameter	Wetland complex 1	Wetland complex 2	Wetland complex 3
		SW4 & SW9	SW5/SW7 & SW1	S4
Mar-09	MEDIAN	2.00	2.00	
	EIS	C	C	
Aug-12	MEDIAN	1.00	4.00	3.00
	EIS	D	A	B
Dec-14	MEDIAN	2.00	4.00	3.00
	EIS	C	A	B

The results are probably a fair reflection, given the extent of land cover transformation that has occurred across the surrounding landscape. It should be noted that most of these criteria are linked to the intactness and heterogeneity of the vegetation communities within the systems

6.3.3 Ecosystem services

Numerous functions are typically attributed to wetlands, which include biodiversity support, nutrient removal (and more specifically nitrate removal), sediment trapping (and associated with this is the trapping of phosphates bound to iron as a component of the sediment), stream flow augmentation, flood attenuation, trapping of pollutants and erosion control. Many of these functions attributed to wetlands are wetland type specific and can be linked to the position of wetlands in the landscape as well as to the way in which water enters and flows through the wetland. Thus, not all wetlands can be expected to perform all functions, or to perform these functions with the same efficiency. Despite this, certain assumptions on the functions supported by wetlands can be made, based on the hydro-geomorphic wetland classification system which classifies wetlands according to the way that water moves through the wetland as well as the position of the wetland within the landscape.

The difference in Ecosystem services from September 2006 to December 2014 was found to be a decrease in nitrate removal, decrease in erosion control, decrease in carbon storage, increase in natural resources, and increase in education and research at wetland complex 1; an increase in flood attenuation, decrease in erosion control, increase in maintenance of biodiversity, and increase in education and research at wetland complex 2; a decrease in sediment trapping, increase in nitrate removal, decrease in toxicant removal, and increase in education and research at wetland complex 2 (Table 6.8, Figure 6.2 and Figure 6.3).

Table 6.8: Ecosystem Services scores for Wilge River sub-catchment B20F area Wetland Complex 1, 2 and 3 for September 2006 and December 2014.

Services	Wetland Complex 1		Wetland Complex 2		Wetland Complex 3	
	Sep-06	Dec-14	Sep-06	Dec-14	Sep-06	Dec-14
Flood attenuation	1.8	1.9	1.8	2.1	1.8	1.9
Streamflow regulation	2.5	2.0	2.5	2.0	2.8	2.4
Sediment trapping	2.2	2.7	2.9	3.0	3.0	1.9
Phosphate trapping	2.1	2.4	2.3	2.4	2.2	1.8
Nitrate removal	3.3	2.3	2.4	2.3	1.7	2.3
Toxicant removal	2.5	2.4	2.4	2.6	2.1	1.5
Erosion control	2.4	1.4	2.3	1.4	2.2	2.0
Carbon storage	2.7	1.7	1.7	1.7	1.7	1.7
Maintenance of biodiversity	3.5	3.8	1.8	4.0	1.8	1.9
Water supply for human use	1.6	1.5	1.7	1.5	1.4	1.6
Natural resources	0.6	1.6	1.1	1.6	2.0	1.6
Cultivated foods	0.0	0.0	1.0	0.0	1.0	0.4
Cultural significance	0.0	0.0	0.0	0.0	0.0	0.0
Tourism and recreation	0.3	0.6	1.1	0.6	1.0	0.9
Education and research	1.0	2.0	0.9	2.0	1.5	3.0
Threats	2.0	2.0	3.0	3.0	0.0	0.0
Opportunities	0.0	0.0	1.0	1.0	0.0	0.0

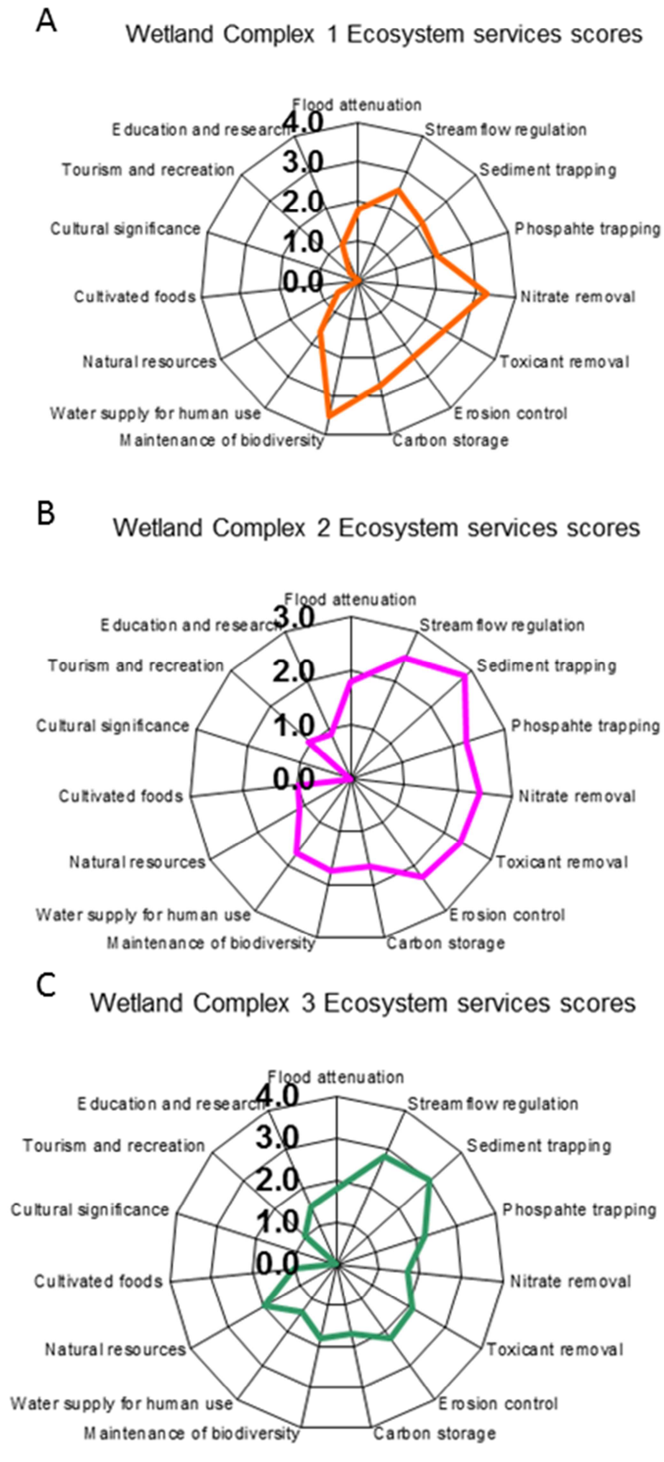

Figure 6.2: Graphical description of ecosystem services data for Wilge River sub-catchment B20F area Wetlands Complex 1, and 3 for September 2006 (A, B and C).

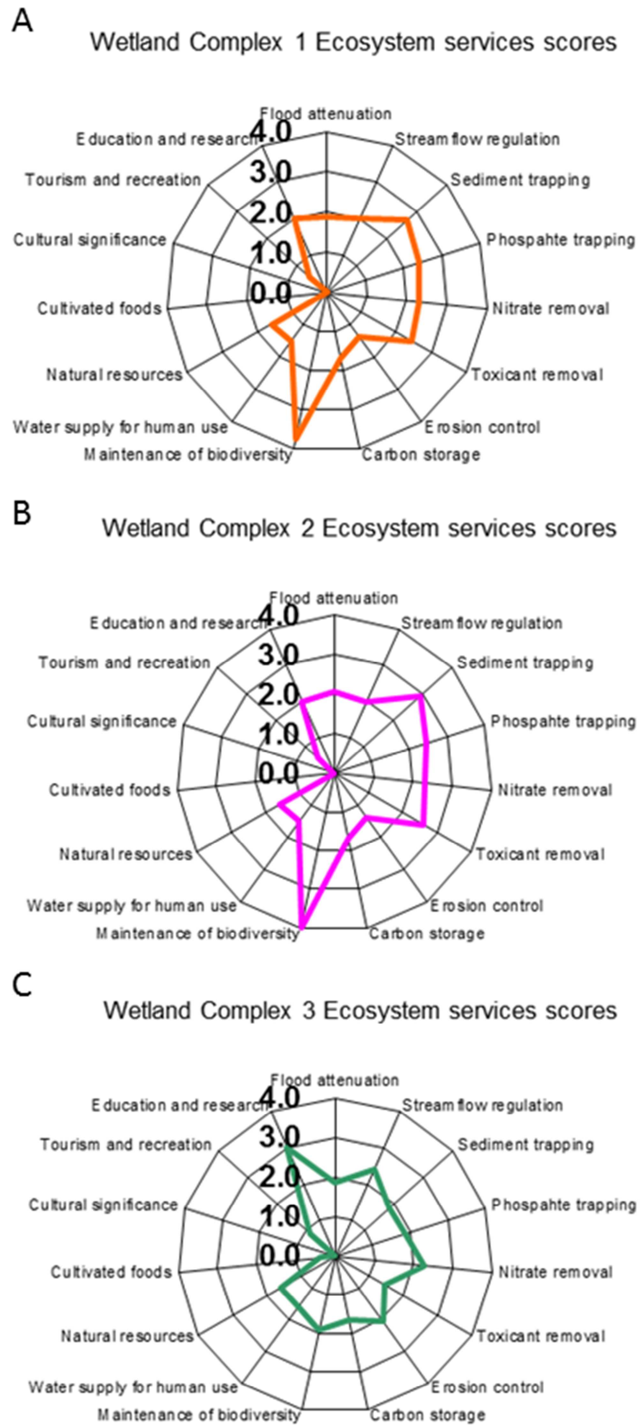

Figure 6.3: Graphical description of ecosystem services data for Wilge River sub-catchment B20F area Wetlands Complex 1, and 3 for December 2014 (A, B and C).

Hillslope seepage wetlands dominate in wetland complex 1 and 2. Hillslope seepage wetlands are mostly maintained by shallow sub-surface interflow, derived from rainwater. Rainfall infiltrates the soil profile, percolates through the soil until it reaches an impermeable layer (e.g. a plinthic horizon or the underlying sandstone), and then percolates laterally through the soil profile along the aquitard (resulting in the formation of a perched water table). The hillslope seepage wetlands are merely the surface expression of this perched water table in those areas where a shallow soil profile results in the perched water table leading to saturation of the profile within 50cm of the soil surface. The importance of individual seepage wetlands in temporarily storing and then discharging flows to downslope wetlands (flow regulation) varies and depends on a number of factors (Reppert et al., 1979).

Generally, seepage wetlands are associated with springs and when located adjacent to terrestrial areas characterised by deep, well-drained soils are more likely to play an important role in flow regulation than seepage wetlands where the wetland and catchment are characterised by shallower soils. Such seepage wetlands are most likely maintained by direct rainfall and lose most of their water to evapotranspiration, and surface run-off during large storm events. Hillslope seeps can support conditions that facilitate both sulphate and nitrate reduction as interflow emerges through the organically rich wetland soil profile, and are thus thought to contribute to water quality improvement and/or the provision of high quality water. The greatest importance of the hillslope seepage wetlands on site is thus taken to be the cleaning of the water as it moves through the hillslope and enters the adjacent valley bottom wetlands. As hillslope seepage wetlands, for the most part, are dependent on the presence of an aquiclude, either a hard or soft plinthic horizon, they are not generally regarded as significant sites for groundwater recharge (Parsons, 2004).

However, by retaining water in the landscape and then slowly releasing this water into adjacent valley bottom or floodplain wetlands, some hillslope seepage wetlands can contribute to stream flow augmentation, especially during the rainy season and early dry season. From an overall water yield perspective there is evidence that seepage wetlands contribute to water loss. The longer the water is retained on or near the surface the more likely it is to be lost through evapo-transpiration (McCartney, 2000, from Kassier, 2013).

Hillslope seepage wetlands are not generally considered to play an important role in flood attenuation, though early in the season, when still dry, the seeps have some capacity to retain water and thus reduce surface run-off. Later in the rainy season when the wetland soils are typically saturated, infiltration will decrease and surface run-off increase. Further flood attenuation can be provided by the surface roughness of the wetland vegetation: the greater the surface roughness of a wetland, the greater is the frictional resistance offered to the flow of water and the more effective the wetland will be in attenuating floods (Reppert et al., 1979). In terms of the hillslope seepage wetlands on site, the surface roughness is taken to be moderately low, given that most of the seepage wetlands are either cultivated of characterised by typical grassland vegetation, thus offering only slight resistance to flow.

The linear nature of valley bottom wetlands within the landscape and their connectivity to the larger drainage system provide the opportunity for these wetlands to play an important role as an ecological corridor allowing the movement and migration of fauna and flora between remaining natural areas within the landscape. Although modified in certain respects, the wetlands still provide a natural refuge for biodiversity, and within the study area and surroundings, the large valley bottom wetlands with associated footslope seepage wetlands represent the most significant extent of remaining natural vegetation, further enhancing their importance from a biodiversity support function.

Channelled valley bottom wetlands, found in wetland complex 1, 2 and 3, indicate through the erosion of a channel through the wetland, that sediment trapping is not always an important function of these wetlands, except where regular overtopping of the channel occurs and flows spread across the full width of the wetland. Under low and medium flows, transport of sediment through, and out, of the system are more likely to be the dominant processes. Erosion may be both vertical and/or lateral and reflect the attempts of the stream to reach equilibrium with the imposed hydrology. A number of the valley bottom wetland systems are significantly eroded (e.g. Sampling site SW5 at the Holfonteinspruit, a tributary to the Klipfonteinspruit), probably as a result of changes in land-use (conversion to cultivated fields) and altered hydrology due to farm road crossings and dams. The Klipfonteinspruit is also currently considered a significant erosion risk due to the increased flows this system is receiving via stormwater from the Kusile construction site. At the same time however, the wetland is currently likely playing an important role in sediment trapping as runoff from the Kusile construction site is extremely sediment rich. However, as flows become more channel bound through further incision and lateral erosion of the channel, the ability of the wetlands to trap sediments decreases.

From a functional perspective channelled valley bottom wetlands can play a role in flood attenuation when it flows over the top of the channel bank and spread out over a greater width, with the surface roughness provided by the vegetation further slowing down the flood flows. These wetlands are considered to play only a minor role in the improvement of water quality given the short contact period between the water and the soil and vegetation within the wetland.

Un-channelled valley bottom wetlands found mostly at wetland complex 3, reflect conditions where surface flow velocities are such that they do not, under existing flow conditions, have sufficient energy to transport sediment to the extent that a channel is formed. In addition to the biodiversity associated with these systems it is expected that they play an important role in retaining water in the landscape as well as in contributing to influencing water quality through, for example, mineralisation of rain water. These wetlands could be seen to play an important role in nutrient removal, including ammonia, through adsorption onto clay particles. The large size of the unchannelled valley bottom wetland associated with the Bronkhorstspruit suggests that this wetland plays an important role in flood attenuation – the temporary storage of flood waters within the wetland.

Given the position of the pans within the landscape, which is usually isolated from any stream channels, the opportunity for pans to attenuate floods is fairly limited, though some run-off is stored in pans. In the cases where pans are linked to the drainage network via seep zones, the function of flood attenuation is somewhat elevated. Pans are also not considered important for sediment trapping, as many pans are formed through the removal of sediment by wind when the pan basins are dry. Some precipitation of minerals and de-nitrification is expected to take place within pans, which contributes to improving water quality. Some of the accumulated salts and nutrients can however be exported out of the system and deposited on the surrounding slopes by wind during dry periods. An important function usually performed by pans is the support of faunal and floral biodiversity. Within the study area however, the small size of most of the pans, together with their seasonal nature and the disturbed vegetation, the biodiversity support of these pans individually is expected to be limited. All of the pans are seasonal systems, though the differences in pan basin size and depth, as well as catchment size and catchment soil characteristics results in pans that fill up and drain at different rates and times. As a consequence a great diversity of habitat is provided by the pans on site and in the surrounding area, and though they are all seasonal systems, the differing hydroperiods result in the fact that at least some of the pans are likely to have water at any one time.

6.3.4 Soil erosion and sediment deposition

Refer to Chapter 4, Table 4.9 for Soil erosion and Table 4.10 for Sedimentation Deposition resuts. Soil erosion scores were found to remain the same or increase at all sample sites from December 2012 to November 2014. Sampling sites downstream on the Wilge River moved from a non-eroded state to a moderately eroded state. Sampling site upstream on the Wilge River remained in a moderately eroded state. On wetland complex 1, sampling site SW9 on the unnamed tributary of the Wilge River remained in a moderately eroded state, while sampling site SW4 on the unnamed tributary of the Wilge River changed from a slightly eroded state to a severely eroded state from December 2012 to November 2014. On wetland complex 2, sampling site SW7 on the unnamed tributary of the Klipfonteinspruit remained in a severely eroded state from December 2012 to November 2014. Sampling site SW1 on the Klipfonteinspruit remained in a largely actively eroded state from December 2012 to November 2014 while sampling site SW5 on the Klipfonteinspruit were found to be steadily eroding from December 2012 to August 2014 with improvement noted in November 2014. On wetland complex 3 on the Klipspruit River was found to change from a non-eroded to a moderately eroded state.

All sampling sites showed an increase in soil deposition with sampling sites: SW5, SW4, SW9 and S4 indicating severe soil deposition. Sampling site SW1 showed largely active soil deposition occurring with large plants extending through the sediment layer in November 2014. The rest of the sample sites had all indicated moderate soil deposition by November 2014.

Excess sediments can in itself cause deterioration of water quality and impact on river ecology but it can also be a means by which pollutants such as metals, and excess nutrient be transported away because of their adsorption onto the sediment. The poor land management practices and changes in land-use have caused erosion, siltation of rivers, unstable river banks and beds and with this mobilisation of sediment associated with pollutants. Sediment and nutrient accumulations likely reflect anthropogenic disturbance within the surrounding catchment. Craft and Casey's (2000) findings suggest that the degree of anthropogenic disturbance within the surrounding watershed regulates wetland sediment, organic C, and N accumulation. Phosphorus accumulation also is greater in floodplain wetlands that have large catchments containing fine textured (clay) sediments that are co-deposited with phosphorous. Better wetland management practices are suggested at sample sites SW5. SW4, SW7, SW1, SW9 and S4 i.e. wetland complex 1, 2 and 3, that have evidence of soil erosion and sedimentation deposition.

6.4 Conclusions

Wetlands can be viewed as complex temporal and spatial mosaics of habitats with distinct structural and functional characteristics. An important aspect of wetland management is to identify ecological risks affecting the area, estimating potential hazards or threats posed by stressors (chemical, physical, or biological) to biotic and/or abiotic components of the wetland (Malekmohammadi and Rahimi Blouchi, 2014).

WET-health is a tool to help planners, stakeholders, and decision makers to understand the linkages between anthropogenic impacts and the ecological condition of a wetland, indicating a novel approach that provides only a snapshot of current wetland conditions or health state (Beuel *et al*., 2016). The majority of wetlands in the study area still considered to be in a natural to largely natural state are small hillslope seepage wetlands located along footslopes of the various drainage lines of the area, especially the wetland system to the north of the power station. In most cases these seepage wetlands are associated with areas of shallow or rocky soils that cannot be cultivated and are thus associated with primary grasslands, which in their own right are of high biodiversity importance. These wetlands are however currently under considerable threat from heavy livestock grazing and trampling, and are likely to deteriorate and change to lower PES categories unless a grazing management plan is compiled by a suitably qualified expert and strictly implemented. The ecological state at wetland complex 1, 2 and 3 all improved to B in May, August and November 2014, this includes wetland complex 2, inspite of part of the wetlands being lost due to Kusile power station construction which started in April 2008.

Wetland conditions were found to have deteriorated from March 2010 to December 2011 in the wetland complexes assessed, and can be attributed to the changing land-use. The modification that the majority of wetlands on site have undergone is as a direct consequence of agricultural activities within the wetlands and their catchments, as well as infrastructure developments such as construction activities and road crossings that alter flow characteristics of the wetlands. Agriculture, specifically cultivation, has a severe impact on the natural vegetation of the wetland areas through the complete removal of vegetation. Such a change in vegetation is considered permanent, even where grasslands are allowed to re-establish on previously cultivated fields, as the species composition is unlikely to ever resemble the unmodified condition again (Kotze *et al*., 2012).

The hydrology of wetland systems that have been cultivated, especially hillslope seepage wetlands that are maintained by infiltration and sub-surface seepage of rainwater, is however often still largely intact, and these systems can still perform important functions in terms of water quality maintenance and flow regulation, as well as supporting productivity. The existing activities within the study area have had significant impacts on the wetland systems, most notably the Klipfonteinspruit which receives a large portion of the stormwater generated by the construction site and discharges from the New Largo Mining Area. In addition to sedimentation and increased turbidity levels being of concern in these systems, the changes in runoff volumes and velocities brought about by the increase on the hardened surfaces pose a risk to these wetlands. Improvements to the wetland ecological status from August 2012 to December 2014 can be due to a deterioration in construction activities and an increase in wetland rehabilitation efforts implemented.

Owing to the high importance of water quality enhancement functions provided by wetlands, it is important to ensure that the wetland services are maintained. As such, a series of seven wetlands in the upper reached of the Wilge River sub-catchment, providing these services, have been prioritised and selected for Resource Quality

Objective (RQO) determination (DWA, 2014). A number of wetlands were also flagged as having a high biodiversity maintenance function. A sub-set of wetlands were therefore also selected to monitor changes to wetlands of high biodiversity value. Wetland management should also be regarded as a key focus in this catchment if wetland protection targets are to be achieved and functional characteristics maintained.

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CHAPTER 7: USING THE RELATIVE RISK MODEL FOR A REGIONAL-SCALE ECOLOGICAL RISK ASSESSMENT OF THE WILGE RIVER SUB-CATCHMENT B20F AREA IN THE UPPER OLIFANTS RIVER CATCHMENT

7.1 Introduction

Ecological services is defined as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life’ (Tilman, 2003). Ecological services are costly and often impossible to replace when aquatic ecosystems are degraded. Yet today, aquatic ecosystems are being severely altered or destroyed at a greater rate than at any other time in human history, and far faster than they are being restored (Tilman, 2003). To ensure the long term sustainability of our limited ecosystem services, resource managers utilise different ways in order to establish the balance between the use and protection of ecosystems as stated in the National Water Act, (Act No 36 of 1998). Risk assessments assign magnitude and probabilities to hazards resulting from anthropogenic activities, or natural catastrophes that have a potential adverse effect on ecosystems (Suter, 1993). The formulation of risk results from the existence of a hazard and the related uncertainty of its effects to components of ecosystems. An Ecological Risk Assessment (ERA) is a structured approach that describes, explains and organises scientific facts, laws and relationships, thereby providing a sound basis to develop sufficient protection measures for the environment, which facilitates the development of utilisation strategies for the environment (Moosa, 2001). Traditional ERA approaches have primarily addressed the potential risk of a single or a small number of chemicals impacting on a limited number of ecological endpoints (Claassen *et al*., 2001). The Regional-Scale Risk Assessment using the Relative Risk Model (RRM) is a form of ERA that is implemented on a large spatial scale and facilitates the consideration of multiple sources of multiple stressors affecting multiple endpoints, including the ecosystem dynamics and characteristics of the landscape that may affect the risk estimate (Landis and Wiegers, 1997). Although ERA guidelines have been developed for use in South Africa (Claassen et al., 2001; O’Brien, 2011), they have not been formally adopted into any water resource management strategies and as such are not being widely used (O’Brien, 2011).

From a South African perspective, the value of the Relative Risk Model (RRM) lies in its potential to be customised to address the threats of multiple sources of multiple stressors to local habitats and endpoints, thereby contributing towards the objective of integrated water resources management (IWRM) in South Africa (DWAF, 2004; O'Brien and Wepener, 2012). O'Brien and Wepener (2012) established a working RRM method for South African conditions. This adapted RRM approach, which has been applied to this study, allows for the assessment of multiple stressors in unique habitats on a spatial scale while allowing for the consideration of ecosystem structure and function dynamics which can contribute towards the management of local surface ecosystems in South Africa. These benefits include the establishment of a validated, structured methodology that is sensitive to the dynamics of individual case studies, relatively simple to apply, extremely informative, locally applicable and internationally comparable with other RRM assessments (O'Brien and Wepener, 2012). The RRM is a suitable water resources management tool that can address the risk assessment of multiple stressors in South African freshwater environments. O'Brien, (2011) showed that the RRM approach is a powerful predictive decision making tool that can be used to address a range of aquatic ecosystem management objectives.

The aim of this chapter is to collate the information from previous chapters and to demonstrate the use of Bayesian Networks (BNs) as a tool to perform a regional-scale, multiple-stressor ecological risk assessment in the Wilge River sub-catchment area in the Upper Olifants River catchment. This work should demonstrate the effect of water resource use scenarios to the socio-ecological system, contribute to achieving Resource Quality Objectives (RQO) and achieve a balance between use and protection.

7.2 The study area

The Olifants River WMA is of major national strategic and economic importance, contributing to a large portion of South Africa's Gross Domestic Product (DWA, 2013). The catchment transcends through a myriad of different ecoregions from the grasslands of the highveld ecoregion in Gauteng to the lowveld in Limpopo and Mpumalanga. A wide variety of ecosystem services exist in the catchment i.e., products (including mineral resources), regulatory services and various aesthetic services. These ecosystem services are being used by a wide variety of users including mining and industrial sector, agriculture sector, tourism and local informal and formal communities (DWA, 2013).

A large amount of coal mining, and industrial activities around the Wilge River, Bronkhorstspruit, Klein Olifants and Olifants Rivers, are the main contributors to poor in-stream and riparian habitat conditions where acid leachate from mines is a primary contributor to poor water quality and instream conditions (DWA, 2011) (Figure 7.1). At present these water quality effects are fairly limited in extent and confined to some specific streams. Poorly treated domestic waste water is causing an increase in nutrients and thereby a change in the trophic state of the dams in the upper catchment. Irrigation return flows also cause a rise in salinity levels downstream of irrigated areas (DWA, 2011).

Continued mining and power generation activities in the Olifants River catchment have led to a general acidification of the water system and the input and mobilisation of heavy metals (Dabrowski *et al*., 2014; Hobbs *et al*., 2008). Elevated levels of heavy metals in an aquatic system pose a significant threat to aquatic life and ultimately to human health, reaching considerable concentrations in plants and biota through bioaccumulation. As the Wilge River currently discharges into the Olifants River and significantly improves its water quality, future mining and development activities in the Wilge River catchment should be carefully managed and monitored to ensure sufficient flows of acceptable quality to prevent further deterioration of water quality in the Olifants River and downstream reservoirs (Dabrowski *et al*., 2014).

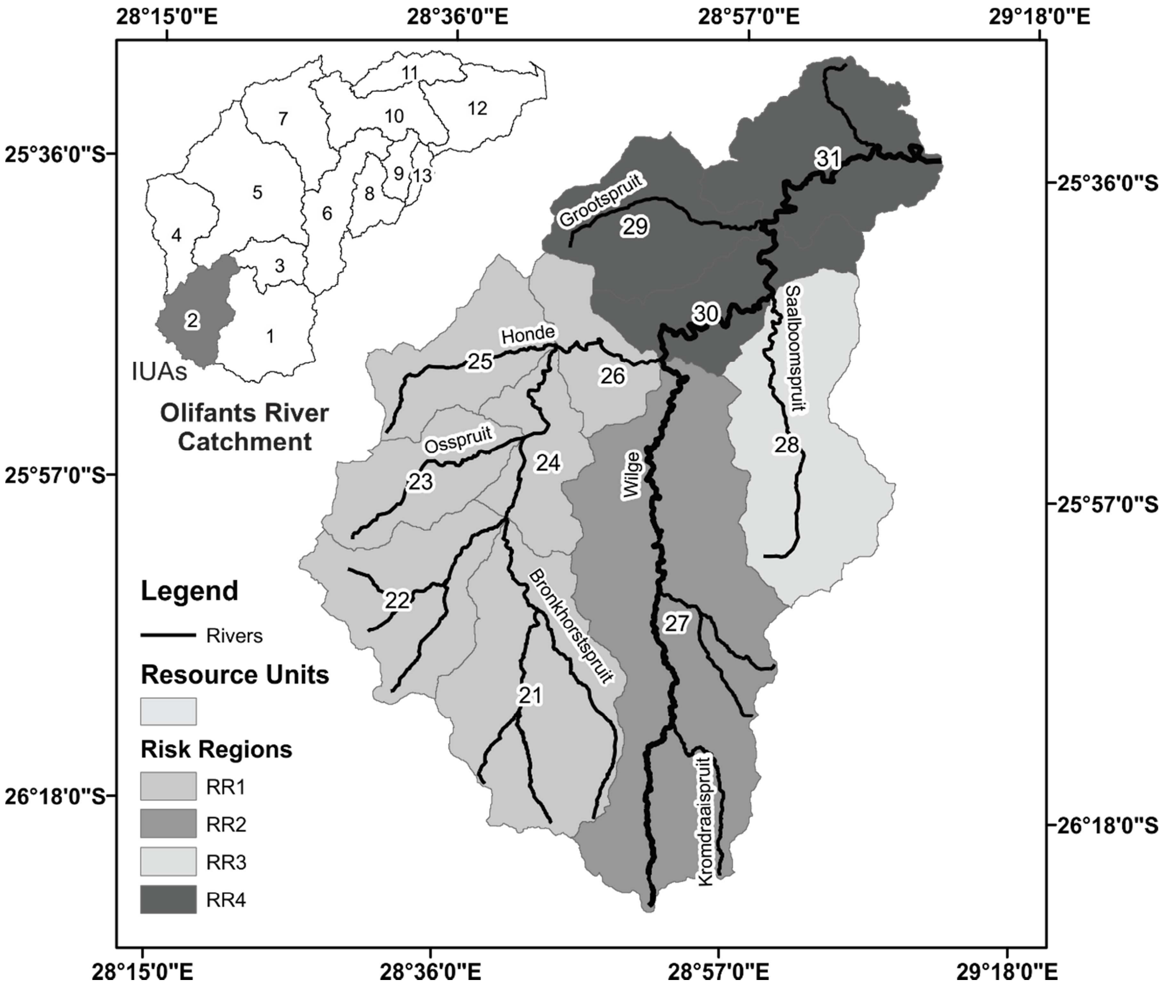

Figure 7.1: River Resource Units (21-31) delineated for the Olifants River WMA and Relative Risk areas delineated for the Wilge River sub-catchment.

7.3 Framework for Relative Risk Method Model

The ten procedural steps established for international and local application of the RRM including the use of Bayesian network modelling techniques has been implemented in this study (Landis, 2005, O'Brien and Wepener, 2012; Ayre and Landis, 2012) (Figure 7.2).

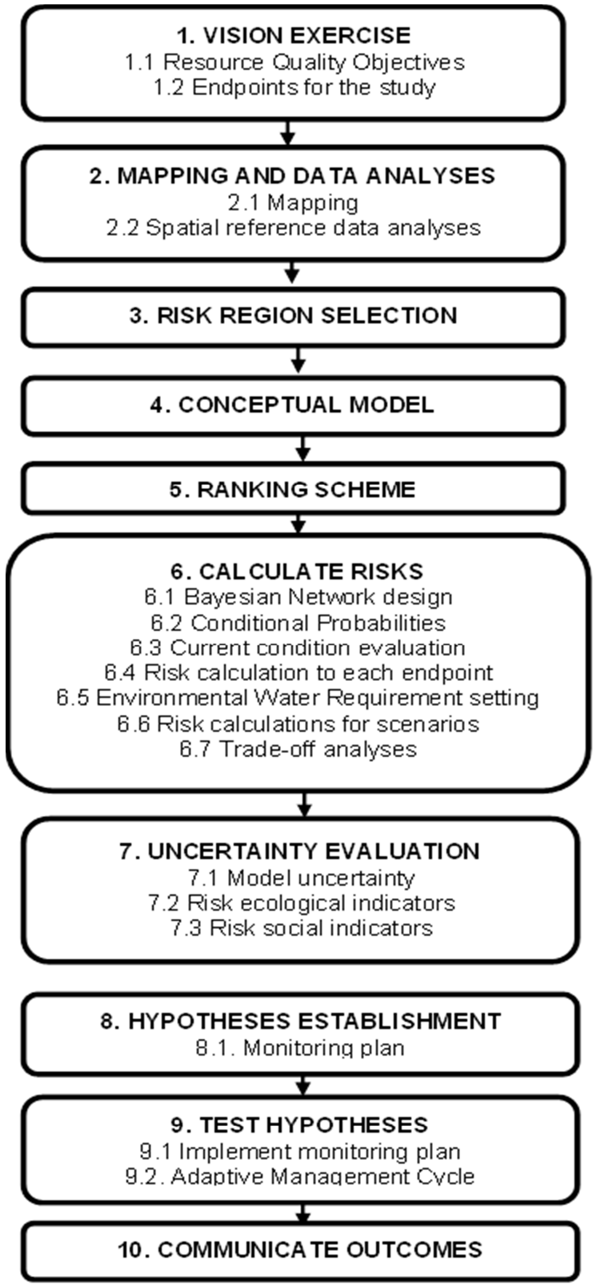

Figure 7.2: The ten procedural steps established for international and local application of the Relative Risk Method (O'Brien *et al*., 2017).

Step 1: Vision exercise

From 2012 the Department of Water Affairs initiated a Water Research Commission study to classify significant water resources in the Olifants River WMA (DWA, 2013). This involved the application of the Water Resource Classification (WRC) system to ensure that the water resources in the Olifants River catchment are protected and sustained while development can continue. The study resulted in the establishment of WRC Management Classes (MC) for 13 Integrated Units of Analyses (IUAs) in the catchment. The purpose of the MC once set, is to establish clear goals relating to the quantity and quality of the relevant water resource in order to facilitate a balance between protection and use of water resources (DWA, 2013). The resulting MCs can now be translated into RQOs that will specify the targets to maintain the MCs. The Wilge River catchment in the Olifants River WMA (IUA 2), consists of Resource Units (RU) 21-31 (Figure 7.1). RU31 is located at the base of the Wilge River catchment, and was selected by stakeholders to ensure that all use of the river ecosystems in IUA 2 could be regulated through RQOs selected for this RU. The rivers in the IUA are in a moderately modified state (category C) with less developed areas in the catchment. Impacts within the catchment are related to agriculture, dams and some mining.

The Wilge River sub-catchment was divided into 4 risk regions based on the Resource Unit Prioritisation Report (DWS, 2014). A total of 121 River Resource Units (RU) were delineated for the Olifants River WMA (Figure 7.1). The location of these River RUs and associated biophysical nodes is shown in Figure 7.1 and Table 7.1.

The IUA, biophysical node and corresponding river as well as the PES, REC (as required by the WRC) for each RU, and the management class for each respective IUA, is detailed in Table 7.1. This RU31 is located at the base of the Wilge River catchment in the Olifants River WMA (IUA 2), and was selected by stakeholders to ensure that all use of the river ecosystems in IUA 2 could be regulated through RQOs selected for this RU (DWS, 2014).

Table 7.1: Summary data for each River Resource Unit delineated in the Olifants River WMA2 - Wilge River catchment area assigned Class II for Integrated Unit of Analyses (IUA) (DWS, 2014).

RU number	River Name	PES	REC
RU21	Bronkhorstpruit (outlet of quaternary)	C	C
RU22	Koffiespruit (confluence with Bronkhorstspruit)	C	C
RU23	Osspruit (inflow to Bronkhorstspruit Dam)	D	D
RU24	Bronkhorstpruit (outlet from Bronkhorstspruit Dam)	C	C
RU25	Hondespruit (confluence with Bronkhorstspruit)	C	C
RU26	Bronkhorstpruit (confluence with Wilge)	C	C
RU27	Wilge (confluence with Bronkhorstspruit	C	C
RU28	Saalboomspruit (confluence with Wilge)	C	C
RU29	Grootspruit (confluence with Wilge)	C	C
RU30	Wilge (outlet of quaternary)	B	B
RU31	Wilge (EWR site – EWR4, outlet of IUA2) (existing)	C	C

Water quality, macroinvertebrates, toxicity, fish and wetland data were analysed in his study for Risk Region 2 (RU27) only. The ecological status for each parameter, for the latest assessments for each, were found to be water quality (B), macroinvertebrates (B), toxicity (B), fish (E) and wetlands (B). Only fish ecological integrity is below status of C as recommended by the DWS RQOs (DWS, 2014).

Step 2: Mapping and data analysis

The basis of the RRM is a conceptual framework that identifies sources of stressors, stressors, effects of stressors on receptors, and describes their interactions and the resulting impacts on endpoints at a regional scale in a single framework (Figure 7.3).

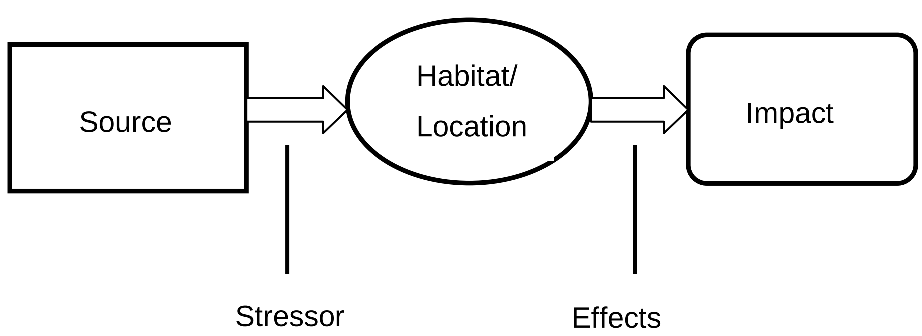

Figure 7.3: Interactions of the Relative Risk Model.

The study area is defined and described, and the locations of potential sources, habitats and impacts are identified and spatially referenced. In addition, source-stressor exposure and habitat/receptor to endpoint pathways/relationships are spatially referenced where possible (O'Brien and Wepener, 2012; Landis *et al*., 2016). Available data describing the ecosystem is to be reviewed and spatially referenced and the uncertainties associated with the availability and quality of data used in the assessment must be documented for evaluation in Step 7. O'Brien and Wepener (2012) provide an approach to delineate ecosystem types, the topological features of importance, the catchment and ecoregion boundaries, the land or water resource use scenarios and the pathways of stressors' exposure. This approach is used to direct the selection of risk regions for assessment. Land-use activities are shown per risk region in Figure 7.4 and Table 7.2.

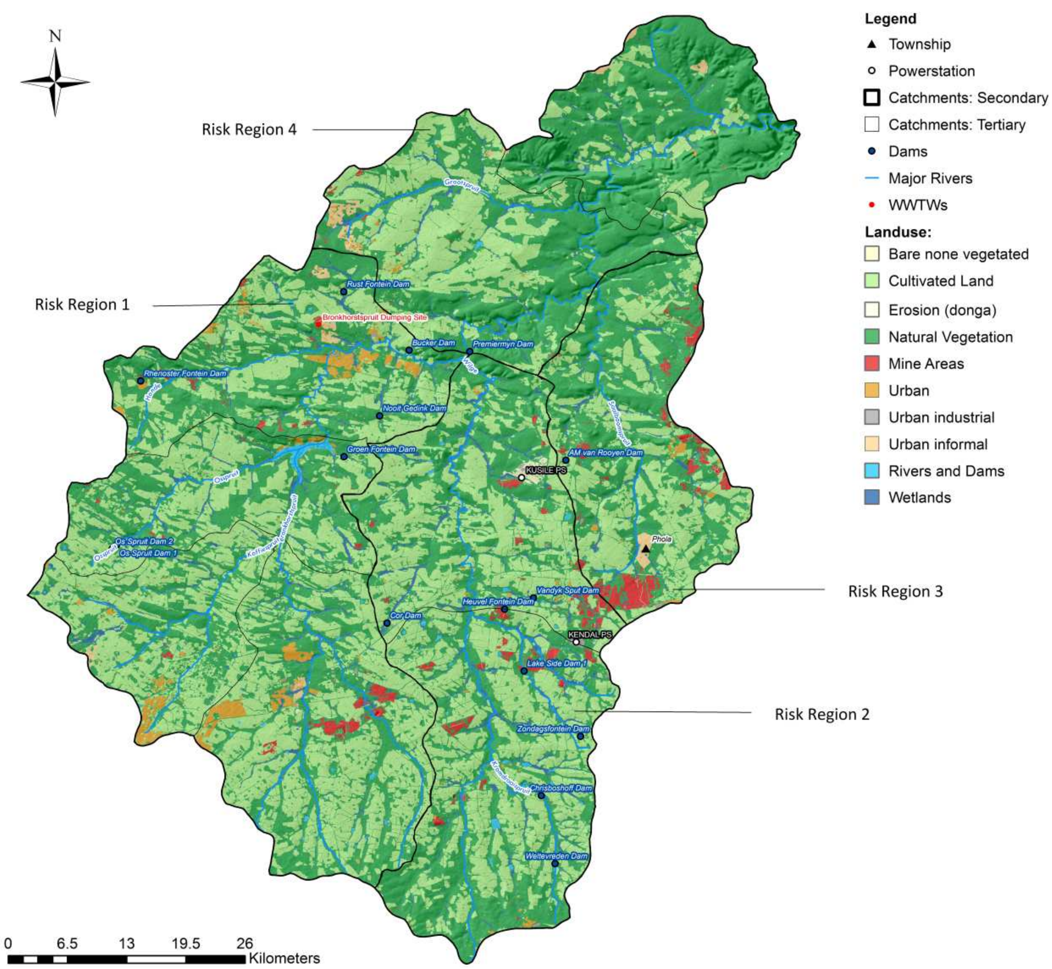

Figure 7.4: Land-use map of the Wilge River sub-catchment in the Upper Olifants River catchment.

Table 7.2: Land-use activities in the Wilge River sub-catchment per risk region.

Risk Regions	Risk Region 1 Bronkhorstspruit	Risk Region 2 Upper Wilge	Risk Region 3 Saalboomspruit	Risk Region 4 Lower Wilge
Land-use Impacts	Agriculture Mines Urban Development Municipality infrastructure Overloaded WWTWs Extensive irrigation	Agriculture Construction Mines Piggery Quarry Power station Some pivot irrigation	Agriculture Mines Informal settlement (Phola park) Some pivot irrigation	Emzemvulo Nature Reserve Agriculture Informal settlement Some pivot irrigation

Step 3: Risk region selection

The Wilge River sub-catchment was divided into 4 risk regions (Figure 7.1). Risk region 1 encompasses the headwaters of the Bronkhorstspruit. Risk region 2 includes the headwaters of the Wilge River. Risk region 3 covers the Saalboomspruit. Risk region 4 follows risk region 2 and encompasses the lower Wilge River. The primary land-use is largely agricultural – commercial irrigated and fertilised land; coal mining i.e. closed New Largo colliery; livestock farming – livestock watering, combination of free range cattle and impounded cattle, chicken and pigs; Kendal power station; Kusile construction site; quarrying and infrastructure, i.e. railway tracks and tar or gravel roads. Monitoring data carried out at Risk region 2 only was used in this study, and was applied to the entire Wilge River sub-catchment.

Step 4: Conceptual model

Four endpoints were used for this risk assessment. The two biotic endpoints were invertebrate and fish wellbeing (Figures 7.5 and 7.6) and the two abiotic endpoints were formal resource water use and eco-tourism and recreational use (Figures 7.7 and 7.8). A conceptual model was created for each endpoint that reflected species-specific pathways stressors in the Wilge River sub-catchment.

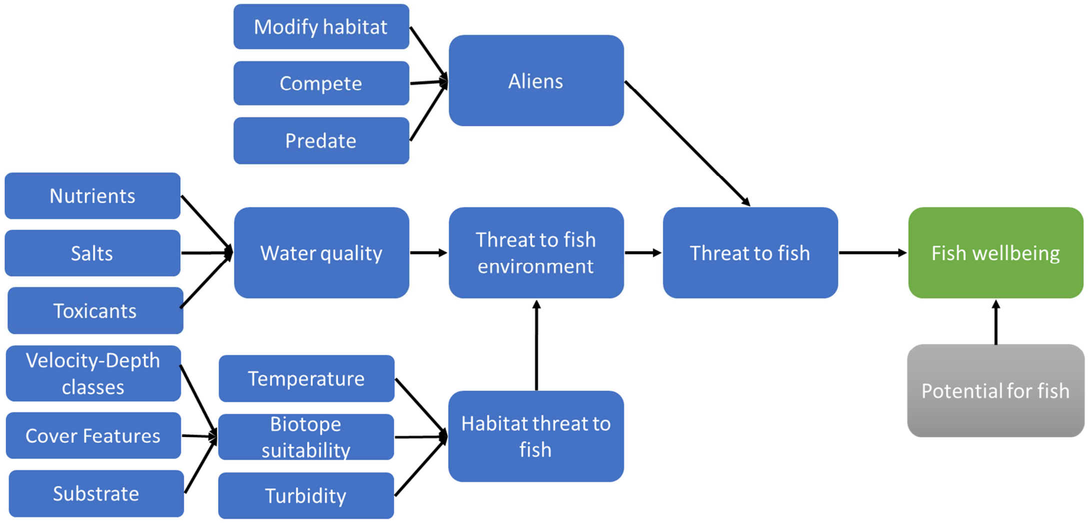

Figure 7.5: Conceptual model to show cause and effect model for Fish wellbeing endpoint.

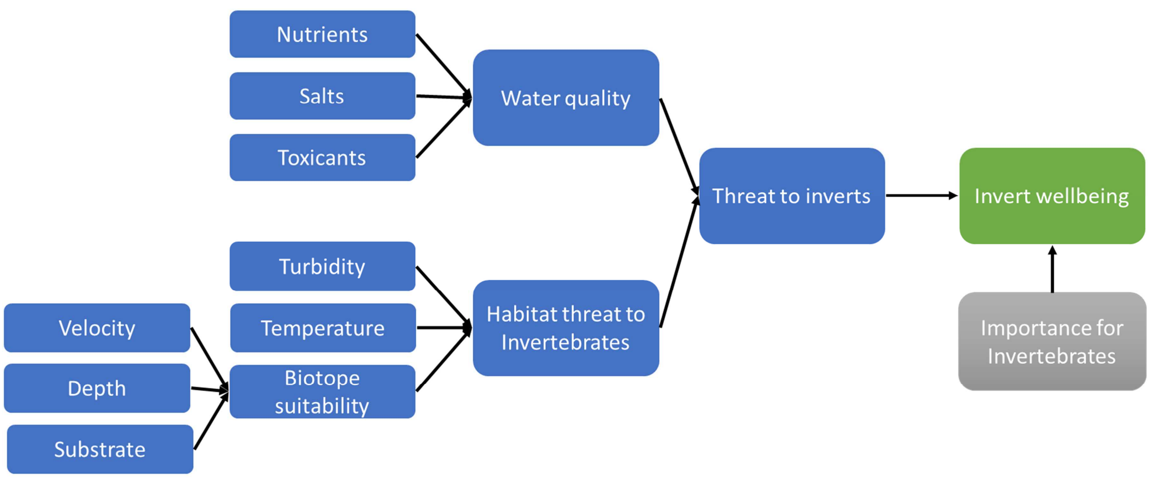

Figure 7.6: Conceptual model to show cause and effect model for Invertebrate wellbeing endpoint.

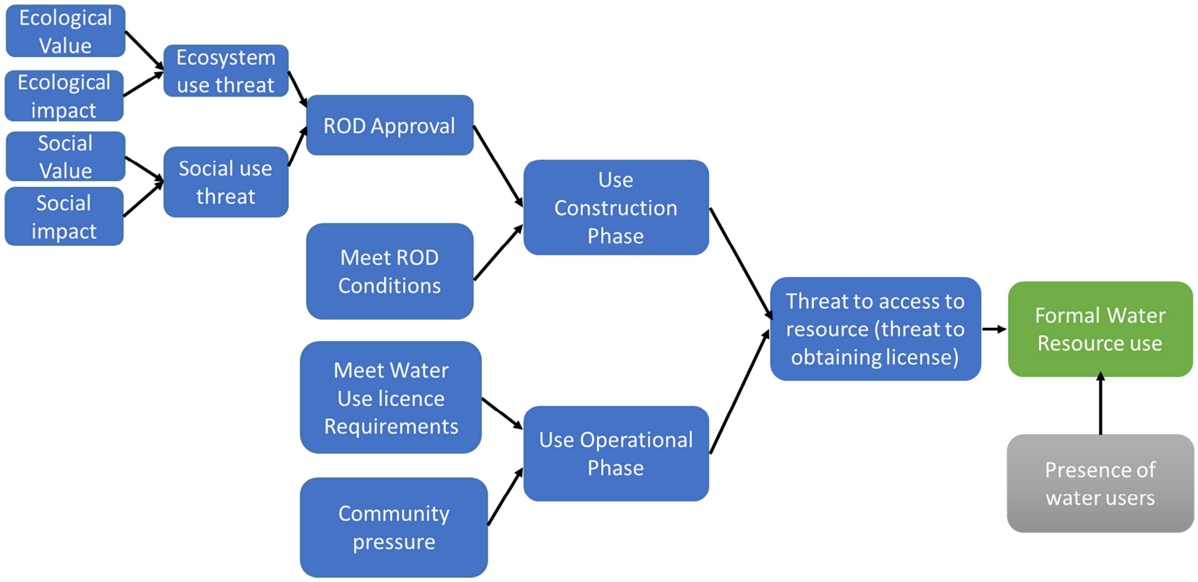

Figure 7.7: Conceptual model to show cause and effect model for Formal Water Resource Use endpoint.

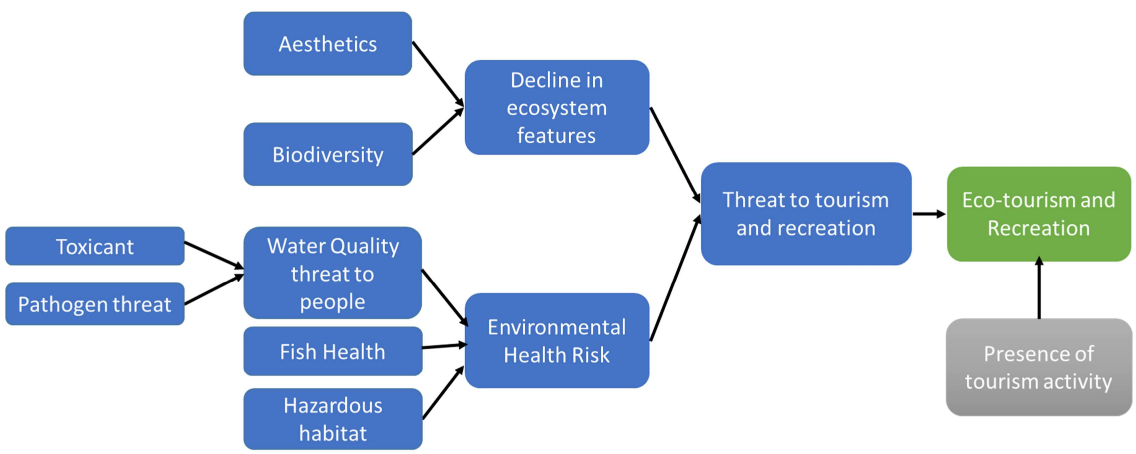

Figure 7.8: Conceptual model to show cause and effect model for Eco-tourism and Recreational Use endpoint.

Step 5: Ranking Scheme

Conceptual models link the source stressors to receptors and to the assessment endpoints. Risk rankings were defined for all the input nodes and additional details on the use of data to inform the model. In this study Bayesian Networks were used to quantify the interactions between the variables and to calculate the relative risk according to Ayre and Landis (2012). Bayesian networks (BN) link cause and effect relationships through a web of nodes using conditional probability to estimate the likely outcome (McCann *et al*., 2006; Tighe *et al*., 2013). INetica software (Norsys software Corp., Vancouver, BD, Canada) was used for the Bayesian network assessment in this study (Figure 7.9).

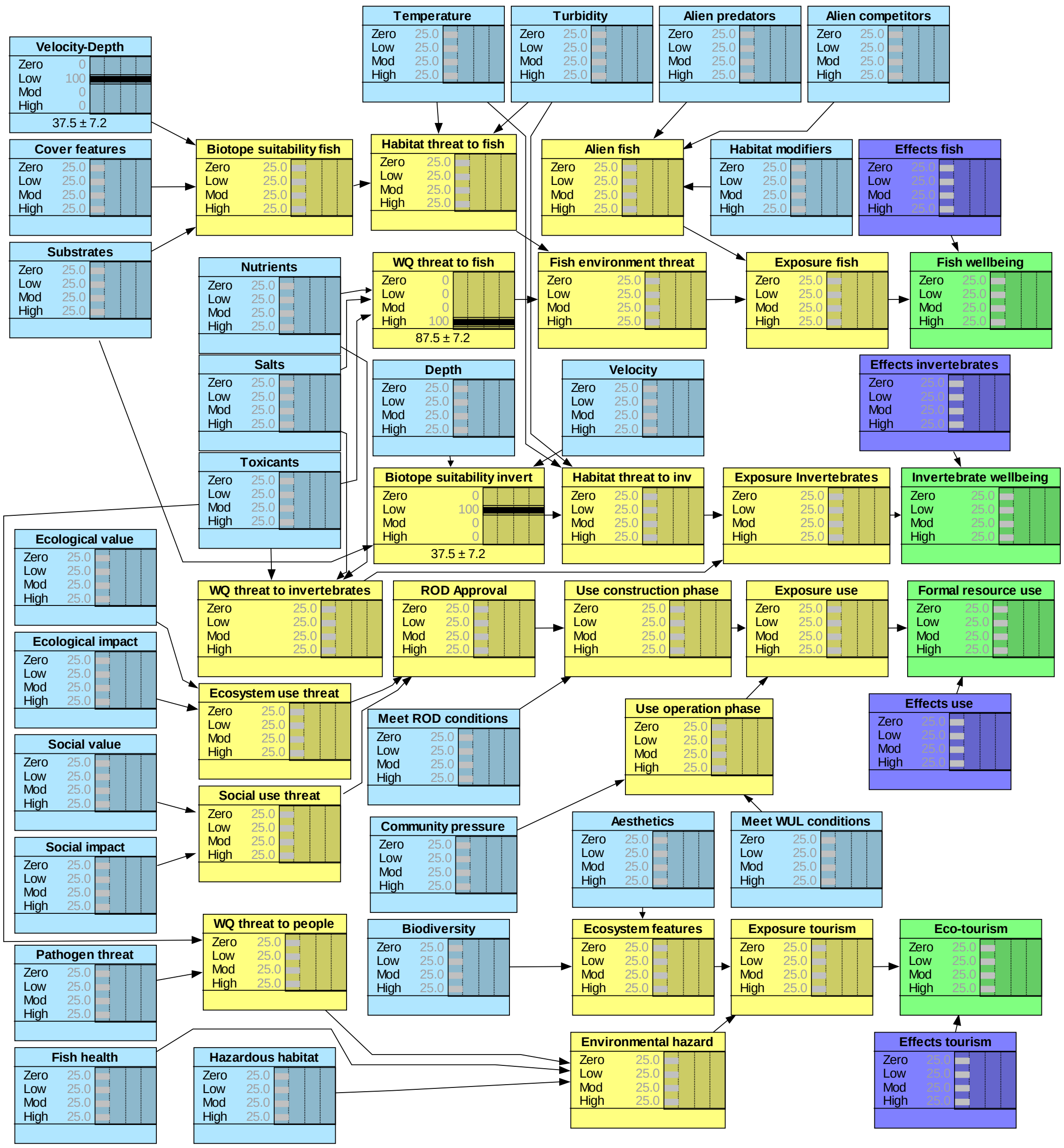

Figure 7.9: Bayesian networks show link between source stressors, receptors and to the assessment endpoints through a web of nodes using conditional probability to estimate the likely outcome.

Ranking schemes are used to represent the state of variables, with unique measures and units to be comparable as non-dimensional ranks and combined in BN-RRMs (Landis, 2004; Landis *et al*., 2016). Four states designated as zero, low, moderate and high as traditionally used in RRMs (Colnar and Landis, 2007; O'Brien and Wepener, 2012; Hines and Landis, 2014; Landis *et al.*, 2016) have been incorporated into this process. The states represent the range of wellbeing conditions, levels of impacts and management ideals as follows:

- Zero: pristine state, no impact/risk, comparable to pre-anthropogenic source establishment, baseline or reference state;
- Low: largely natural state/low impact/risk, ideal range for sustainable ecosystem use;
- Moderate: moderate use or modified state, moderate impact/risk representing threshold of potential concern or alert range;
- High: significantly altered or impaired state, unacceptably high impact/risk.

This ranking scheme selected for this model represents the full range of potential risk to management options. Low risk states usually represent management targets and moderate risk states represent partially suitable conditions that usually warrant management/mitigation measures to avoid high risk conditions. The incorporation of BN modelling, allows the approach to incorporate the variability between ranks for each model variable, represented as a percentage for each rank. Indicator variables representing the socio-ecological system being evaluated in this assessment are selected, and unique measures and units of measurement are converted into, and represented by ranks for integration in BN assessments. Assessments using the RRM have been completed for a variety of stressors and combinations of stressors including salts, nutrients, toxicants, and environmental parameters (Table 7.3).

Table 7:3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references.

Input parameter	Rank	Range	Justification
FALIENCOMP is an important driving variable considered in the fish wellbeing network. Aliens compete for food and habitat resources and threathen current fish populations. Node refers to the presence and abundance of competing fish species.	Zero	0-5	The presence of alien fauna is known to be an important determinant in the change of the wellbeing of ecosystems. The threat of aliens includes the potential of species to competition, hybridise and predate on indigenous spp. A major threat to indigenous populations is the competition by alien invasive species for food and habitat availability (WRC , 2009). Three species are alien species that have the potential to impact on the indigenous fish community structures in the form of competition for food or predation (such as the Largemouth bass – *Micropterus salmoides*) or alter habitats indirectly affecting fish communities (such as the Common carp – *Cyprinus carpio*) (Wepener *et al* ., 2011).
	Low	5-10	
	Moderate	10-15	
	High	>15	
FALIENPRED is an important driving variable considered in the fish wellbeing network. Aliens predate on current fish populations and affect current fish population numbers. Node refers to the presence and abundance of predating fish species.	Zero	0	A major threat to indigenous populations is the predation on larval indigenous fish by alien invasive species (FOSAF, 2011).
	Low	0-4	
	Moderate	5-10	
	High	>10	
TURBIDITY is an important driving variable considered in the invertebrate wellbeing network. Increase in turbidity will result in decrease in aesthetic and fish and invertebrate health. Turbidity that affects the wellbeing of invertebrate and fish.	Zero	<2	Turbidity occurs as a result of 'suspensoids' in the water column. This suspended matter, which may include clay, silt, dissolved organic and inorganic matter, plankton and other microscopic organisms, causes the water to appear turbid (Davies and Day, 1998). This causes light to be scattered and absorbed rather than transmitted in straight lines through a water sample and may: reduce light penetration, smother habitat, interfere with the feeding mechanisms of filter-feeding organisms such as certain macroinvertebrates and, reduce visibility, thus leading to a reduction in biodiversity and a system which is dominated by a few tolerant species (Davies and Day, 1998). Increased turbidity typically results in increased daily temperature maxima (Paaijmans *et al* ., 2008), leading to shorter macroinvertebrate generation times. Rivers-Moore *et al* ., (2013) concluded that reduced life cycle completion times will benefit multivoltine pest species and that aquatic macroinvertebrate communities could become increasingly dominated by warm water eurythermic, generalist species. Consequently, outbreaks of multivoltine pest species are likely to become more frequent and more severe under scenarios of increased water temperatures. Fish predators exert strong regulating forces on lower trophic levels through predation. As most fish are visual foragers, visual conditions in the water may alter the strength of this regulation. *In situ* and laboratory feeding trials coupled with stomach content analysis of largemouth bass (*Micropterus salmoides*) were performed to examine how turbidity influences the size selectivity and capture rates of prey. No significant differences in piscivory were apparent between juvenile largemouth bass collected from turbid and clear habitats. Stomach content comparisons of juvenile largemouth bass seined from six clear and turbid habitats suggest that piscivory is primarily regulated by the availability of vulnerable size-classes of prey fish, as opposed to water clarity (Reid *et al* ., 2011). Contrast degradation theory predicts that increased turbidity decreases the visibility of objects that are visible at longer distances more than that of objects that are visible at short distances. De Robertis *et al* ., (2011) results suggest that turbid environments may be advantageous for planktivorous fish because they will be less vulnerable to predation by piscivores, but will not experience a substantial decrease in their ability to capture zooplankton prey.
	Low	>2-16 NTU	
	Moderate	>16-59 NTU	
	High	>59NTU	

Table 7.3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references (continued).

Input parameter	Rank	Range	Justification
TEMPERATURE is an important driving variable considered in the fish and invertebrate wellbeing network. Effects of temperature on physical habitat and effects on invertebrates and fish. Node refers to the change in temperature that affects the wellbeing of invertebrate and fish.	Zero	18-24°c	Water temperatures play an important role in aquatic ecosystems by affecting the rates of chemical reactions and therefore also the metabolic rates of organisms (DWAF, 1996 and Chapter 3). Temperature varies with seasons and affects the rate of development, reproductive periods and emergence time of aquatic macroinvertebrates (Thirion, 2007). The temperatures (5-30°C) were considered normal for these systems and clearly reflected seasonal variation over time. Therefore, temperatures were not expected to have a limiting effect on aquatic biota.
	Low	12-<18°C and <24-26°C	
	Moderate	10-<12°C and <26-28°C	
	High	<10°C and >28°C	
VELOCITY-DEPTH (VD) is an important driving variable considered in the fish wellbeing network. Diversity of VD present to sustain reference fish populations. Node refers to the diversity VD that affects the wellbeing of fish.	Zero	0	Change from reference conditions(diversity of VD presence) changes suitable conditions and affect fish reference populations. Fish spp have a preference to certain velocity-depth conditions i.e. fast-deep, fast-shallow, slow-deep or slow shallow. Usually, species with a preference for fast flow are also those most intolerant and indicative of environmental changes (Kleynhans, 2008).
	Low	1-50	
	Moderate	51-80	
	High	>80	
COVER is an important driving variable considered in the fish wellbeing network. Presence of various cover types to sustain reference fish populations. Node refers to the change in the presnece of various cover types that affects the wellbeing of fish.	Zero	0	Change from reference conditions (presence of various cover types) affect fish reference populations. Overhanging vegetation, undercut banks and root wads, stream substrate, aquatic macrophytes and water column are cover features that are considered to provide fish with the necessary cover (such as refuge from high flow velocity, predators and high temperatures) to utilise a particular velocity-depth class (Kleynhans 1999).
	Low	1-50	
	Moderate	51-80	
	High	>80	
SUBSTRATES is an important driving variable considered in the fish and invertebrate wellbeing network.Substrates present to sustain reference fish and invertebrate populations. Node refers to the change in substrates from reference conditions that affects the wellbeing of invert and fish.	Zero	0	Substrates (bedrock cobbles, vegetation, sand, gravel and mud) is an aspect of habitat that provides invertebrates with its requirements for each specific life stage at a particular time and locality (Thirion, 2007). Substrate presence is one of the aspects of habitat that provide a fish with its life-stage requirements at that particular point in time and geographically (Kleynhans, 2008). Fish species do best in the habitats in which they evolved and that changes in habitat extent and characteristics can impose various levels of stress on populations. Changes in habitat may, therefore, be indicators of the well-being or condition of particular species or assemblages. A particular combination of habitat features may not necessarily provide optimum conditions for the specific life-history stage requirements of a fish at the time, frequency, duration and place when they are required. This may be the result of anthropogenic impacts on the habitat (such as flow reduction, sedimentation of habitat, and physico-chemical changes), or it may even be a situation where a particular species occurs only marginally and is, even under natural conditions existing under sub-optimal conditions. This relates to stress, and will result in compensatory mechanisms being activated in order to establish homeostasis in response to these stressors. It follows that species occurring marginally, and which are naturally already subject to higher stress than under optimal conditions, will have a narrow stress buffer. In such case, a relatively "small" decrease in the flow may, for example, already result in a pronounced stress effect due to particular critical habitats or critical habitat features being in limited supply naturally. It follows that differences in the requirements of different species constituting the fish assemblage may result in a change in the assemblage when the natural flow regime changes (including natural disturbance regimes) (Kleynhans, 2008).
	Low	1-50	
	Moderate	51-80	
	High	>80	

Table 7.3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references (continued).

Input parameter	Rank	Range	Justification
Effects of nutrients on water quality. Node refers to the N -P concentrations that affects the wellbeing of invert and fish.	Zero	N<3 and P<0.02	Eutrophication resulting from nutrient enrichment is globally considered to be one of the most serious threats to freshwater ecosystem services such as water quality and biodiversity. The principal anthropogenic point sources of inorganic nitrogen and phosphorus in aquatic ecosystems are: municipal sewage effluents and overflows of storm and sanitary sewers, wastewater from livestock farming, industrial wastewater effluents, and runoff from waste disposal sites, working mines and unsewered industrial sites. The main anthropogenic diffuse sources are: agricultural activities (use of manure and nitrogenous fertilizers, cultivation of N2-fixing crops), runoff from nitrogen saturated and burned forests and grasslands, urban runoff from unsewered, sewered and failed septic systems, runoff from construction sites and abandoned mines, polluted ground waters, anthropogenic atmospheric deposition loads (such as from fossil fuel combustion) and biomass burning. Globally, the dominant source of the increase, by a factor of about 4, in nutrient levels are widespread agricultural intensification and increased discharge of domestic wastes (Villiers de S and Thiart C., 2007). The Wilge IWRMP RQWO for nitrate is 6 mg/l. The Wilge IWRMP RWQOs for phosphate is 0.05 mg/l (DWA, 2009; DWS, 2014).
	Low	N=3-4.5 and P=0.02-0.03	
	Moderate	N =4.5-6 and P=0.03-0.05	
	High	N>6 and P>0.05	
SALTS is an important driving variable considered in the fish and invertebrate wellbeing network. Effects of Salts on Water Quality. Node refers to electrical conductivity concentrations as an indicator of salt presence.	Zero	0-10	Changes in salt concentrations in the river water may negatively affect crop growth and production as the water is used for irrigation. It is supposed that fertilizer, stormwater runoffs and urban wastewater discharge represent the main sources of ions (e.g. sodium, sulfate, chloride and magnesium) from manmade origin in the water. It is assumed that drought further increases the salt concentration. The measure chosen for this variable is electrical conductivity concentrations. Overall salt concentrations need to be improved to so that they do not threaten the ecosystem or agricultural users. The Wilge IWRMP RWQOs for electrical conductivity for the Wilge sub-catchment is 40mS/m (DWA, 2009; DWS, 2014).
	Low	11-25	
	Moderate	26-40	
	High	>40	
TOX is an important driving variable considered in the fish and invertebrate wellbeing network. Effects of toxicants on water quality. Node refers to toxicant inputs which affect biota or key biotic indicators.	Zero	0-1	A cumulative WHS for all tests was calculated for each sample by adding the individual WHS. A hazard category was then assigned to the cumulative WHS for each sample. As each sample is weighted according to its level of toxicity (no hazard to extreme hazard) observed for each test performed, if none of the tests detected any toxicity, the sample would have a weight of 0%, while extreme hazards detected by all tests will result in a weight of 100%. The weighting system therefore provides a measure to compare relative toxicity on a scale between 0 and 100, and toxicity hazards can therefore be compared between samples that fall within the same class. This hazard category can then be assessed in terms of ecological and management viewpoint (DWAF, 2003).
	Low	2-5	
	Moderate	6-10	
	High	>10	
VELOCITY is an important driving variable considered in the invertebrate wellbeing network. Velocity requirements met to sustain invertebrate populations. Node refers to velocity that affects the wellbeing of invertebrates.	Zero	<0.1 m/s	Low and high flows must be suitable to maintain the river habitat and ecosystem condition. Flow and thermal alterations are associated with ecological change and the risk of change increases with increasing magnitude of alteration. For example, changes in magnitude of alteration typically lead to stabilized flows (loss of extreme high and/or low flows) with ecological impacts showing as loss of sensitive species, loss of diversity and changed community structure (Thirion, 2007; Thirion, 2015; Poff and Zimmerman, 2010).
	Low	0.1-0.3 m/s	
	Moderate	0.3-0.6 m/s	
	High	>0.6 m/s	
DEPTH is an important driving variable considered in the invertebrate wellbeing network. Depth requirements met to sustain invertebrate populations. Node refers to depth that affects the wellbeing of invertebrates.	Zero	10-20cm	Depth is not a significant factor in determining the distribution of insects investigated (Thirion, 2015). Although some variation was noted.
	Low	20-30cm	
	Moderate	30-40cm	
	High	<10cm and >40cm	

Table 7.3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references (continued).

Input parameter	Rank	Range	Justification
FALIENHAB is an important driving variable considered in the fish wellbeing network. Aliens change habitat and affect reference fish populations. Node refers to the change in habitat conditions caused by aliens that affects the wellbeing of fish.	Zero	0-25	Exotic fish species affect indigenous fish populations by competing for food and habitat. Over time exotics may pose a problem to indigenous populations (Minshull, 1993).The Olifants River has, been described as one of the most polluted rivers in southern Africa, due to the number of anthropogenic stressors that are present in the catchment. These stressors include intensive coal mining activities, coal-fired power generation industrial activities (e.g. chemical manufacturers, chrome and steel smelters) and agriculture, combined with a general decline in the operation and management of wastewater treatment infrastructure, especially sewage treatment. The pollutants generated by these activities include general acidification of the system and the input or mobilisation of heavy metal ions plus sulphates and other contaminants via acid mine drainage (Hobbs *et al* ., 2008); potential acid rain resulting from poor air quality; industrial effluent containing a variety of potential pollutants; excessive nutrient inputs (phosphorus and nitrogen) from agricultural activities and sewage effluent; and microbiological pollution from intensive agriculture (e.g. feedlots) and sewage effluent. Associated with these key water quality parameters are threshold concentrations which, if regularly exceeded, can result in harmful impacts on aquatic ecosystems and human health.(Dabrowski and De Klerk, 2013).
	Low	26-50	
	Moderate	51-75	
	High	76-100	
ECOVAL is an important driving variable considered in the formal water resource use network. Ecological value to catchment. Environmental study (water/air terr) evaluated in EIA. Area with high ecological value will increase the threat to ecosystem use.	Zero	0-25	Ecological value of the environment is high then risk to obtaining the Water Use License (WUL) for the activity will be high. Broad variables. Low confidence. Limited amount of available literature (National Water Act, (Act No 36 of 1998); DWAF WUL, 2009; DWAF WUL, 2011; DWAF WUL, 2012; Eskom EIA, 2006 (Agricultural impact study, Air Quality study, Aquatic ecological assessment, Heritage impact study, Hydrogeological investigation, Socio-economic impact assessment, Terrestrial ecological investigation, Town planning investigation, and Visual impact assessment; Expert opnion -Kusile Environmental Manager).
	Low	26-50	
	Moderate	51-90	
	High	91-100	
ECOIMP is an important driving variable considered in the formal water resource use network. Ecological impact to catchment. EIA shows activity to have high ecological impact to the environment. Area with high ecological impact will decrease the threat to ecosystem use.	Zero	0-25%	Ecological impact of proposed activity to have low impact in the environment then risk for obtaining WUL for activity is low. Broad variables. Low confidence. Limited amount of available literature (National Water Act, (Act No 36 of 1998); Kusile WUL, 2009; Kusile WUL, 2011; Kusile WUL, 2012; Eskom EIA, 2006 (Agricultural impact study, Air Quality study, Aquatic ecological assessment and, Hydrogeological investigation and, Terrestrial ecological investigation; Expert opinion -Kusile Environmental Manager).
	Low	26-50%	
	Moderate	51-75%	
	High	76-100%	
SOCVAL is an important driving variable considered in the formal water resource use network. Social impact to catchment. Heritage/ visual impact/ socio - economic impact evauated in EIA. Area with high socal value will increase the threat to ecosystem use.	Zero	0-25%	Impact of proposed activity to have low social impact then risk for obtaining WUL for activity is low. Broad variables. Low confidence. Limited amount of available literature (National Water Act, (Act No 36 of 1998); Kusile WUL, 2009; Kusile WUL, 2011; Kusile WUL, 2012; Eskom EIA, 2006 (Heritage impact study, Socio-economic impact assessment, Town planning investigation and, Visual impact assessment; Expert opinion - Kusile Environmental Manager).
	Low	26-50%	
	Moderate	51-75%	
	High	76-100%	

Table 7.3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references (continued).

Input parameter	Rank	Range	Justification
SOCIMP is an important driving variable considered in the formal water resource use network. Social impact to catchment. Heritage/ visual impact/ socio - economic impact evauated in EIA. Area with high socal impact will decrease the threat to ecosystem use.	Zero	0-25%	Impact of proposed activity to have low social impact then risk for obtaining WUL for activity is low. Broad variables. Low confidence. Limited amount of available literature (National Water Act, (Act No 36 of 1998); Kusile WUL, 2009; Kusile WUL, 2011; Kusile WUL, 2012; Eskom EIA, 2006 (Heritage impact study, Socio-economic impact assessment, Town planning investigation and, Visual impact assessment; Expert opinion - Kusile Environmental Manager).
	Low	26-50%	
	Moderate	51-75%	
	High	76-100%	
CONLICENCE is an important driving variable considered in the formal water resource use network. Node refers to the requirements for obtaining construction licence met. EIA / ROD approved.	Zero	100	National Water Act, (Act No 36 of 1998)
	Low		
	Moderate		
	High	0	
COMMUNITY is an important driving variable considered in the formal water resource use network. Community pressure for land use to be present and operate. Node refers to Number of stakeholder complaints.	Zero	0	Stakeholder participation is viewed as critical in the current water sector reforms taking place in the Southern African region. A sttudy undertaken in Zimabawe revealed widespread use of indigenous knowledge and practice by communities. Such knowledge is based on smell, taste, colour and odour perceptions. Residents are generally more concerned about the physical parameters than the bacteriological quality of water. They are aware of what causes water pollution and the effects of pollution on human health, crops, animals and aquatic ecology. They have ways of preventing pollution and appropriate interventions to take when a source of water is polluted, such as boiling water for human consumption, laundry and bathing, or abandoning a water source in extreme cases. Stakeholder participation and ownership of resources needs to be encouraged through participatory planning, and integration between the three government departments (water, environment and health) (Narea *et al* ., 2006). Broad variables. Low confidence. Limited amount of available literature.
	Low	1-5	
	Moderate	6-20	
	High	>20	
FHEALTH is an important driving variable considered in the eco-tourism and recreation network. Good fish health will result in lower environmetal health risk. Node refers to FRAI scores.	Zero	90-100	Ecological Categories (EC) used in interpreting RHP data (DWAF, 2008; Kleynhans, 1999). FRAI Score 90-100 (A) Natural, FRAI Score 80-89 (B) Good, FRAI Score 60-79 (C) Fair, FRAI Score 40-59(D) Poor, FRAI Score 20-39(E) Seriously modified, FRAI Score 0-19(F) Critically modified.
	Low	60-89	
	Moderate	20-59	
	High	0-19	
HAZHAB is an important driving variable considered in the eco-tourism and recreation network. Presence of hazardous habitat. Node refers to presence and abundance of hazardous habitat affects environmental health risk.	Zero	0	Presence and abundance of hazardous habitat (holes/rubble/ tailings dam/crocodile/diseases) will increase environmental health risk. Areas with a higher number of mines, industries, chemical plants, coal-fired power stations, construction sites etc would have a higher environmental health risk. Broad variables. Low confidence. No available literature.
	Low	0-1	
	Moderate	1	
	High	>1	

Table 7.3: Biotic model table describing input parameters, rank, ranking schemes, justification, and data sources or references (continued).

Input parameter	Rank	Range	Justification
BIODIVERSITY is an important driving variable considered in the eco-tourism and recreation network. Biodiversity and richness of fish and invert species. Node refers to number of sensitive invertebrate and fish species.	Zero	FRAI 90-100 SASS >124 ASPT > 5.6	Modelled reference conditions for the Highveld Ecoregion (11) based on SASS5 and ASPT values (Dallas, 2007). SASS Score >124 ASPT Score>5.6 (A) Unimpaired. High diversity of taxa with numerous sensitive taxa, SASS Score 83-124ASPT Score 4.8-5.6 (B) Slightly impaired. High diversity of taxa, but with fewer sensitive taxa. SASS Score60-82 ASPT Score4.6-4.8 (C) Moderately impaired. Moderate diversity of taxa, SASS Score52-59 ASPT Score 4.2-4.6 (D) Considerablely impaired. Mostly tolerant taxa present, SASS Score30-51 ASPT Score Variable<4.2 (E) Severely impaired. Only tolerant taxa present, SASS Score<30 ASPT Score Variable (F) Critically impaired. A few tolerant taxa present. Ecological Categories (EC) used in interpreting RHP data (DWAF, 2008; Kleynhans,1999). FRAI Score 90-100 (A) Natural, FRAI Score 80-89 (B) Good, FRAI Score 60-79 (C) Fair, FRAI Score 40-59(D) Poor, FRAI Score 20-39(E) Seriously modified, FRAI Score 0-19(F) Critically modified.
	Low	FRAI 60-89 SASS 60-124 ASPT 4.6-5.6	
	Moderate	FRAI 20-59 SASS 30-59 ASPT 4.2-4.6	
	High	FRAI 0-19 SASS <30 ASPT Variable	
AESTHETICS is an important driving variable considered in the eco-tourism and recreation network. Node refers to changes in aesthetics affect ecosystem features.	Zero	0-25%	A landscape with a high sensitivity would be one that is valued for its aesthetic attractiveness and/or have ecological, cultural or social importance through which it contributes to the inherent character of the visual resource. (Eskom EIA, 2006). The extent, effects and mitigation measures to reduce and\or alleviate the potential adverse landscapes needs to be investigated (Oberholzer, 2005). Pollution may cause nuisance for local residents and tourists as well as environmental problems and may lessen the psychological benefits of tourism (WHO, 1980). Expanding urbanisation and industrialisation has resulted in changes of the surrounding enviroment from its natural state. Anthropogenic activities i.e. land disturbance, pollution, overpopulation, landfills, deforestations and natural causes i.e. storms and wild fires degrade the environment physically decreasing animal and plant groups to a point where they can no longer survive. Decrease in aestheics will result in a decline for ecosystem features. Decrease in aesthetic may affect human health, biodiversity, air quality, loss of tourism and negative ecnomic impact -by having to restore green cover, clean up landfills, protection of endangered species. Broad variables. Low confidence. No available literature.
	Low	26-50%	
	Moderate	51-75%	
	High	76-100%	
OPLICENCE is an important driving variable considered in the formal water resource use network. Requirements for maintaining operational licence met. Node refers to monitoring reqirements met.	Zero	0	National Water Act, (Act No 36 of 1998)
	Low		
	Moderate	80	
	High	100	
PATHTOUR is an important driving variable considered in the eco-tourism and recreation network. Threat of pathogen on tourism. Abundance of pathogens which may affect human health. Node refers to faecal coliforms presence.	Zero	0	The natural ability of rivers and reservoirs to trap toxic chemicals and nutrients in their sediments enables these systems to accumulate contaminants, altering the natural balance in environmental water quality, thereby raising a plethora of public and environmental health concerns. Impaired water quality has been linked to an array of problems in South Africa including massive fish mortalities, altered habitat template leading to the thinning of riverine macroinvertebrate diversity, shifts in microbial community structures with drastic ecological consequences and evolvement of antibiotic resistance genes that, under natural conditions, can be transferred to waterborne pathogens. Urban wastewater discharge has also been implicated in increased bioaccumulation of metals in edible plant parts, elevated concentrations of endocrine-disrupting compounds (EDCs), which are blamed for reduced fertility and increased cancer risk, excessive growth of toxic cyanobacteria and an increase in concentrations of pathogenic microorganisms which constitute a potential health threat to humans.(Sibanda *et al*., 2015; Eskom, 2016). Microbial infection and infectious disease transmission - causes gastro-intestinal disease, diarrhoea in human health. Livestock health effects - risk of infection particularly in young stock. The measure selected to represent microbial contamination potential for this assessment is the faecal coliform data.
	Low	1-60	
	Moderate	61-130	
	High	>130/100mL	

Step 6: Calculate risks

The risk calculated may be compared between individual endpoints per risk region or by management scenario, and the cumulative risk of all endpoints within risk regions or management scenario can be determined using Monte Carlo simulations, Oracle's Crystal Ball software, Oregon. This approach was also used to evaluate the effects of uncertainty on risk projections. The RRM uses spatially distinct risk regions to organise the information into cause and effect pathways and the ranking schemes are used to combine variables with different units. Relative-risk scores are calculated for assessment endpoints and can be compared across risk regions (spatial gradients) and between endpoints.

Four scenarios were chosen: Present status, Pre-anthropogenic status, Increased mining and increased urban development. Increased mining, due to the expansion of coal mining in the upper Olifants River catchment is a concern in the future in the Wilge River sub-catchment B20F area (DWA, 2011). There has been rapid growth in the urban sector especially in the metropolitan area of Emalahleni and Middelburg, while the growth in more rural areas has been limited by the lack of water supply infrastructure. The estimated growth from 2010 to 2035 is from 93 to 113 million m^3/annum (DWA, 2010), making increased urban development a concern in the future in the Wilge River sub-catchment B20F area as well.

Data for determining present status for RR2 was used from Chapter 2 to 5 of this study. Present status for RR1, RR3 and RR4 were determined by referring to land-use maps. For scenario 3 (increased mining) the only variables that were changed from present state include: salts, toxicity, turbidity, aesthetic, biodiversity, hazardous habitat, community, ecological value, ecological importance, social value, social importance and fish health. For scenario 4 (increased urban development) the only variables that were changed from present state include: salts, toxicity, nutrients, pathogen threat, turbidity, aesthetic, biodiversity, community, ecological value, ecological importance, social value, social importance and fish health.

The mean probability of a change in risk distributions for 2 biotic endpoints (fish wellbeing and invertebrate wellbeing) and 2 abiotic endpoints (formal water use and eco-tourism) are presented in Figure 7.10.

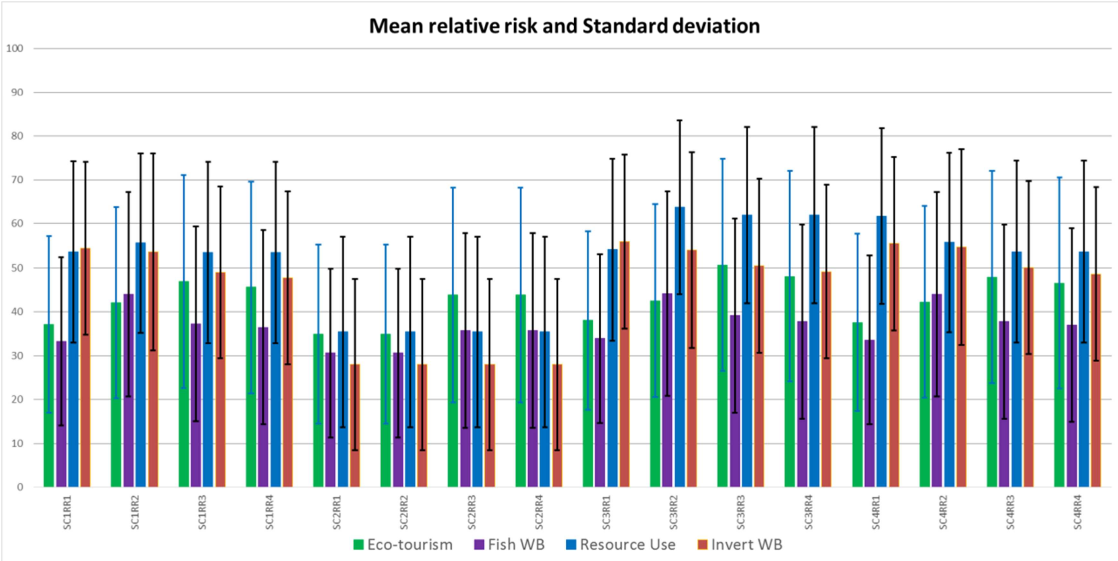

Figure 7.10: Mean relative risk and standard deviation for each endpoint for each scenario. SC represents scenario and RR represents risk region. Error bars represent the sensitivity and uncertainity analyses.

The mean probability of change in risk distributions for all endpoints were shown to be lowest in all risk regions in Scenario 2 which is the pre-anthropogenic state. This is expected as this state has the least amount of land-use impacts. The mean probability of change in risk distributions for all endpoints for all risk regions increased from scenario 2 (pre-anthropogenic state) to scenario 1 (current state). This is expected due to the increased land-use activities in all risk regions. Currently RR1 is impacted on by agriculture, urban development, municipality infrastructure, overloaded waste water treatment works (WWTW), extensive irrigation and mining activities. Risk region RR2 is impacted on by agriculture, livestock farming i.e. piggery, power station, construction site, quarry, some pivot irrigation and mining activies. Risk region RR3 is impacted on by agriculture, informal settlement (Phola park), some pivot irrigation and mining activity. Risk region RR4 is impacted on by agriculture, informal settlements and some pivot irrigation acivities. Emzemvulo Nature Reserve is located on RR4.

Scenario 3 (increased mining) and scenario 4 (increased urban development) shows mean probability of change in risk distributions for all to risk region 1-4 for all endpoints. Abiotic endpoints, formal water use and eco-tourism showed increased risk distributions for all risk regions for scenario 3. No significant change in risk distributions for scenario 4 for all risk regions.

The probability of zero, low, moderate and high risk is presented for endpoints fish wellbeing, invertebrate wellbeing, formal water use and eco-tourism use in Figure 7.11-7.14. In RR2 the probability of high risk increased for fish and invertebrate wellbeing from scenario 2 (pre-anthropogenic state) to scenario 1 (current state). This is due to the increase in land-use activities from scenario 2 to 1 (Figure 7.11 and Figure 7.12).

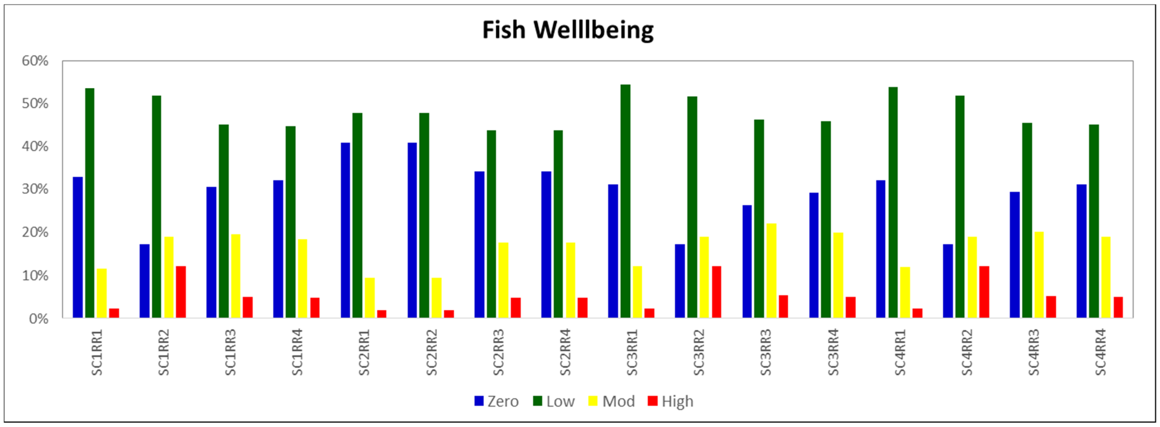

Figure 7.11; Fish Wellbeing risk distributions for each risk region for each scenario. SC represents scenario and RR represents risk region.

Fish and invertebrate wellbeing showed increased risk distributions for all risk regions for scenario 3. Fish and invertebrate wellbeing showed no significant change in risk distributions for scenario 4 for all risk regions (Figure 7.11 and Figure 7.12).

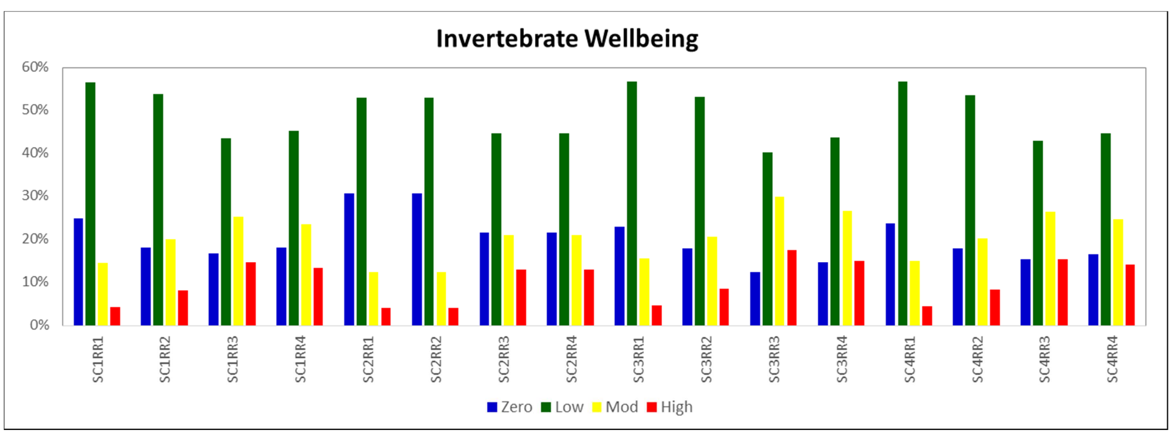

Figure 7.12: Invertebrate wellbeing risk distributions for each risk region for each scenario. SC represents scenario and RR represents risk region.

The probability of high risk increased for abiotic endpoints formal water use and eco-tourism use from scenario 2 (pre-anthropogenic state) to scenario 1 (current state). This is due to the increase in land-use activities from scenario 2 to 1 (Figure 7.13 and Figure 7.14).

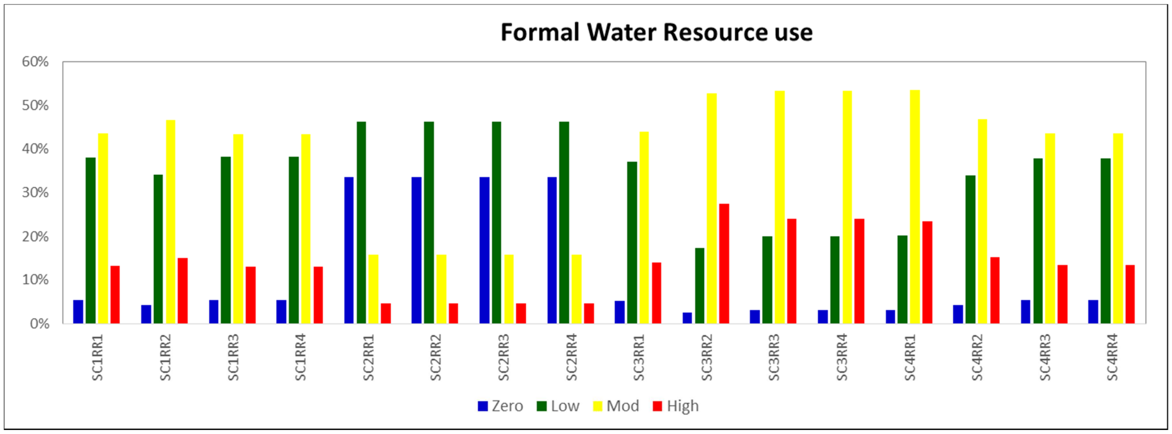

Figure 7.13: Formal Water Resource use risk distributions for each risk region for each scenario. SC represents scenario and RR represents risk region.

Formal water use and eco-tourism use showed increased risk distributions for all risk regions for scenario 3 and 4 (Figure 7.13 and Figure 7.14).

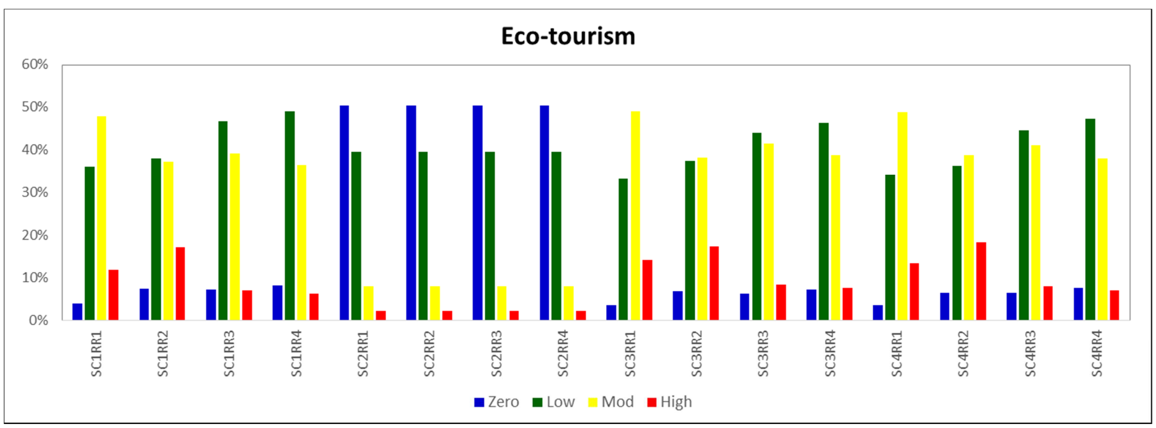

Figure 7.14: Eco-tourism endpoint for each risk region for each scenario. SC represents scenario and RR represents risk region.

Step 7: Uncertainty evaluation

The Monte Carlo permutation process was used to provide a range of values to simulate uncertainty (Landis, 2005; Colnar and Landis, 2007). The ranks and filter components with medium and high classifications are assigned with discrete statistical distributions to represent the uncertainty. The range of statistical distributions used to address the uncertainty of each rank and filter must be documented. The ranks and filter components with low uncertainty classifications retain their original values. After the uncertainty classifications are assigned, the Monte Carlo simulations can be run using sufficient iterations, usually >1000, to account for all variability in the model (Colnar and Landis, 2007). The equations to tables, in this study were written, with 1000 random iterations per cell.

Sensitivity analysis

The process is completed by conducting a sensitivity analysis of the model and revising the data with new evidence of knowledge. Various statistical methods can be used to carry out correlation coefficients, including Crystal Ball™ 2000 software, for example (Colnar and Landis, 2007).

Step 8: Hypotheses establishment

The fundamental adaptive management approach and RRM approach to improving our understanding of socio-ecological risk relationships is to reduce uncertainties and to confirm the risk rankings in risk assessments (Landis 2004). These adaptive management principles acknowledge that socio-ecological systems are dynamic and that our limited understanding of these processes necessitates the incorporation of many assumptions. In many case studies, uncertainties associated with the outcomes need to be mitigated before they can be used to inform decision making (O'Brien *et al*., 2017).

The following hypothesis was established and tested:

(1) Increase in construction activities will not change or influence the endpoints associated with Risk Region RR2.

The probability of high risk increased for all biotic and abiotic endpoints for RR2 from preanthropogenic state to current state, thus declaring the hypothesis null (Figure 7.10).

However, construction activity was not the only land-use activity occurring within the RR2 so the impact observed may not be due to construction activities only. Also, the probability of high risk increased for all biotic and abiotic endpoints for RR1, RR3 and RR4 from proanthropogenic state to current state which experienced various degrees of other land-use activities (Table 7.2 and Figure 7.4). The biggest impacts to RR2 were biotic endpoint fish wellbeing and biotic endpoint eco-tourism (Figure 7.11 and Figure 7.13). This hypothesis was not accepted.

(2) Increase in mining will not change or influence the endpoints.

Increase in mining activities scenario show a decrease on the low risk probability and an increase in moderate and high risk probability for all risk regions for formal water use, eco-tourism and invertebrate wellbeing. The least change in risk was found to be for the fish wellbeing endpoint. The hypothesis was not accepted.

(3) Increase in urban development will not change or influence the endpoints.

Increase in the urban activities scenario show an increase in moderate to high risk probability for RR1, almost no change to risk for the invertebrate and fish wellbeing and eco-tourism for the other risk regions. The hypothesis was not accepted.

Step 9: Test hypotheses

The existing monitoring programme in the Wilge River sub-catchment B20F area, used in this study and continuous monitoring of the Wilge River sub-catchment can be used to test the accuracy of the risk projections for the scenarios presented and improve the validity of the assessment and understanding of the aquatic ecosystem relationships. The present scenario indicates that the RQOs (DWS, 2014) at RR2 are not being met for water quality and fish ecological integrity. Continued monitoring and remedial action should be taken to meet the RQO targets for RR2. The RRM clearly shows an increase in the probability of high risk for RR2, in comparison to the other risk regions, from preanthropogenic to current state for the fish endpoint (Figure 7.10). The RRM shows an increase in the probability of high risk for all risk regions for the formal water use endpoint. From this study, the RQOs (DWS, 2014) at RR2, even though they are within their limits, have deteriorated indicating that the increased formal water use has resulted in the deterioration of the water quality. As the formal water use for all risk regions has increased, the resulting high risk probablility has become evident (Figure 7.12). Comprehensive water quality monitoring should be undertaken in the other risk regions in order to determine if there has been a deterioration in water quality.

Step 10: Communicate outcomes

The importance of clear and careful communication of the outcomes in the context of the uncertainty identified in an assessment is highlighted in this RRM approach (Hayes and Landis, 2004). The communication of the aquatic ecosystem outcomes and associated socio-ecological consequences of the impacts to aquatic ecosystems must be presented to relevant stakeholders with information that can be easily understood (O'Brien and Wepener, 2012). In this study the outcomes were represented as risk profiles.

7.4 Conclusions

The RRM is implemented on a large spatial scale and facilitates the consideration of multiple sources of multiple stressors affecting multiple endpoints, including the ecosystem dynamics for risk analysis, decision making and adaptive management (Johns *et al*., 2016). The effects of management alternatives under consideration can be incorporated into the initial conceptual model to evaluate the changes for sources, stressors, and habitats to endpoint risk. The RRM has been used to examine how different management strategies change risk (Ayre *et al*., 2014; Hines and Landis 2014; Herring *et al*., 2015).

The BN-RRM was applied as a tool to perform a regional-scale, multiple-stressor ecological risk assessment in the Wilge River sub-catchment area in the Upper Olifants River catchment. The RRM model has shown to be a robust, transparent, adaptable and flexible probalistic modelling tool that can make a positive contribution to the sustainable management of the aquatic ecosystem in the Wilge River sub-catchment. The results of this study demonstrate that BN can be used to calculate risk for multiple stressors, and that they are a powerful tool for informing future management strategies for aquatic ecosystem management in the Wilge River sub-catchment. The evidence based outcomes can facilitate informed environmental management decision-making in the context of social and ecological aspirations.

By evaluating the scenarios, this study showed that the RRM approach is a powerful predictive decision making tool that can be used to address a range of aquatic ecosystem management objectives. Furthermore the approach has been shown to be versatile and adaptable to unique case studies while maintaining its validity.

This showed that the RRM approach could contribute towards the establishment of holistic, integrated management plans for aquatic ecosystems on various scales. The versatility of the approach allows for a range of different aquatic ecosystem management objectives to be considered. These include the establishment of sustainable balances between the use and protection of surface aquatic ecosystems, such as this study, which demonstrates the effect of water resource use scenarios to the socio-ecological system, and to achieving RQO. This study demonstrated that the RRM methodology has the potential to contribute greatly to the management of aquatic ecosystems.

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CHAPTER 8: CONCLUSIONS AND RECOMMENDATIONS

8.1 General conclusions

The aims of this study is to use existing environmental monitoring and biomonitoring tools that are routinely applied in environmental assessments in South Africa and to interpret the results in a uniform risk-based format, in order to demonstrate whether the RRM provides an efficient framework to identify the risks posed on the local aquatic resources to allow for informed decision-making relating to the potential risk of impacts of construction activities.

The aims of the study were established as follows:

- Development of a RRM framework based on activities related to construction activities of Kusile power station;
- Using the Kusile construction site as a case study, assess the impacts on local aquatic environments in terms of water quality and aquatic ecosystem health;
- Integrate data from existing aquatic assessment tools into an RRM protocol;
- Integrate the RRM-based outputs to refine the existing Environmental Management Plan of the Kusile construction site, based on data generated;
- Recommend environmental guidelines that would be implemented for construction activities at Kusile sites.

The following hypotheses were tested in this study:

Hypothesis 1: Land-use activities do not change the water quality and there is no influence of these water quality parameters on the toxicity to aquatic organisms.

The findings showed that the combination of land-use impacts has affected the water quality in the Wilge River sub-catchment B20F area. The two main sets of stressors in the upper Olifants River catchment therefore, is acidic water containing heavy and trace metal ions and sulphate that is attributable to abandoned mining and nutrient concentrations originating from agricultural and livestock runoff, and from untreated or poorly treated sewage.

The four-tiered toxicity assessments were found to applicable and appropriate for measuring the change in toxicity hazards due to a range of land-uses and produced additional information when considering the relative health of a water resource under stress. The hazard categories of the sampling sites were found to have a predominantly moderate hazard to toxicity. Thus implying that the cumulative effects of the impacts, i.e. agriculture, livestock farming, mining, the construction site and the quarry are contributing to the increasing toxicity in the catchment. This hypothesis was therefore not accepted.

Hypothesis 2: Land-use activities do not change the integrity of macroinvertebrate community structures.

Comprehensive macroinvertebrate studies showed that considerable variations occurred with regard to the families found between the various surveys and between each of the sampling sites. These variations probably resulted from a combination of environmental and physical conditions, which include the conditions of the habitat, habitat availability changing with flow, seasonal variation in macroinvertebrate communities and various anthropogenic influences such as flow modifications (impoundments), agriculture and mining. A dominance of families found has a preference for low to very low water quality, probably due to the changes in land-use. Ultimately, in order to manage and conserve the Wilge River, it is essential to understand the catchment and its impeding land-uses (Farrell *et al.,* 2015). Macroinvertebrate assemblage within the Wilge River catchment show that it is in a poor state of health and it is therefore imperative to maintain the ecological integrity of the Wilge River and strive to improve it. This hypothesis was therefore not accepted.

Hypothesis 3: Land-use activities do not change the integrity of fish community structures.

The fish assessments showed that highest species diversity was found at sampling reaches 4 and 5, with all fish species on the FFROC list being found. Sampling site SW7 in sampling reach 2 showed the lowest biodiversity with only *Enteromius anoplus* being found for the duration of the study period. Anthropogenic factors such as, impeding structures i.e. dams, bridges and roads have affected fish migration in sampling reaches 1, 2 and 3 showing lower species diversity and higher fish species absences in theses reaches. *Enteromius. paludinosus* which is found abundantly in the Wilge River sub-catchment B20F area is completely absent at sampling reach 3 for the study period. *C. pretoriae* and *L. cylindricus* are completely absent at sampling reaches, 1, 2 and 3 for the study period. *L. marequensis* is completely absent at sampling reaches 2 and 3 for the study period. Many of the species (such as *L. marequensis* and *L. cylindricus*) are known to migrate tens of kilometres. The migration and movement of species into/from the tributaries may be influenced by these structures. Certain taxa such as *L. marequensis* and *L. cylindricius* are also not expected to occur in the smaller tributaries due to a lack of habitat. Fish assemblage structures were shown not to have altered due to changes in land-use as the ecological categories remained similar from assessments carried out from 2006 to 2014. This hypothesis was therefore accepted.

Hypothesis 4: Land-use activities do not change the wetland integrity.

WET-health assessments of the wetlands, indicated that the majority of wetlands still considered being in a natural to largely natural state are small hillslope seepage wetlands located along footslopes to the various drainage lines of the area, especially the wetland system to the north of the power station. In most cases these seepage wetlands are associated with areas of shallow or rocky soils that cannot be cultivated and are thus associated with primary grasslands, which in their own right are of high biodiversity importance. These wetlands are however currently under considerable threat from heavy livestock grazing and trampling, and are likely to deteriorate and change to lower PES categories unless a grazing management plan is compiled by a suitably qualified expert and strictly implemented. The ecological state at wetland complex 1, 2 and 3 all improved to B in May, August and November 2014, this includes wetland complex 2, inspite of part of the wetlands being lost due to Kusile power station construction which started in April 2008. Wetland conditions were found to have deteriorated from March 2010 to December 2011 in the wetland complexes assessed, and can be attributed to the changing land-use. Improvements to the wetland ecological status from August 2012 to December 2014 can be due to a decrease in construction activities and an increase in wetland rehabilitation efforts implemented. This hypothesis was therefore not accepted.

Hypothesis 5: The RRM is effective in achieving management goals.

The findings showed that when the Bayesian Network-RRM was applied as a tool to perform a regional-scale, multiple-stressor ecological risk assessment in the Wilge River sub-catchment area in the Upper Olifants River catchment, it seems to be a robust, transparent, adaptable and flexible probalistic modelling tool that can make a positive contribution to the sustainable management of the aquatic ecosystem in the Wilge River sub-catchment.

The following hypothesis was established and tested by the RRM: (1) Increase in construction activities will not change or influence the endpoints associated with Risk Region RR2; (2) Increase in mining will not change or influence the endpoints and; (3) Increase in urban development will not change or influence the endpoints. All the hypotheses were tested and not accepted. The present scenario indicates that the RQOs (DWS, 2014) at RR2 are not being met for water quality and fish ecological integrity. The RRM clearly shows an increase in the probability of high risk for RR2, in comparison to the other risk regions, from preanthropogenic to current state for the fish endpoint. The RRM shows an increase in the probability of high risk for all risk regions for the formal water use endpoint. From this study, the RQOs (DWS, 2014) at RR2 are not being met for water quality indicating that the increased formal water use has resulted in the deterioration of the water quality status. As the formal water use for all risk regions has increased, the resulting high risk probablility has become evident.

Comprehensive water quality monitoring should be undertaken in the other risk regions in order to determine whether they meet the RQO requirements. The results of this study demonstrate that BN can be used to calculate risk for multiple stressors, and that they are a powerful tool for informing future management strategies for aquatic ecosystem management in the Wilge River sub-catchment. The evidence based outcomes can facilitate informed environmental management decision-making in the context of social and ecological aspirations.

The use of BNs as a tool to perform a regional-scale, multiple-stressor ecological risk assessment in the Wilge River sub-catchment area in the Upper Olifants River catchment demonstrates the effect of water resource use scenarios to the socio-ecological system, and contribute to achieving RQO and achieve a balance between use and protection. This hypothesis was accepted.

8.2 General recommendations

The careful management and mitigation of pollutant sources are essential to ensure compliance to the Wilge River IWRMP RWQOs thus, achieving long-term sustainability and acceptable water quality and ecosystem health in the upper Olifants River catchment. Increased land-use activities have resulted in excessive utilisation of the ecological services of aquatic ecosystems in South Africa. The resulting decline of the integrity status of this system and the consequent loss of key ecosystem services suggest that the national requirements to maintain a sustainable balance between the use and protection of these systems are not being met. In order to address this issue, the following recommendations need to be taken into consideration:

- When large changes in land-use or large construction activities are planned, a buffer region need to be demarcated and comprehensive turbidity and sedimentation management plans implemented to ensure proper protection and management of the water resources;
- Management plans need to be integrated and take a wide range of conservation and use objectives for specific ecosystems into account;
- All stakeholders of these systems need to become more closely engaged in the social and institutional decision making processes to manage these systems;
- The RRM approach that allow the unique characteristics of the ecosystem taken into consideration should be used in other case studies in South Africa to address the risks of multiple stressors affecting various endpoints in South African freshwater environments.

8.3 References

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APPENDIX A:

Yearly graphs over different Sampling Sites

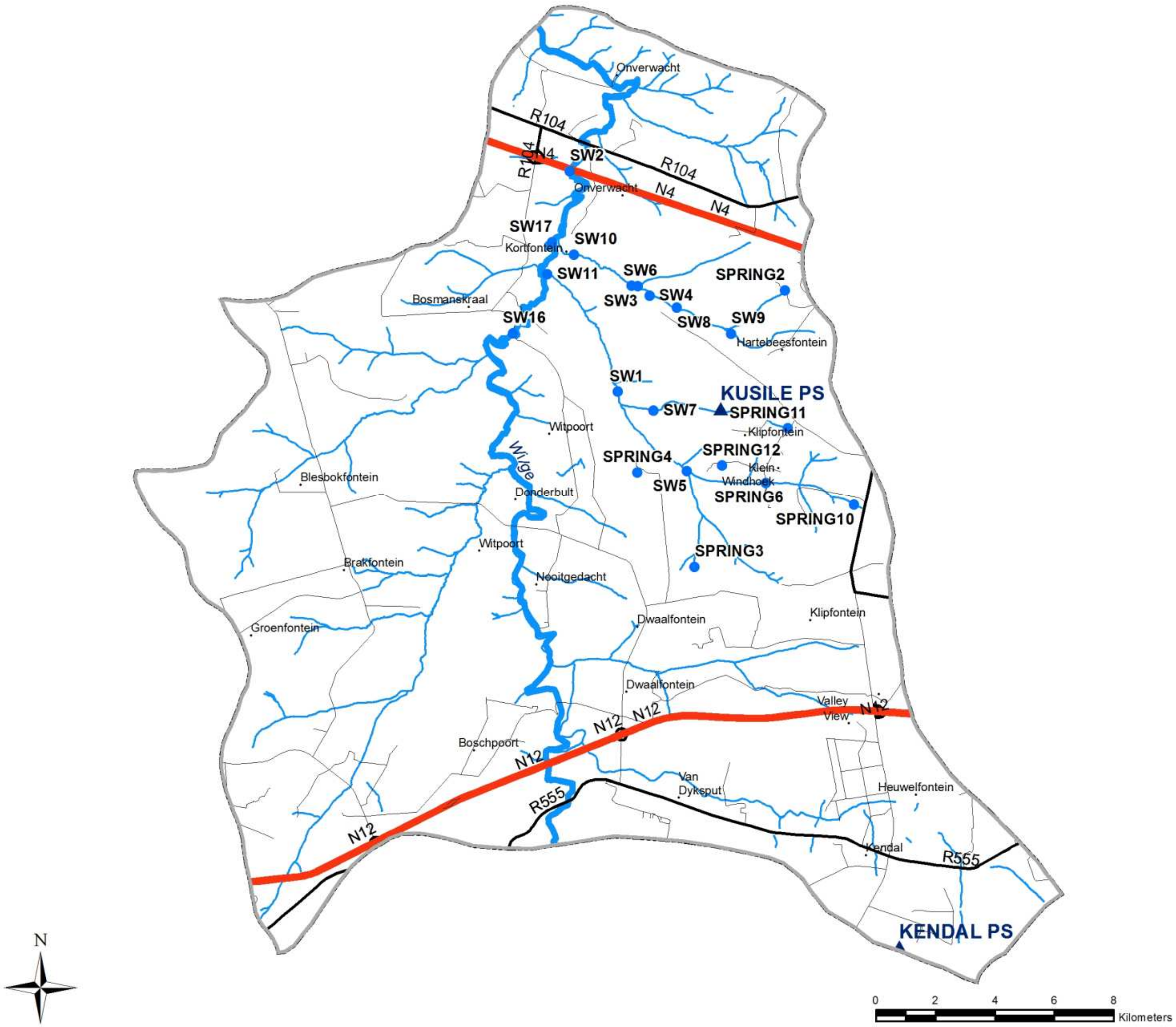

Figure A.1: Yearly pH data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

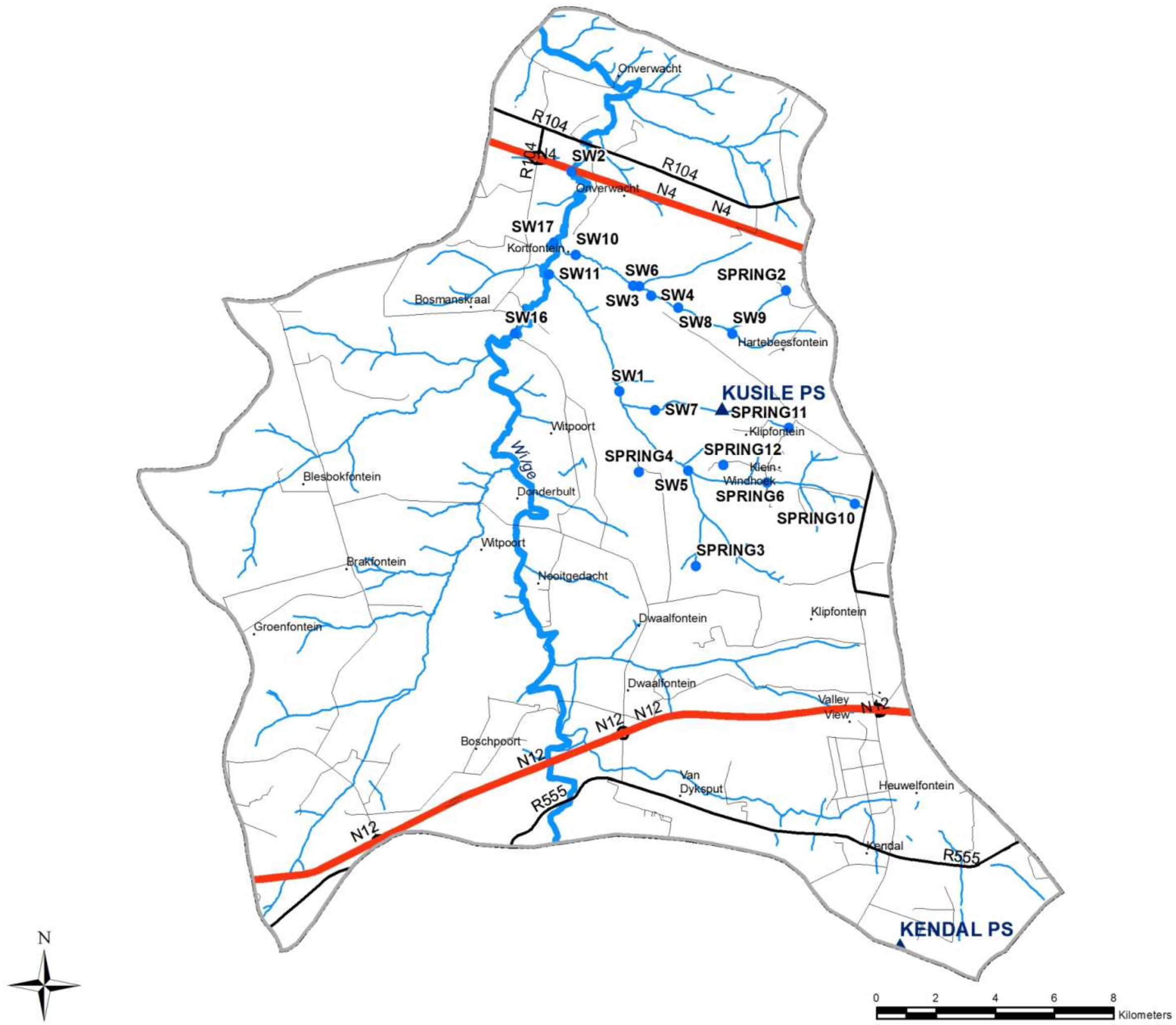

Figure A.2: Yearly Conductivity data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

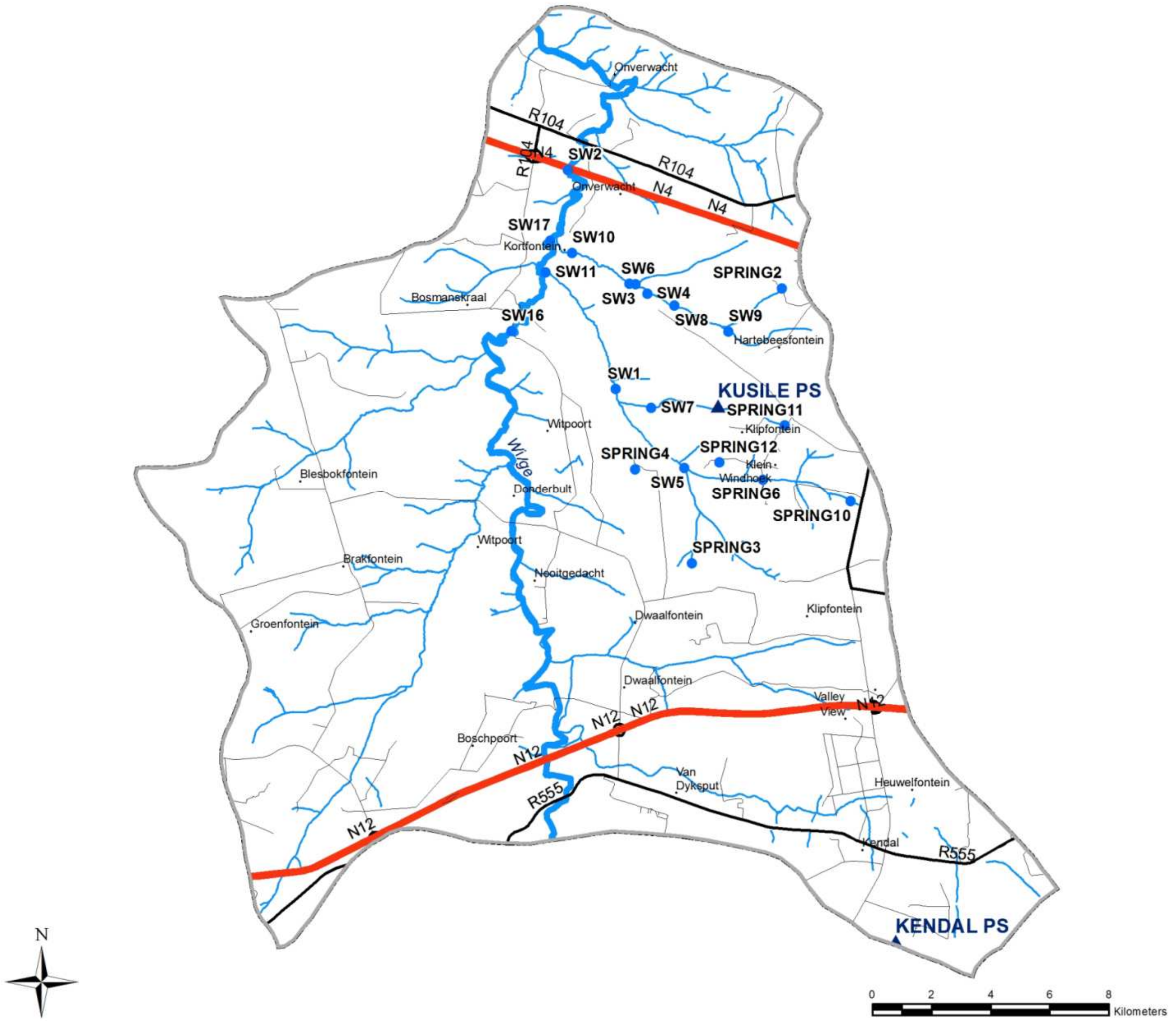

Figure A.3: Yearly Turbidity data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

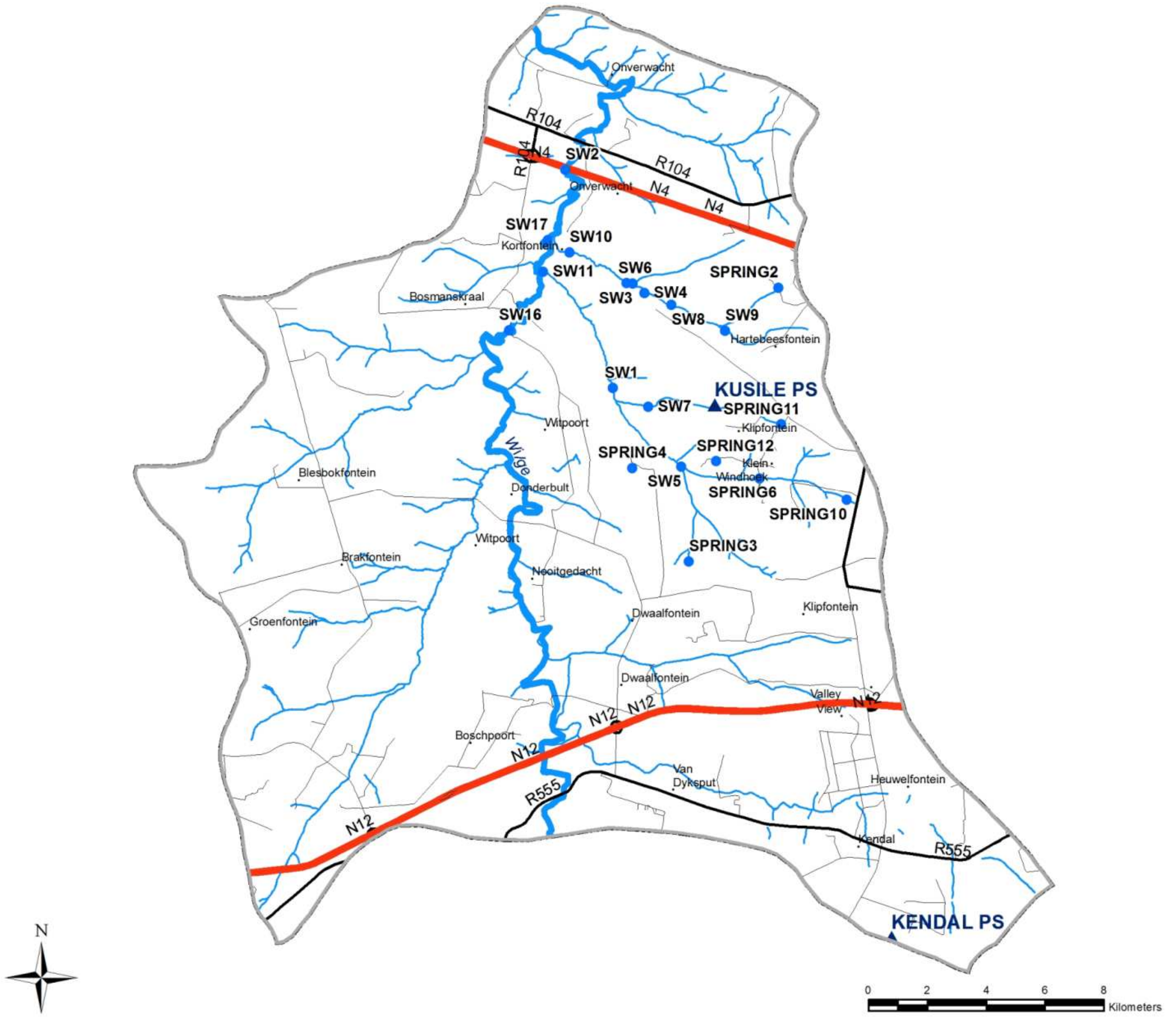

Figure A.4: Yearly Dissolved Oxygen data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

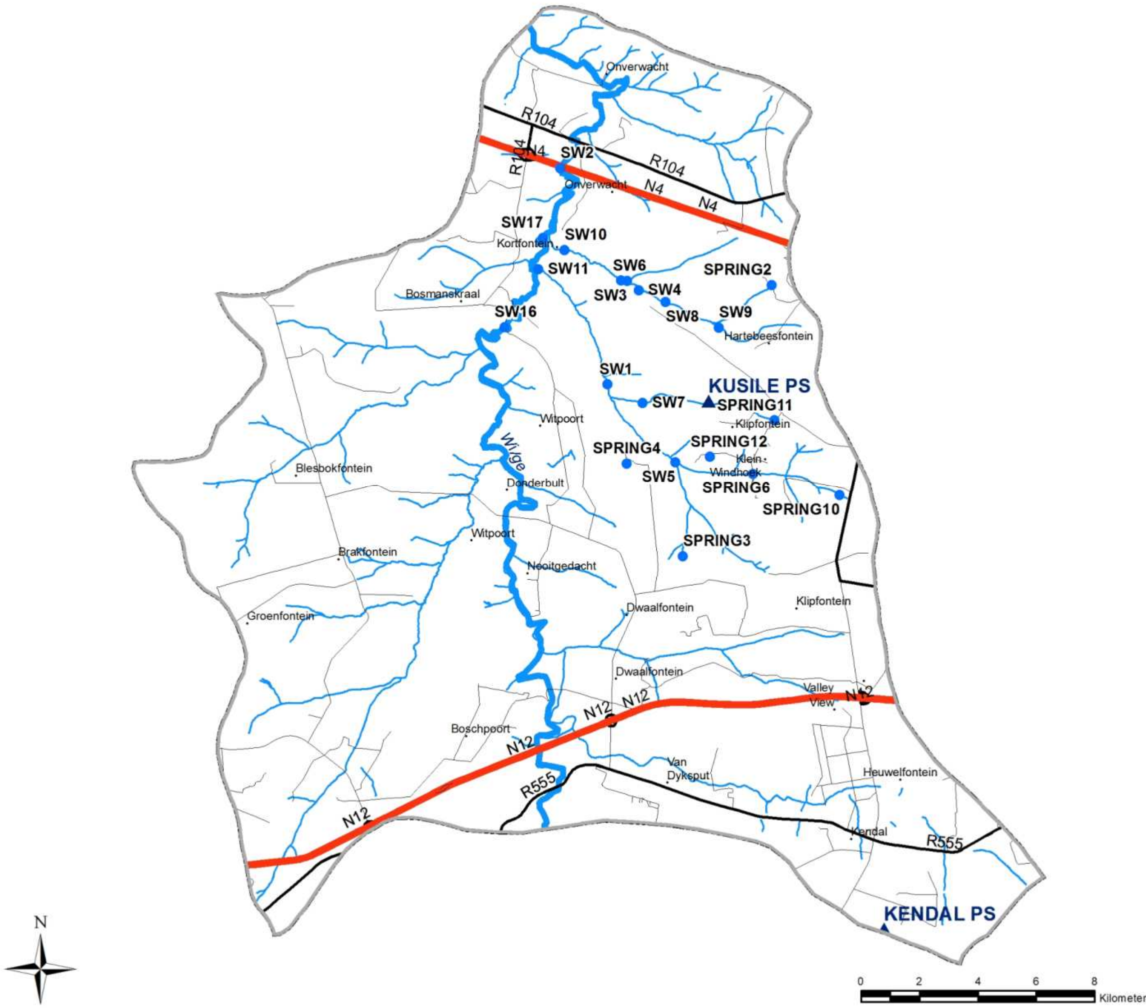

Figure A.5: Yearly Suspended Solids data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

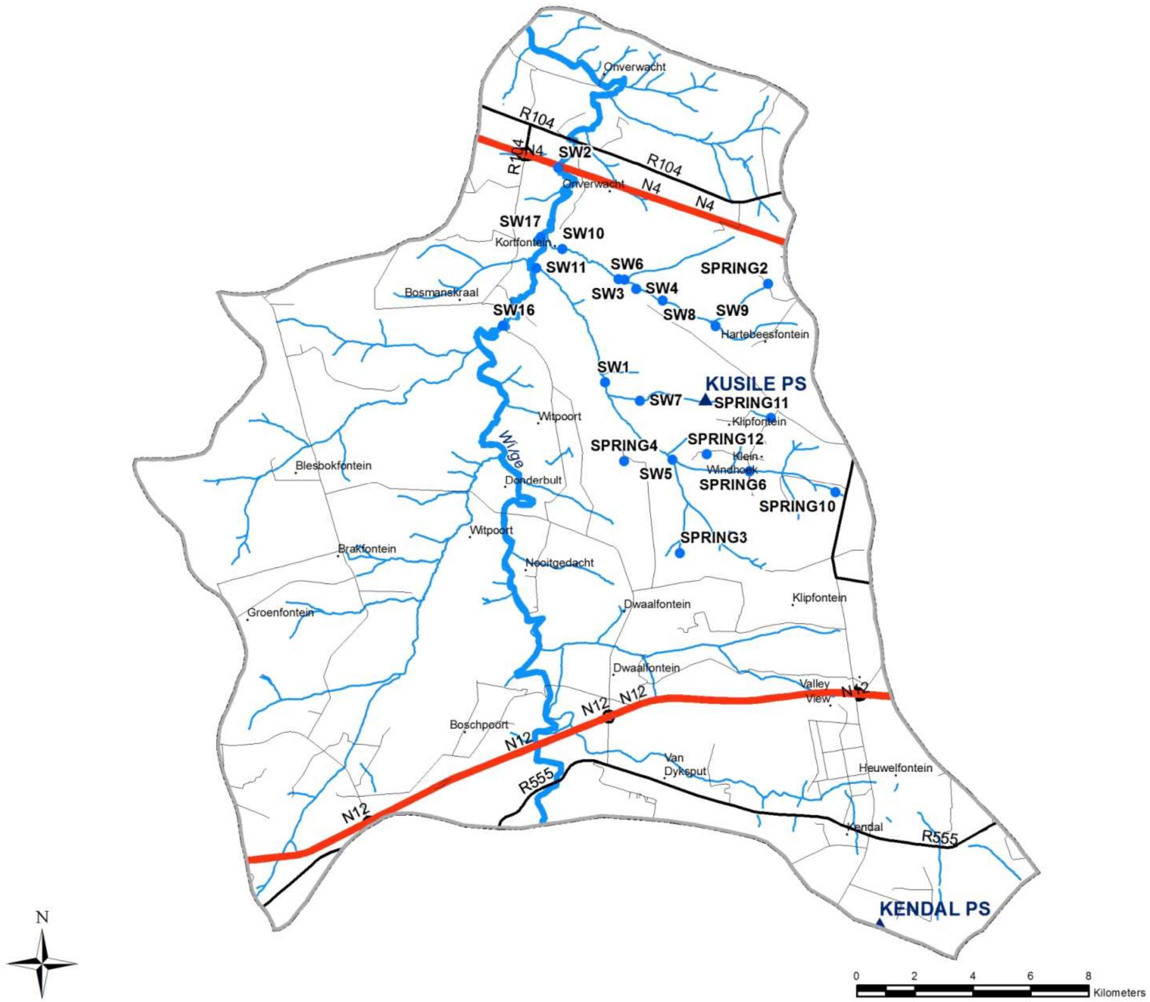

Figure A.6: Yearly Ammonia data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

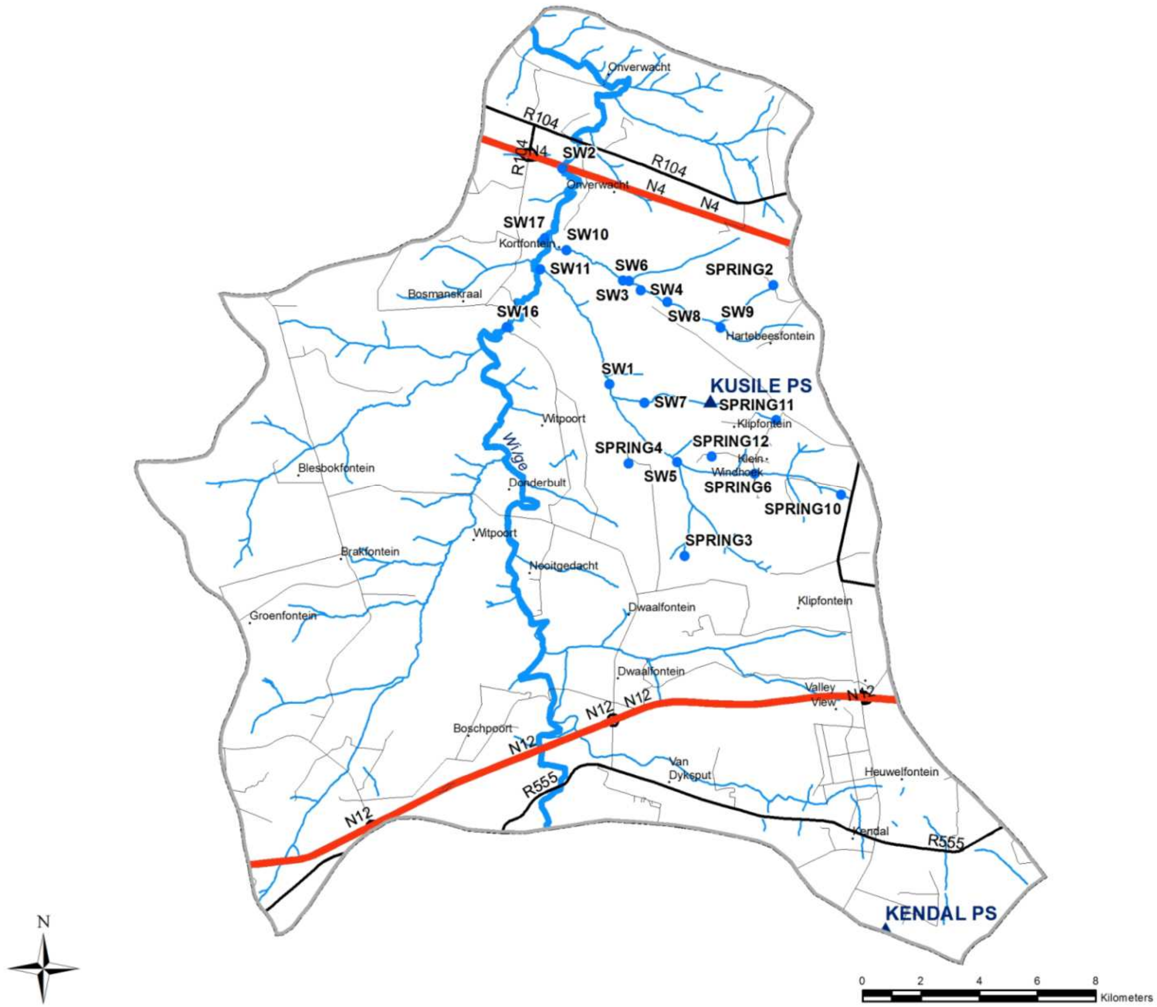

Figure A.7: Yearly Nitrate data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

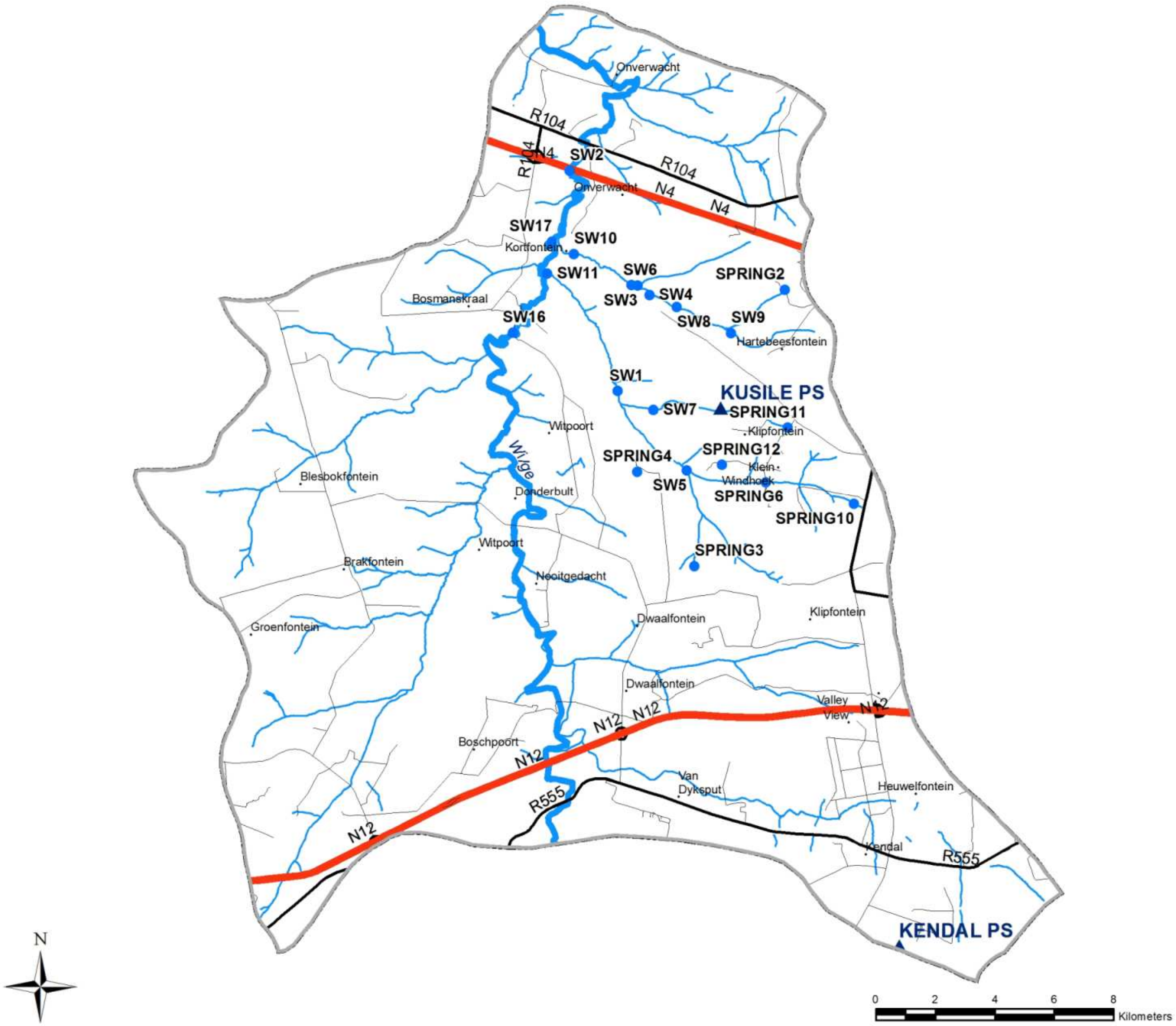

Figure A.8: Yearly Phosphate data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

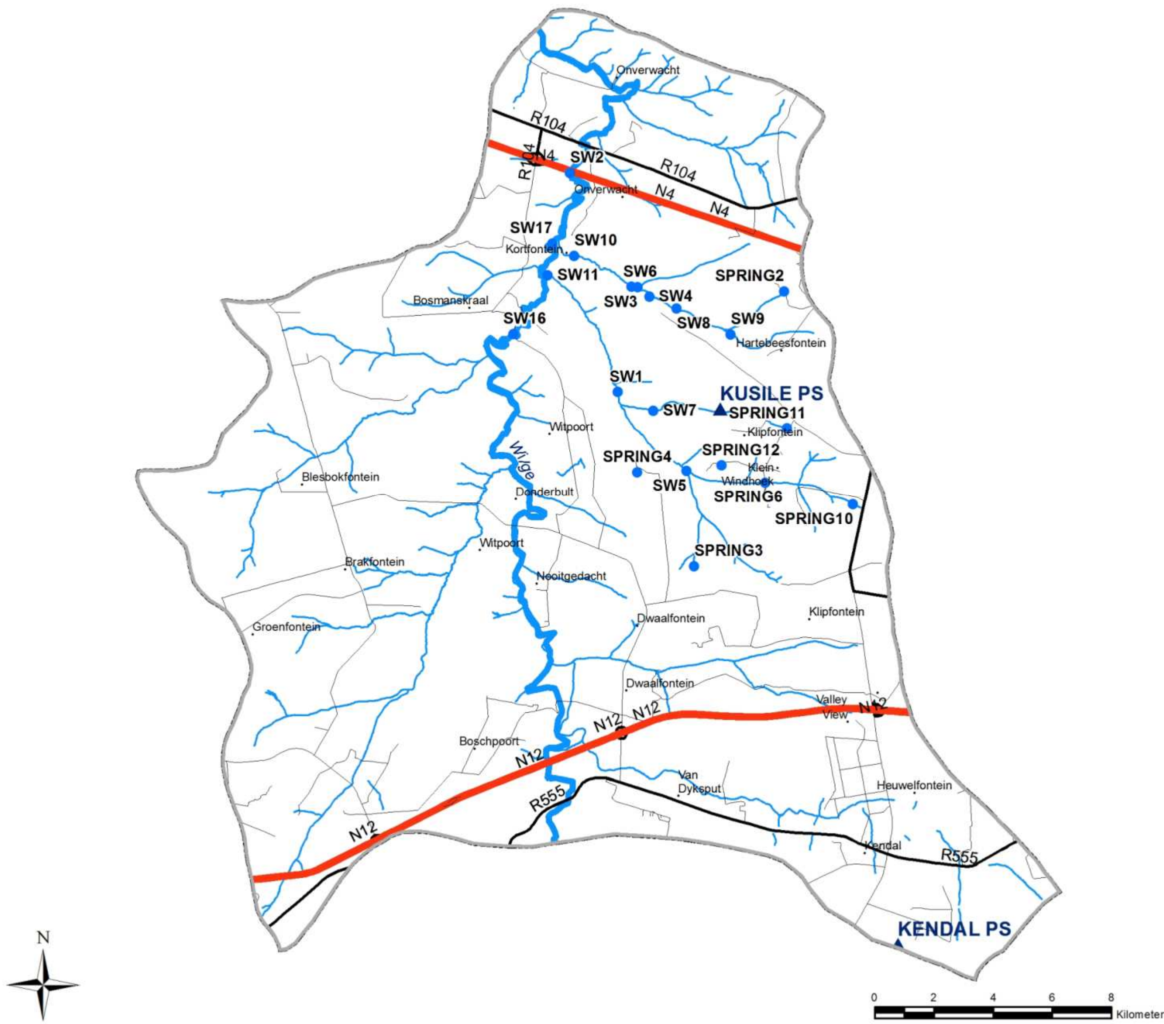

Figure A.9: Yearly Sulphate data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

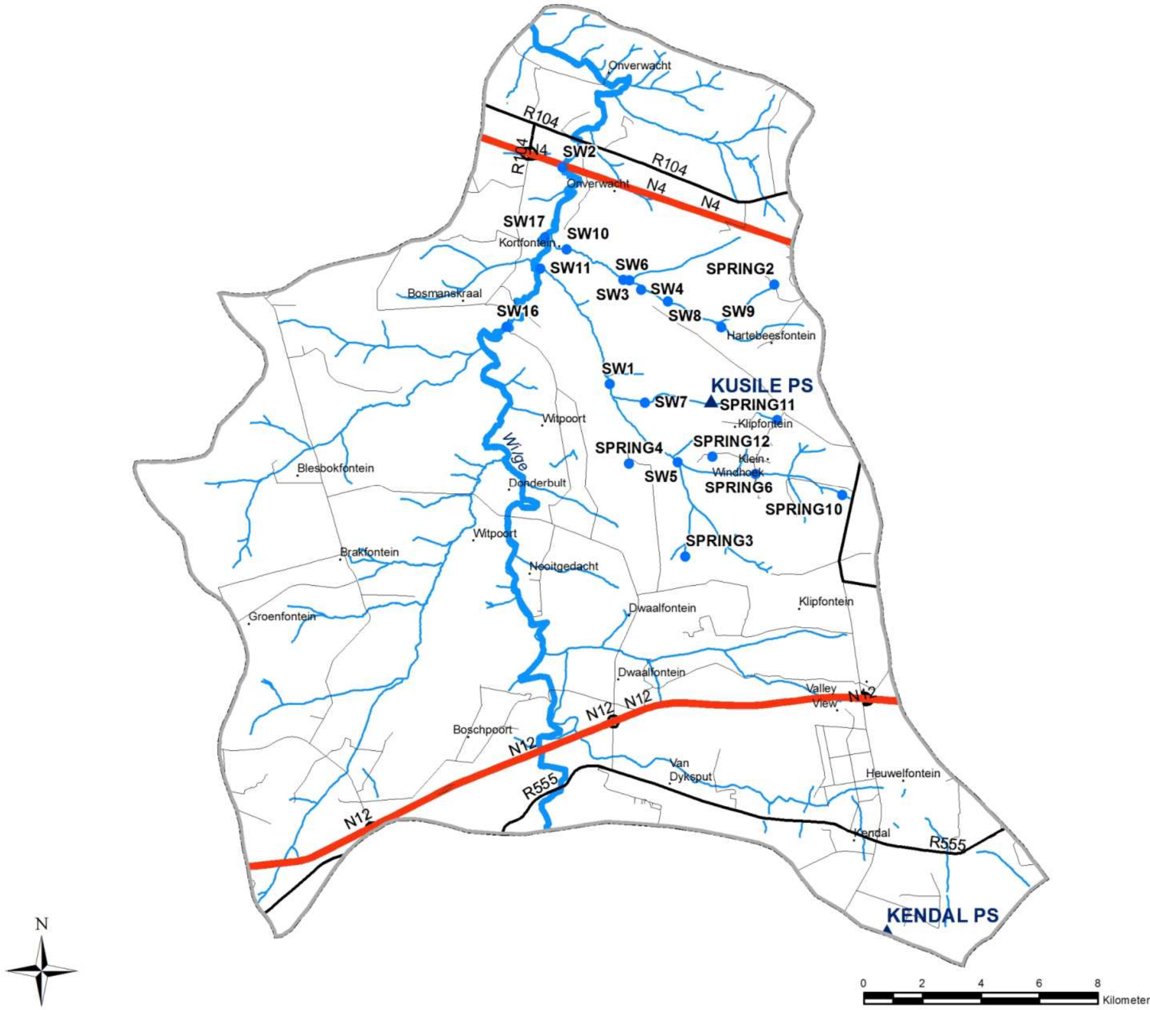

Figure A.10: Yearly Aluminium data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

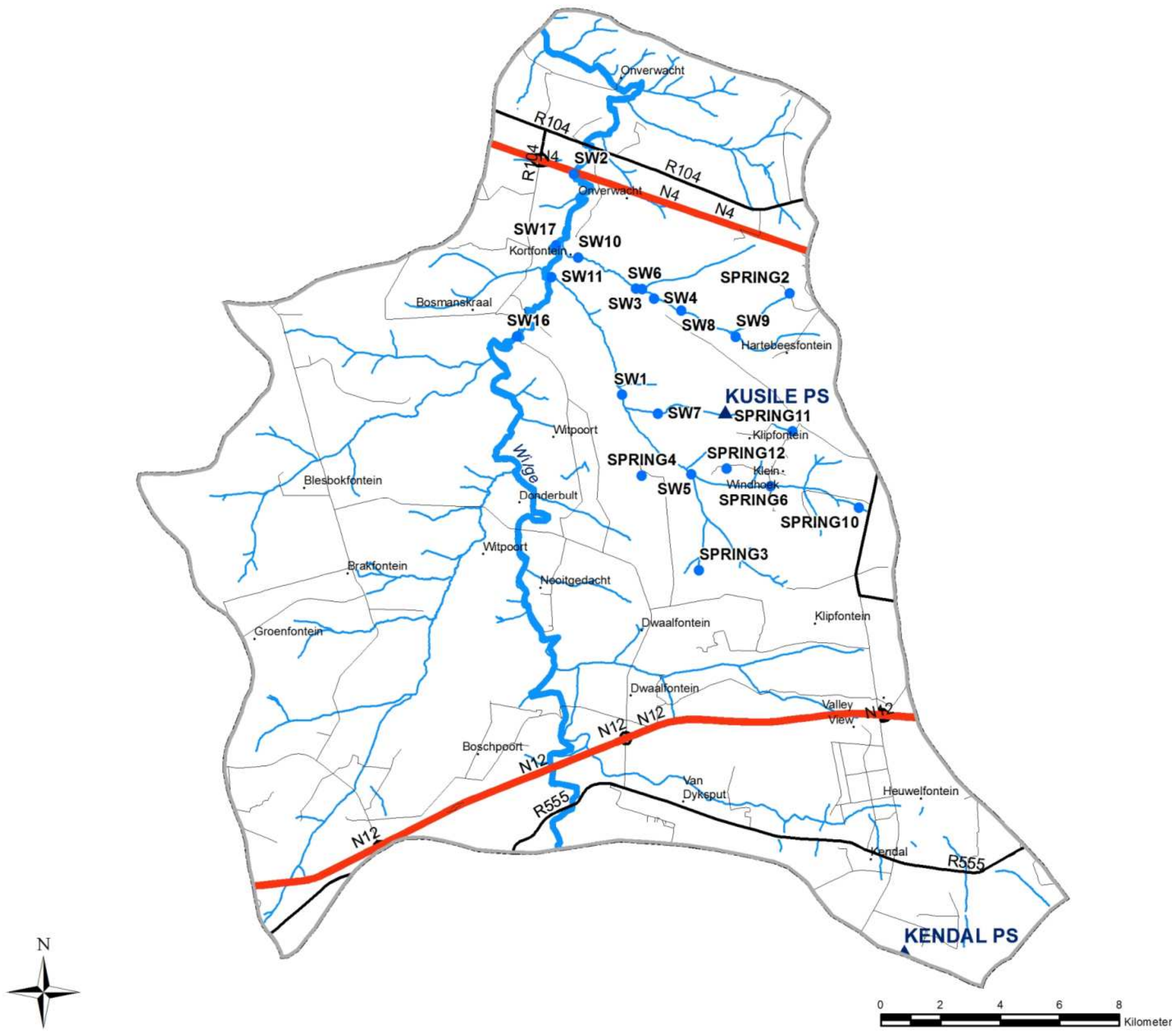

Figure A.11: Yearly Iron data for sampling sites in Wilge River sub-catchment B20F area for the period 2008-2014.

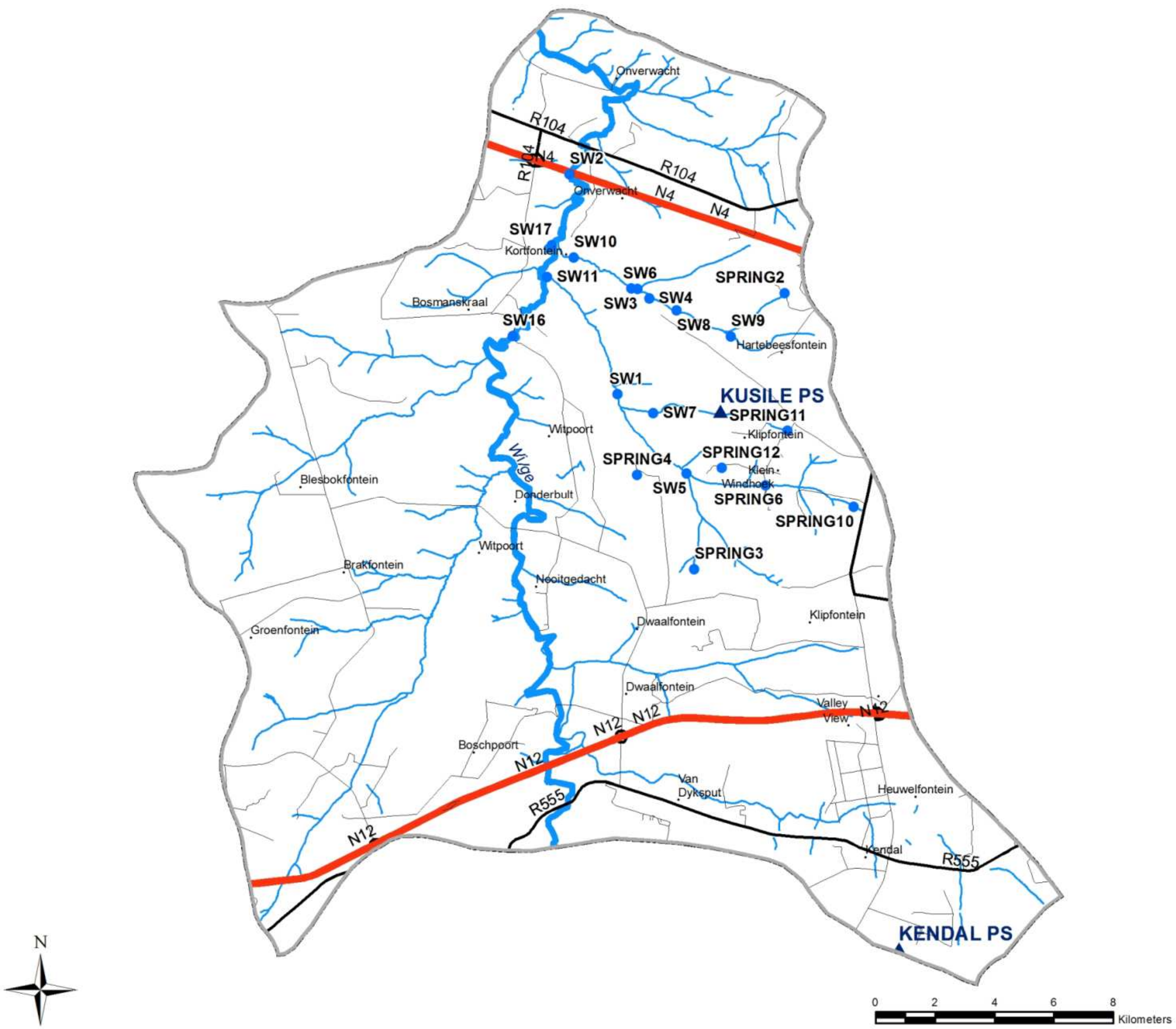

Figure A.12: Yearly Manganese data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

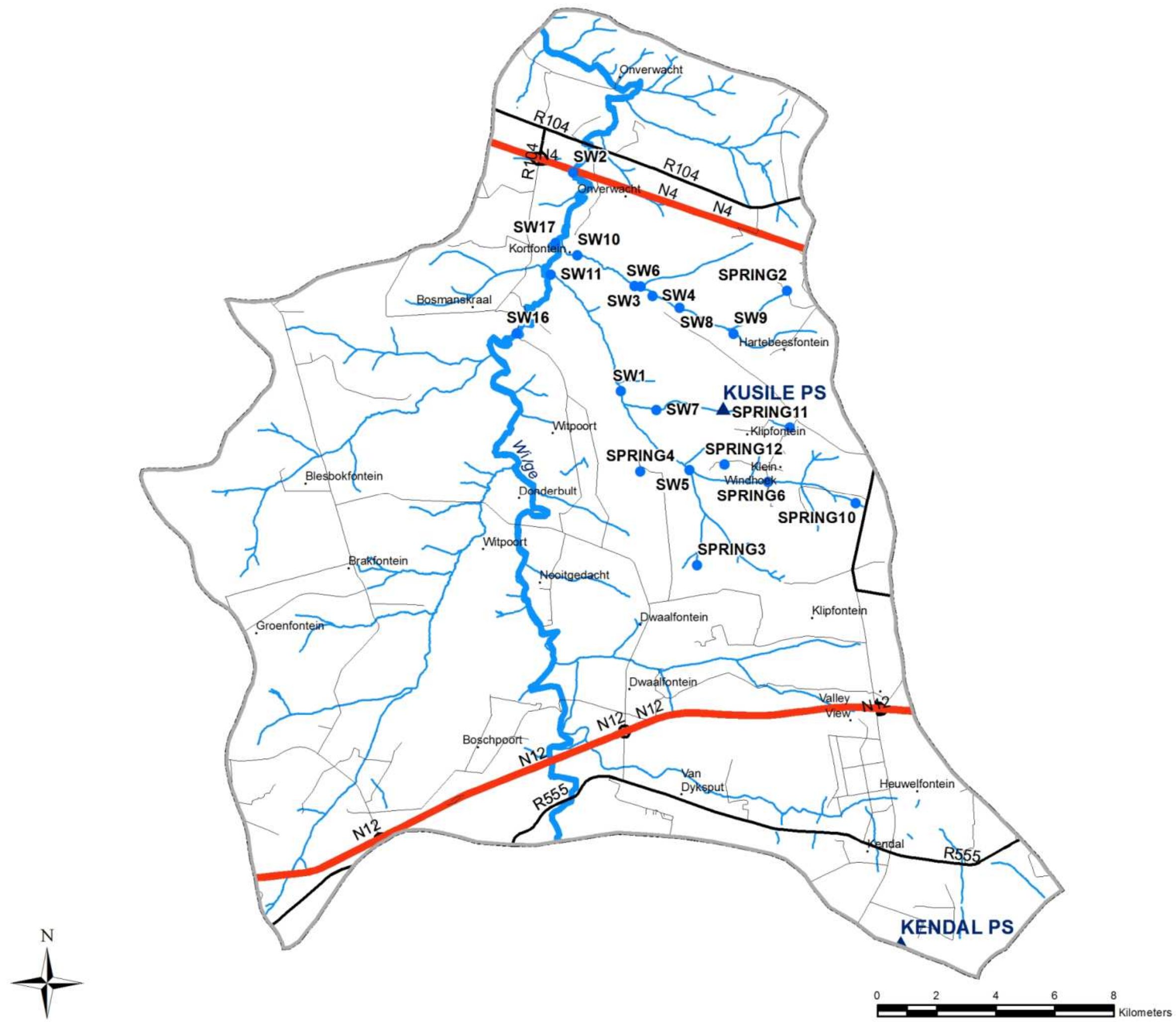

Figure A.13: Yearly Zinc data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

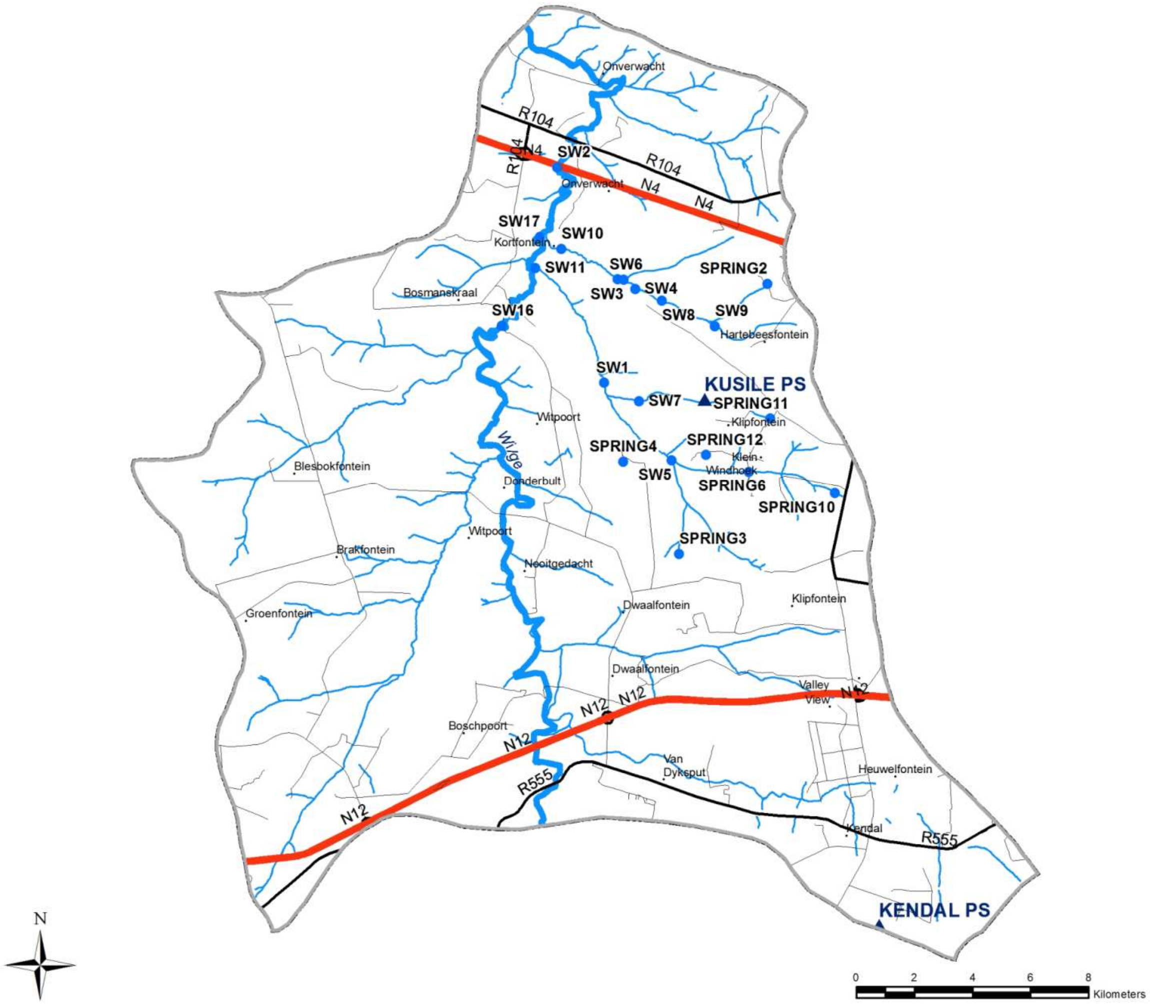

Figure A.14: Yearly Bromine data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

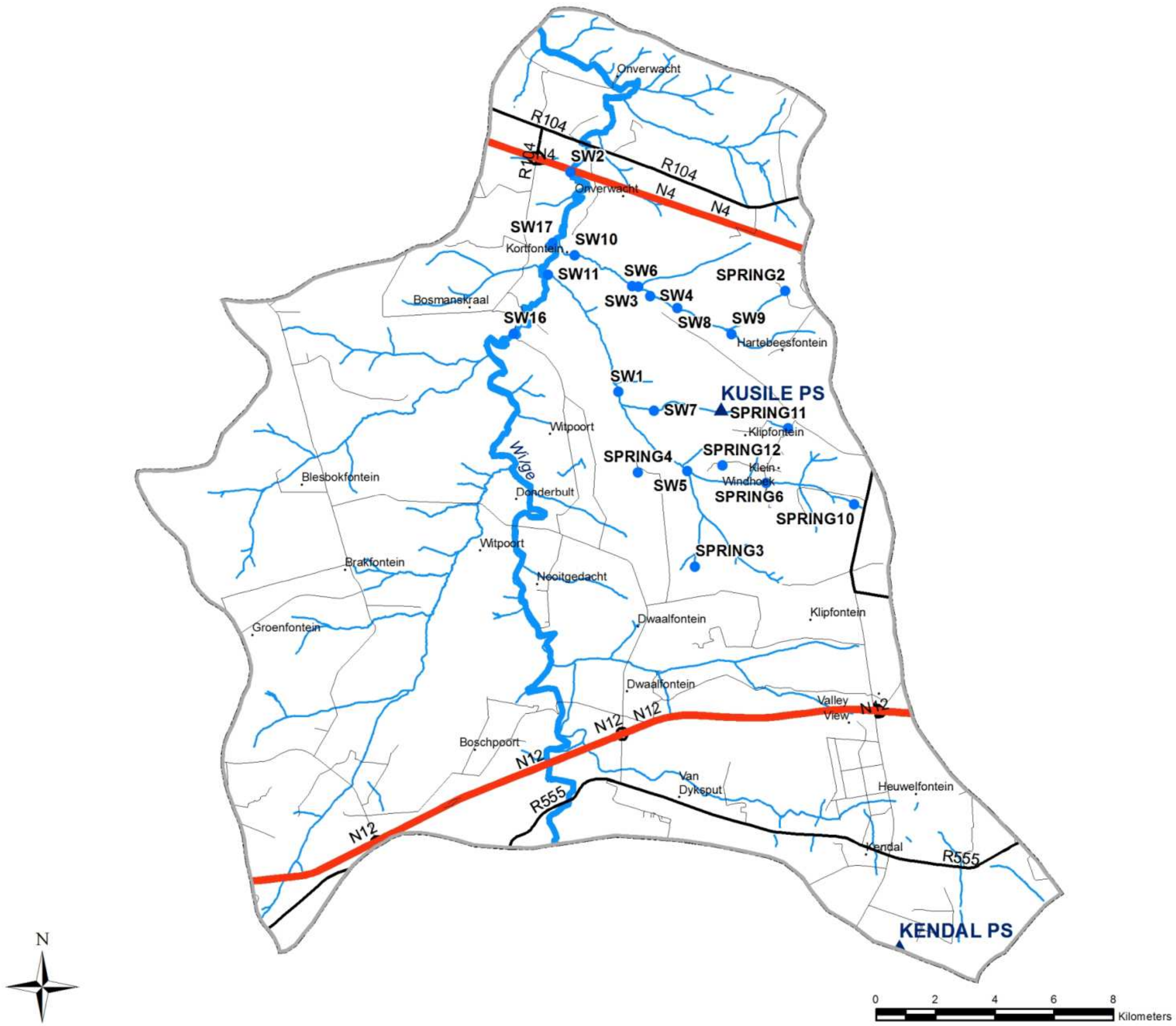

Figure A.15: Yearly Cadmium data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

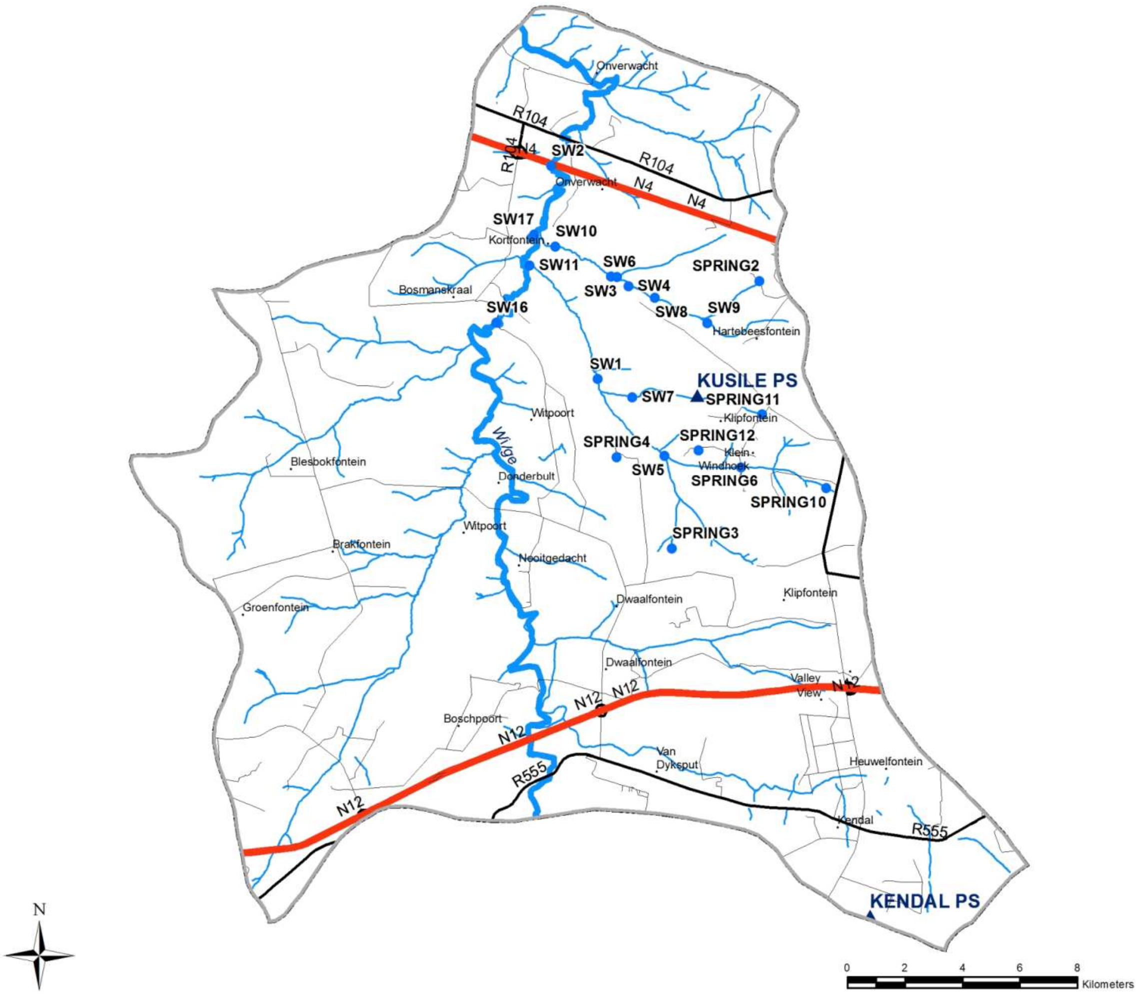

Figure A.16: Yearly Lead data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

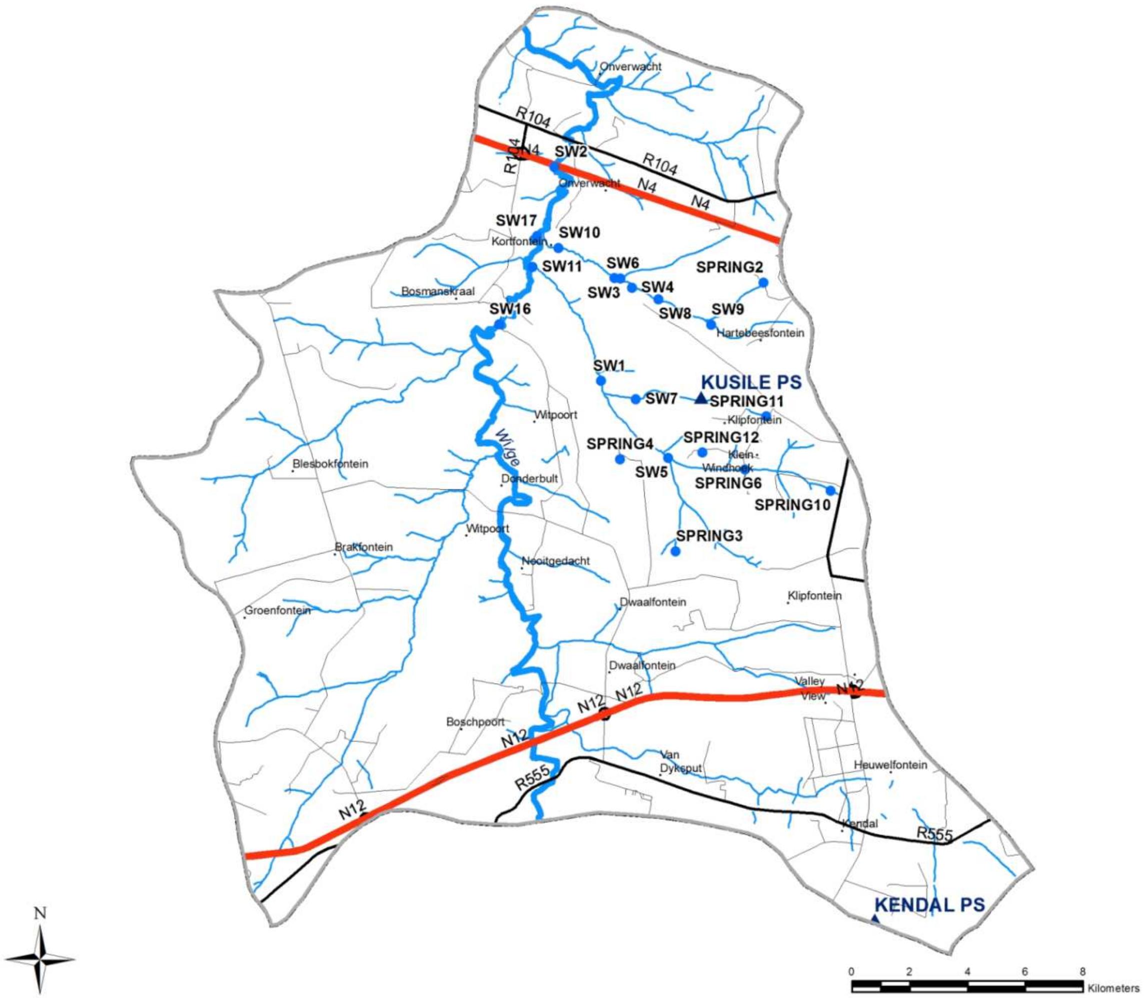

Figure A.17: Yearly Mercury data for sampling sites in the Wilge River sub-catchment B20F area for the period 2008-2014.

APPENDIX B:

PCA tables for combined years (2008-2014) and individual years 2008-2014

Table A.1: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2008-2014.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 10	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6	SW 7	SW 8	SW 9	SW 10	SW 11	SW 16	SW 17
pH	6.61±0.56	6.87±0.42	5.92±0.62	7.17±0.47	5.36±0.87	6.99±0.51	7.16±0.40	7.53±0.33	7.83±0.32	7.86±0.41	7.51±0.30	7.05±0.87	7.43±0.42	7.88±0.25	7.27±0.53	7.64±0.26	7.81±0.35	7.61±0.22	7.82±0.40	7.84±0.36
Conductivity (mS/m)	8.01±4.42	10±3.38	9.91±9.12	39±25	39±44	8.42±4.58	11±2.74	28±13	31±8.24	20±5.78	14±2.90	35±17	14±6.93	16±4.21	12±2.74	13±7.84	13±1.98	19±6.63	34±7.55	31±7.69
Turbidity (NTU)	3.49±5.33	36±53	2.77±10	5.55±4.35	4.72±5.79	47±84	45±98	199±320	74±123	43±51	210±234	11±8.66	158±162	221±318	233±237	267±297	120±148	275±510	17±43	75±162
Dissolved Oxygen (%Sat)	61±35	50±17	61±22	63±19	56±19	58±27	47±25	64±22	65±26	78±120	63±19	67±20	63±27	72±27	67±25	70±25	73±28	70±24	63±22	64±23
Suspended Solids (μg/L)	15672±42814	59590±71610	19130±55251	9033±20728	19129±24768	31848±65729	36258±119882	113830±166781	56458±65717	31138±49883	110032±150963	77081±485952	66721±50633	117287±159427	82400±58474	72806±69351	52373±49151	77757±104833	19248±28485	51689±72509
Nutrients																				
Ammonia [NH_3] (μg/L)	619±1454	303±427	186±241	294±507	496±378	138±212	229±373	176±214	123±154	158±175	241±399	148±137	205±255	190±207	328±498	215±221	219±187	243±233	124±131	185±190
Nitrate [NO_3-N] (μg/L)	4716±4616	793±891	4393±2423	461±292	12674±7726	1235±767	203±173	447±364	386±233	181±204	286±353	270±251	272±193	764±785	371±269	461±368	310±218	584±390	664±1561	447±252
Phosphate [PO_4] (μg/L)	438±727	319±150	339±200	349±183	400±0.00	355±200	534±840	409±357	351±187	355±202	353±194	355±202	432±740	113±217	50±0.00	125±149	50±0.00	104±131	308±208	331±257
Sulphate [SO_4] (μg/L)	3242±2933	8438±6457	8903±38066	160067±130443	169715±299796	8465±17460	2615±2032	85936±62685	54591±21319	17390±24189	12020±6496	130686±80862	22387±27027	10114±5444	15062±6129	17426±18451	14920±8763	41093±30821	52431±26176	54023±23061
Metals																				
Aluminium [Al] (μg/L)	73±71	150±183	82±232	378±2007	334±225	369±537	574±1066	724±997	595±873	1644±2305	2254±2920	236±331	1010±1386	482±712	729±513	849±643	491±234	592±595	215±371	518±789
Iron [Fe] (μg/L)	105±230	4988±21432	189±659	528±767	67±110	718±2299	1601±1493	1259±2511	900±1419	1673±1175	2963±3551	1325±2589	3058±3820	2865±3674	3555±4358	3537±5244	2495±2334	3819±13307	747±1821	1314±3128
Manganese[Mn] (μg/L)	91±210	308±1250	39±63	182±196	555±574	107±276	404±378	149±146	99±107	112±242	175±390	287±333	148±94	271±179	185±199	161±118	64±38	108±116	147±94	121±92
Zinc [Zn] (μg/L)	23±48	14±17	13±25	17±66	25±1.41	12±17	16±29	17±25	16±27	11±18	13±16	11±15	14±20	6.56±7.45	10±12	14±17	8.68±11	24±61	7.23±11	20±65
Bromine [Br] (μg/L)	86±33	161±103	84±19	62±35	36±1.41	100±0.00	150±112	59±34	98±36	64±43	61±39	65±36	58±37	68±39	57±37	57±36	51±39	55±37	91±24	91±29
Cadmium [Cd] (μg/L)	0.46±0.19	0.50±0.00	0.38±0.21	0.44±0.16	0.11±0.01	0.50±0.00	0.75±0.55	0.32±0.23	0.43±0.16	0.42±0.18	0.49±0.03	0.37±0.20	0.87±1.83	0.43±0.17	0.40±0.20	0.40±0.20	0.42±0.18	0.39±0.20	0.41±0.18	0.44±0.18
Lead [Pb] (μg/L)	3.42±11	0.56±0.19	0.98±2.62	0.48±0.48	0.04±0.01	0.50±0.00	0.94±1.08	2.53±5.39	1.14±1.41	0.81±0.64	1.58±1.59	0.44±0.34	2.16±1.72	1.35±1.98	2.24±2.07	2.23±2.88	1.33±1.33	2.19±3.19	0.71±1.13	1.11±1.71
Mercury [Hg] (μg/L)	0.49±0.14	0.50±0.00	0.47±0.18	0.44±0.14	0.44±0.14	0.50±0.00	0.50±0.00	0.52±0.12	0.86±1.29	0.54±0.09	0.57±0.20	0.48±0.15	0.61±0.30	0.93±1.49	0.69±0.62	0.62±0.37	0.50±0.02	0.50±0.02	0.47±0.11	0.47±0.10

Table A.2: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2008 only.

Physical Parameters	Spring 2	Spring 6	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5
pH	6.71±0.44	7.49±0.29	6.86±0.38	7.27±0.51	7.67±0.22	7.88±0.09	7.91±0.37	7.70±0.29	7.54±0.20
Conductivity (mS/m)	6.5±0.58	48±26	6.25±0.5	11±1.71	24±20	22±15	19±6.16	16±3.87	50±20
Suspended Solids (μg/L)	5667±0.00	1400±0.00	7000±0.00	15667±0.00	231533±0.00	50400±0.00	76667±0.00	180867±0.00	3600±0.00
Nutrients									
Nitrate [NO_3-N] (μg/L)	3668±0.00	243±0.00	2125±0.00	150±0.00	138±0.00	228±0.00	138±0.00	135±0.00	197±0.00
Phosphate [PO_4] (μg/L)	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00
Sulphate [SO_4] (μg/L)	1028±0.00	185235±0.00	3065±0.00	1215±0.00	113283±0.00	48293±0.00	8578±0.00	6313±0.00	198387±0.00
Metals									
Aluminium [Al] (μg/L)	50±0.00	50±0.00	150±0.00	723±0.00	1313±0.00	930±0.00	2635±0.00	1820±0.00	100±0.00
Iron [Fe] (μg/L)	31±0.00	170±0.00	40±0.00	1028±0.00	706±0.00	496±0.00	1853±0.00	7243±0.00	237±0.00
Manganese[Mn] (μg/L)	31±0.00	170±0.00	40±0.00	1028±0.00	706±0.00	496±0.00	1853±0.00	7243±0.00	237±0.00

Table A.3: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2009 only.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 10	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6
pH	6.72±0.39	7.02±0.31	6.43±0.82	7.21±0.23	5.19±0.52	6.85±0 .45	7.17±0.28	7.46±0.17	7.78±0.14	7.88±0.31	7.50±0.25	7.4 9±0.18	7.54±0.00
Conductivity (mS/m)	6.61±1.05	9.29±1.38	10±3.89	45±13	15±3.91	7±0.77	12±1.33	3 0±4.05	30±4.05	20±4.03	14±2.9	36±7.29	30±0.00
Turbidity (NTU)	1.5±0.84	9.01±4.34	30±43	13±3.84	9.84±0.00	16±23	29±24	494 ±556	207±38	32±13	472±426	14±10	225±0.00
Suspended Solids (µg/L)	2073±3650	26571±25156	84886±121118	7891±8453	16720±2408 8	10436±19910	17945±13374	190964±277865	63418±62298	3990 9±90808	241582±288812	8327±16910	104800±0.00
Nutrients													
Nitrate [NO_3-N] (µg/L)	3387±705	687±817	3937±6375	571±409	13636±6685	1856±411	15 2±72	210±131	311±186	113±74	379±743	164±128	100±0.00
Phosphate [PO_4] (µg/L)	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400 ±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0. 00
Sulphate [SO_4] (µg/L)	2030±2007	6207±3392	2236±1876	177277±91507	1964±1890	418 5±1240	1548±1311	94999±40866	58893±23228	11806±6541	8783 ±3558	129072±46626	74750±0.00
Metals													
Aluminium [Al] (µg/L)	58±81	190±131	363±545	120±191	190±83	407±401	853±1850	1243 ±1210	1475±1755	1928±1623	5363±5017	251±424	1520±0.00
Iron [Fe] (µg/L)	23±14	787±515	478±579	155±178	100±179	223±252	1164±761	633 ±432	985±965	1462±778	3623±3008	405±322	960±0.00
Manganese[Mn] (µg/L)	0.00±0.00	17±24	104±123	138±93	187±85	24±29	376±305	133±12 0	12±13	74±83	51±20	106±85	310±0.00

Table A.4: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2010 only.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 10	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6	SW 9	SW 16	SW 17
pH	6.75±0.19	6.80±0.63	6.33±0.62	7.51±0.17	5.13±0.74	6.78±0 .38	7.26±0.41	7.63±0.20	7.86±0.11	7.92±0.17	7.58±0.14	7.5 2±0.17	7.73±0.00	8.00±0.03	7.87±0.13	7.84±0.12
Conductivity (mS/m)	6.93±0.31	9.09±1.30	14±19	70±7.52	41±50	7.80±0.63	12±1.08	47±10	35±4.09	21±4.72	15±2.10	55±8.64	46±0.00	36±2.08	31±4 .57	35±3.89
Turbidity (NTU)	1.60±0.98	9.06±7.96	4.49±6.31	3.76±2.10	5.49±7.88	30±40	1 7±7.97	84±107	37±49	29±18	289±291	7.50±7.00	6.80±0.00	80±9 4	6.35±2.44	18±25
Dissolved Oxygen (%Sat)	57±10	52±13	46±15	61±11	53±20	53±21	58±25	63±14	61±15	62±12	62±12	6 5±12	68±0.00	54±1.00	58±20	62±16
Suspended Solids (µg/L)	7382±10774	25800±25221	17436±33397	6467±9948	21200±3349 5	46420±49649	11650±6337	45583±57027	29183±38801	16883±1 3278	105917±103377	5983±9362	3000±0.00	50200±79953	6075±6150	13775±16828
Nutrients																
Nitrate [NO_3-N] (µg/L)	3282±1107	1337±955	2010±1261	670±309	14274±6772	1828±280	264±259	413±321	495±326	197±199	265±277	297±306	150±0.00	2 53±75	734±312	668±249
Phosphate [PO_4] (µg/L)	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	679 ±	567±	445±	400±0.00	400±0.00	400±0.00	400±0.00	400±0.00	40 0±0.00	400±0.00
Sulphate [SO_4] (µg/L)	4351±2633	6737±4556	25372±76326	334628±44188	140116±307 398	4449±1741	2895±2678	174882±54429	68817±19738	8922±44 49	8402±2599	225614±57708	168370±0.00	71917±14190	40591± 20583	74958±19118
Metals																
Aluminium [Al] (µg/L)	35±25	64±36	51±14	65±86	450±190	111±130	473±569	572±991	393 ±364	1726±1707	2643±2076	99±82	130±0.00	647±499	85±31	179± 179
Iron [Fe] (µg/L)	34±37	100±77	79±55	90±37	75±98	70±82	1489±780	347±515	394±2 50	1390±821	1934±1239	327±286	540±0.00	543±184	160±66	229± 109
Manganese[Mn] (µg/L)	15±22	25±18	29±17	95±47	572±670	54±98	427±264	119±81	58±96	6 6±83	83±81	151±61	90±0.00	12±13	108±123	108±123

Table A.5: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2011 only.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 10	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6	SW 11	SW 16	SW 17
pH	6.69±0.61	7.06±0.18	6.09±0.62	7.34±0.15	5.17±1.10	7.40±0 .71	7.18±0.46	7.51±0.24	7.77±0.32	7.83±0.27	7.64±0.26	7.3 6±0.26	7.54±0.42	7.25±0.69	7.73±0.63	7.80±0.28
Conductivity (mS/m)	10±5,43	10±3	11±11	54±5.89	46±47	11±9.38	13±2.25	34±6.83	36 ±3.66	20±4.61	17±2.99	43±8.34	16±3.91	21±19	38±3.67	35±2.4 9
Turbidity (NTU)	7.52±8.84	35±33	1.22±2.05	7.40±6.50	4.66±4.18	89±130	61±1 52	218±452	72±185	55±72	129±112	10±6.02	76±53	13±0.00	14±14	77±193
Dissolved Oxygen (%Sat)	78±59	52±22	56±15	60±11	52±21	70±40	47±26	64±13	59±26	129±236	60±1 1	66±14	58±13	55±7	59±18	54±20
Suspended Solids (µg/L)	22350±33182	72255±80624	650±1458	6067±5635	19550±19279	6 8800±123756	87167±229476	111217±219172	49550±100969	268 17±33394	56850±52841	11683±10591	44160±22191	5000±4243	1 5383±17720	50417±96798
Nutrients																
Nitrate [NO_3-N] (µg/L)	5345±5592	511±500	3531±1120	568±233	15310±8467	806±693	21 4±176	444±307	346±247	257±370	150±0.00	278±211	150±0.00	54 5±559	390±262	377±278
Phosphate [PO_4] (µg/L)	400±0.00	400±0.00	400±0.00	435±120	400±0.00	438±126	400±0 .00	453±185	400±0.00	400±0.00	400±0.00	400±0.00	554±344	40 0±0.00	400±0.00	400±0.00
Sulphate [SO_4] (µg/L)	4210±4917	8616±5922	14623±47678	223417±36565	164684±325 547	17000±36921	2886±1927	107350±33376	58050±12555	10190 ±4777	9998±4802	159950±59120	14120±2932	52580±66921	5735 0±15838	60617±14651
Metals																
Aluminium [Al] (µg/L)	85±21	126±55	85±0.00	85±0.00	503±227	861±990	250±232	323±4 54	391±521	1068±939	2289±1964	88±45	1630±1195	325±233	213± 333	696±1027
Iron [Fe] (µg/L)	55±105	223±210	40±35	77±60	25±0.00	429±591	1658±1450	270±3 06	320±357	1328±771	2161±1494	278±301	1454±942	295±120	175 ±248	433±709
Manganese[Mn] (µg/L)	380±127	60±0.00	60±0.00	139±110	613±646	90±0.00	513±452	70 ±20	160±0.00	163±170	156±122	185±199	50±0.00	50±0.00	50±0. 00	50±0.00

Table A.6: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2012 only.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 10	Spring 11	Spring 12	SW 1	SW 2	SW 3
pH	6.29±1.13	6.44±0.17	5.57±0.53	7.03±0.27	7.00±0.00	6.67±0 .44	6.46±0.25	7.43±0.61	7.71±0.54	7.82±0.81
Conductivity (mS/m)	10±10	7.50±1.27	6.92±2.01	17±11	134±0.00	7.23±1.19	12±1.2 7	18±5.63	26±9.39	18±8.66
Turbidity (NTU)	2.76±2.51	33±25	0.15±0.14	5.76±3.28	0.00±0.00	29±29	12±4. 09	330±312	73±90	60±71
Dissolved Oxygen (%Sat)	64±23	50±0.00	67±19	68±21	78±0.00	55±10	2.75±3.18	61±31	72± 31	68±31
Suspended Solids (µg/L)	18120±7815	54900±63498	14600±5079	37167±55463	37167±0.0 0	30100±21422	21100±15698	178022±107258	80520±54169	4746 7±36269
Nutrients										
Ammonia [NH_3] (µg/L)	1525±2367	5.00±0.00	115±204	310±537	763±0.00	11±6.56	67±8 8	150±201	161±232	164±226
Nitrate [NO_3-N] (µg/L)	7569±10123	250±141	5011±476	298±132	880±0.00	343±42	150±0 .00	580±324	420±255	264±103
Phosphate [PO_4] (µg/L)	2327±2644	400±0.00	806±10	667±231	400±0.00	806±11	4990±0. 00	1007±358	667±231	907±185
Sulphate [SO_4] (µg/L)	3090±2200	4530±792	2623±2197	56134±52112	664600±0.00	735 3±2409	2750±141	37055±20595	36363±16921	22511±43239
Metals										
Aluminium [Al] (µg/L)	95±107	145±64	19±12	59±60	64±0.00	249±253	580±0.00	825±109 7	537±636	2595±4433
Iron [Fe] (µg/L)	143±275	280±198	463±1475	1053±1317	46±0.00	905±1443	2300± 2871	1895±2952	980±1113	1756±1459
Manganese[Mn] (µg/L)	239±388	48±0.00	48±92	212±265	1237±0.00	740±976	1370±0.00	113±86	147±161	56±58
Zinc [Zn] (µg/L)	54±71	44±26	26±41	9.08±8.59	24±0.00	36±15	52±38	26±27	34±38	18±24
Bromine [Br] (µg/L)	14±0.00	13±0.00	71±18	24±10	13±0.00	13±0.00	13±0.00	26±18	7 0±19	15±7.11
Cadmium [Cd] (µg/L)	0.30±0.44	0.03±0.00	0.16±0.21	0.17±0.20	0.12±0.00	0.50±0 .00	0.03±0.00	0.11±0.16	0.17±0.20	0.04±0.01
Lead [Pb] (µg/L)	14±24	0.5±0.00	0.53±0.55	0.39±0.52	0.05±0.00	0.50±0.00	0. 50±0.01	2.13±2.36	1.66±1.76	1.17±0.84
Mercury [Hg] (µg/L)	0.43±0.48	0.12±0.00	0.68±0.32	0.16±0.00	0.16±0.00	0.12±0 .00	0.12±0.00	0.59±0.36	3.24±2.88	0.72±0.08

Physical Parameters	SW 4	SW 5	SW 6	SW 7	SW 8	SW 9	SW 10	SW 11	SW 16	SW 17
pH	7.26±0.5	6.96±0.33	7.31±0.62	7.91±0.511	7.48±0.31	7.63±0.81	7.80±0.35	7.69±0.22	7.71±0.46	7.69±0.55
Conductivity (mS/m)	11±1.25	20±11	10±3.25	14±5.58	10± 3.52	9.15±3.47	12±1.14	17±1.81	29±10	26±10
Turbidity (NTU)	262±236	14±10	222±214	291±360	349±25 9	322±340	200±224	513±750	16±25	72±98
Dissolved Oxygen (%Sat)	70±26	73±21	64±36	79±25	73±26	77±27	85±29	81±25	66±2 2	68±20
Suspended Solids (µg/L)	111200±73400	19167±10284	88250±59680	131333±107 866	118125±45041	87375±64957	82800±49595	150000±0.00	287 25±25919	84956±65153
Nutrients										
Ammonia [NH_3] (µg/L)	217±270	177±210	202±253	193±274	3 66±285	247±296	465±300	457±311	155±207	241±256
Nitrate [NO_3-N] (µg/L)	402±235	268±101	346±223	554±34 2	458±319	543±374	537±205	770±478	448±201	430±229
Phosphate [PO_4] (µg/L)	870±121	907±185	1650±1472	800± 0.00	400±0.00	400±0.00	400±0.00	400±0.00	667±231	927±219
Sulphate [SO_4] (µg/L)	1509 5±4704	67270±52559	14960±6736	10031±6160	14071±6080	1065 8±5587	14840±10290	31030±9184	38444±18980	36179±18131
Metals										
Aluminium [Al] (µg/L)	1599±1946	149±89	1550±2202	366±468	748 ±406	655±428	499±238	931±945	302±633	944±1260
Iron [Fe] (µg/L)	2067±1630	3292±5378	212 9±742	3263±5365	2137±1218	2395±886	1552±583	1378±795	453± 407	1123±945
Manganese[Mn] (µg/L)	89±59	366±311	125±62	307±289	167±185	2 20±152	65±35	124±113	103±60	133±121
Zinc [Zn] (µg/L)	21±18	11±11	24±29	10±6.99	16±19	17±16	16±15	22±24	15±1 6	50±109
Bromine [Br] (µg/L)	23±15	24±13	20±9.11	37±33	22±16	23±18	13±7	22±1 5	74±15	75±10
Cadmium [Cd] (µg/L)	0.40±0.00	0.2 3±0.25	0.13±0.15	0.16±0.21	0.05±0.00	0.03±0.02	0.03±0.01	0.04±0.01	0.12±0.15	0.23±0.32
Lead [Pb] (µg/L)	2.28±1.76	0.49±0.5 7	2.53±1.52	1.83±2.39	3.48±1.93	2.74±2.60	2.25±1.69	4.02± 4.49	0.59±0.71	1.06±1.42
Mercury [Hg] (µg/L)	0.94±0.31	0.5 7±0.37	1.19±0.49	0.59±0.42	1.33±1.22	1.30±0.72	0.45±0.00	0.45±0.00	0.12±0.00	0.18±0.00

Table A.7: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2013 only.

Physical Parameters	Spring 2	Spring 4	Spring 6	Spring 10	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6	SW 7	SW 8	SW 9	SW 10	SW 11	SW 16	SW 17
pH	6.90±0.00	5.65±0.44	6.73±1.03	6.50±0.00	7.57±0.39	8.09±0 .33	7.80±0.00	7.43±0.22	5.22±1.26	7.36±0.31	7.92±0.25	7.1 6±0.23	7.56±0.23	7.80±0.46	7.54±0.13	7.99±0.32	8.08±0.34
Conductivity (mS/m)	6.58±0.00	8.87±4.04	23±28	13±0.00	22±8.93	31±10	25±0.00	12 ±2.05	34±16	12±1.38	18±3.78	11±0.95	11±1.87	13±1.59	22±6.9 9	36±5.51	32±7.79
Turbidity (NTU)	1.60±0.00	0.10±0.12	4.98±3.21	0.70±0.00	203±334	90±161	17 ±0.00	264±226	12±12	249±140	158±271	289±251	336±332	129±13 2	285±433	35±87	118±244
Dissolved Oxygen (%Sat)	82±0.00	88±16	93±6.78	66±0.00	88±14	90±10	97±0.00	87±7.81	9 0±13	87±11	93±13	82±16	91±13	94±6.20	89±10	85±16	91±10
Suspended Solids (µg/L)	76636±0.00	63000±0.00	63000±0.00	63000±0.00	99250±57188	93333±62353	99750±0.00	113500±42532	63000±75545	99750±4 0987	86500±69702	102857±51860	76636±63833	64667±31835	87 333±69206	75000±65962	80200±67500
Nutrients																	
Ammonia [NH_3] (µg/L)	130±0.00	150±9.19	406±487	228±0.00	297±237	66±8.08	160±0. 00	262±228	176±52	160±95	360±260	303±219	291±112	209±81	180 ±104	110±62	105±65
Nitrate [NO_3-N] (µg/L)	2910±0.00	5175±406	384±230	1110±0.00	674±598	343±167	401± 0.00	462±193	506±477	401±204	688±844	416±256	564±428	290±2 31	592±460	1384±3451	406±296
Sulphate [SO_4] (µg/L)	1990±0.00	2301±2499	92459±172864	681710±0.00	58253±4599 6	46875±20002	61320±0.00	23575±8330	138654±54812	21011±4 909	8971±5526	19202±1644	12584±4283	13835±8796	56321±423 88	46978±20940	46664±23198
Metals																	
Aluminium [Al] (µg/L)	26±0.00	12±3	1737±4704	61±0.00	866±1254	355±249	477±0.00	5 96±362	629±554	644±300	393±422	871±701	1000±722	525±236	57 1±537	240±298	406±436
Iron [Fe] (µg/L)	14±0.00	13±25	932±594	46±0.00	1055±990	730±474	824±0.00	13 62±298	1546±703	2240±470	1157±668	1916±1121	1794±778	1511 ±502	902±662	525±380	709±479
Manganese[Mn] (µg/L)	1.20±0.00	17±1.54	203±287	1401±0.00	251±228	109±88	25±0.0 0	54±24	806±506	131±29	261±130	174±233	136±66	53±23	88±65	16 2±88	119±81
Zinc [Zn] (µg/L)	4.70±0.00	4.29±2.02	60±142	26±0.00	17±27	7.37±7.94	1.60±0 .00	16±17	24±20	11±11	6.88±10	11±9.16	17±16	7.26±8.64	17±27	2.88±3.04	4.54±6.07
Bromine [Br] (µg/L)	9.40±0.00	75±18	43±17	36±0.00	46±25	99±35	6.7±0.00	37±32	52 ±29	37±19	55±40	34±19	34±14	27±12	31±17	95±31	94±41
Lead [Pb] (µg/L)	0.03±0.00	0.04±0.02	0.56±1.03	0.03±0.00	4.90±8.65	1.38±1 .68	0.58±0.00	2.36±1.91	0.40±0.23	2.90±0.88	1.71±2.65	2.8 ±2.36	3.56±3.87	1.51±1.33	2.54±3.22	1.08±1.90	1.84±2.59

Table A.8: PCA Table showing Mean and Standard Deviation for all sampling sites for study period 2014 only.

Physical Parameters	Spring 2	Spring 3	Spring 4	Spring 6	Spring 11	Spring 12	SW 1	SW 2	SW 3	SW 4	SW 5	SW 6	SW 7	SW 8	SW 9	SW 10	SW 11	SW 16	SW 17
pH	6.44±0.20	6.73±0.34	5.67±0.27	6.96±0.30	7.10±0.19	7.02±0.30	7.56±0.18	7.73±0.23	7.82±0.22	7.56±0.16	7.19±0.09	7.54±0.19	7.82±0.32	7.22±0.78	7.62±0.32	7.83±0.25	7.68±0.10	7.82±0.14	7.81±0.18
Conductivity (mS/m)	7.63±0.98	14±4.85	9.54±0.93	13±3.30	8.90±1.38	4.88±1.31	17±1.94	30±6.85	22±5.60	14±1.82	16±2.45	15±1.87	16±2.47	14±1.61	12±1.31	14±2.12	17±5.60	36±7.89	28±6.99
Turbidity (NTU)	2.21±3.01	75±83	0.55±0.38	3.77±2.88	34±70	97±89	79±126	64±86	31±21	60±74	12±8	67±71	232±348	103±147	210±262	60±74	162±426	11±19	69±140
Dissolved Oxygen (%Sat)	42±9.32	47±18	44±15	44±6.44	51±15	43±8.64	45±9.30	41±10	49±19	47±8.55	44±8.36	46±12	44±8.44	49±19	47±11	44±10	49±12	44±11	43±9.38
Suspended Solids (µg/L)	31891±86666	106880±95199	1200±1910	2945±3666	10473±13627	31840±40791	39982±53149	46455±59627	16855±18763	19345±22375	343527±1105934	31800±24877	126418±216546	43400±50271	64545±81667	31836±50647	59491±134956	10945±19429	40473±68682
Nutrients																			
Ammonia [NH_3] (µg/L)	169±229	362±446	238±281	258±525	185±232	294±434	169±229	100±0.00	153±123	256±519	112±39	225±307	156±151	315±663	184±199	154±104	158±146	100±0.00	156±133
Nitrate [NO_3-N] (µg/L)	5265±1080	730±1135	6436±465	315±173	503±85	198±160	455±201	451±181	86±64	228±153	193±118	180±112	980±920	287±251	369±337	205±105	465±208	426±178	420±136
Phosphate [PO_4] (µg/L)	50±0.00	86±114	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00	50±0.00
Sulphate [SO_4] (µg/L)	3319±1326	12457±9313	2748±1528	36718±12236	10229±4768	4704±996	30064±4205	61591±21549	33691±27384	14110±5823	41018±9201	17215±9795	11429±4944	12395±7025	12329±5211	16056±8449	31564±14501	76882±34347	59100±23408
Metals																			
Aluminium [Al] (µg/L)	94±67	209±309	56±41	79±46	283±373	628±828	298±145	336±287	608±439	399±199	189±102	439±177	674±1067	598±408	894±746	452±245	444±326	172±214	308±202
Iron [Fe] (µg/L)	320±406	19487±41618	149±104	1131±566	2270±4664	2872±3277	3689±4979	2245±3005	2489±1695	4514±4252	2446±1964	5815±6371	4403±3474	5926±6247	6928±8193	4079±3217	8666±21719	2360±3714	3935±6412
Manganese[Mn] (µg/L)	32±35	649±1846	30±23	305±226	78±104	21±14	156±149	105±84	216±438	492±766	218±147	181±132	253±108	206±196	185±115	75±50	116±157	180±111	116±77
Zinc [Zn] (µg/L)	5.00±3.94	7.65±6.03	8.82±8.78	1.05±0.70	5.93±10	1.87±1.28	5.22±12	7.78±18	4.70±7.14	3.39±5.81	1.28±1.44	5.03±6.91	3.76±3.72	3.86±3.91	9.78±19	4.75±8.15	33±97	3.09±4.05	5.63±7.92
Bromine [Br] (µg/L)	100±0.00	143±92	99±2.41	93±22	100±0.00	100±0.00	94±17	112±39	96±14	95±17	96±14	94±18	95±17	94±18	95±17	94±20	94±18	96±11	96±13
Cadmium [Cd] (µg/L)	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.75±0.56	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	1.17±2.12	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00
Lead [Pb] (µg/L)	0.5±0.00	0.56±0.20	1.57±3.56	0.5±0.00	0.5±0.00	1.03±1.18	0.5±0.00	0.5±0.00	0.5±0.00	0.57±0.21	0.5±0.00	1.24±2.08	0.71±0.48	0.73±0.48	0.64±0.46	0.5±0.00	0.60±0.32	0.5±0.00	0.5±0.00
Mercury [Hg] (µg/L)	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00	0.5±0.00

APPENDIX C:

Macroinvertebrate Datasheets showing South African Scoring System (SASS) score, Average Score Per Taxa (ASPT), Integrated Habitat Assessment Score (IHAS) and Number of Taxa for years 2008-2014.

Table A.9: Macronvertebrate temporal and spatial data showing invertebrate indices South African Scoring System (SASS) score, Average Score Per Taxa (ASPT), Integrated Habitat Assessment Score (IHAS) and Number of Taxa.

		Sep-06	Mar-10	Jun-10	Sep-10	Dec-10	Mar-11	Jun-11	Sep-11	Nov-11	May-12	Aug-12	Nov-12	Dec-12	Feb-13	May-13	Jul-13	Aug-13	Nov-13	Jan-14	Mar-14	May-14	Jun-14	Aug-14	Nov-14	Dec-14
	SASS5 score	47	27	57	40	14	37	12	55	47		46		25	18	36		17	41		82	80		44		54
[redacted]	Number of Taxa	10	6	10	10	3	9	2	13	11		10		6	7	10		4	12		18	17		12		13
	ASPT	4.7	4.50	5.70	4.00	4.67	4.11	6.00	4.23	4.27		4.60		4.17	2.59	3.60		4.25	3.42		4.56	4.71		3.70		4.15
	IHAS	52	45	56	37	42	41	40	46	41		25		42	29	32		40	29		51	50		39		38
	SASS5 score		40	55	53	67	23	9	66	39		56		20	16	63		61	47		47	65		48		58
[redacted]	Number of Taxa		10	14	13	15	4	2	12	9		8		4	4	16		11	9		12	16		11		13
	ASPT		4.00	3.93	4.08	4.47	5.75	4.50	5.50	4.33		7.00		5.00	4.00	3.94		5.55	5.22		3.92	4.06		4.40		4.46
	IHAS		34	45	34	39	35	35	39	33		39		53	40	40		25	50		56	56		58		65
	SASS5 score	38	40	50	41	33	22	52	76	83		92		67	92	96		82	58		87	102		120		101
[redacted]	Number of Taxa	10	7	11	10	8	5	9	14	17		17		12	15	18		15	12		15	22		19		19
	ASPT	3.8	5.71	4.55	4.10	4.13	4.40	5.78	5.43	4.88		5.41		5.58	6.13	5.33		5.47	4.83		5.80	4.64		6.30		5.32
	IHAS	42	34	44	31	36	32	47	48	51		38		65	60	60		63	56		73	71		69		75
	SASS5 score		74	97	102	95	75	38	92	54				69	80	83		86	44		108	103		103		100
[redacted]	Number of Taxa		17	19	20	18	17	8	21	13				14	17	18		17	11		23	21		20		21
	ASPT		4.35	5.11	5.10	5.28	4.41	4.75	4.38	4.15				4.93	4.71	4.61		5.06	4.00		4.70	4.90		5.20		4.76
	IHAS		65	60	60	40	45	47	39	35				47	38	32		34	41		59	60		58		45
	SASS5 score		67	118	81	79	90	63	79	109		37		69	113	91		63	56		56	53		48		51
[redacted]	Number of Taxa		12	20	14	17	15	10	17	21		8		15	21	17		12	12		13	13		11		12
	ASPT		5.58	5.90	5.79	4.65	6.00	6.30	4.65	5.19		4.63		4.60	5.38	5.35		5.25	4.67		4.31	4.08		4.36		4.25
	IHAS		46	67	60	50	45	62	45	50		27		64	62	35		40	60		42	32		41		44
	SASS5 score	145	101	99	103	118	97	97	101	115		95		81	95	121		112	119		128	130		106		139
[redacted]	Number of Taxa	28	18	17	19	23	18	17	18	22		19		16	16	22		19	23		23	24		19		26
	ASPT	5.2	5.61	5.82	5.42	5.13	5.39	5.71	5.61	5.23		5.00		5.06	5.94	5.80		5.89	5.17		5.57	5.42		5.58		5.35
	IHAS	72	63	63	62	62	63	65	70	60		46		64	49	55		61	51		83	76		75		70
	SASS5 score	116	85	118	113	138	129	101	126	107		97		84	95	120		126	165		155	141		116		111
[redacted]	Number of Taxa	20	11	22	20	26	22	16	21	21		17		15	18	21		22	27		28	28		23		23
	ASPT	5.8	7.73	5.36	5.65	5.31	5.86	6.31	6.00	5.10		5.71		5.60	5.28	5.50		5.73	6.11		5.54	5.04		5.04		4.83
	IHAS	67	59	62	61	58	62	62	64	60		14		86	67	55		58	58		78	67		67		73
	SASS5 score	128	90	126	106	110	105	111	114	132		121		109	138	121		143	128		153	128		137		130
[redacted]	Number of Taxa	25	13	22	20	21	16	17	20	26		20		21	26	21		25	23		26	24		27		23
	ASPT	5.1	6.92	5.73	5.30	5.24	6.56	6.53	5.70	5.08		6.05		5.19	5.31	5.76		5.72	5.57		5.88	5.33		5.07		5.65
	IHAS	55	66	70	68	67	58	43	53	51		49		63	65	54		62	53		83	72		73		63
	SASS5 score		139	114	114	155	47	90	133	112	85	59	67	108	108	104	115	111	100	79	113	136	114	135	88	134
[redacted]	Number of Taxa		23	19	20	30	7	15	20	22	19	15	15	20	20	20	20	20	22	17	22	24	19	25	20	27
	ASPT		6.04	6.00	5.70	5.17	6.71	6.00	6.65	5.09	4.47	3.93	4.47	5.40	5.40	5.20	5.75	5.55	4.55	4.65	5.14	5.67	6.00	5.40	4.40	4.96
	IHAS		63	66	69	69	41	72	62	57	83	49	79	82	73	58	78	55	73	73	77	74	73	80	68	72
	SASS5 score				90	86	28	56	81	100		88		34	61	62		50	56		135	112		137		158
[redacted]	Number of Taxa				16	13	5	9	14	17		16		9	13	11		10	13		25	18		22		29
	ASPT				5.63	6.62	5.60	6.22	5.79	5.88		5.50		3.78	4.69	5.64		5.00	4.31		5.40	6.22		6.23		5.45
	IHAS				45	56	46	40	44	49		30		51	45	21		36	38		75	74		75		77
	SASS5 score		67	96	108	110	30	91	123	143	130	129	82	77	109	146	116	139	100	101	129	123	117	139	103	158
[redacted]	Number of Taxa		13	13	20	19	6	14	18	25	24	23	16	13	18	25	20	23	19	19	22	20	19	22	20	27
	ASPT		5.15	7.38	5.40	5.79	5.00	6.50	6.83	5.72	5.42	5.61	5.13	5.92	6.06	5.84	5.80	6.04	5.26	5.32	5.86	6.15	6.16	6.32	5.15	5.85
	IHAS		41	62	70	61	41	61	66	57	70	54	68	82	72	72	81	59	74	63	75	73	66	73	63	73