LAND USE PRACTICES AND THEIR IMPACT ON THE WATER QUALITY OF THE UPPER KUILS RIVER (WESTERN CAPE PROVINCE, SOUTH AFRICA)

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Keywords

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South African Scoring System version 5 (SASS5)

Ecological state of the river

ACRONYMS AND ABREVIATIONS

AMD: Acid Mine Drainage

ANOVA: One-Way Analysis of Variance

ASPT: Average Score Per Taxa

BMI: Benthic Macroinvertebrate

BOD: Biochemical Oxygen Demand

CPOM: Coarse Particulate Organic Matter

COD: Chemical Oxygen Demand

CSO: Combined Sewer Overflow

DO: Dissolved Oxygen

DWAF: Department of Water Affairs and Forestry

EC: Electrical Conductivity

EPT: Ephemeroptera Plecoptera and Trichoptera

FPOM: Fine Particulate Organic Matter

NEMP: National Eutrophication Monitoring Programme

NoT: Number of Taxa

P/R: Photosynthesis/Respiration

RCC: River Continuum Concept

RHP: River Health Programme

SASS: South African Scoring System

TDS: Total Dissolved Solid

WCR: Water Research Commission

ABSTRACT

The water quality in many Cape Town Rivers and streams is a major challenge. Kuils River is subject to multiple land use impacts from upstream to downstream because of rapid urbanization in its catchment area. The main pollution sources are urban and industrial, organic matter from litter under the road-bridge, and golf course.

However no systematic efforts have been made to evaluate and improve the health of the river in term of management. To assess impacts on water quality, this study was conducted from 4th September to 27th November 2012 in 5 selected sites in the upper reach of the Kuils river. The main aim was to compare the health of the river in 2012 with that found in 2005 using physical and chemical characteristics and the South Africa Scoring System (SASS). The statistical analysis showed a significant difference between and within sites.

The water temperature, pH, dissolved oxygen concentration, total dissolved solids (TDS), and salinity were collected in situ by YSI 30 meter. To evaluate nutrient (nitrate and phosphorus) concentrations water samples were analyzed at UWC laboratory using spectrophotometer.

In addition human activities, basic conditions (7.13 to 8.76), high total dissolved solids (416 to to 916.5 mg L^{-1}) and salinity (0.31 to 0.71 mg L^{-1}) concentrations were influenced by Malmesbury shales. Nitrate (0.1 to 3.1 mg L^{-1}) and phosphorus (0.11 to 5.27 mg L^{-1}) concentrations and the decrease in dissolved oxygen in November 2012 showed eutrophic conditions of the river. In the tributary site phosphorus (1.32 to 3.62 mg L^{-1}) concentrations revealed hypertrophic condition compared to South Africa guideline.

Macroinvertebrates sampled showed a total of 28 taxa grouped in 11 orders were sampled. Poor habitat diversity and water quality degradation were principal causes of low species diversity. The South Africa Score System version 5 (SASS5) and Average Score per Taxon (ASPT) indicated that the river is seriously impacted in 2012 compared to 2005 where water quality was in poor condition. The SASS and the ASPT scores were less than 50 and 4.2 at all sampling sites in most part of sampling period.

DECLARATION

I declare that *The land use practices and their impact on the water quality of the upper Kuils River (Western Cape Province, South africa)* is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references and that this work has not been submitted before for another degree anywhere, other than the University of the Western Cape.

Full name: Francois NGERA MWANGI

Signed:

Date:

DEDICATION

I dedicate this thesis:

- To my eldest brother Mwendambali Ngera Mwangi for looking after me with love. His wish is accomplished while he is unconscious in a sick bed.

- To my wife Georgette Mayela Munkete for her assistance in managing to provide all what I needed and I still need. May this work be the result of her great patience, courage and love after three years of separation.

- And to my beloved children, Mushaara Ngera, Heroine Ngera Sikujua, Lumiere Ngera, Aron Ikwa, Merveil Litangwa Ngera, and Imelda Ngera for their support in good wishes and prayers to see me go forward in my endeavour to build my scientific being.

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CHAPTER ONE: HISTORICAL PERSPECTIVES ON THE WATER QUALITY DEGRADATION

1.1 INTRODUCTION

Freshwater watercourses are characterized by patterns of physical and chemical parameters. They differ from one continent to another and even from region to region because these characteristics are determined largely by climatic, geomorphological, geology and soils conditions, as well as by the aquatic biotas (Davies and Day, 1998). The physical and chemical quality of pristine water would normally be as occurred in pre-human times, i.e. with no signs of anthropogenic impacts.

However, it is very difficult to find physical and chemical qualities of pristine water because of direct human impact on water sources and atmospheric transport of contaminants to remote areas (Chapman, 1996). Human activities affect a high proportion of watercourses in virtually all countries. They are responsible for much of the alteration in landuse or landcover worldwide, and rivers and streams are the most affected ecosystems by these changes (Helmens, 2008).

Rivers usually shaped by natural events, are additionally stressed by human activities which generate disturbances that lead to the modification of rivers and their biota (Downes, *et al.* 2002). In many countries including South Africa, agriculture and urbanization are common types of landuse, and disturbances from each type may apply its own unique suite of pressures on receiving streams (Helmens, 2008). Disturbances due to the discharge of substances by humans may affect rivers over a range of temporal and spatial scales. Uncontrolled land use has undesirable and devastating effects on the aquatic environment. According to Luger and Brown (undated**)** the effects of pollutant into freshwater ecosystems depend on the quality and quantity of the effluent, and on the condition, type and resilience of the receiving ecosystems. Perturbations consist usually of two events, namely, the application of disturbing force (or pollutant agents) to the biota of the system, and the response of the affected biota to such changes (Downes, *et al.* 2002; Chapman, 1996). Some of these impacts may be hazardous to human health and to the biota reducing diversity and abundance of aquatic species.

To limit effluent discharges from municipal and industrial sources into water bodies in order to prevent damage to human health and aquatic life, water quality criteria and standards are currently used across the world (Novotny, 2003; Perry and Vanderklein, 1996; Bilotta and Brazier, 2008). Quality standards are, in effect, a regulatory tool that list specific quality aims associated with specific uses and are based on scientific experiments and observations (Perry and Vanderklein, 1996).

However, environmental and pollution control policies are also guided, to a higher degree, by moral issues and ethical standards (Novotny, 2003). Each society has cultural values that determine its attitudes and the ways it values natural resources. These attitudes are sometimes expressed as explicit goals for water quality management (Perry and Vanderklein, 1996).

Despite the promulgation of water quality criteria and standards, and mitigation in place, the pollution of watercourses remains a major concern throughout the world. Because of population pressures and migration, land-use conversion and its pollution consequences on freshwater resources appear to be the major diffused pollution problem today (Novotny, 2003). In the U.S for example, it has been demonstrated by recent studies of stream and river health that water quality continues to be degraded by nonpoint pollutant sources (Kenney *et al.* 2009) despite national water quality standards and a very effective control agency put in plave by the US Environmental Protection Agency.

In South Africa, freshwater resources are under increasing stress. The main factors contributing to the deterioration of water quality in South Africa Rivers are salinization, eutrophication, acidification, and microbial pathogens (CSIR, 2010). South Africa's climatic conditions, coupled with these discharges of treated and untreated sewage effluent from settlements and industrial effluents, excessive nutrient loads in return flows from agriculture, as well as modification of river flow regimes and changing land use or land cover patterns, have resulted in large-scale changes to aquatic ecosystems (Oberholster and Ashton, 2008). This situation is aggravated in urban areas where river health has suffered because buildings have been erected close to their banks Riparian vegetation has been cleared and rivershore has been canalized in places. They receive in-flows from storm water drains; they are constricted by bridges and exotic vegetation is planted.

In the Cape Metropolitan area, treated and untreated sewage effluent from urban areas is one of the most common types of pollution found in the rivers (CSIR, 2010; Pool, 2008; Luger and Brown, undated). Furthermore, the rivers have channel modifications, infilled floodplains and beams (berms) /levees. This has resulted in the loss of indigenous instream, riparian and floodplain vegetation, loss of indigenous fauna, and invasion by exotic flora and fauna. Hence the current degraded state of many rivers and wetlands in Cape Metropolitan Area (Luger and Brown, undated). Recent research confirms that in Cape Town's rivers, there is contamination by runoff from urban and informal settlement areas (CSIR, 2010). Taking into account the importance of water in the scope of the human economy, the deleterious consequences attributed to waterborne diseases and numerous changes observed in the South African rivers, it is necessary to identify the origin and type of pollutants, and to know the manner in which they affect water quality.

1.2 HISTORICAL PERSPECTIVE

All species can survive only in certain limited ranges of environmental conditions. The survival of any species to the present day implies that it has been, and still is, able to adapt to particular living conditions. As for aquatic organism, each species is adapted to living in water containing a particular suite of chemicals within certain concentration limits (Davies and Day, 1998).

Water quality reflects the composition of water as affected by nature and human activities. In its pristine state, water draining in the forest is clean but it contains chemicals, microorganisms, and sediments from the contact of rainwater with vegetation, soils, decaying vegetation, and animal and insect droppings, among others (Novothny, 2003).

However, all human process produce waste products that can negatively affect water quality. In history, human beings do not have a good record regarding pollution (Novotny, 2003). When humans decide to develop land areas that are pristine or near pristine, a cascading series of events occur that impact the quality of water bodies (Ahuja, 2009). Nevertheless, most rivers and lakes were still relatively clean during the Middle Ages, though urban settlement were highly polluted, causing frequent epidemics (Novotny, 2003). Two hundred years ago, deterioration of watercourses due to organic pollution was not a serious problem for, a relatively small human population lived in scattered communities (Mason, 2002). When human population was small, and technologies were simple, pollutants were confined to human and animals wastes (Davies and Day, 1998).

Water pollution became a severe problem with industrialization coupled with rapid acceleration in population growth (Mason, 2002). Population increase and improvement of living standards caused accelerated water quality changes, and led to water stresses and severe diffuse pollution problems (Downes *et al.* 2002). Each additional person represents an additional demand on productive resources, and additional wastes (Novotny, 2003).

When urbanization increased, governments were unable to manage natural resources in a suitable manner. The provision of clean water and safe disposal of wastewater and storm water for the towns of developing countries became increasingly more complex and serious (Biswas, 2004 and 2006). Domestic wastes from the rapidly expanding towns and wastes from industrial processes were all poured untreated into the rivers causing gross pollution. This was hazardous for human health (cholera) and noxious odors rise from the rivers (Mason, 2002).

With respect to urban runoff, problems and concerns regarding polluted date to ancient Rome, where sewers were built primarily for storm water disposal. As a result of building sewers without treatment, many rivers became heavily overloaded with nutrients and gave off a putrid smell which was caused by decomposition of sewage and garbage in the river (Novotny, 2003).

In the mid-nineteenth century it was observed that the filth of the cities and urban contamination of the water supplies were the major reasons for water borne epidemics of cholera and typhoid fever in many parts of the world (Perry and Vanderklein, 1996; Novotny, 2003). To protect human health, cleanup efforts focused primarily on point sources and removed pollutants dangerous to human health (Novotny, 2003). In many developing countries (Africa, Asia, and Latin America) where human and animal waste are not yet adequately collected and treated fecal contamination, it is still the primary water issue in rivers (Ahuja, 2009).

In South Africa, water quality degradation in rivers deals a major challenge. From the earliest days of water crisis in South Africa, it was plain that the potential danger of water quality was overexploitation of rivers due to climatic conditions associated with population increases. The impoundment, extraction and transfer of waters from rivers, domestic and industrial waste disposal, agricultural runoff, catchment degradation, and introduction of exotic species were major causes of South Africa's rivers degradation (O'Keeffe, 1986).

In regard of the overexploitation and deteriorating of South Africa's rivers, a number of structures and programmes of research in certain rivers have been initiated since the 1950s.
The first official expressions of concern for the water degradation of rivers and human health, specifically bilharzia took place in the 1950's (O'Keeffe, 1986). Numerous studies have shown that wastewater contains a wide range of pathogens and sometimes heavy metals and organic compounds that are hazardous to human health and the aquatic environment. Many of the rivers have been impacted by effluent discharged from wastewater treatment works (WWTW) and agriculture runoff causing nutrient enrichment (CoCT, 2011).

In Cape Town's rivers, Heydorn and Grindley (1982) observed that pollution in Kuils River was a real and rapidly growing threat. The first sign of Kuils River degradation was predicted in 1946 when the Department of Water Affairs suggested a possible canalization to facilitate irrigation and afford a measure of flood protection. Also, the sewage disposal sites as well as waste disposal facilities that exist in Bellville since the 1930s and 1960s were respectively to be discharged into the Kuils River. These constitute fundamental factors that decrease water quality. It was shown that the effluents from the Bellville WWTW exceeded general effluent standards (Parson, 2002).

Decades ago, studies in Kuils River catchment area reported that the entire course of the river was subjected to the multiple impacts associated with the rapid urbanization of its catchment area. The water quality deteriorated significantly due to organic pollution from multiple pathways namely farms, urban settlement, wasterwater treatment works (WWTW), storm water and industrial waste. Also a large number of road-bridge and variable channel conditions in the courses impede the free flow of water and increase upstream water levels (CoCT, 2011; Heydorn and Grindley, 1982; Ninham, 1979).
Note that before the advent of anthropogenic influences, the Kuils River was a seasonal river, drying summer into a series of small pools, or kuils, and then flowing torrential during the winter rains. Because of treated effluent from wastewater treatment works (WWTW) that it receives from Scottsdene, Bellville, Zandvleit and Macassar, Kuils River has a perennial flow (Li Rui, 2005). The sewage effluent discharged is probably the main source of pollution of the Kuils River. The change in flow from a seasonal to a perennial system is due to the addition of sewage effluent that has severely impacted the system (Ewart-Smith and Ractliffe, 2002).

1.3 PROBLEM STATEMENT

"Rivers are complex self-regulating system" (Davies and Day, 1998). If "left alone'' they support a range of processes and organisms that maintain the rivers in a healthy state. However, human intervention in any part of the catchment area does have a negative impact on river health. The Kuils River is impacted by activities emanating from human settlements, road and bridge, and agriculture and this has resulted in the health the river diagnosed as poor in some places and unacceptable in other places (River Health Programme, 2005).

Large-scale manipulation of sections of the river course through canalization, the loss of indigenous riparian vegetation and a reduction in water quality through agricultural and industrial runoff and particularly waste water effluent discharges have resulted in a dramatic loss of natural ecological functioning along its entire length (Brown and Magoba, 2009).
The vision for the health of the Kuils River has been expressed to be fair. From observation, it seems to suggest that no concerted efforts have beeb made by water management organizations to improve the health of the Kuils River.

1.4 AIM OF THE STUDY

The overall aim of this study is to compare the state of the Kuils River in 2012 to that of 2005 using two river health indices—the index of Water Quality and the South Africa Scoring System.

The specific objectives are:

1. To identify and describe the main sources of pollution in the Kuils River Catchment area.
2. To determine the water quality (pH, water temperature, total dissolved salts, dissolved oxygen, phosphates, and nitrate) and invertebrate diversity at selected sites from headstream to the confluence with the Bottelary River.
3. To compare the water quality and invertebrate diversity:
   3.1 From the upstream in Durbanville to the confluence with the Bottelary River
   3.2 Over time for each sampling site.
   3.3 With the 2005 outcome.
   3.4 With historic data from DWAF

1.5 RESEARCH QUESTIONS

Currently, what is the ecological state the upper the Kuils River?

1. What are the main sources of pollution of the river course?
2. What is the influence of each landuse pollution type on water quality upstream?
3. Which pollutant contributes more to the pollution of the river?
4. What is the actual state of habitat integrity, water quality and invertebrate diversity as compared to 2005 with regards to spatial and temporal scale for each sampling site?

1.6 SIGNIFICANCE OF THIS STUDY

Poor water quality has been principally associated with human health concerns through the transmission of water-borne diseases. These diseases are still major problem in many regions of developing countries. The deterioration of rivers not only results in loss of aquatic habitat and aquatic life but also degrades the ability of the systems to provide the goods and services that people depend on. This study will determine the current state of the upper Kuils River and provide updated information to enhance the effective management of the river. It will also assist the authorities to put in place effective mitigation mechanisms for the effluents disposal to the receiving water bodies.

1.7 CHAPTER OUTLINE

Chapter One – Introduction: Historical Perspective of Water Quality

Chapter Two – Literature review: presents certain notion of river continuum, describes the physic-chemical and biological characteristics of the rivers, and pollution sources and their consequences in South Africa aquatic ecosystems.

Chapter Three – Research design and methodology: presents the study areas, describes sampling points and method to evaluate water quality.

Chapter Four – Results: carry on physical, chemical and biological parameters upper of the Kuils River.

Chapter Five - Discussion

Chapter Five – Conclusion and recommendation

CHAPTER TWO: LITERATURE REVIEW

2.1 INTRODUCTION

This chapter describes the physical and chemical parameters, its natural state, perturbation from human activities and the impacts on water quality and aquatic life. The different sources of pollution and their consequences on aquatic systems are also reviewed.

Although freshwater has been recognized as an increasingly important resource, it is under threat from human activities. Increased population has led to landscape transformations that have a number of documented effects on stream ecosystems (Allan, 2004). Land use described by many authors as hazardous to aquatic ecosystems, are urbanization and agriculture activities. In comparison with urban land use, agriculture occupies the largest portion of land and constitutes the major cause of stream impairment in many developed countries catchments (Allan, 2004; Paul and Meyer, 2001). Numerous studies have documented declines in water quality, habitat, and biological assemblages as the extent of agricultural land increases within catchments (Allan, 2004). Despite the fact that urban land use may occupy a low percentage of the total catchment, numerous studies have shown that ever-increasing urbanization represents a threat to stream ecosystems because of population concentration. Urbanization impacts alter water quality and constitute a threat to aquatic life (Paul and Meyer, 2001).

Undoubtedly, by changing the landscapes of stream catchments, human activities alter stream ecosystems in various ways (Allan, 2004). Human actions at the landscape scale are a principal threat to the ecological integrity of river ecosystems, impacting habitat, water quality, and the biota. Water quality studies are used to describe the physical and chemical characteristics of water affected by human activities. Chemical assessment does not provide direct information on the effects of pollution on the biological quality or ecosystem health of the river. For that fact, to obtain more complete information on the water quality, the assessment has been extended to biological assessment (Knoben *et al.*, 1995). The impacts of population growth and rapid urbanization constitute a major issue in Africa, including South Africa.

Rapid urbanization and human activities in urban and rural areas pose a serious threat to water quality in rivers due to an increased risk of pollution in South Africa. Some studies (Ninham, 1979; Heydorn and Grindley, 1982; River Health Programme, 2005; Nel *et al.* 2013) and student these (Fisher, 2003) have been carried out in Kuils River catchment areas. The River Health Programme (2005) reported that many Greater Cape Town Rivers including Kuils River are exposed to several kinds of pollution such as waste effluent from urban and industrial areas, stormwater, agriculture run-off, and spilled oil. Major pollution sources and their impact on Kuils River have been studied by Ninham, (1979) and channelization impacts on geomorphology and ecology have been studied by Fisher (2003). In the ensuing section, the concept of river continuum is introduced and discussed.

2.2 RIVER CONTINUUM

The first attempt to categorize the Stream Zonation Concept started in 1963 by Illies Botosaneanu defining a series of distinct communities along river systems (Maiolini and Bruno, 2007). About three decades ago, Vannote and colleagues introduced the River Continuun Concept (RCC) according to which the biotic stream community adapts its structural and functional characteristics to the abiotic environment from headwater to downstream (Maiolini and Bruno, 2007; Vannote *et al.*, 1980). Changes in physical habitat and food base from source to mouth profoundly influence biological communities.

Based on considerations of stream size and progressive changes in biological communities along a river system, Vannote *et al.* (1980) divided river orders into three major categories namely, headwater (low order stream), medium-sized stream, and large rivers.
The headstream is characterized strongly by forest canopies which decreases autotrophic production by shading and contributing significant amounts of detritus (Vannote *et al.* 1980). In this part of the stream, macroinvertebrate are usually dominated by shredders (Vannote *et al.* 1980) and collectors (Maiolini and Bruno, 2007; McCabe, 2010). Common shredders include the stonefly (Plecoptera), cranefly (Tipulidae: Diptera) larvae, and caddisflies (Limnephilidae: Trichoptera) that feed directly on coarse particulates organic matter (CPOM), ingesting falling leaves and converting them to fine particulate organic matter (FPOM) which become the food for collectors (Fang, 2010). However in the Southern hemisphere, streams

including South Africa, although many headstreams are without forest canopy but dominated by in-stream plant communities (some fynbos streams in Western Cape for instance), there are still shredders and collectors (Davies and Day, 1998). Owing to groundwater supply or infiltration sources areas and riparian cover, headwater streams present little variation of temperature and have a restricted nutritional base, and therefore biological communities show very low diversity of species (Vannote *et al.* 1980).

Moving downstream, the stream size increases and the influence of forest canopy decreases allow sunlight penetration, which favors significant production of periphyton and macrophyte (Fang, 2010; Lévêque, 1996). Due to forest canopy reduction, they note that the coarse particulate organic matter (CPOM) contribution decreases, fine particulate organic matter (FPOM) occurs and systems become more autotrophic, and the temperature may attain its maximal variance because of increased solar input (Fang, 2010; Lévêque, 1996).

The macroinvertebrate diversity becomes important in medium size stream for temperature variations tend to be maximized (Vannote *et al.* 1980). Grazers including caddisflies (for instance: *Glossossoma* and *Dicosmoecus*) and mayflies (example: *Stenonema*) having a mouth adapted to feeding on periphyton from rock surface dominate midsized rivers with P/R>1 (Photosyntesis/respiration= P/R ratio) (Fang, 2010; Vannote *et al.* 1980; Maiolini and Bruno, 2007).

When stream size increases, the influence of forest canopy becomes insignificant and several major hydrological phenomena may occur (Fang, 2007). The primary production is often limited by depth and turbidity, flows drop, and bottom substrate become not only smaller but also more and more homogenous (Vannote *et al.* 1980). High turbidity which reduces sunlight penetration and unstable sandy riverbeds limit photosynthesis (root plants or algal development) and the system reverts to heterotrophy (P/R<1) due to abundant fine particulate organic matter (FPOM) from upstream (Maiolini and Bruno, 2007; Lévêque, 1996). In high order streams temperature observed is often greatly diminished due to the buffering effect of the large volume of water in the channel (Vannote *et al.* 1980). Macroinvertebrate communities are dominated by collectors (e.g *Tricorythides*, *Baetis* and *Epmerella*) (Fang, 2010; Vannote *et al.* 1980) and shredders (Maiolini and Bruno, 2007).

Although the physical zonation of rivers are explained as longitudinally linked systems in which ecological processes in upstream are correlated to those in downstream, there are some exceptions because human influences increase unpredictability. Human influences such as riparian removal, logging, damming, and dumping interrupt the pristine conditions in a river continuum leading to sediment and nutrient increased and loss of aquatic habitat which can affect biodiversity and ecological functions (Fang, 2010). In South Africa, the uses and abuses of rivers and their waters by humans have become such that the natural communities of organisms often cannot survive. Many rivers become no more than a dirty, sluggish drain, flowing over a concrete bed through a landscape covered by brick, concrete, roof tile, or tarmac from upstream to downstream (Davies and Day, 1998).

2.3 PHYSICO-CHEMICAL PARAMETERS

Water quality assessment is the overall process of evaluating the physical, chemical and biological nature of the water (Chapman, 1996). Physicochemical variables have been well investigated in monitoring and assessment of rivers and streams (Downes *et al.*, 2002). The need of water quality assessment is to verify whether the observed water quality is suitable for intended uses, to determine trends in the quality of the aquatic environment and how that quality is affected by the release of contaminants due to anthropogenic activities, and/or by waste treatment operations (Chapman, 1996).

2.3.1 Temperature

Water bodies are characterized by temperature variations along with normal climatic fluctuations. Natural variations of the water temperature often are influenced by factors such as hydrological, climatilogical, spatial and temporal scale, geomorphic variations, and structural features of the region and catchment areas (Dallas and Rivers-Moore, 2011; Dallas, 2008; Chapman, 1996). At the river scale geomorphological variation, riparian vegetation cover and different type of habitat determine the temperature fluctuation longitudinally in the river. Headwaters covered by riparian vegetation usually present lower temperature than downstream where temperature is often high (Dallas, 2008). Temperature is important because it influences physical, chemical and biological processes in water bodies. Its degree of predictability in a stream provides an indication of the degree of structure and functional predictability of invertebrate communities (Vannote and Sweeney, 1980). The development of temperature criteria is important for the effective protection and management of aquatic ecosystems (Dallas and Rivers-Moore, 2011). For that reason, South African guidelines suggest that the water temperature for aquatic ecosystems should not be allowed to vary from the background average daily water temperature considered to be normal for that specific site and time of day, by > 2°C, or by > 10 %, whichever estimate is the more conservative.

However, in South African inland water temperatures vary between 5ºC and 30ºC (DWAF, 1996a). Spatial and temporal variations in water temperature have been recorded in many South African rivers. Altitude has usually been indicated as a fundamental parameter which determines significantly water temperatures in many rivers (River-Moore *et al.*, 2008; Jacobsen, 2000). Dallas (2008) for example, reported that the water temperature in the Kuils river was lower (from 7.5 to 15°C with a mean of 10.9°C) at high altitude (671 m) than at low altitude (335 m) where a higher water temperature (from 7.0 to 20.0°C, with a mean of 13.1°C) has been observed. Temperature fluctuations may also be caused by seasonal and daily variation of climates in the catchment. The minimum temperature (e.g 6.5°C in Mpumalanga Rivers) is often recorded in winter whereas the higher temperature (e.g 29.9°C in Mpumalanga Rivers) is observed in summer. Although the water temperature may shift with season and size of the river, this may also vary daily. During the night and early morning, the water temperature is often lowest and increased from mid to late afternoon (Dallas, 2008).

Inter-basin transfer schemes also impact on water temperatures in so far as many effluents increase flow volumes and may lead to ecosystem variability (Rivers- Moore *et al.*, 2008). Water temperature is recognized as an important abiotic driver of aquatic ecosystems (Dallas and Rivers-Moore, 2011). However, human activities constitute a main cause for temperature modification. Many human activities such as water abstraction, hot effluents from industrial processes, land-use change, returning irrigation waters, removal of riparian vegetation, increased storm water runoff, power generation, and climate change and global warming can cause temperature increases in the receiving water of 10°C or more (Dallas and Rivers-Moore, 2011; Dallas, 2008; Abel, 2002).

Effect of temperature on water quality: Elevated water temperature is more common and widely documented in the literature, although its studies in South African rivers are relatively less known (Dallas, 2008; DWAF, 1996a). Numerous authors explain that, the rise of water temperature alter many physical and chemical characteristics of water including the solubility of oxygen and other gases, chemical reaction rates and toxicity, and microbial activity. In freshwater the physical environment in terms of a reduction in density of water, a decrease in pH, a reduction in solubility of dissolved oxygen followed by an increase in BOD by stimulating organic decomposition by microorganisms are observed as temperature increases (CWT, 2010; Dallas and Day, 2004; Abel, 2002; Rivers-Moore *et al.*, 2008; Mason, 2002; Chapman, 1996). Duffus, (1980) cited by Dallas, (2008) shows that the increasing water temperature decreases the dissolved oxygen concentration in water and therefore its availability to aquatic organisms.

Effects on aesthetics: Higher temperature favors the growth of sewage fungus and also the growth of macrophyte and algal blooms when nutrient conditions are suitable (Dallas and Day, 2004). It leads also to rapid bacteria and phytoplankton growth (Chapman, 1996). These factors reduce the environmental quality of the water; affect the suitability of drinking water and aesthetic values for recreation (Dallas and Day, 2004).

Effects on biological process: Temperature is one of the most important environmental variables affecting aquatic biota activities (Helmens, 2008). Its modification influences many aspects of an individual specimen's existence, including its metabolic, growth and feeding rates; fecundity; emergence; behavior and survival. Aquatic organisms are susceptible to changes in water temperature since a 10°C increase results in doubling of the organism's metabolic rate (Hellawell, 1986 in Dallas, 2008). The growth of aquatic insects has been shown to be strongly correlated with temperature in several taxa such as mayflies, stoneflies, and isopods (Dallas, 2008).

Effects on aquatic biota: Changing the thermal regime of a river significantly alters a component of the environment for which river organisms are adapted (Rivers-Moore *et al.* 2008) and can lead to changes in the abundance of specimens, species richness, diversity and composition of aquatic community (Dallas, 2008; Dallas and Day, 2004). Many species intolerant of warm conditions may disappear from heated waters and replaced by heat-tolerant species which increase in number and supplant the original species in the ecosystem (Abel, 2002). Because temperature decreases linearly with increasing altitude thus, the changes in stream invertebrate community composition and species richness may also be attributed to decreasing water temperature at higher altitudes (Jacobsen, 2000). However, many stream macroinvertebrates are adapted so that seasonal changes in temperature act as cues for the timing of migration, spawning or emergence, cyst formation or to change diet, to produce flowers or to set seed (Hauer and Lamberti, 2006; Davies and Day, 1998).

2.3.2 Electrical conductivity, Total dissolved salts/solids (TDS) and Salinity

The total amount of material dissolved in water sample is commonly measured as conductivity, as total dissolved solids, or as salinity (Davies and Day, 1998). Numerous authors define conductivity as the capacity of water to conduct an electrical charge. The total dissolved salts concentration is considered as a measure of the quantity of all dissolved compounds in water able to carry an electrical current (DWAF, 1996a). It has been found that conductivity is often correlated with the concentration of the total dissolved salts (TDS) in solution (O'Harye and Amendola, 2010; Dougall, 2007; Davies and Day, 1998). The total dissolved salts (TDS) concentration is directly proportional to electrical conductivity (DWAF, 1996a). Le Roux, *et al.,* (2007), for instance, converted electrical conductivity to total dissolved salts (TDS) according to Richards, (1969) method. This method has become more practical to use because EC is easy to measure. Naturally, in streams and rivers, conductivity is dependent primarily on the geology of the area through the water flows. The degree of the dissociated ions, particularly with mineral salts, the amount of electrical charge on each ion and its mobility, the distance from upstream, organic matter from decomposing plants, and the temperature of the water all play a role (Dougall, 2007; Chapman, 1996; DWAF, 1996a).

Streams that flow through a surface characterized by clay soils present higher conductivity because of the presence of materials that ionize when dissolved into the water. On the other hand, rivers that run through areas with granite bedrock show lower conductivity because granite is composed of more inert materials that do not dissolve (Dougall, 2007). In South Africa, the waters draining on the Table Mountain Series may be low in TDS for these rocks contain very little leachable material. All natural freshwaters flowing on rocks adjacent of Malmesbury Shales are characterized by a high TDS concentrations because these rocks have considerable quantities of leachable ions (Brown and Magoba, 2009). Mineral salts elements which provide ability to water to conduct electrical current include dissolved inorganic ions such as Mg^{+2}, Ca^{+2}, K^{+}, Na^{+}, Cl^{-}, SO_4^{-2}, HCO_3^{-} and CO_3^{-2} in the aquatic environment (Leske and Buckley, 2003). The salinity may also be influenced by natural phenomena, namely evapotranspiration and rainfall (DWAF, 1996a).

As regards the distance from upstream to downstream, studies carried out by Dougall (2007) in many glacial and non-glacial in many rivers revealed that conductivity was lower in headwaters and was elevated downstream, perhaps, due to greater abundance of proglacial sediment. On the other hand, higher conductivity observed in the headwaters of Sandy River was probably due to the chemical weathering of rock from Sand Glacier volcano and sulfate concentration. Electrical conductivity decreased with distance because of a sulfate concentration in the headwaters diluting with distance.

The majority of freshwaters usually have TDS levels between 0 and 1 000 mg L^{-1}, but it is undoubtedly true that it can exceed 1,000 mg L^{-1} in polluted waters or those receiving large quantities of land run-off (Davies and Day, 1998; DWAF, 1996a). The rivers which flow on Paleozoic and Mesozoic sedimentary rock formations TDS concentrations vary between 200-1 100 mg L^{-1} and may exceed 1 100 mgL^{-1} at high evapoconcentration (DWAF, 1996a). According to Thomas and Tris (1996), the TDS levels in the lower reaches of the Sundays ranged from 1 000 to 15 000 mgL^{-1}. These values are not suitable for benthic macroinvertebrate fauna (Dougall, 2007; Thomas and Tris, 1996). In the Namibian Desert for example, it has been recorded in a Gypsous spring a value reaching 150 000 mg L^{-1} (24 800 mSm^{-1}) which maintains a limited but flourishing fauna and flora (Dallas and Day, 2004). Studies conducted by Statistic South Africa (2005), showed that many South Africa rivers which drain the dry interior regions may have a high TDS varying from 53 to 9059 mg L^{-1}.

Anthropogenic impacts may cause increased conductivity values of aquatic ecosystems worldwide, particularly in semi to arid regions. High conductivity in many natural freshwater systems arises in discharging saline domestic and industrial effluents into the rivers. Surface runoff from urban, industrial and cultivated areas, irrigation, clear-felling, and return of large quantities of sewage effluent also contribute to increased salts in the rivers (Brown and Magoba, 2009; DWAF, 1996a).

Effects of conductivity on aquatic organisms: several authors support the hypothesis that there is relationship between conductivity and various parameters of macroinvertebrate populations in streams, particularly with adverse impacts to mayflies (Howard *et al.* 2000; Chambers and Messenger 2001; Hartman *et al.* 2005; Merricks *et al.* 2007). Pond *et al.* (2008b) cited by GEI, (2009) explain that the population reductions in mayflies observed may likely be due to the effects of sediment ponds or changes in vegetation rather than high conductivity. According to O'Hayre and Amendola, (2010) there is no scientific evidence for conductivity as a toxicity factor to benthic organisms at the low levels. Toxicity to aquatic organisms can occur at very high conductivity levels and varies depending on the specific aquatic organism and relative mix of ions such as sulfate and chloride in the water. According to DWAF, (1996a) the changes in TDS concentrations affect adaptations of individual species, community structure, metabolism rates and nutrient cycling. According to the literature review of Dallas and Day (2004) there is little information available on salinity tolerances of aquatic organisms. EC in natural freshwater varies so widely that no absolute values can be recommended and therefore, no national standards for preservation of aquatic life have been proposed in the literature.

2.3.3 pH

The concentration of proton (H^+), hydroxyl (OH^-), bicarbonate (HCO_3^-) and carbonate (CO_3^{-2}) ions are some of the most important attributes determining the composition and quality of water. The concentration of hydrogen ions is an important factor. Its value varies from 0 to 14 with pH = 7 representing a neutral condition, pH < 7 indicating acid condition and pH > 7 as a basic condition. Acid waters (pH < 7) can have measurable alkalinity, and alkaline waters (pH>7) can have measurable acidity (Chapman, 1996).

Natural state: The pH is principally controlled by the balance between carbon dioxide, carbonate and bicarbonate ions as well as other natural compounds such as humic and fulvic acids. In natural freshwaters pH varies from 3.0 to 11.0 and sometimes more. The values between 5.0 and 9.0 generally support a diverse assemblage of aquatic species (Abel, 2002).

Important factors that influence pH include geology, biotic activities, type of vegetation, atmospheric influences, acid-neutralizing or buffering capacity, and cation exchange capacity (Belcher, 2009; Abel, 2002). In the catchment, the geology is the major influence on the pH. Rivers and streams which flow on the Malmesbury system rock present alkaline conditions in the South Western Cape (Ndiitwani, 2004). Diurnally change in pH can be influenced by the photosynthesis and respiration cycles of photoautotrophs in eutrophic waters and other effluents. The photosynthic process may alter the balance between carbonate and bicarbonate by taking away CO_2 from surface water. Klerk *et al.* (2012) reported that an increase in pH in spring for example may be attributed to increased photosynthesis activities of aquatic plants, namely macrophytes and algae. DWAF (1996a) associates seasonal fluctuation to the hydrological cycle, especially for rivers which flow in catchments dominated by fynbos.

According to Struyf *et al.* (2012) riparian vegetation characterized by fynbos plants leads to low pH. Dead plant litter from fynbos plants produce organic compounds leading to acidic (Brown and Magoba, 2009). Nevertheless, all natural waters have some buffering capacity, which is the ability to absorb acid or alkaline inputs without undergoing a change in pH. Where the buffering capacity of water is exceeded by the input of an effluent, the pH of the water will change.

Anthropogenic effects: The source of the changes in pH of the natural water has been well documented and constitutes a serious water pollution problem through the world. Human activities influence acidification of aquatic ecosystems by diverse point-source effluents. Alkaline pollution in rivers is less common than acid pollution. Many untreated effluents impact water quality in term of pH which may be strongly acidic or alkaline. High biological activities due to alkaline effluents from certain industries increase pH values in the rivers under eutrophic conditions (Dallas and Day, 2004; DWAF, 1996a). A very common form of acid pollution involving extreme pH in many developing countries, including South Africa is acid mine drainage (Abel, 2002) which causes very considerable stream and river pollution problems (Moon and Lucostic, 1979; Ross, *et al.* 2007). According to Ochieng *et al.* (2010) numerous studies have shown that excess in H^+ in many South African watercourses result from mine drainage which alters significantly the ecology of the river and impacts numerous economic activities. The effects of acidity vary between streams because of variability in buffering capacity and land use. Some streams have relatively high concentrations (>10 mg/l) of $CaCO_3$, which buffers acids; these streams have an average pH of ~6.0. The streams that have lower concentrations of $CaCO_3$ show a low mean value of pH leading to high concentrations of soluble aluminium which is toxic under acid conditions.

Effect on pH: There are several factors which affect pH: biological activities, temperature, total dissolved salts, concentrations of organic and inorganic ions (Gueade *et al.,* 2009; DWAF, 1996a). According to Gueade *et al.,* (2009) lower pH values often are related to higher conductivity. In natural fresh water, the pH value declines by 0.1 of a unit when temperature increases by 20°C (DWAF, 1996a).

Water quality: In many aquatic ecosystems, the changes observed in the concentration of metallic complexes leading to increase in toxicity of most metal are attributed to small variations in pH (DWAF, 1996a). At low pH, streams and acid precipitation may liberate toxic heavy metals. The most probably heavy metal increases which result from a low pH include Ag, Al, Cd, Co, Cu, Hg, Mg, Ni, Pb and Zn (Kimmel *et al.,* 1985).

A non-metallic ion that can be similarly affected by changes in pH is the ammonium ion (NH_4^+). Lowering pH can also decrease the solubility of certain elements such as selenium. Leske and Buckley, (2003) reported that a very high or a low pH does not affect TDS concentration in water.

Effect on biota: The combination of elevated hydrogen ion concentrations and heavy metals in solution can eliminate many types of aquatic life (Kimmel *et al.*, 1985). Higher pH values as well as lower pH affect aquatic biota. The high concentrations of Al at low pH are one of the primary causes of aquatic organisms' mortality (Schofield and Trojnar, 1980). Several studies revealed that low pH may influence the structure of macroinvertebrate community and species diversity (Abel, 2002; Soulsby *et al.* 1997; Wade *et al.* 1989; Kimmel *et al.* 1985; Haines, 1981; Moon and Lucostc, 1979). Numerous searches have shown that mayflies are more sensitive taxa in acidified waters (Weatherley *et al.*, 1987; Kimmel *et al.*, 1985; Friberg *et al.* 1980).

2.3.4 Dissolved oxygen (DO)

To assess dissolved oxygen is fundamental for it influences almost all chemical and biological processes within water bodies (Chapman, 1996). Oxygen availability is recognized as a key factor in aquatic ecology influencing the composition of freshwater communities because its depletion in water bodies affects the distribution of many species, community structure and local richness (Jacobsen, 2008; Connolly *et al.*, 2004). Dissolved oxygen can be used to indicate the degree of pollution due to organic matter, the destruction of organic substances and the level of self-purification of the water. In natural freshwaters, dissolved oxygen at sea level ranges from 15 mg L^{-1} at 0° C to 8 mg L^{-1} at 25° C (Chapman, 1996), and from 12.77 mg L^{-1} at 5°C to 9.09 mg L^{-1} at 20°C according to DWAF, (1996a). In unpolluted water, dissolved oxygen concentrations range usually close to, but less than, 10 mg L^{-1}. Dissolved oxygen below 5 mg L^{-1} may negatively affect the functioning and survival of biological communities and below 2 mg L^{-1} may have harmful effects on aquatic organisms (Chapman, 1996). Oxygen enters the water by absorption directly from the atmosphere, by aquatic plant and algae photosynthesis and is removed from the water by respiration and decomposition of organic matter (Novotny, 2003; Jacobsen, 2008). However, the level of dissolved oxygen concentrations may vary in water bodies. The factors that influence the DO variation in water bodies have been thoroughly documented (e.g Novotny and Bendoricchio, 1989; Kolar and Rahel, 1993; Jacobsen, 2000; Connolly *et al.*, 2004; Kaller and Kelso, 2007, Van der Geest, 2007; Jacobsen and Marin, 2008; Jacobsen, 2008). Theoretical, the solubility of oxygen in stream water may be influenced by three main parameters, namely altitude, temperature and photosynthic activity by aquatic plants and algae. Numerous literatures showed that solubility of oxygen increases as temperature decreases and decreases with decreasing atmospheric pressure (Jacobsen, 2008; Hauer and Lamberti, 2006; Jacobsen, 2000). Tropical high mountain streams are more oxygen rich than warmer lowland streams (Jacobsen, 2008; Jacobsen and Marin, 2008). Dissolved oxygen concentrations fluctuate daily in stream water because photosynthesis takes place during the daylight in shallow reaches and euphotic zones, while respiration occurs during the night and in deep zones (Novotny and Bendoricchio, 1989).

Dallas (2008) shows that the solubility of oxygen in water is inversely related to both temperature and salinity. Higher temperatures and salinities reduce the solubility of dissolved oxygen in water, decreasing its concentration and thus its availability to aquatic organisms while low temperature and salinities increase the solubility of oxygen in water (Mason, 2002). The structure of a stream or river may also affect dissolved oxygen contents. Turbulence of water, depth and degree of exposure of the substratum on surface water influence the re-aeration of water. In fast-moving streams, rushing water is aerated by bubbles as it churns over rocks and falls down hundreds of tiny waterfalls. These streams, if unpolluted, are usually saturated with oxygen. In slow, stagnant waters, oxygen only enters the top layer of water, and deeper water is often low in DO concentration due to decomposition of organic matter by bacteria that live on or near the bottom (Dallas, 2008). Seasonally, dissolved oxygen concentrations are usually higher in the winter than in the summer. During rainy seasons, oxygen concentrations tend to be higher because the rain interacts with oxygen in the air as it falls. Whereas during dry seasons, water levels decrease and the flow rate of a river slows down. As the water moves slower, it mixes less with the air, and the DO concentration decreases (Mason, 2002).

Anthropogenic impacts have increased the frequency, duration, and intensity of hypoxia in many aquatic systems, resulting in changes in community composition and often a loss of aquatic diversity (Connolly *et al.*, 2004). Oxygen depletion depends on total and nature of organic material load in the rivers, and the numbers and types of bacteria which degrade waste discharges into the river (Mason, 2002).

The organic pollution such as municipal sewage treatment discharge, industry wastes, storm waters from urban areas, and farm effluents can lead to decreases in DO concentrations as a result of the increased microbial activity occurring during the degradation of the organic matter (Dallas and Day, 2004; Mason, 2002). The potential for organic wastes to deplete oxygen is commonly measured as the biological oxygen demand (BOD) and chemical oxygen demand (COD) (Dallas and Day, 2004). Both, BOD and COD directly affect the amount of dissolved oxygen in rivers and streams. The greater the BOD, the more rapidly oxygen is depleted in the stream, because microorganisms are using up the DO. The consequences of high BOD are the same as those for low dissolved oxygen: aquatic organisms become stressed, suffocate, and die (Canadian Council of Ministers of the Environment 1999). Waste streams also contain inorganic plant nutrients, namely nitrogen and phosphorus that stimulate primary productivity, indirectly affecting oxygen concentrations. Increased primary productivity results in increased dissolved oxygen during the day. In contrast, too many plants may reduce the DO levels, because of either night-time respiration by plants, algae, and decaying process by heterotrophic micro-organisms causing oxygen declines (Perry and Vanderklein, 1996).

Effect on biota: When a river system has relatively stable levels of DO, it is usually considered as a healthy ecosystem able to support lots of different kinds of aquatic organisms. However, the absence of oxygen (hypoxic) in water may be a sign of severe pollution having severe consequences for the stream biota. Generally, the decrease in dissolved oxygen in aquatic ecosystems may have adverse effects on many aquatic organisms (e.g micro-organisms, invertebrates and fish), which depend upon oxygen for their efficient functioning. The significant effect of depletion in DO on aquatic organisms depends on the frequency, timing and duration of such depletion (DWAF, 1996a). The oxygen requirements of benthic macroinvertebrates vary with type of species (warm or cold species), with life stages (eggs, larvae, nymphs, adults) and with different life processes (feeding, growth, reproduction) (Alabaster and Lloyd, 1982 cited by NWQMS, 2000), and size. The impact may lead to acute, physiological, and behavioral effects or the possibility to avoid anoxic or oxygen depletion zones (van der Geest, 2007; Canadian Council of Ministers of the Environment, 1999). Very low concentrations of dissolved oxygen are lethal to aerobic organisms, while relatively low concentrations may cause changes in behavior, blood chemistry, structure deformity, growth rate and food intake (Davies and Day, 1998; Canadian Council of Ministers of the Environment, 1999).

Kolar and Rahel (1993) examined the response of benthic invertebrates to low oxygen and found that oxygen depletion affects the distribution and activity of benthic organisms and species-specific mortality resulting from hypoxia. The sensitive benthic macroinvertebrates such as Ephemeroptera (mayflies), Trichoptera (caddisflies), and Plecoptera (stoneflies) which respire with gills or by direct cuticular exchange decline and may be entirely eliminated with oxygen depletion (Abel, 2002; Dallas and Day, 2004). While Tubificidae (worms), Hirudina (leeches), and Chironomidae (Diptera) are typically tolerant of low dissolved oxygen levels and muddy substrata, other benthic macroinvertebrates are more seriously affected by low dissolved oxygen levels and muddy substrata (Couceiro *et al.*, 2007; Abel, 2002). Shift mechanism is a key behavioral response used by lotic macroinvertebrates to avoid poor environmental conditions due to oxygen depletion (Connolly *et al.*, 2004). Kolar and Rahel (1993) indicate that high mobile taxa unable to tolerate hypoxia (mayflies and amphipods) respond behaviorally to declining oxygen concentrations by migrating upward in the water column. This situation may also decrease taxa abundance and diversity of sensitive benthic macroinvertebrates.

Water quality: In general, a low dissolved oxygen concentration lead to an increased in the toxicity of poisons. It has been observed that low dissolved oxygen may increase slightly the toxicity of zinc (Abel, 2002).

2.3.5 Nutrients

Nutrients are the necessary elements for the growth and reproduction of plants. The most common are nitrogen and phosphorus which lead to nutrient enrichment (eutrophication) of the aquatic ecosystem. They cause excessive plant and algal growth. Most nutrients are not toxic; however elevated concentrations affect the structure and functioning of biotic communities (Neda *et al.* 2011).

2.3.5.1 Nitrogen

Nitrogen is essential for living organisms as an important constituent of proteins and genetic material (Neda *et al.* 2011). Nitrogen undergoes biological and non-biological transformations in the environment as part of the nitrogen cycle. Plants and bacteria convert inorganic nitrogen to organic forms. In the environment, inorganic nitrogen occurs as nitrate (NO_3^-) and nitrite (NO_2^-), the ammonium ion (NH_4^+) and molecular nitrogen (N_2). Of these forms, nitrate is usually the most stable and commonest form often found in aquatic environments (CWT, 2010).

2.3.5.1.1 Nitrate

Nitrate is the end product of the oxidation of ammonia or nitrite. It is the most stable of the three forms, and usually, by far, the most abundant in the soil and water environment (DWAF, 1996a). The nitrate ion (NO_3^-) is the most oxidized form of nitrogen (N) present in the environment, with an oxidation state of +5. By nitrification microbial process, ammonium undergoes an oxidation to nitrite and then nitrate including two stages under aerobic condition:

- ammonium is oxidized to nitrite: $NH_4^+ + 3/2O_2 \rightarrow NO_2^- + H_2O + 2H^+$
- oxidation of nitrite to nitrate: $NO_2^- + ½O_2 \rightarrow NO_3$

To reduce nitrate levels in aquatic systems, denitrification process provides an important pathway for nitrogen removal. Denitrification involves several kind of bacteria (*Pseudomonas*, *Micrococcus*, *Bacillus*) which transform nitrate to nitrite and then to molecular (N_2) under extremely low oxygen conditions (0.2 mgL^{-1}) before it is released into the atmosphere as N_2 gas (DWAF, 1996a). Biotic assimilation by algae and macrophytes may also remove large quantities of nitrate from surface waters (Mason, 2002).

In surface water, sources of nitrate are wet and dry deposition of HNO_3 or NO_3^-, which are formed through nitrogen cycling in the atmosphere. Furthermore, igneous rocks, volcanic activity, mineralization of native soils, organic nitrogen, and the complete oxidation of organic nitrogen from vegetable and animal debris in native soil contribute to supply natural waters with nitrate (Environment Canada, 2003; DWAF, 1996a). Nitrate concentrations rarely exceed 4 mgL^{-1} in non impacted Canadian and European Rivers (Crouzet *et al.* 1999 cited by Environmental Canada, 2003). In streams where primary productivity is low, nitrate concentrations are generally < 0.4 mgL^{-1} (NRC, 1978; Nordin and Pommen, 1986 in Environmental Canada, 2003). Studies conducted by De Villiers and Thiart (2007) indicated <0.040 mgN L^{-1} as a value indicative of near pristine or low natural background levels. DWAF (1996a) reported that in South Africa, inorganic nitrogen concentrations in unimpacted, natural surface waters are usually below 0.5 mgN L^{-1} but may increase to above 5 - 10 mgN L^{-1} in highly enriched waters.

Anthropogenic: High nitrate levels recorded in surface waters originate from human activities and differ with land use. Several studies (e.g De Villiers and Thiart, 2007; Environment Canada, 2003; Mason, 2002; Novotny, 2003; Fatoki *et al.* 2001; Fleming and Fraser, 1999) reported that high nitrate concentrations observed in many river systems may be due to diffused source from urban and agricultural runoff and to point discharge from sewage treatment plants. Atmospheric deposition including gases released from agriculture and burning of fossil fuels can add significant amounts of N to surface waters (Carpenter *et al.* 1998). Elevated levels of nitrate may also result from the reduction of vegetative cover through forest fires, and logging, or insect defoliation. To protect and maintain aquatic ecosystems against pollution of nitrate several countries evoked a guideline of nitrate associated with its use. For aquatic animals, water quality criteria suggested ranged between 2 and 3.6 $mgNO_3^- L^{-1}$ (De Villiers and Thiart, 2007). South Africa water quality guidelines suggest nitrate concentrations less than 0.5 $mgNO_3^- L^{-1}$ as oligotrophic conditions and from 0.5 to 2.5 $mgNO_3^- L^{-1}$ as mesotrophic conditions (DWAF, 1996a).

Variation of nitrate level: Temporal and spatial trends have been observed in term of nitrate concentrations. Numerous investigations in Europe and United States rivers show increasing temporal trends in term of nitrate concentrations associated with agricultural activities. In many surface waters, seasonal variations occur with high nitrate concentrations in winter and spring and declined in summer and autumn due to greater biological productivity and nitrate uptake (Environment Canada, 2003). According to Mason (2002), nitrate levels are low during the summer, even when fertilizer is being added, because growing plants utilize nitrogen and the high rates of evaporation and transpiration are often observed. Whereas, in winter decrease in transpiration and evaporation are noted as nitrate is leached from the soil and levels in rivers rise. Also, a decline in late winter is because soluble nitrate reserves are depleted and low temperatures reduce the rate of nitrification.

In South Africa, studies carried out by De Villiers and Thiart (2007) showed that in several catchments, diffused nutrient sources produce seasonal concentration profiles coincident with river runoff, that is, concentrations that peak during high runoff conditions associated with fertilizer application. Dominant point sources for elevated NO_3^- (>0.400 $mgN L^-$) result in seasonal concentration profiles that have no relation to runoff, they provide a relatively constant input throughout the year, or have an inverse relation to river runoff.

Effect of nitrate on water quality: Temperature, dissolved oxygen, and pH affect rates of nitrification. It has been reported that most strains of nitrifying bacteria grow optimally at pH of 7.5 – 8.0, in water temperature varying between 25 and 30°C, and in darkness (Dong *et al.* 2011). Numerous researchers have found that denitrification rates increase with increasing temperature (Cavari and Phelps, 1977; Holmes, *et al.*, 1996). Other factors that affect rates of denitrification in aquatic systems include oxygen concentration and the supply of nitrate and organic matter. High nitrate concentrations (i.e exceeding 4 mg NO_3^- L^{-1}) tend to be associated with eutrophic conditions and algal growth blooms (DWAF, 1996a; NRC, 1978) which cause oxygen depletion.

Effect of nitrate on aquatic organisms: nitrate is considerably less toxic to aquatic organisms than ammonia or nitrite due to its limited uptake and absence of major physiological effects (Camargo *et al.* 2005). In general, based on acute concentrations, amphibians (from 73 to 7752 mg NO_3^- L^{-1}) and invertebrates (from 24 to 3070 mg NO_3^- L^{-1}) are typically more sensitive than fish (from 847 to 9344 mgNO_3^- L^{-1}). Harmful effects observed in aquatic organisms include: mortality, growth reduction, reduced feeding rates, reduced fecundity, reduced hatching success, lethargy, behavioral signs of stress, bent spines and other physical deformities (Environment Canada, 2003).

Effect on human health: Water supply with a high nitrate level (~100 mg L^{-1}) presents a potential threat to human health. Nitrate in water is toxic at high concentrations and has been linked to methemoglobinemia in infants for which digestive bacteria are able to reduce nitrate to nitrite causing conversion of hemoglobin into methemoglobin (Mason, 2002; Carpenter *et al.* 1998).

2.3.5.2 Phosphorus

Phosphorus is an essential nutrient, it forms part of the primary energy (adenosine tri-phosphate) carrier for living organisms, and constitutes an integral part of DNA (Davies and Day, 1998; Chapman, 1996). In aquatic ecosystems and in wastewaters phosphorus exists as both dissolved and particulate species which account for 70% of total phosphorus found in fresh waters (Chapman, 1996). It occurs mostly as dissolved orthophosphates and polyphosphates, and organically as the phosphate ion (PO_4^{3-}). According to Ahuja, (2009) phosphorus can enter streams either via surface runoff, groundwater contamination and subsequent lateral movement.

Because phosphorus is an essential component of the biological cycle in water bodies, it is often included in basic water quality surveys or background monitoring programmes (Chapman, 1996; Carpenter *et al.* 1998).

The major natural source of phosphorus includes weathering of rocks, decomposition of organic matter, and atmospheric deposition. In mountainous regions characterized by crystalline rocks, phosphorus level is lowest while it increases in lowland waters dominated by sedimentary deposits (DWAF, 1996a). Phosphorus associated with organic and mineral constituents of sediments in water bodies can be mobilized by bacteria and released to the water column (Dallas and Day, 2004). The high concentrations of phosphorus in freshwaters are rarely found in non-polluted water as phosphorus is actively consumed by aquatic plants or is adsorbed onto suspensoids or bonded to ions such as Fe^{+2}, Al^{+3}, Ca^{+2} and a variety of organic compounds (Dallas and Davies, 2004). In natural freshwater, phosphorus concentrations vary from 0.005 to 0.020 mg L^{-1}, and sometimes it decreases to 0.001 mg L^{-1} in certain pristine waters (Mason, 2002; Chapman, 1996).

Anthropogenic effects: High phosphate concentrations due to human activities are carried by domestic waste-waters, as detergents, industrial effluents and fertilizers run-off in surface waters (Ahuja, 2009; Jones and Lee, 1984). Intensive animal production may also contribute to increase phosphorus concentrations in water bodies (Mason 2002). Domestic sewage typically contains high levels of phosphate largely because detergent washing powder formulations normally contain high levels of phosphate (Abel, 2002). In South Africa, Jones and Lee, (1984) estimated that approximately 35% to 55% of phosphorus in domestic wastewater treatment plant effluents is from household detergent.

Effect of phosphorus on aquatic ecosystems: Primary production in fresh waters is generally limited by low phosphorus levels. High concentrations of phosphates and nitrates result in an increase in productivity (Mason, 2002) and are largely responsible for eutrophic conditions (Chapman, 1996). Total phosphorus (TP) concentrations exceeding 0.100 mg-P L^{-1} (3.2µM) are sometimes considered problematic in fresh and estuarine waters.

According to Carpenter *et al.* (1998) phosphorus in water is not considered to be directly toxic to humans and animals. However, toxicity caused by phosphorus in freshwaters may have an indirect effect. Eventually, overproduction can lead to toxic algal blooms and hypoxic waters with reduced biotic diversity.

2.4 BIOLOGICAL PARAMETERS

2.4.1 Macroinvertebrates

Streams and rivers fauna may include several hundred benthic macroinvertebrates (BMI) species from numerous groups such as arthropods including insects (larvae or adult forms), mites (hydracarina), scuds and crayfish, mollusks including snails, limpets, mussels, and clams, annelids (segmented worms, leeches), nematodes (roundworms) and turbellarians (flatworms) (Hauer and Lamberti, 2006; Tachet *et al.* 2003; Thirion, 2007; Davies and Day, 1998).

Macroinvertebrate communities may vary both spatially and temporally into the rivers following environmental factors (Reece and Richardson, 2000) which include flow regime, physical habitat structure (channel and substrate distribution), water quality, and energy inputs from watershed (Thirion, 2007). According to Dallas, (2007a) the diversity, abundance and nature of biotope (stone, gravel, sand, vegetation) at a site or in the river may influence macroinvertebrate assemblages due to biotope preferences of macroinvertebrates. It has been established that small streams have greater relative abundance and species richness due to more complex habitats than large rivers (Reece and Richardson, 2000).

Anthropogenic effects: Anthropogenic activities in many aquatic systems may alter streamflow patterns, channel morphology, water quality, and lead to changes in benthic macroinvertebrate community structures through loss of certain species and increases of others (Thirion, 2007; Kasangaki *et al.* 2006; Couceiro *et al.* 2007). For instance, studies carried out by Wang and Kanehl (2003) indicated that urban land use closer to stream was negatively correlated with macroinvertebrate sensitive taxa. Studies carried out by Fisher, (2003) showed that channelization, a reduced diversity of aquatic habitats in the Kuils River resulting to a low diversity of benthic macroinvertebrates.

Many studies (e.g Makoba *et al.* 2008; Silveira *et al.* 2006; Paul and Meyer, 2001) show that urban effects on macroinvertebrates reduces invertebrate diversity dramatically, resulting in a community dominated by Chironomidae (Diptera), Oligochaeta and tolerant gastropods. Declines in macroinvertebrate abundance and diversity often occur in sensitive families belonging to Ephemeroptera, Plecoptera and Trichoptera orders. The links between macroinvertebrate community structures and environmental variables have been the subject of numerous investigations throughout the world to determine water quality (Arimoro *et al.* 2007; Dallas, 2007a; Duran, 2006; Duran and Suicmez, 2007; Ogbeibu and Oribhabor, 2001; Reece and Richardson, 2000; Silveira *et al.* 2006).

The effects of human activities resulting in degradation of environmental characteristics of streams are the main cause of alteration structures and functions of aquatic biota leading to the need for water quality assessment. Certain benthic macroinvertebrates recognized as sensitive to perturbation in their environment and habitat characteristics, have been widely considered as best biological indicators (Stoyanova *et al.* 2010; Ngera *et al.* 2009; Makoba *et al.*, 2008; Sundermann *et al.* 2008; Arimoro *et al.* 2007; Dallas, 2007; Duran, 2006; Hauer and Lamberti, 2006; Chapman and Chapman, 2002; Abel, 2002; Mason, 2002; Davies and Day, 1998; Olomukoro and Ezemonye, 2007). These organisms reflect the intensity of anthropogenic stress and respond to the totality of environmental conditions which they have experienced throughout their lives. Their responses to environmental conditions usually depend on the nature and severity of the pollution (Abel, 2002). The presence of certain species such as mayflies (Ephemeroptera), caddisflies (Trichoptera), and stoneflies (Plecoptera) often indicates that the water is well oxygenated although their absence does not necessarily indicate the converse (Stoyanova *et al.* 2010; Lorion and Kennedy, 2009; Robertson, 2006) whereas the dominance of aquatic worms, chironomids, leeches and pouch snails usually signifies poor water quality (Robertson, 2006; Fisher, 2003; Abel, 2002). Following their response to organic or inorganic pollutants (Duran, 2006), diverse biotic indices were developed to evaluate the water quality in rivers (Chutter, 1972; Chapman, 1996; Abel, 2002; Duran, 2006). In this regard, Kolkwitz and Marsson (1902 and 1909) cited by Abel, (2002) and Chapman (1996), set the pace in Europe to explore the response of macroinvertebrates using the Saprobic System. Currently, over 100 different biotic indices have been developed throughout the world (Ziglio *et al.*, 2006). The South African Scoring System (SASS) based on the British Biological Monitoring Working Party (BBMWP) method has been initiated and adapted for South African conditions originally by Dr F. M. Chutter in 1994 (Davies and Day, 1998; Dallas, 2000 ; Dickens and Graham, 2002).

2.4.2 RIPARIAN VEGETATION

The riparian zone is the area adjacent to a river or water body that forms part of the river ecosystem (River Health Programme, 2005). It includes vegetation which improves water quality and provides ideal habitats for many fauna species. The riparian zone is characterized by higher biodiversity, both in terms of flora and fauna, and plays an important role in the ecological functioning (Table 2.1) of the river (CES, 2004). According to Dallas and Day, (2004) riparian vegetation modifies energy input into streams and rivers in two ways, supplying organic matter and reducing light availability and thermal energy to primary producers.

Table 2.1 Summary of riparian zone functions that potentially buffer conditions and inputs streams from various landuse effects (Collier *et al*., 1995)

Riparian zone function	Potential in-stream effects
-Buffers banks from erosion	-Reduces fine sediment levels
-Buffers channels from localized changes in morphology	-Maintains water quality
-Buffers input of nutrients, soil, microbes and pesticides in overland flow	-Reduce contaminant loads
	-Encourages growth of bryophyte and thin periphyton films
-Denitrifies groundwater	-Maintains lower summer maximum temperature
-Buffers energy inputs	-Increases in-stream habitat features and terrestrial carbon inputs
-Provides in-stream food supplies and habitat	
-Buffers flood-flows	-Maintains food webs
-Maintains microclimate	-Reduces flood-flow effects
-Maintains dispersal corridors	-Increases biodiversity

According to Vannote *et al.* (1980), many headwater streams are often influenced by riparian vegetation which reduces autotrophic production by shading and contributes to inputs of large amounts of allochthonous detritus. The authors observed that as stream size increases, there is a decline of terrestrial organic inputs with increased significance of autochthonous primary production and organic transport from upstream. In semi-arid to arid regions, for example much of South Africa, riparian zones are important for biodiversity because they provide habitats and refuges for a diversity of aquatic organisms (Cleaver *et al.*, 2003). Numerous studies indicate that streams draining primary humid forest are characterized by higher species richness and diversity of benthic macroinvertebrates fauna dominated by clean water taxa namely, Ephemeroptera, Plecoptera, and Trichoptera (EPT) (Couceiro *et al.*, 2007; Lorion and Kennedy 2009; Kasangaki *et al.* 2008; Chapman and Chapman, 2003), and Odonata (Kasangaki *et al.* 2008).

The forest canopy improves water quality leading to low conductivity, low acidity, low turbidity, low temperature due to shading, and low TDS, high water transparency and high dissolved oxygen (Collier *et al.* 1995; Chapman and Chapman, 2003; Kasangaki *et al.* 2006 and 2008; Water and River Commission, 2000). Unfortunately, according to different sources (e.g Chapman and Chapman, 2003; Couceiro *et al.* 2007; Benstead and Pringle, 2004), it is undoubtedly true that the area of forest remaining is drastically reduced. Internationally, the influence of landuse impacts on stream and river health is a subject to several studies (Arthur, 2010). A decade ago, Chapman and Chapman, (2003) reported that the impacts of deforestation and land conversion on aquatic systems are largely unstudied in Africa. The tropical region has received little attention from conservation organizations, managers, and local governments. For instance, Couceiro *et al.* (2007) showed that 22,360 km² of stream banks in tropical forest were affected annually by deforestation and very little is known about the ecological effects of this impact on the aquatic community. In Africa and the world at large, deforestation along the edges of the streams and rivers in many countries is associated with agricultural practices, such as logging (Couceiro *et al.* 2007; Lorion and Kennedy, 2009; Benstead and Pringle, 2004; Benstead *et al.* 2003; Kasangaki *et al.* 2008; Chapman and Chapman, 2003), and human settlement (Chapman and Chapman, 2003; Arthur, 2010). Deforestation changes the hydrological, geomorphological, and biochemical states of streams (Coe *et al.* 2011).

Effects on water quality: Several authors (for example: Couceiro *et al.* 2007; Lorion and Kennedy, 2009; Kasangaki *et al.* 2006 and 2008; Paul and Meyer, 2001) argue that riparian clearing and canopy opening may have many effects on water quality of streams and rivers ecosystems including increased electrical conductivity, turbidity, pH, temperature, and reduced transparency and dissolved oxygen. Similar studies conducted by Paul and Meyer, (2001) and Couceiro *et al.* (2007) reported that riparian deforestation associated with urbanization reduces food availability, affects stream temperature, and disrupts sediment, nutrient, and toxin uptake from surface runoff. Where riparian vegetation has been removed in the catchment, many streams and rivers present high nutrient inputs which favor largely growths of phytoplankton at levels to be considered indicative of eutrophication (KIMO, 2011; Nijboer and Verdonschot, 2004). Recently, Virbickas *et al.* (2011); Lorion and Kennedy, (2009); Kasangaki *et al.*, (2008) and (2006); Lorion, (2007); Couceiro *et al.*, (2007); Allan, (2004); Benstead and Pringle, (2004); Benstead *et al.*, (2003); Derleth, (2003); Storey and Cowley, (1997) have demonstrated that conversion of forest to agricultural lands can have significant impacts on stream biodiversity. Many results indicate that high levels of deforestation can alter the taxonomic composition of benthic macroinvertebrates communities, reduce macroinvertebrate diversity and eliminate the most sensitive taxa belonging to EPT groups.

In South Africa including Cape Town, land use consists largely of agricultural (livestock farming, dryland farming), and urbanization (settlement, canalization, industry, road, bridge).

Clearing of indigenous riparian vegetation have resulted in the invasion of alien plants, increased sedimentation which leads to modification of the river bed, and reduced water quality (River Health Programme, 2005). Alien vegetation may lead to instability of the river banks, elevated nutrient loads, clogging the water channel, flow modification, and low dissolved oxygen content (River Health Programme, 2006, 2005 and 2003). Water hyacinth in the Black River, for example, led to oxygen depletion, smothering of aquatic life, mosquitoes and restricted water flow (River Health Programme, 2005).

2.5 POLLUTION SOURCES AND THEIR CONSEQUENCES IN SOUTH AFRICAN AQUATIC ECOSYSTEMS

This section presents different sources of pollution, their consequences on aquatic systems and how they impact on aquatic life and socio-economic activities. The problem of pollution and its consequences on South African river and stream systems is well documented. Numerous research studies have shown that as a semi-arid country, South Africa is facing a water supply crisis due to low rainfall, high evaporation rates, and increased economic development and population growth. The combined effects of the natural environment (geology, climate) and a large variety of land use and land management practices have accelerated water quality degradation leading to numerous consequences such as salinization, eutrophication, acidification and pathogenic organisms.

2.5.1 Sources of pollution

The major sources of pollution in South African freshwaters include industry, urbanization, mining, agriculture, and power generation which may be categorized as both non-point and point sources.

Pollutants entering water bodies from non point sources such runoff from urban areas, seepage from mines, agricultural runoff and atmospheric pollutions are diffused and therefore, difficult both to quantify and to control. Human activities are responsible for pollutants generated by those non-point sources which enter the rivers and streams from terrestrial sources, through runoff (agricultural runoff or urban runoff), leaching, direct dumping, and livestock manure, drainage and interflow, or via groundwater and atmosphere deposition (Itoba, 2010; Davies and Day, 1998).

Modern agricultural practices including various processes such as land preparation, irrigation, fertilizer application, livestock handling and pesticide application may influence water quality (CISR, 2010; Dallas and Day, 2004). Through these processes, agricultural runoff and soil erosion transfer soil particles, nutrients including nitrogen and phosphorus, pesticides and herbicides, and pathogenic organisms to adjacent water bodies. Subsurface irrigation water may also alter surface and groundwater quality through salinization or potentially toxic trace elements (Dallas and Day, 2004). Urban South African rivers have been disturbed because of buildings erected too close to the river banks, riparian vegetation being cleared, canalization, inflows from stormwater drains, spills, and unauthorized dumping or washing, and exotic vegetation planted on the banks (Dallas and Day, 2004; Davies and Day, 1998). Runoff from urban areas include numerous pollutants and have adverse effects on aquatic ecosystems, namely flooding, erosion, sedimentation; physico-chemical effects such as elevated temperatures, dissolved oxygen depletion, nutrient enrichment, toxicity and biological effects. Most of greater Cape Town's rivers including Kuils River suffer from habitat loss due to canalization, informal settlement, and agriculture along the rivers (River Health Programme, 2005; Fisher, 2003). Urban run-off from streets and surrounding areas for instance, may be a major source of derivatives of fossil fuel combustion, bacteria, metals (e.g lead) and industrial organic pollutants. Pesticides from urban gardening, landscaping, horticulture and their regular use on railways, airfields and roadsides also contribute to the water pollution of many rivers.

In contrast, pollutants entering into river systems from point sources may be discharged legally under controlled or semi-controlled conditions, while others are discharged deliberately and illegally, or accidentally. The major point sources include discharges from untreated or inadequately sewage disposal, mines, industrial effluents, and fish farms (Downes *et al.* 2002; Davies and Day, 1998; Chapman, 1996).

A large proportion of sewage emanating from South African urban areas is not treated properly prior to discharge, because the sewer systems are incomplete or broken, or sewage treatment plants are overloaded and mismanaged. Many industrial processes produce waste products that contain hazardous chemicals, and these are sometimes discharged directly into sewers, rivers or wetlands (CSIR, 2010). Many sources of pollution in Kuils River were reported by Ninahm Shand (1979).

2.5.2 Major consequences of pollution in South Africa

This subsection reviews the major causes and consequences of pollution on water quality in South Africa in relation to aquatic biota, human health and socio-economic impacts. The main consequences include salinization, eutrophication, pathogen organisms and acidification.

2.5.2.1 Salinization

Salinization refers to increase concentration of dissolved inorganic salts or compound in natural water or in soil caused by the dissolution of minerals in rocks, soils and decomposing plant material. The influence of salinity in a river, depends on the geology and climate, pavent rock, evaporation and rainfall (Du Preez *et al.* 2000; DWAF, 1996a and b).

The natural lowest TDS values in South Africa rivers (0.9-3.6 mSm^{-1}) (10-27 mg L^{-1}), and 1.8-3.1 mSm^{-1} (17-37 mg L^{-1}) were observed in Waterkloof (Transvaal) and Swartboskloof (near Stellenbosch) streams, respectively. The natural highest salinities have regularly been recorded in the Sak River near Williston in the Karoo (maximal values 84 020 mg L^{-1}) (Dallas and Day, 2004). A study carried out in Berg River reported 60 mg L^{-1} as TDS concentrations at its source on Table Moutain Sandstone, while tributaries rising on Malmesbury shale, present a higher TDS concentrations above 3500 mg L^{-1} (De Villier *et al.*, 2003).

Anthropogenic effects: According to Dallas and Day, (2004) human activities have severely increased the TDS concentration of inland waters worldwide, particularly in arid regions. In South Africa**,** salinization of rivers is recognized as one of the major threats to water resources. In addition to natural condition such as geology and climate, human induced causes of salinization include discharge of municipal and industrial effluent, irrigation return flows, urban storm-water runoff, surface mobilization of pollutants from mining and industrial operations, and seepage from waste disposal sites, mining and industrial operations (CSIR, 2010; Le Roux, *et al.* 2007).

On a global scale, it has been estimated that over a million hectares of land have been lost to agriculture as a result of salinization of soils (Dallas and Day, 2004). Poor irrigation management systems are the primary cause of high levels of soil and water salinity. Surveys conducted by Nell and Van den Berg (2001) in Le Roux *et al*., (2007) showed land potentially available for irrigation in South Africa represents a total of 1.6 x 10^{6} ha with 1.1 x 10^{6} ha for temporary irrigation and 0.5 x 106 ha for permanent irrigation (sugar-cane included).

In most of South African rivers, such as the Berg and Breede rivers in the south-western Cape, and Sundays (TDS levels exceed 1000 mg L^{-1}) and Fish rivers in the Eastern Cape, although naturally occurring geological characteristics contribute to salinity to some extent, elevated concentrations of dissolved salts are aggravated by intensive agricultural land-use (CSIR, 2010; Davies and Day, 1998). The change in water quality of the Lower Vaal River and its tributaries may be due to high soil salinity because the river served as water source for a significant portion of the countris irrigated lands (LeRoux *et al.* 2007).

Fifteen years ago, Davies and Day, (1998) reported that during the previous 25 years salinity had increased in most South Africa Rivers that receive saline mine effluents. In the Vaal Dam, for instance, the concentration of TDS is rising at a rate of 2.5 mgL^{-1} every year and an increase in Vaal Barrage has been noted from less than 200 mg L^{-1} in the 1930s to more than 550 mg L^{-1} in the early 1980s (Davies and Day, 1998). According to water quality guidelines for aquatic ecosystems, salinity is recognized as non-toxic inorganic constituent that may cause toxic effects only at high concentrations (DWAF, 1996). However, the heavy metals are considered as toxic because they may cause toxic effects at low concentrations. The common ions such as Na^{+}, Ca^{++}, Mg^{+2}, Cl^{-}, SO_4^{-2} and HCO_3^{-} present toxic and other effects only at high concentrations compared to normal background levels. In general, these common ions make up the major fraction on the total ionic concentration in many South Africa waters. Note that under salinity waters and groundwaters in South Africa are a significant problem, of national concern (Leske and Buckley, 2003)

Effect on water quality: High salt levels in surface water may modify oxygen, temperatures, sediment inputs and organic material sources (Leske and Buckley, 2003).

Effect on community: Plants and animals possess a wide range of physiological mechanisms and adaptations to maintain the necessary balance of water and dissolved ions in cells and tissues (DWAF, 1996). However, changes in the dissolved salt concentration can have effects on individual species, community structures and on microbial and ecological processes such as rates of metabolism and nutrient cycling (Leske and Buckley, 2003; DWAF, 1996a).

Invertebrates are more sensitive to increasing salinities. The most sensitive insects include stoneflies, some mayflies, caddisflies, dragonflies and waterbugs. The most sensitive molluscs are pulmonate gastropods. Larval fish are more sensitive than eggs and adults. Fish are generally tolerant to salinities in excess of 10 000 mg L^{-1} TDS. Salinity tolerance studies of selected macro-invertebrates of the Sabie River have linked mortality to increasing salinity and the nature of the salt that elevated the salinity (Leske and Buckley, 2003).

High salt levels in surface water may also cause a decrease in the abundance and diversity of riparian vegetation. Salinity in the root zone can adversely affect plant growth due to a decrease of the osmotic potential caused by the high concentration of soluble ions (Leske and Buckley, 2003).

Economic impacts: High levels of salinity can lead to diminished crop yields, increased scale formation and corrosion in domestic and industrial water pipes and increased requirement for pre-treatment of selected industrial water uses. As regards agriculture, high salinity leads to a reduction of yield, and of the quality of crops (CSIR, 2010; DWAF, 1996b and d).

2.5.2.2 Eutrophication

Eutrophication is a process whereby water bodies receive excess inorganic nutrients, especially N and P, which stimulate excessive growth of macrophyte and algae or cyanobacteria. In most fresh waters, the major nutrients that contribute to eutrophication are nitrogen which occurs as nitrate (NO_3), nitrite (NO_2) and ammonia (NH_3), and phosphorus as ortho-phosphate (PO_4) (CSIR, 2010; De Villiers, 2007; Nijboer and Verdonschot, 2004). Both N and P (in organic and inorganic forms) could be important determinants of autotrophic and heterotrophic activities in rivers and streams (Dodds, 2006). Nitrates and phosphates are discharged into the aquatic environment from natural and human sources and these nutrients alter ecosystems' function and structure (KIMO, 2011). Natural eutrophication due to natural influxes of nutrients is considered as not reversible or controllable, and will therefore continue slowly and inevitably (Van Ginkel, 2011).

In South African natural water bodies' nitrogen (N) and phosphorus (P) concentrations vary with local geology, climate, and natural characteristics of the catchment (Frost and Sullivan, 2010; Davies and Day, 1998). In mostly unimpacted surface waters, inorganic nitrogen concentrations are usually below 0.5 mg N L^{-1} but may increase to above 5-10 mg N L^{-1} in highly enriched waters (DWAF, 1996a). Phosphorus is rarely found in high concentrations in unimpacted surface waters because it is actively taken up by plants (Davies and Day, 1998; DWAF, 1996a). For instance, in certain non-polluted waters, soluble inorganic phosphorus concentration may be as low as 1 mg L^{-1} or even <0.01 mg L^{-1} (Dallas and Day, 2004; DWAF, 1996). Oberhoster and Ashton (2008) reported that the average phosphorus concentration in natural water resources of South Africa (as orthophosphate) may be evaluated at 0.73 mg L^{-1}. Nevertheless concentrations between 10 and 50 mg L^{-1} are commonly recorded, and values as high as 200 mg L^{-1} of total phosphorus are often found in some enclosed saline waters (DWAF, 1996a).

Anthropogenic effects: Increased discharges of nitrogen (N) and phosphorus (P) recorded in most aquatic ecosystems since the 20th century are attributed to human activities (KIMO, 2011; Gooday *et al.* 2009) such as agricultural intensification and increased discharge of domestic waste from urban areas (Van Ginkel, 2011; De Villiers, 2007).

It has been shown that South Africa's climatic conditions, combined with various factors, namely storm water runoff, discharge of treated and untreated sewage effluent from urban areas and industrial development, excessive nutrient loads in return flows from modern agriculture practices, modification of river flow regimes, and changing land use or land cover patterns have resulted in large-scale changes to aquatic ecosystems and subsequent eutrophication of rivers and water storage reservoirs (CSIR, 2010; Oberholster and Ashton, 2008). South Africa has some of the most enriched surface water in the world where eutrophication presents a major problem (Frost and Sullivan, 2010). During the past 40 years, eutrophication has become an increasing threat to the usability of South African freshwater resources. Van Ginkel (2011) indicated that numerous studies recognized the problem of eutrophication in the late 1970s. Although many eutrophication management options such as effluent discharge standards in 1980, National Eutrophication Monitoring Programme (NEMP) in 1985 and Reservoir Eutrophication Model (REM) since 1985 to 1988 were suggested, South Africa eutrophication remains a real challenge.

Until the mid-1980s South Africa was recognized as a world-leader in eutrophication research (Oberhoster and Ashton, 2008). To prevent eutrophication, an effluent standard measure of 1 mg/l orthophosphate for wastewater discharge from point sources, an eutrophication control guideline, and several structures of surveillance were initiated by DWAF. Inorganic phosphorus and inorganic nitrogen concentrations of less than 0.005 mgP L^{-1} and 0.5 mgN L^{-1} are considered to be sufficiently low to reduce the likelihood of algal and other plant growth. However, in the absence of sufficient available phosphorus, nitrogen-fixing organisms will be able to fix atmospheric nitrogen, thereby compensating for any deficit caused by low inorganic nitrogen concentrations (Dallas and Day, 2004; DWAF, 1996). The classification of trophic status of South Africa's aquatic ecosystems associated with aquatic ecosystems is presented in Table 2.2 based on the work of Van Ginkel (2011) and DWAF (1996a)

Table 2.2 Classification system used by the DWAF to classify the National Eutrophication Monitoring Programme sites regarding their trophic status (DWAF, 1996a; Van Ginkel, 2011)

Parameter	Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Inorganic Nitrogen	<500 µg N L^{-1} or <0.5 mgL^{-1}	500 - 2500 µg N L^{-1} Or 0.5 – 2.5 mgL^{-1}	2500-10000 µgN L^{-1} Or 2.5 – 10 mgL^{-1}	>10 000 µg NL^{-1} or >10 mgL^{-1}
Inorganic Phosphorus	<5 µg L^{-1} or <0.005 mgL^{-1}	5 - 25 µg L^{-1}or 0.005-0.025 mgL^{-1}	25 - 250 µg L^{-1} 0.025-0.250 mgL^{-1}	>250 µg L^{-1} >0.250 mgL^{-1}
Mean annual total Phosphorus mgP/l	<0. 015 mgL^{-1}	0.015–0.047 mgL^{-1}	0.047 – 0.130 mgL^{-1}	>0.130 mgL^{-1}

Despite numerous research programs carried out by NEMP, WCR, and RHP, South Africa eutrophication management presents a relatively low priority and an inability to transform policy into practice (CSIR, 2010). Two important consequences result from this situation, namely stimulation of the growth of blooms of cyanobacteria carrying the threat of cyanotoxin contamination, and excessive growth of macrophytes, which clog water-supply structures and reduce the recreational value of aquatic resources (Van Ginkel, 2011).

Impacts: In South African rivers and reservoirs, the dominant phytoplankton are usually the cyanobacteria: *Microcystis* and *Anabaena* (Oberholster and Ashton, 2008). The excessive algal blooms (cyanobacteria) can cause various effects, namely economic impacts, environmental impact (deteriorating water quality and loss of biodiversity), social impacts (aesthetic and recreational), and human health impacts (Frost and Sullivan, 2010).

-Economic impacts: The economic impacts of eutrophication may be categorized across treatment measures, alternative water sources and agriculture. In South Africa, most of the drinking water that is supplied to communities is obtained from surface water sources and needs to be treated before human consumption. The excessive growth of toxic cyanobacteria increases the costs of water treatment. Excessive algal blooms can also clog filters and increase maintenance costs. South African surface waters used by rural population are often subject to cyanobacterial contamination and alternative sources are needed which usually require costly investment. The phenomenon of eutrophication also has significant negative impacts on both farming and fishing (CSIR, 2010; Frost and Sullivan, 2010).

- **Environmental impacts:** Excessive algal blooms decrease the amount of dissolved oxygen in the water which leads to the death of fish, shrimps, and immobile bottom dwellers. A decline in macroinvertebrate abundance and variation in composition and species richness also occur. Increased abundance of pollution-tolerant invertebrates such as *Tubifex*, *Chironomus* and Hirudinea, and alteration of assemblage structure may occur because of their ability to tolerate the lower oxygen concentrations (Frost and Sullivan, 2010; Nyenje *et al.* 2010; Dodds, 2006; Rast and Thornton, 1996). The variety and nature of species within a certain catchment provide an important indicator for environmental change due to eutrophication (Frost and Sullivan, 2010).

Another adverse effect of eutrophication is an increase in macrophytes such as water hyacinth, phytoplankton abundance, bacteria biomass, and suspended inorganic particles which lead to a reduction of water clarity (Nyenje *et al.*, 2010). Macrophyte or algal and cyanobacteria invasion impede the growth of other aquatic plants. Nutrient enrichment causes an intensification of all biological activity and typically leads to dramatic changes in the composition and structure of aquatic food webs. With regards to water quality, the major consequences of eutrophication include the availability of oxygen and changes in pH. The lack of dissolved oxygen is linked to the accumulation and decomposition of dead organic matter which consumes oxygen and generates harmful gases such as methane and hydrogen sulphide that impact biologically on communities (Frost and Sullivan, 2010; Nyenje *et al.* 2010). Certain algal blooms release toxins which cause death in animals and humans alike. It has also been observed that other cyanobacterial products, such as mucopolysaccharides lead to high concentrations of metal ions such aluminium in the potable water supply (CSIR, 2010).

Social impacts: In most South African rivers, social impacts from eutrophication include aesthetic and recreational impacts. Frost and Sullivan (2010) report excess algal bloom and surface scum that are ugly to see and give off noxious odors, which affect the aesthetics of water bodies. The presence of macrophytes such as water hyacinth and *Typha capensis* decrease the fitness for use of the water for water sports such as swimming, skiing, yachting and fishing (Frost and Sullivan, 2010; Van Ginkel, 2011).

-**Health impact:** Water hyacinth and algal provide ideal breeding habitat for mosquito larvae (*Anopheles*) vector of *Plasmodium* (malaria) and snail such as *Biomphalaria* and *Bulinus* vectors of Schistosomiasis. Davies and Day (1998) estimate that about 2 million people in South Africa have Schistosomiasis, out of which 10 % show severe symptoms of the disease. Malaria has, on the other hand, been considered endemic in the northern parts of South Africa including northern KwaZulu-Natal. Mostiquo larvae and pupa usually collect in sheltered, slow-moving or still water from irrigation canal, reservoirs, pond and vleis. Cyanobacterial toxins can cause a great hazard to human health. According to Codd *et al.* (2005) cyanobacterial toxins are grouped according to the physiological systems, organs, tissues, or cells which are primarily affected. These toxins lead to serious threats to human health such as the death from liver haemorrhage, gastrointestinal and other hepatic illnesses.

2.5.2.3 Pathogen organisms

In many parts of the world today, waterborne diseases remain a major hazard. They are endemic in those countries which have not yet established systems for sanitary disposal of waste (Abel, 2002). In 1978, over 7.5 million people, principally children, in Africa died of diarrhoeal infections due to poor sanitation and inadequate water supplies (Davies and Day, 1998). Fecal contamination is still the primary water quality issue in rivers, especially in many developing countries where human and animal wastes are not yet adequately collected and treated (Ahuja, 2009).

Water constitutes both a source and vehicle agent for many of these pathogenic organisms, which emerge when environments are favorable to their development (CSIR, 2010). Bacterial densities are usually higher in urban streams, especially after storms. Much of this is attributable to increase coliform bacteria, especially in catchments with wastewater treatment plant (WWTP) and combined sewer overflow (CSO) effluent (Paul and Meyer, 2001). Pathogenic organisms which are spread in polluted water include bacteria, viruses, protozoan parasites, parasitic worms, and fungi (CSIR, 2010; Abel, 2002; DWAF, 1996). Diseases caused by the use of water contaminated with those pathogenic organisms include cholera, typhoid, dysentery, gastroenteritis, schistosomiasis, salmonellosis, eye, skin and nose infections, and other diarrheal diseases (Ahuja, 2009; DWAF, 1996b). The lack of treated potable water remains an important issue in many rural communities in the developing world, including South Africa (Bessong *et al.* 2009). Infections are often contracted by drinking contaminated water, recreational exposure to contaminated water, inhaling contaminated aerosols or the consumption of raw food (irrigated vegetables and shellfish) exposed to polluted water.

Faecal coliforms and total coliform bacteria are primarily used to indicate the presence of bacterial pathogens (e.g *Shigella, Salmonella, Vibrio cholerae*, *Campylobacter*, pathogenic *E. coli*). Those pathogens are transmitted via the faecal/oral route by contaminated or poorly-treated drinking water (Abel, 2002; DWAF, 1996b). According to South African water quality Guidelines for drinking water the number of total coliforms in drinking water should be less than 10 colonies per 100 ml, while the number of faecal coliforms should be zero per 100 ml. Higher concentrations of faecal coliforms in water will indicate a higher risk of contracting waterborne disease, even if small amounts of water are consumed (DWAF, 1996b). In South Africa, the major sources of microbiological contamination of water in rivers include human settlement, inadequate sanitation and waste removal practices, storm water wash-off and sewage spills (CSIR, 2010). Because the availability of clean water for drinking and washing is still inadequate in most South Africa's rural areas, townships and informal settlements (Davies and Day, 1998) the spread of disease such as cholera, cryptosporidiosis, dysentery and typhoid remain a major concern (CSIR, 2010). Studies carried out by Bessong *et al.* (2009) indicated that Tshikuwi community in Venda use untreated water from Khandanama River, and wells for drinking, cooking, and laundry. The presence of *Vibrion*, *Salmonella*, and *Shigella* species and the detection of total coliforms, faecal coliform, and enterococci recorded in water sampling were associated with diarrhea (41% of 145 individuals) observed in Tshikuwi community. Similar studies were carried out by several authors in South Africa, for instance Theron and Cloete, (2002); Germs *et al.* (2004); and Zamxaka *et al.* (2004) showed that both the total and faecal coliform counts in all the sites under consideration were above the South African recommended limits for drinking water.

As noted earlier by Davies and Day (1998) Bilharzia (schistosomiasis) is not only a major health in South Africa, the Africa continent at large. In Africa, three species of the disease infect humans, namely *Schistosoma mansoni* and *S. intercalatum* which parasitise the blood vessels and cause intestinal bilharzias, while *S. haematobium* lives in blood vessels of the bladders and causes urinary bilharzias. The snail vector of *S. mansoni* is *Biomphalaria pfeifferi*, while *S. haematobium* has for intermediary host *Bulinus (Physopsis) globosus*.

2.5.2.4 Acidification

Acidification of freshwater is a critical problem in many regions of the world. Acid deposition affecting fresh water originates from sulfur dioxide and nitrogen which are released into the atmosphere through burning of fossil fuels (Perry and Vanderklein, 1996). Acidification of natural water is also determined by bedrock (CSIR, 2010), biotic activities (Davies and Day, 1998), and dominant wind and climate patterns that are responsible for the deposition patterns of acidifying elements (Perry and Vanderklein, 1996). Acidification effects are influenced by the timing of acid deposition, the source and rate of acid deposition and buffering capacity of soils and bedrock (Perry and Vanderklein, 1996). Many South Africa fresh water resources are relatively well buffered (CSIR, 2010) and more or less neutral, with pH value around 6 to 8 (Dallas and Day, 2004). However, human activities such as industrial effluents from mine drainage, and acid-rain may decrease the pH value in many South African aquatic ecosystems (CSIR, 2010). Although the South African coal mining industry contributes large revenues for the country (e.g 16% of export revenue in 2003), it presents a great environmental risk from the coal fields. Acid mine drainage (AMD) has major consequences such as ground water, soil quality, and surface water degradations (CSIR, 2010). Acid mine drainage from active and abandoned mines contribute to lowered pH levels , elevated concentration of metal ions and dissolved salt dominated by sulphate (CSIR, 2010; Coetzee, undated).

Dallas and Day (2004) note that water from acidic coal mine drainage is normally colored to deep orange, contain sulphates of ferrous and ferric iron, aluminium, calcium, magnesium and usually sodium, and has a very low pH (down to 2) and a high TDS. The low pH itself has adverse effects on the receiving water flora and fauna. It also promotes the solubilisation of heavy metals, which exert their own toxic effects. Aluminium for example, a metal which does not commonly cause serious problems of toxicity to aquatic life, has received particular attention. In general, the result is a marked reduction in species diversity and biomass in the affected areas (Abel, 2002). Acid rain has been recognized as a substantial problem in other parts of the world including South Africa. It occurs when sulphur dioxide, carbon dioxide and oxides of nitrogen are emitted into the atmosphere by fossil fuel burning, iron smelting, chemical extraction processes and motor vehicle exhaust emission (Coetzee, undated; DWAF, 1996a). Acid precipitation has been noted in many South African regions where the lowest pH value for rain recorded was 2.9 (Davies and Day, 1998). A study carried out in the catchment area of the Pienaars River north of Pretoria found a mean rain pH of 4.5. The rain pH ranged from 3.9 to 4.6 in Eastern Transvaal Highveld in 1984 (Davies and Day, 1998). Ochieng *et al.* (2010) revealed that water affected by acid mine drainage systems temperatures reached of 47°C with a pH as low as 3.6. Consequently it affected aquatic biota in term of diversity and community structure (Perry and Vanderklein, 1996). Simmons, *et al.*, (undated) reported that streams receiving treated AMD showed in terms of macroinvertebrate diversity significantly few taxa, low Shannon-Weiner diversity indices, and low percentages of EPT organisms.

CHAPTER THREE: METHODOLOGY

3.1 STUDY AREAS

3.1.1 The location of the City of Cape Town

The City of Cape Town is located at the Northern end of the Cape Peninsula, Western Cape Province, South Africa. The peninsula consists of a dramatic mountainous spine jutting southwards into the Atlantic Ocean, ending at Cape Point. Most of the suburbs of the city are on the large plain of the Cape Flats, which join the peninsula to the mainland. The City of Cape Town is sheltered in a valley, overhang by a series of mountains, with the touristic Table Mountain whose elevation is 1086m. In addition to these mountains, a Group of Aquifers on the South serve as important points in term water resources. The Devil's Peak (elevation: 1000 m) is situated to the Southeast and the Lion's Head (elevation: 669 m) to the Southwest. It presents a Mediterranean climate type characterized by hot and dry summer (November to March), and cold wet winters (end May to August). The spring (September and October) and autumn (April and May) seasons characterize the City of Cape Town with daytime temperatures rising to 20°C. Runoff is mostly generated in the mountain ranges of the Hottentots Holland Mountains in the Southeast and Table Mountain and Cape Peninsula mountains in the Southwest. Its principal catchments include in the East (588 km^2): Steenbras, Sir Lowry's Pass, Lourens and Eerste/Kuils; in the South (471 km^2) Sand, Zeekoe, Sylvermine; centrally (327 km^2) Hout Bay and Salt; and in the North (1087 km^2) Diep and Sout catchments. In these catchment areas, most rivers arise in the Sandstone Mountains ranges of the Hottentots Holland Mountains in the East, and Table Mountain and Cape Peninsula Mountains in the Southwest (Adelana *et al.* 2010; Brown and Magoba, 2009**).**

3.1.2 Kuils River catchment description (Figure 3.2)

3.1.2.1 Location

Kuils River catchment, located in the City of Cape Town, measures 261km^2 from the source to the confluence with the Eerste River (Ninham, 1999 in Fisher, 2003). It extends into Durbanville area north-westwards and Bellville to the West. In the south-west, the catchment area is limited by Cape Town International airport and Mitchells Plain, while to the northeast it is bordered by Kraainfontein urban area which is dominated by agricultural activities in the whole eastern area (Ninham, 1979). The Southern part extends to the False Bay coast where the Southern side of the dunes drains to the sea rather than into the Kuils River (Ninham, 1979). Kuils River is a short river, 30 km in length that flows southwards. It originates from Durbanville Hills (elevation between 300 m and 500 m) between the Tygerberg (300 m and 500 m) and the Bottelary Hills (350 m) east of Kanonkop (450 m) (Fisher, 2003; Heydorn and Grindley, 1982). Kuils River crosses the Eastern Cape Flats before joining Eerste River near the False Bay estuary in the south and receives Bottelary River as its main tributary in upper reaches. A large number of road-bridges and variable channel conditions in the course affect the free flow of waters and increase upstream water levels (CoCT, 2011; Heydorn and Grindley, 1982). Furthermore, minor tributaries of the Kuils River are dammed by Sonstraal Dam and Edward Dam both on the boundary of the Durbanville Municipality (Heydorn and Grindley, 1982).

3.1.2.2 Vegetation

In the major extent of Kuils River catchment area, the original vegetation has been replaced by agricultural activities, urban development and alien vegetation. The alien vegetation was dominated by Port Jackson willow (*Acacia saligna*), white poplar (*Populus canescensus*), Rooikrans (*Acacia cyclops*) and black wattle (*Acacia mearnsii*) (Heydorn and Grindley, 1982). Where the N1 crosses the river the riparian zone is edged by sedges (e.g. *Cyperus brevis*, *Juncus punctorius*), grasses (e.g. kikuyu grass, *Pennisetum clandestinum*), bulrushes (*Typha capensis*), sparse shrubs (e.g. *Cliffortia strobilifera)* and alien vegetation such as, *Sesbania punicea*. Vleibos (*Cliffortia strobilifera*) are common riverbank plants found near the N1/R300 interchange. Wetland systems are dominated by low grasses and sedges with occasional emergent plants such as bulrush (Typha) (Ewart-Smith and Ractliffe, 2002).

3.1.2.3 Geology

The geology of Kuils River catchment area includes the Malmesbury group of rocks formed from Pre-Cambrian age (Ninham, 1979). The Malmesbury group of rocks consist of a variety of shales, greywackes, phyllite, siltstones, quartzites and grits (CoCT, 2011; Conradie *et al.*, 2002; Heydorn and Grindley, 1982; Ninham, 1979). Those rocks are covered with recent thin layer of turf and loamy sands with some alluvial deposition (Ninham, 1979). In the area, there are occasional bands of gravel, conglomerate, limestone, dolomite, chert, basic lavas and tuff rocks (CoCT, 2011; Conradie *et al.*, 2002; Ninham, 1979). Below the confluence of the Kuils and Bottelary rivers, the geology is characterized by the considerable depths of tertiary and recent deposits of loose sand and dune formations underlained by extensive clay lenses.

3.1.2.4 Climate and Hydrology

The Kuils River catchment is located in a Mediterranean climate region with hot dry summers and cold wet winters. Generally, the Kuils River has high peak winter flows and low summer flows (Li Rui, 2005). Winter temperatures vary from 7 to 18°C and from 15 to 27° in summer which may sometimes attain a maximum of 40°C in February and March. The Kuils River receives a mean annual runoff which is estimated to be 22,000,000 m³ excluding sewage effluent discharged into the river. The urban area and sewage effluent discharged into the river constitute favorable factors to increase inundation in the catchment area (Ninham, 1979). The Kuils River catchment area is quite dry. The average yearly precipitation recorded reaches around 800 mm in the Tygerberg Hills to about 500 mm vicinity of the coast (Heydorn and Grindley, 1982; Ninham, 1979).

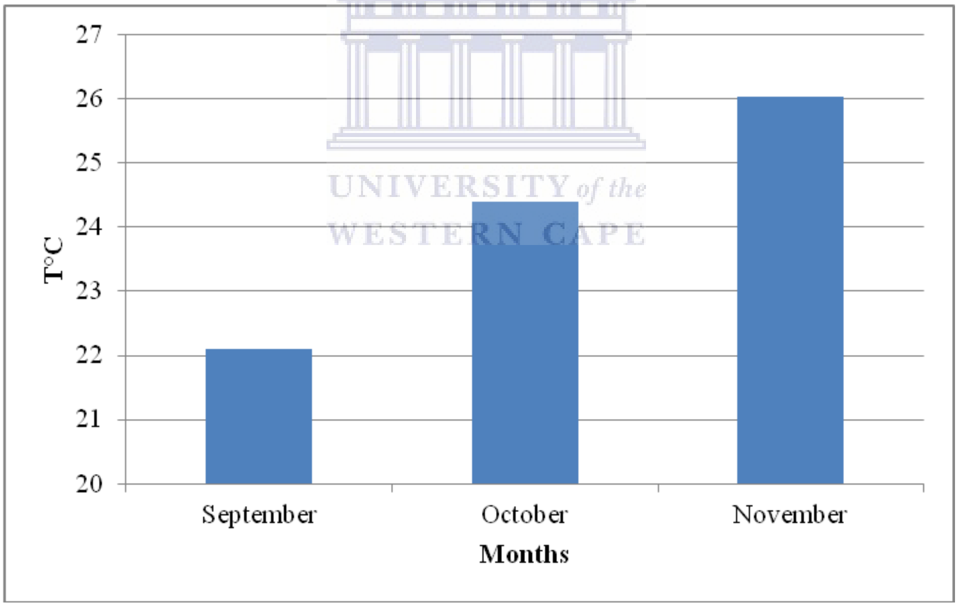

Figure 3.1 : Air temperature variations in the study area from September to November 2012

The geological features of catchment area influence the surface runoff characteristics. Runoff is relatively high in the upper reaches with little subsurface flow (Heydom and Grindley, 1982). The main flow of the river crosses a part of the Vergenoegd farm both North and South of the N2 freeway. The natural flow of the river is characterized by high flood peaks and low base flows. The base flow is highly saline in the headwaters. The sand areas of the lower reaches result in little surface runoff (Heydorn and Grindley, 1982). The main tributary of Kuils River is the Bottelary River in the upper reach. It receives other minor tributaries such as the Langverwacht and Swart Rivers farther south which drains the slopes of the Bottelary Hills (Heydorn and Grindley, 1982).

3.1.2.5 Soils

The soil is characterized by shale and granite clays, and rocky soils. It arises from the weathering of Malmesbury Group shale, a result of deposition and compression of silt and clay. These soils are not often vulnerable to erosion. They provide a good basis for agriculture due to high nutrients and their fertility (CoCT, 2011; Conradie *et al.*, 2002).

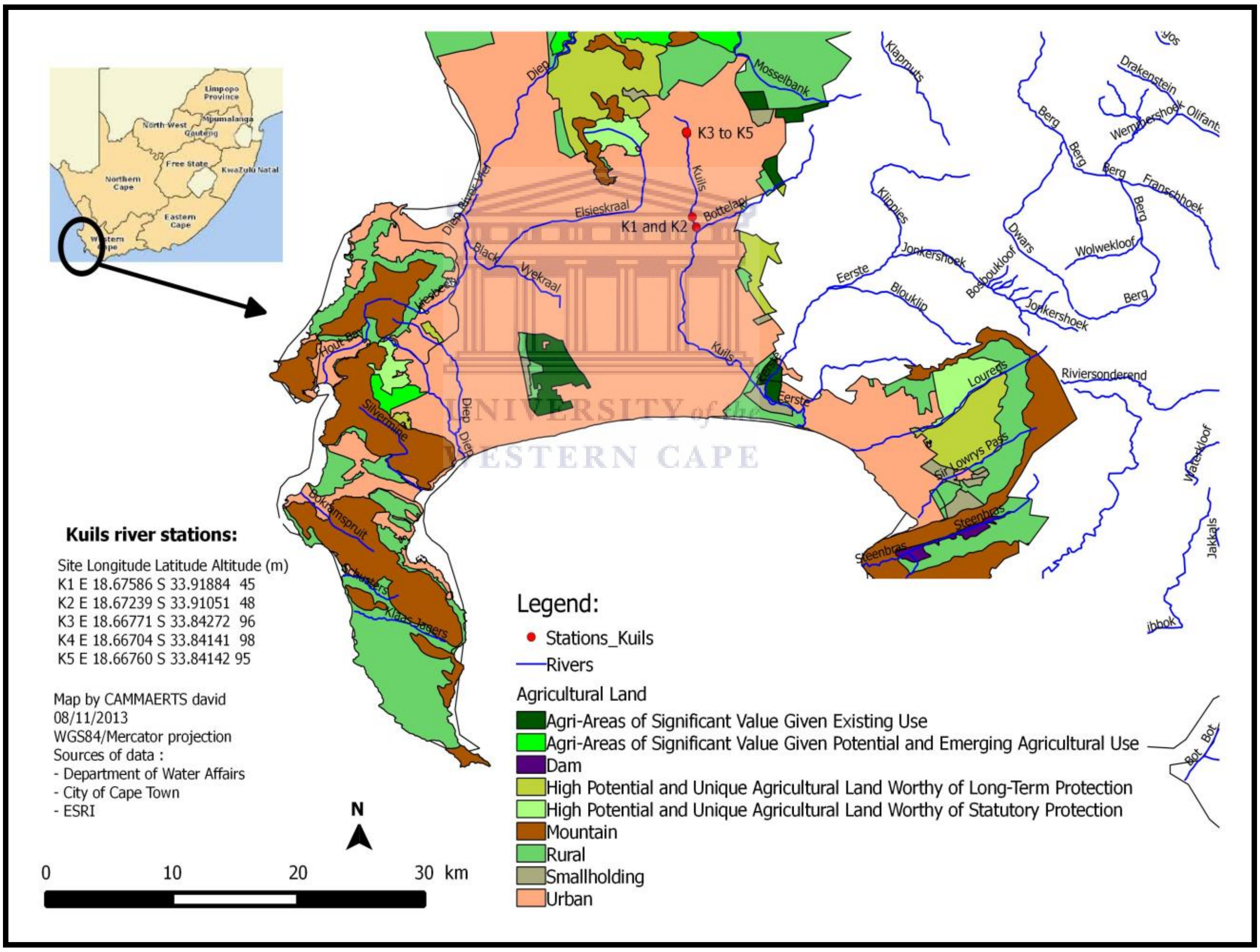

Figure 3.2: Kuils River and sampling site locations

3.2 SAMPLE POINTS SELECTION AND DESCRIPTION

3.2.1 Sample sites description (figure 3.3)

To evaluate land use effects on Kuils River a total of 5 sampling sites (two downstream sites: K1 and K2, and three upstream sites K3, K4 and K5 have been selected in the upper reach of the river. The K4 is situated in the small branch of the Kuils River between K3 and K5. Sites were selected according to the access points and identified sources of pollution, namely storm water, residential and industrial areas, recreation sites, and road-bridge. For those pollution sources, the major consequences include salinization and eutrophication which affect water quality, aquatic biota and socio-economic activities.

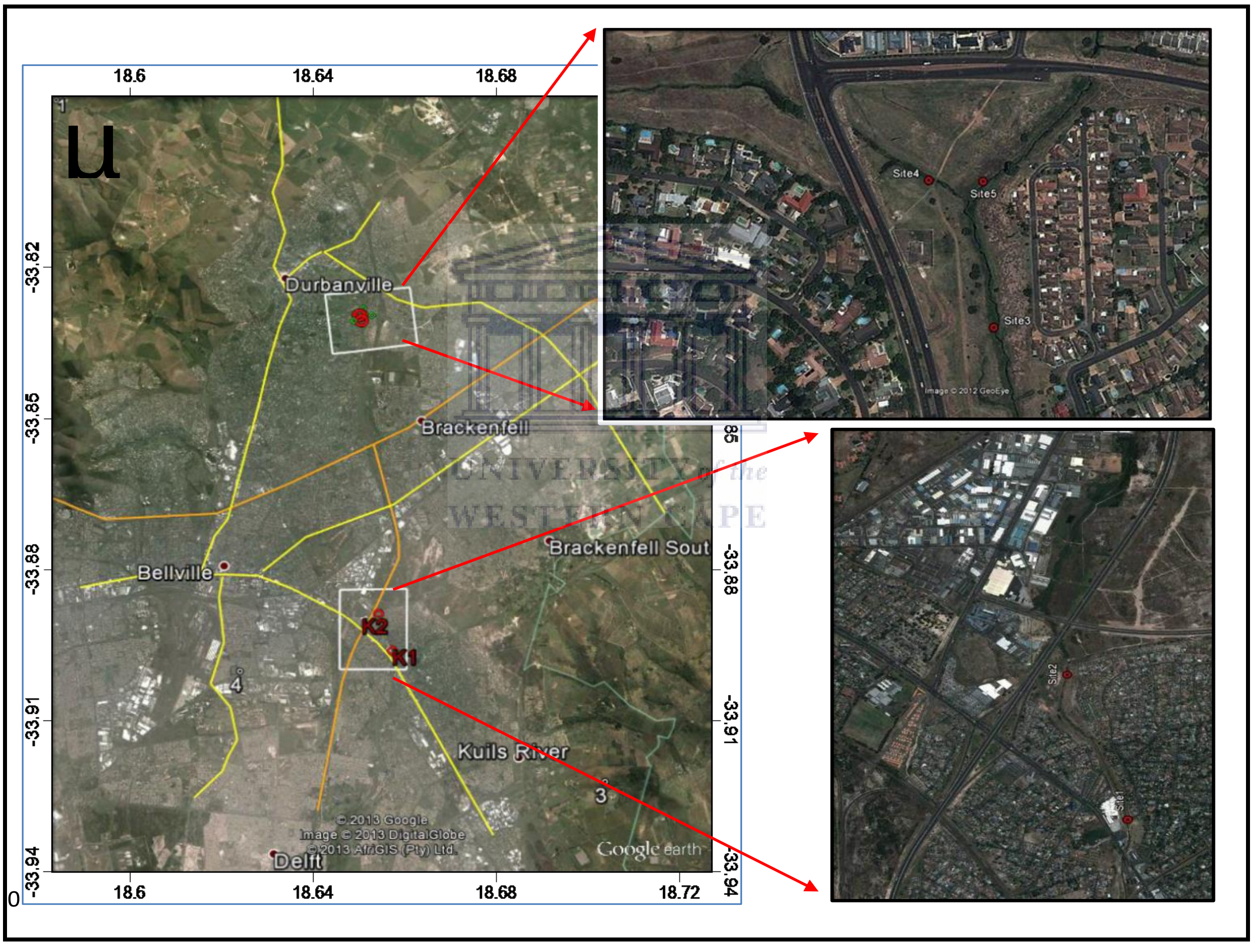

Figure 3.3: Sites selected upper of the Kuils River (K1, K2, K3, K4 and K5)

At each site, the location (latitude, longitude) and elevation (altitude) were measured using a Global Positioning System (GPS) *eTrex* SUMMIT (Garmin, USA). The bottom substrate, canopy cover, aquatic and marginal vegetations, and sources and nature of pollutants were visually estimated. The air temperature was recorded using maxima and minima thermometer.

The width was measured using a tape, and stream depth was obtained weekly using a graduated ruler. All these characteristics are summarized in table 3.1. The principal pollution sources affecting selected sites include both point and nonpoint sources in table 3.2.

Table 3.1 Location and description of sites selected upper of the Kuils River

	Downstream sites		Upstream sites		
Characteristics	K1	K2	K3	K4	K5
Location	S33.91884, E18.67586	S33.91051, E18.67239	S33.84272, E18.66771	S33.84141, E18.66704	S33.84142, E18.66760
Altitude	45 ± 4 m	48 ± 4 m	96 ± 3 m	98 ± 3 m	95 ± 4 m
Air temperature (°C)	22.8	23.4	24.6	25.0	25.4
Width (m) average	2.9	4.8	2.01	1.5	2.96
Depth (cm) average	26.0	21.3	14.9	9.5	18.7
Flow	0.74	0.86	0.20	very slow	0.40
Substratum (%)	Sand (100 %)	Sand (80 %), gravel covered by algae (20%)	Sand (70 %) gravel (30 %)	Sand mixed with debris (100 %)	Stones and solid blocks (60 %), sand (30 %) Gravel (10 %)
Decaying plant and		Organic detritus from	Decaying plant (20 %)	Decaying plant in	Organic detritus
organic detritus		road-bridge		September and mid- October	from urban area
Aquatic vegetation	Marginal vegetation covered by algal (10 %)	Marginal (10 %), Aquatic vegetation covered by algal (5 %)	Aquatic vegetation (10 %), filamentous algal	Aquatic vegetation (100 %) from mid-October to November	Marginal plant (10 %) Filamentous algal
Riparian vegetation	Open canopy (100 %) Edge dominated by grass	Open canopy (100 %) Edge dominated by grass	Open canopy (100 %) Edge dominated by grass	Open canopy (100 %) Edge dominated by grass	Open canopy (100 %) Edge dominated by grass
Sources of pollution	Residential, industrial and Hospital areas	Residential & industrial area road-bridge,	Residence area, storm water from urban area, effluent (tributary)	Redisential area, storm water from urban area road-bridge, dead plants poison to kill root, Golf course	Residential area stormwater from urban area

Table 3.2 Type of pollution sources at sampling sites

Sites	Point sources	Non point sources
K1		Surface runoff from residence and hospital areas
K2	Wastewater effluent from residence and industrial areas	Surface runoff from residence area
	Stormwater runoff from urban and industrial areas	Surface runoff from residence and industrial areas
	Organic detritus from bridge runoff	
K3	Stormwater runoff from residence area	Surface runoff from residence areas
	Small temporary effluent (tributary) from residence area	
K4	Stormwater from residence area	Surface runoff from residence area, Surface runoff from golf course
K5		Surface runoff from residence area

3.2.2 Data sampling

To evaluate and quantify changes in water quality a total of 65 samples of physical and chemical parameters were collected weekly from 4th September 2012 to 27th November 2012 at sampling stations upper reach of the river. As regards biological parameters, 55 samples were collected from 18th September to 27th November 2012.

3.2.2.1 The physical and chemical parameters

From the five stations, physical and chemical parameters namely, pH, water temperature (°C), Total Dissolved Solids (mg L^{-1}), Dissolved Oxygen (mg L^{-1}), Oxygen saturation (%), and Salinity were recorded immediately (in-situ) with an YSI 30 meter.

At each site, a water sample was collected at the middle of the river using a sterile white plastic bottle of 250 ml. Before sampling, the container was rinsed three times with the water to be sampled. And then, the container was plunged into the stream and filled up with water. Once filled, the container was tightened to prevent air from entering. The samples were transported to University of Western Cape (UWC) laboratory and kept in the refrigerator at 4°C before analysis.

3.2.2.2 Macroinvertebrate sampling

With respect to current ecological state, benthic macroinvertebrates as bioindicators of water quality were used referring to the South African Scoring System version5 (SASS5). This technique is based on the British system, adapted by Chutter in 1994 to assess water quality in rivers in South Africa (Dicken and Graham, 2002). It is based on the presence or absence of benthic macroinvertebrates of which a score is attributed to each taxa. The sensitive taxa have a high score while a low score is attributed to tolerant and less sensitive taxa.

Data collecting and identification

The sampling of benthic macroinvertebrates was carried out from 18th September 2012 to 27th November 2012 using the rapid bioassessment protocol South Africa Scoring System version (SASS5) (Dallas, 2005; Dickens and Graham, 2002). It consists of using a *D-frame net* (500-µm mesh net) which is placed downstream of the area sampled oriented perpendicularly to the stream current. All available SASS5 biotopes, namely, aquatic and marginal vegetations, stones and solid objects, gravel, sand and mud (GSM) were sampled separately. Stones and solid objects (blocks) found in the stream were sampled vigorously by kicking, turning and scraping them using the feet and hands so that the disturbed particles and macroinvertebrates dislodged are carried into the net by stream current. In marginal, aquatic vegetation and debris biotopes the net was pushed strongly under the water into the emergent plants, and many time through the submerged vegetation and into the roots of aquatic plants. The sand and mud also were shifted with the feet to disturb all area so that all organisms dislodged are carried into the net. After using the net, picking-hand has been used for supposed missed specimens.

The content of each sample was then washed in the bottom of the net and carefully emptied into a separate white tray with water by inverting the net. To reduce sample volume, the content was washed with water to separate the specimens from debris. After cleaning, all organisms were transferred into a plastic bottle and carried at University laboratory.

The organisms collected were sorted, counted and identified at family level using dichotomous key of Gerber & Gabriel, (2002) and Tachet *et al.* (2003) to family level except for Oligochaeta, and Hirudina for which a higher taxonomic level of order was used.

3.3 DATA ANALYSIS AND INTERPRETATION

3.3.1 Water quality analysis

To estimate nutrient concentrations, a 10 ml water sample was analyzed in the laboratory at UWC using a DR 2700 spectrophotometer of HACH. The nitrate concentration was measured following the Cadmium Reduction Method using *NitraVer 5 Nitrate reagent* powder pillow for a 10 ml sample. The result was expressed in mg $L^{-1}NO_3^{-}$-N was recorded.

The phosphate concentration was estimated according to PhosVer 3 (Ascorbic Acid) Method using *PhosVer 3 Phosphate Reagent* Powder Pillow in 10 mL water sample. The result was expressed in mg $L^{-1}PO_4^{3-}$.

3.3.2 Macroinvertebrate

3.3.2.1 South African Scoring System version 5 (SASS5)

According to Dicken and Graham, (2002) and Dallas, (2007) there are three principal indices calculated namely, SASS5 Score, Number of Taxa (No. Taxa), and Average Score per Taxa (ASPT). A quality score based on its susceptibility to pollution was allocated for each taxon per sample. The score attributed to benthic macroinvertebrates varies between 1 and 15. High score is attributed to greater sensitive organisms and the low score correspond to tolerant organisms (Dallas, 2000). These values were then added up to calculate the SASS5 score. The total number of taxa found in a sample corresponds to the sum of number of taxa (No Taxa) per sample. While the Average Score per Taxon (ASPT) was calculated by dividing the SASS5 scores by number of taxa for each sample at each site.

The SASS5 and ASPT results were used to evaluate the biotic integrity and ecological state of the sites using the SASS Data Interpretation Guidelines (Dallas, 2007) (Table 3.3). Good water quality usually is associated to high SASS and ASPT scores due to the presence of Ephemeroptera, Plecoptera and Trichoptera which indicate that water is very well oxygenated. These taxonomic groups are sensitive to water pollution, therefore they have high score (Stoyanova *et al.* 2010; Lorion and Kennedy, 2009; Robeston, 2006; Bredenhand, 2005; Hart and Campbell, 1994).

Table 3.3 Ecological categories for the interpretation of SASS data (modeled reference conditions for the Highveld Ecoregion).

SASS Score	ASPT	Class	Description
>124	>5.6	A (Natural)	Unimpaired. High diversity of taxa with numerous sensitive taxa.
83 – 124	4.8 – 5.6	B (Good)	Slightly impaired. High diversity of taxa, but with fewer sensitive taxa.
60 – 82	4.6 – 4.8	C (Fair)	Moderately impaired. Moderate diversity of taxa.
52 – 60	4.2 – 4.6	D (Poor)	Considerably impaired. Mostly tolerant taxa present.
30 – 51	Variable <4.2	E (Seriously modified)	Severely impaired. Only tolerant taxa present.
<30	Variable	F (Critically modified)	Critically impaired. A few tolerant taxa present.

Sources: Golder Associates (2009). Aquatic specialist study for the proposed construction of the Kusile Rail project.

3.3.2.2 Species richness and species diversity

Although used by numerous authors, both species richness and species diversity seem to be confused.

a. Species richness: According to Klemm, *et al.* (1990) species richness reflects the health of the community through measurement of the variety of taxa present. It has been estimated as the total number of taxa (families and higher taxonomic levels) in a given sample.

b. Species diversity: Numerous indices and scores have been used to evaluate variation of species diversity in studies carried out on biological diversity and ecological monitoring (Spellerberg and Fedor, 2003). The most widely used diversity index is that referred to Shannon diversity *(H')* because of its stability in any spatial distribution and its insensitivity to rare species.

A Shannon diversity ***(H')*** index per week was calculated at each site to assess the equilibrium of the community (Spellerberg and Fedor 2003). In natural areas it varies between 0.05 for a low diversity site and 4.5 for a high diversity site. The Formula of Shannon diversity ***(H')*** index is:

$$H = -\sum_{i=1}^{N} pi\,\mathbf{ln}\,pi$$ (Spellerberg and Fedor, 2003) (1)

Where: H= Shannon and Weaver index

$Pi = \frac{ni}{n}$

n = Total individuals of specimens and n¡ Total individuals of 1 species

The mean diversity (đ) was calculated using Klemm *et al.*, (1990) method known as the Shannon-Weaver mean diversity.

$$(đ) = \frac{C}{N}\,\mathbf{(Nlog10\ N - \sum ni\ log10\ ni)} \qquad (2)$$

Where C= 3.321928

N= total number of individuals

ni= total number of individuals in the species i

3.3.2.3 Similarity Indices (Abel, 2002; Klemm *et al.*, 1990).

There are numerous kinds of similarity indices available to calculate the degree of similarity between samples. In the present study, the similarity between sites was calculated using the formula for Sorensens's coefficient.

$$S = \frac{2c}{a+b} \qquad (3)$$

Where a = the number of taxa in community a; b = the number of taxa in community b; and c = the number of taxa common to both.

3.3.2.4 Accumulation curve

The species accumulation curve is a plot of cumulative numbers of recorded species as a function of effort. The accumulation curve of species at each site was established using Gaston, (1996) method.

3.3.2.5 Statistical Analysis

To analyze data from different sources of pollution, one-way analysis of variance (ANOVA) was used to evaluate whether or not there is a difference between groups in physical and chemical parameters, and number of taxa, and SASS and ASPT scores. To find out if there are significant differences between groups, the post-hoc tests, namely Fisher's PLSD, Bonferroni, and Tukey alpha were used to compare the sites. The leve of statistical significance used for all tests was 0.05. There is a significant difference when the value is less than 0.05.

3.4 HISTORICAL WATER QUALITY DATA AQUISITION

Historic data were obtained from DWAF and City of Cape Town in excel files. Two monitoring stations located in the upper river were selected, data were used for September, October and November of each year from 1989 to 2012.

CHAPTER FOUR: RESULTS

The section presents the results of the study. It discusses in detail the physical and chemical parameters of the upper Kuils River in the ensuing sections.

4.1 PHYSICAL AND CHEMICAL PARAMETERS OF THE UPPER THE KUILS RIVER

The Physical and chemical data are given in appendix 1 and statistical analysis in appendix 3. It includes water temperature, pH, dissolved oxygen (DO), oxygen saturation, total dissolved solids (TDS), salinity, phosphate (PO_4^{-3}) and nitrate (NO_3^-).

4.1.1 Temporal variations of physical and chemical parameters of the upper Kuils River

4.1.1.1 pH variations

The mean weekly values of pH ranged from 7.61 to 8.22 on the 13th November 2012 and 9th October 2012 respectively (**Figure 4.1**). The slight decrease in pH observed in November 2012 may be attributed to seasonal variation.

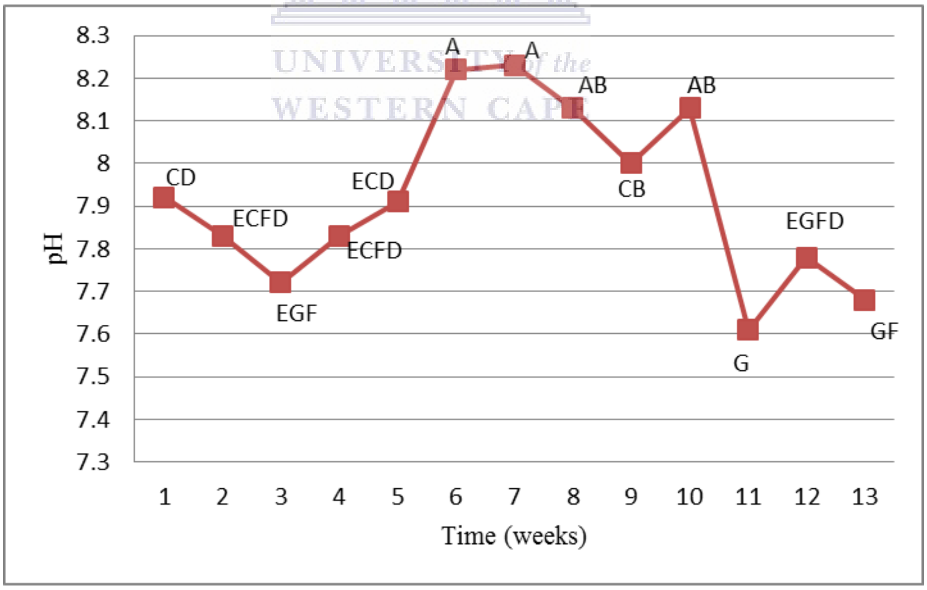

Figure 4.1: Temporal variation in pH from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.2 Temperature variations

The water temperature averages varied from one week to another (**Figure 4.2**). The highest mean (23.96°C) water temperature was measured on the 6th November 2012 while the lowest mean (14.84°C) was obtained on the 14th September 2012. November, early summer, is a dry and hot period in the study area. September (late winter/early spring) 2012 characterized by low water temperature may be due to wet period.

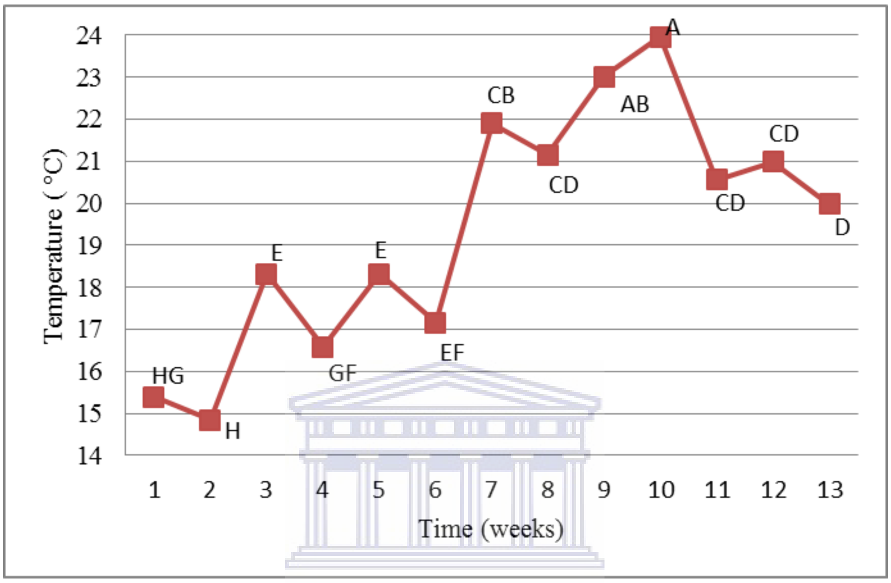

Figure 4.2: Temporal variation in water temperature from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.3 Total Dissolved Solids variations

Total dissolved solids (TDS) mean recorded during the whole study period fluctuated between 620 mg L^{-1} on the 18th September 2012 and 834 mg L^{-1} on 4th September 2012 (**Figure 4.3**). The major reason for the high TDS in the Kuils River is attributable to the geological characteristics of soil over which the river flows. Perhaps, runoff and storm water from urban areas may also contribute to increased salts in the water body.

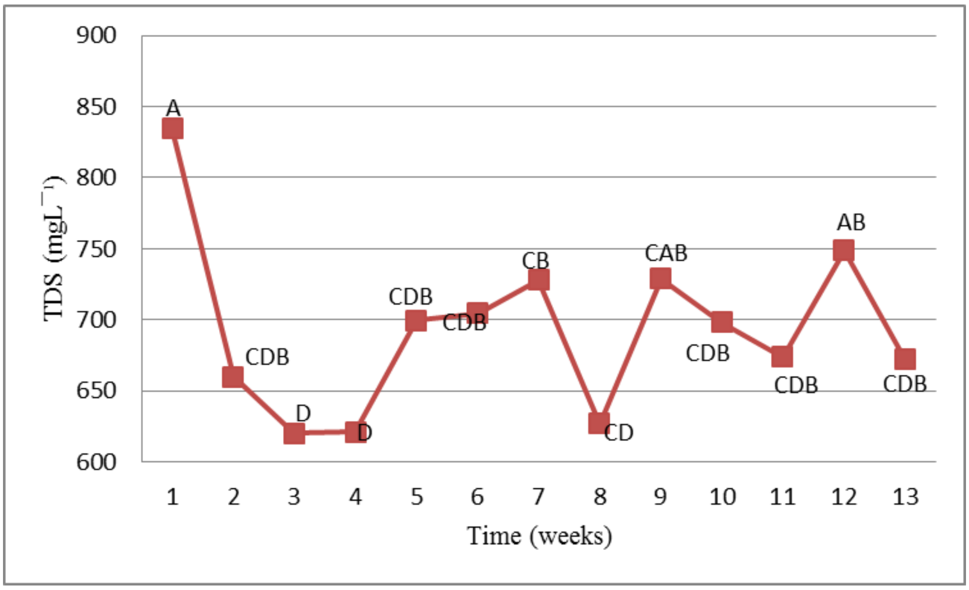

Figure 4.3: Temporal variation in TDS from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.4 Salinity variations

The mean salinity shows similar patterns of distribution in TDS concentrations. The highest mean of salinity was recorded on the 4th September 2012 while the lowest mean of 0.474 mg L^{-1} was obtained on the 18th September 2012 (**Figure 4.4**).

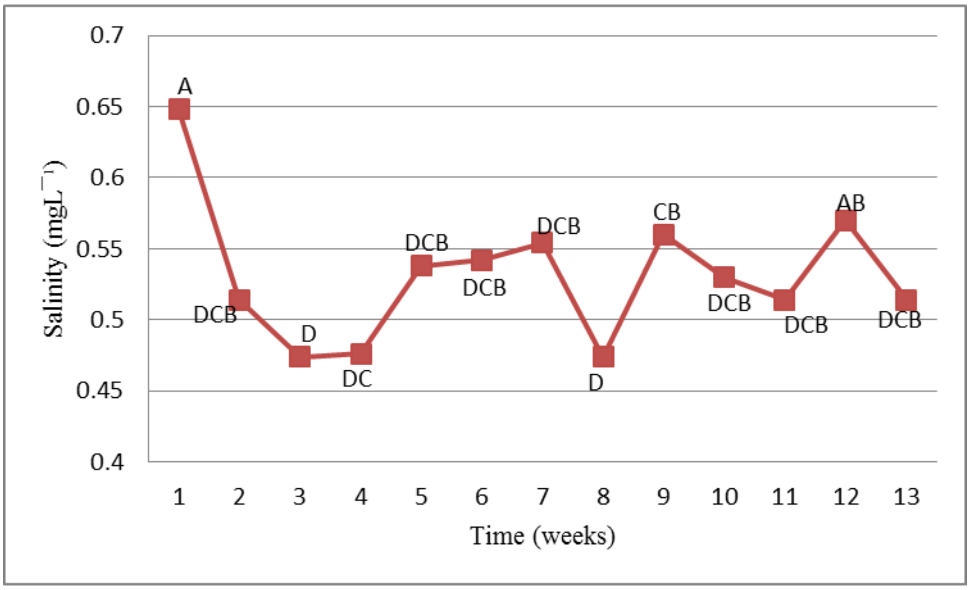

Figure 4.4: The temporal variation in salinity from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.5 Dissolved oxygen variations

Dissolved oxygen in aquatic ecosystems constitutes an important factor that determines the quality of water and support an important number of aquatic organisms. The mean weekly concentration of dissolved oxygen of the upper Kuils River decreased from September to November 2012 (**Figure 4.5).** The highest mean (12.472 mg L^{-1}) dissolved oxygen was recorded on the 4th September. This figure may be attributed to the rain season in the catchment area. The lowest mean (5.404 mg L^{-1}) concentration was evaluated on the 27th November 2012 probably due to dry and hot period.

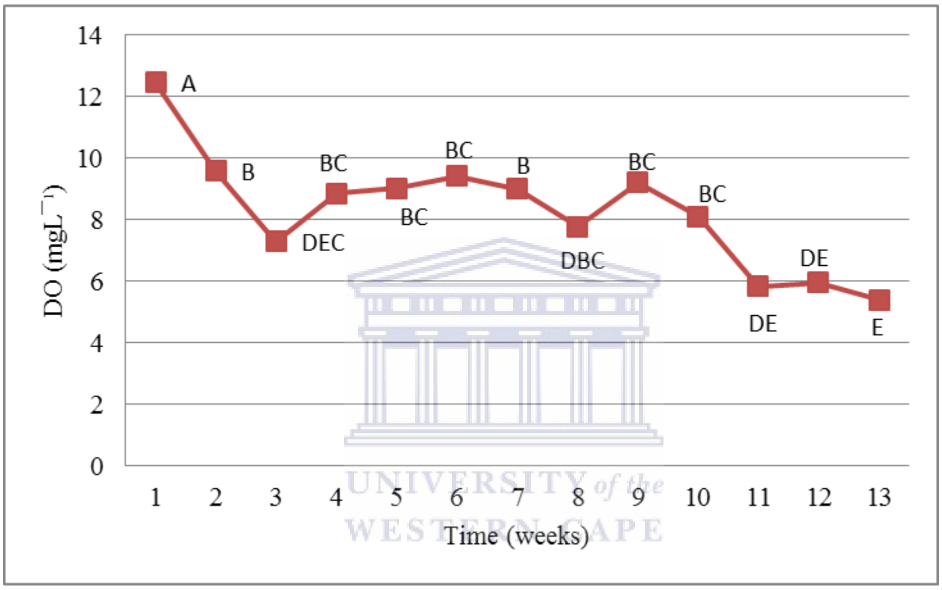

Figure 4.5: Temporal variation in dissolved oxygen from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.6 Oxygen saturation

The mean weekly variations of oxygen saturation present a similar pattern of distribution as dissolved oxygen. The highest mean (124.92 %) was recorded on September 4th while the lowest mean (65.41 %) was recorded on the 13th November 2012 (**Figure 4.6**).

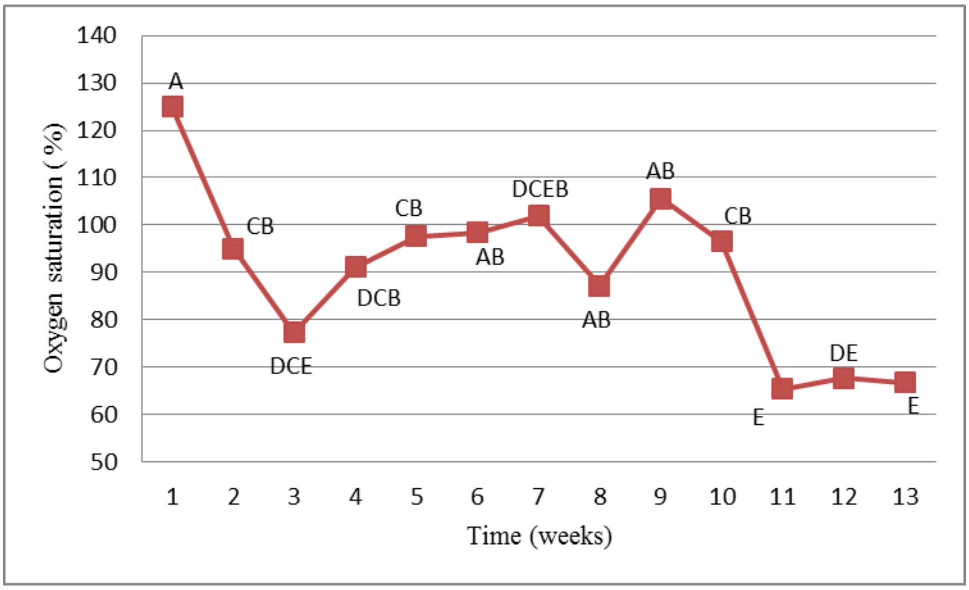

Figure 4.6: The temporal variation in oxygen saturation from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.7 Phosphate concentration variations

The means of phosphate concentrations assessed on the upper Kuils River are shown in **figure 4.7.** The temporal variations increased between September and November 2012. The mean weekly distribution of phosphate concentrations varied from (0.514mg L^{-1}) on the 18th September to (2.145 mg L^{-1}) on the 20th November 2012. These values exceed the recommended limits by ecosystem health criteria in South Africa. A slight increase in phosphate concentrations observed in November 2012 coincides with algae proliferation.

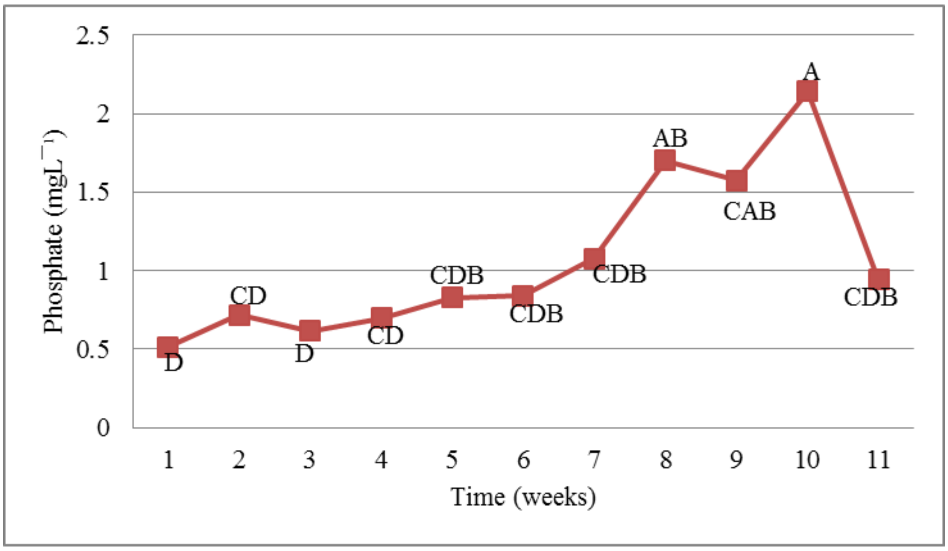

Figure 4.7: The temporal variation in phosphate concentrations from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.1.8 Nitrate concentration

The mean fluctuations in concentrations ranged between (0.680 mg L^{-1}) on the 27th November 2012 and (1.340 mg L^{-1}) on the 2nd October 2012 (Figure 4.8). A general observation shows that the mean values of nitrate concentrations were above 0.5 mgL^{-1} during the all sampling periods. These concentraions were higher than the recommended criteria set by the South African water quality guidelines for aquatic ecosystem.

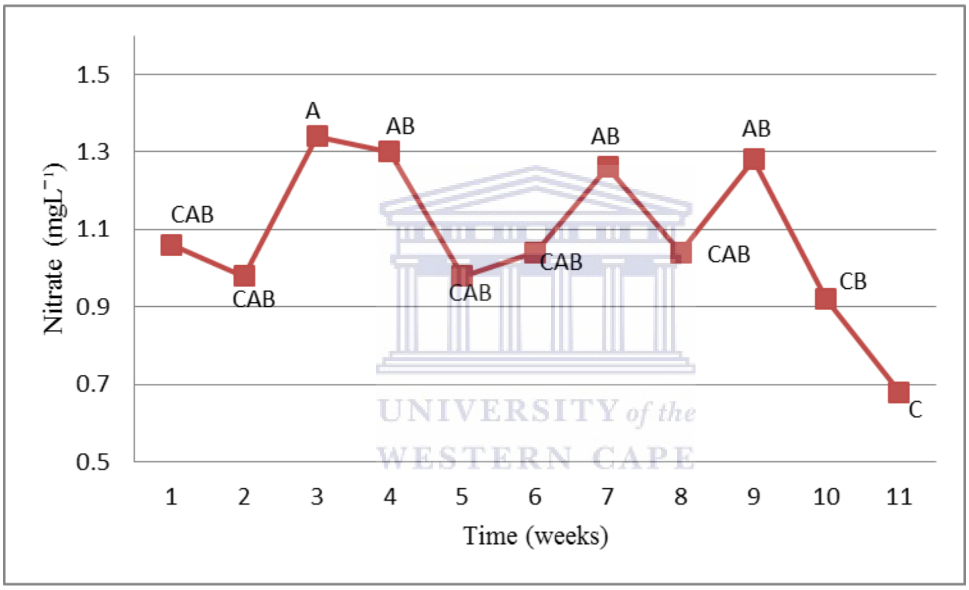

Figure 4.8: The temporal variation in nitrate concentrations from September to November 2012. Means with the same letter are not significantly different ($p \leq 0.05$).

4. 1.2 Spatial variations of physical and chemical parameters of the upper Kuils River

4.1.2.1 pH

The average of pH calculated at all sampling sites decreases from upstream sites to downstream sites (Figure 4.9). The highest mean pH value was observed at K5 (8.30) while the lowest mean value was recorded at K1 (7.68). Low pH values evaluated at downstream sites may be due to abundance of algal, surface runoff and storm water from industrial area, and organic matter from road-bridges. In the tributary site the mean pH value (7.60) was lower compared to most of the main stream sites. At site K4, storm water from residence area, macrophytes (photosynthesis and respiration), and organic matter from road-bridge and dead plant litters may also hav had an influence on pH levels.

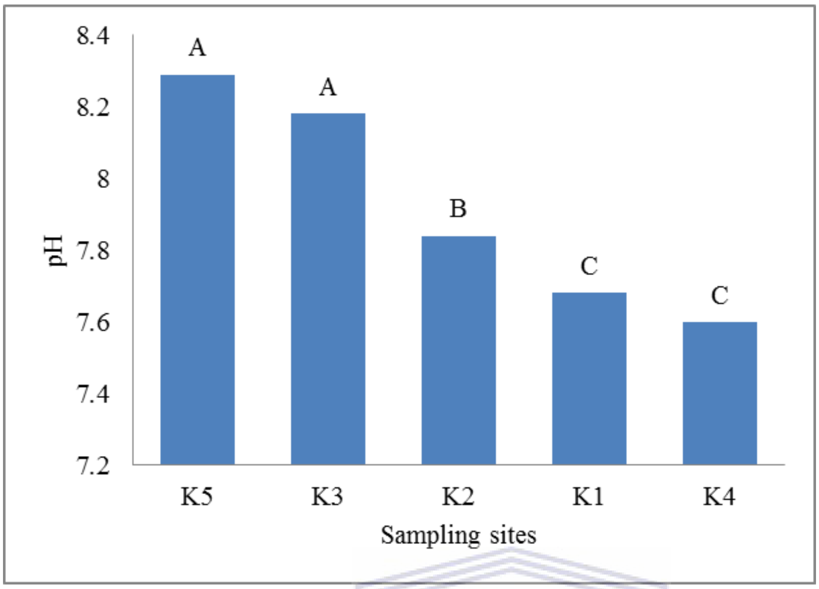

Figure 4.9: pH variations at different sampling sites. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.2.2 Average of water temperature between sampling sites

The mean of water temperature measured at each sampling site during the whole period decreased from upstream sites to downstream sites (Figure 4.10). The mean water temperature was highest at K5 (21.81°C) while the lowest was recorded at K1 (17.33°C). This difference may be due to daily temperature variations. At downstream sites (K1 and K2) water temperature was measured in the morning while at upstream sites (K3, K4 and K5) the temperature was collected during mid-day. Overall, shallow water, riparian clearing, and loss of vegetation cover that characterize the Kuils River may have influenced the variations in water temperature.

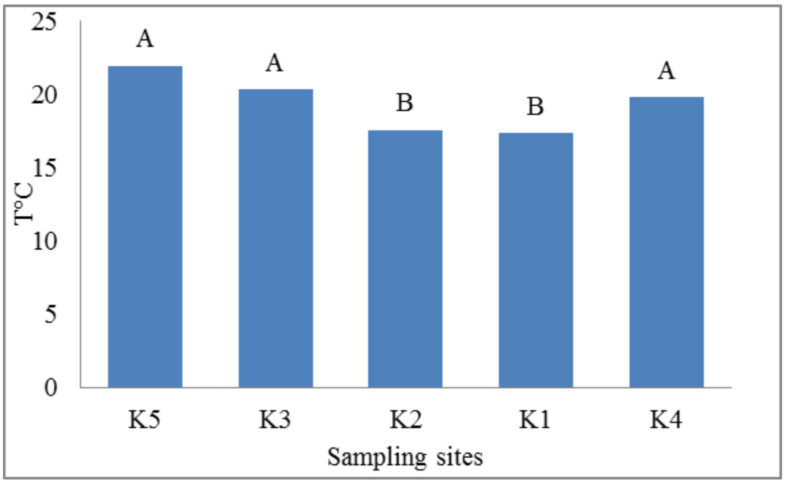

Figure 4.10: Water temperature variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.3 Average of TDS between sampling sites

Mean values of TDS calculated for the upper Kuils River decreased from upstream sites to downstream sites (Figure 4.11). The highest mean TDS concentration was obtained at site K5 (747.6 mg L^{-1}) while the lowest mean was recorded at site K1 (611.4 mg L^{-1}). The mean value of TDS concentration measured at site K4 was (727.34 mg L^{-1}). The lowest TDS concentrations recorded at downstream sites may be due to water dilution effect.

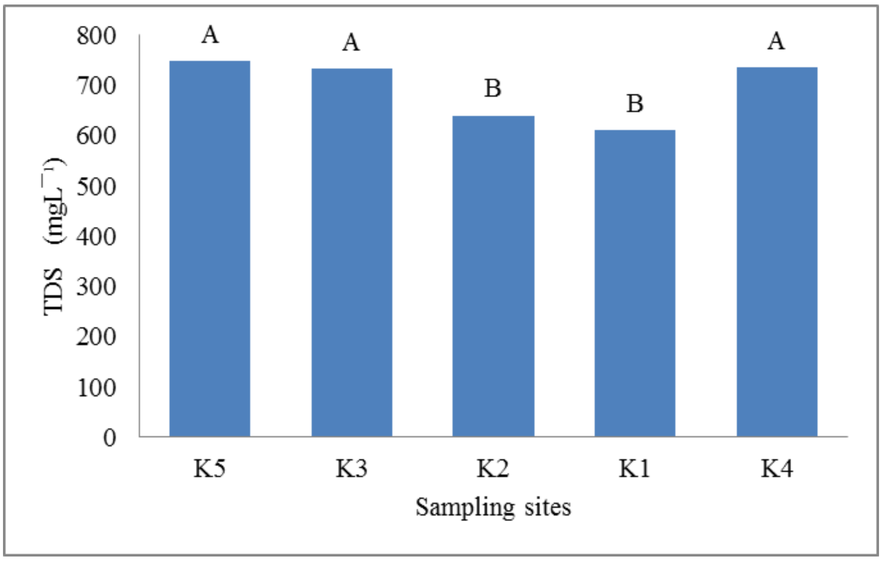

Figure 4.11: TDS variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.4 Average of Salinity between sampling sites

Salinity is defined as total dissolved solids or as conductivity. In water samples, salinity is related to TDS concentration or electrical conductivity. Therefore, it shows the similar pattern of distribution as TDS concentrations. The salinity averages evaluated at each sampling station in the main stem during the whole sampling period decreased from upstream sites to downstream sites (Figure 4.12). The highest mean salinity was recorded at K5 (0.573 mg L^{-1}) while the lowest concentration was evaluated at K1 (0.472 mg L^{-1}).

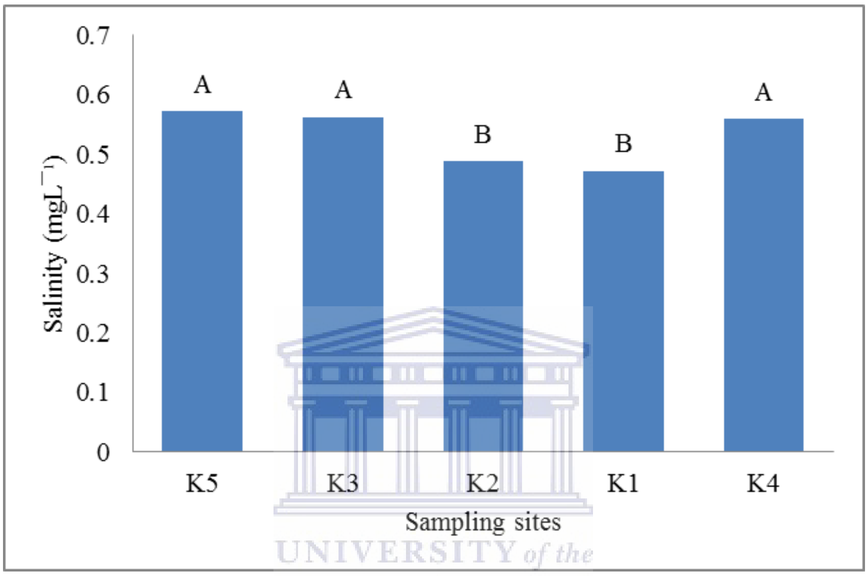

Figure 4.13: Salinity variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.5 Dissolved oxygen variation between sampling sites

In the main stem, dissolved oxygen concentrations ranged from 8.85 mgL^{-1} at K1 to 9.48 mgL^{-1} at K5. The mean weakest value (4.52 mgL^{-1}) was observed at site K4 a tributary of the Kuils River (Figure 4.14). Apart from pollutants from residential area, the site K4 is very shallow, slow and stagnant in early summer, and overrun by aquatic and decaying plants. All these factors may lead to low DO concentration in this site.

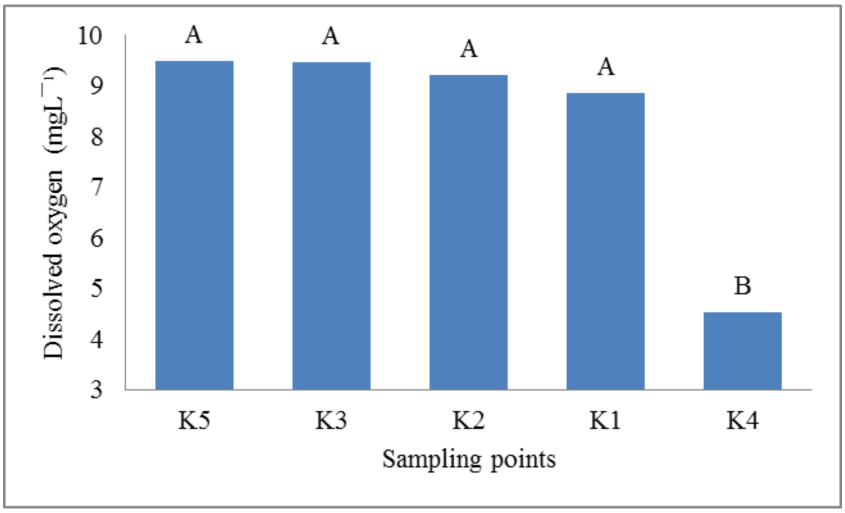

Figure 4.14: Dissolved oxygen variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.6 Oxygen saturation between sampling sites

Mean percentages of oxygen saturation show a slight decrease from upstream sites to downstream sites (Figure 4.15). The mean variations of oxygen saturation at different sites show a similar pattern of distribution as dissolved oxygen.

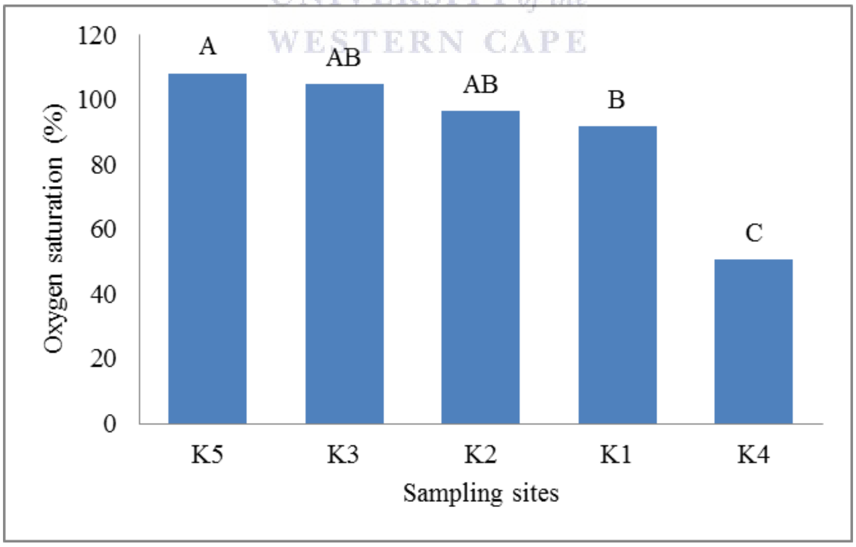

Figure 4.15: Oxygen saturation variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.7 Nitrate variation at different sampling sites

In the main stem, the trend of the curve shows that mean values of nitrate concentrations increased from upstream to downstream (Figure 4.16). The highest mean value was obtained at K1 (1.71 mgL^{-1}) while the lowest mean concentration was recorded at K5 (0.74 mgL^{-1}). The sites K1 and K2 both receive pollutants from hospital, residential and industrial areas, and organic matter from the road-bridge. Upstream sites receive pollutants from residential area, storm water, and soil bank erosion. In comparison with other sites, K4 had the lowest nitrate concentration (0.09 mgL^{-1}) which may be due to rapid growth of aquatic plants, and shallow and slow water which often dries in summer.

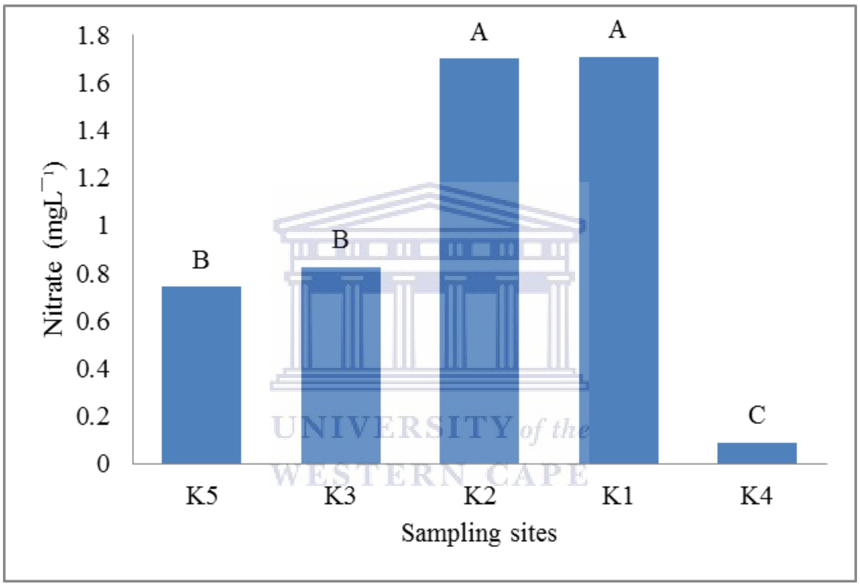

Figure 4.16: Nitrate variations at different sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.1.3.8 The average of phosphate at different sampling points

In the main river, the highest mean value (1.750 mg L^{-1}) of phosphate concentrations was evaluated at site K1 while the lowest mean (0.350 mg L^{-1}) concentration was obtained at K5. In addition to residential and industrial areas, the site K1 receives pollutants from hospital area, susceptible to increased phosphate concentrations. At the tributary site (K4) the mean value (2.166 mg L^{-1}) of phosphate concentration was high compared to main stream sites. Because there is a significant difference between K3 and K4, one might reasonably suggest that high concentration observed at site K4 did not affect the river in terms of phosphate. The K4 site characterized by shallow and slow flow of water includes organic matter from death plants, and storm water from residential area which may probably be sources of phosphate.

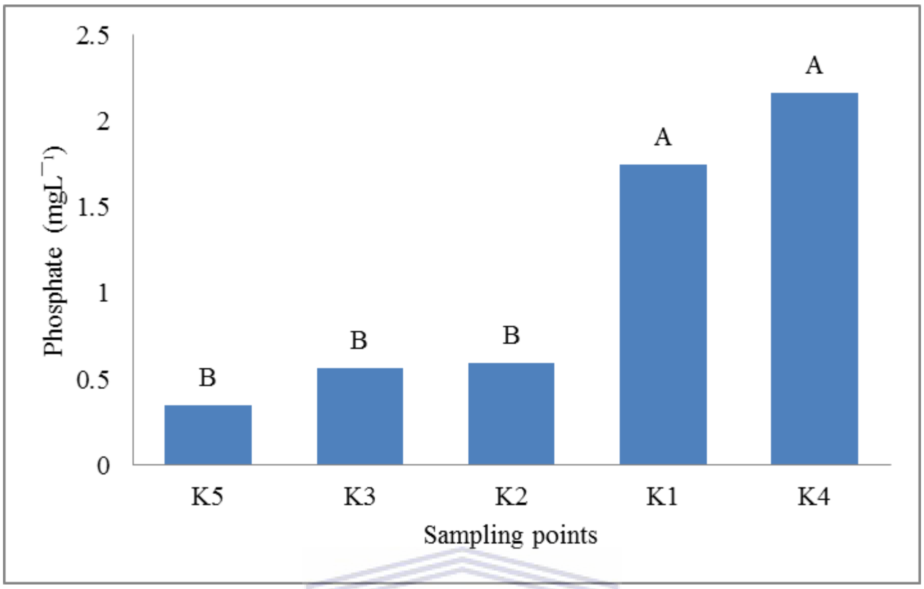

Figure 4.17: Phosphate variations at sampling sites of the upper Kuils River. Means with the same letter are not significantly different ($p \leq 0.05$).

4.2 BIOLOGICAL PARAMETERS

4.2.1 DISTRIBUTION OF BENTHIC MACROINVERTEBRATES

Benthic macroinvertebrates abundance, specific richness and diversity collected per week at different sites are presented in appendix 2. The statistical analysis is showed in appendix 3.

4.2.1.1 List of macroinvertebrates between sites

Benthic macroinvertebrates (BMI) collected from the upper Kuils River are listed in **table 4.1**. A total of 8409 specimens belonging to 28 taxa (families), and 11 orders were sampled from 18th September (late winter/early spring) to 27th November (early summer) 2012. The weekly numbers varied from one taxon to another and from one site to another. Some taxa were present at all sampling points during the whole sampling period, while others were absent or partially recorded in some sites during the whole sampling period. The Physidae showed the highest records of (00%) at all sampling sites. The Oligochaeta had (00%) of weekly frequency at all sampling sites except K4 that had (72.7%). Other taxa such as

Chlorocyphidae, Aeschnidae, Syrphidae, occurred in some sampling sites (1 to 4 sites) with a frequency ranging between (90.09 %) and (9.09 %). The most abundant taxa are Physidae with (32.33 %), followed by Simuliidae (21.807 %), and Chironomidae (18.59 %) whereas Chlorocyphidae, Belastomatidae, Gerridae, Naucoridae, Veliidae, Psychodidae, and Planorbidae each had the lowest proportion of specimens of (0.01%).

Table 4.1 List of benthic macroinvertebrates collected at different sites upper stream of the river

Systematic Group		Score	Downstream sites				Upstream sites						Tot	Total (%)
			K1		K2		K3		K4		K5			
ORDER	Taxa		Eff	Fr	Eff	Fr	Eff	Fr	Eff	Fr	Eff	Fr		
ACARINA	Hydrachnellae	8	0	0	0	0	1	9.09	0	0	1	9.09	2	0.02
OLIGOCHAETA	Oligochaeta	1	170	100	210	100	240	100	32	72.7	135	100	787	9.36
HIRUDINA	Hirudina	3	190	100	34	63	68	90.9	3	18.18	73	81.8	368	4.38
CRUSTACEA														
	Potamonautidae	3	7	55.54	24	81,8	43	100	2	18.18	33	90.9	109	1.31
ODONATA														
	Coenagrionidae	4	3	18	0	0	24	90.9	0	0	13	63.6	39	0.47
	Aeschnidae	8	0	0	0	0	4	18.18	0	0	1	9.09	5	0.06
	Chlorocyphidae	10	0	0	0	0	1	9.09	0	0	0	0	1	0.01
	Libellulidae	4	1	9,1	2	18.18	15	54.5	1	9.09	6	36.3	25	0.30
HEMIPTERA														
	Belostomatidae	3	0	0	0	0	0	0	0	0	1	9.09	1	0.01
	Corixidae	3	1	9,1	0	0	11	54.5	141	81.8	5	27.2	158	1.88
	Gerridae	5	0	0	0	0	1	9.09	0	0	0	0	1	0.01
	Naucoridae	7	0	0	0	0	1	9.09	0	0	0	0	1	0.01
	Notonectidae	3	0	0	1	9.09	2	18.18	0	0	2	18.18	5	0.06
	Veliidae	5	0	0	0	0	1	9.09	0	0	0	0	1	0.01
EPHEMEROPTERA														
	Baetidae	4	1	9.09	7	27.2	221	72.7	3	27.27	336	63.6	568	6.75
COLEOPTERA														
	Dytiscidae	5	1	9.09	2	18,1	22	72.7	21	81.81	51	72.7	97	1.15
	Hydrophillidae	5	0	0	0	0	9	45.4	1	9.09	3	27.2	13	0.15
DIPTERA														
	Ceratopogonidae	5	0	0	7	18.18	14	54.5	0	0	33	54.5	54	0.64
	Chironomidae	2	53	54.5	82	72,72	456	100	594	100	379	100	1564	18.60
	Ephydridae	3	0	0	1	9.09	7	18.18	1	9.09	6	36.6	15	0.18
	Simuliidae	5	0	0	4	27.2	626	100	481	72.7	723	100	1834	21.81
	Psychodidae	1	0	0	0	0	0	0	1	9.09	0	0	1	0.01
	Syrphidae	1	0	0	1	0	1	9.09	0	0	2	18.18	4	0.05
	Tipulidae	5	0	0	0	0	0	0	6	54.5	4	27.2	10	0.12
GASTROPODA														
	Physidae	3	1028	100	1278	100	80	100	91	100	242	100	2719	32.33
	Limnaeidae	3	1	9.09	0	0	4	18.18	1	9.09	18	63.6	24	0.29
	Planorbidae	3	0	0	0	0	0	0	1	9.09	0	0	1	0.01
TURBELLARIA														
	Turbellaria	3	0	0	2	18.18	0	0	0	0	0	0	2	0.02
			1455		1655		1852		**1380**		2067		8409	100.00

4.2.1.2 Number of taxa per systematic group

A total of 28 taxa were collected during the study period (**Figure 4.18**). The number varies from one group to another. The highest number of taxa was in the order Diptera (7 families), followed by Hemiptera (6 families), and Odonata (4 families). Orders Gasteropoda and Coleoptera were represented by 3 and 2 families respectively. The other groups (Ephemeroptera and Decapoda) had only 1 taxon each. For Oligochaeta, Hirudina, Turbellaria, and Hydracarina only their presence was noted.

Figure 4.18: Number of taxa per systematic group

4.2.1.3 Distribution of families and orders at different sites

The number of taxa varies between sites (**Figure 4.19)**. The highest numbers were collected at upstream sites (K5, K3). The highest number was collected at K3 (10 orders and 23 families) while the lowest number was collected at downstream sites. Upstream sites consisted of vegetable debris, marginal plants, gravel, stones and solid materials. These substrates are favorable settlement to benthic macroinvertebrates. At site K4 the number of taxa recorded was 9 Orders and 16 families. The decrease in macroinvertebrate at sites K2 and K1 may be due to sand and filamentous algae which dominate those sites.

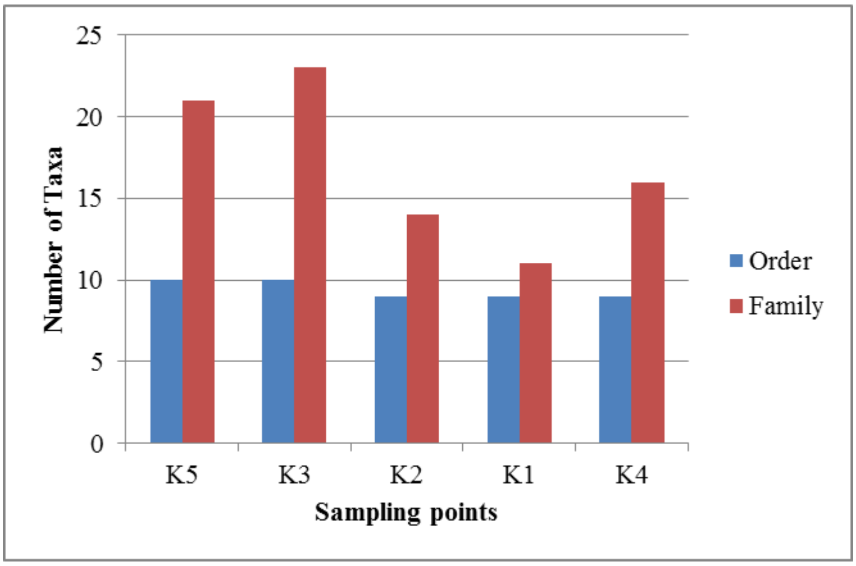

Figure 4.19 : The number of families and order per site over the sampling period

4.2.1.4 Abundance of macroinvertebrate specimens per order on the whole sampling sites

Macroinvertebrate abundance varies between groups (Figure 4.20).The most abundant macroinvertebrates identified were the order Diptera (41.4 %), followed by Gastropoda (32.62 %), Oligochaeta (9.35 %), Ephemeroptera (6.75 %), and Hirudina (4.37 %). Hydracaena (Acarina) and Turbellaria were less than 0.1 %. For each order, the abundance is linked to abundance of 1 or 2 families. Diptera are dominated by Simuliidae (1834 specimens) and Chironomidae (1564 specimens). Gasteropoda were dominated by Physidae (2719 specimens), and Ephemeroptera were dominated by Baetidae (568 specimens)

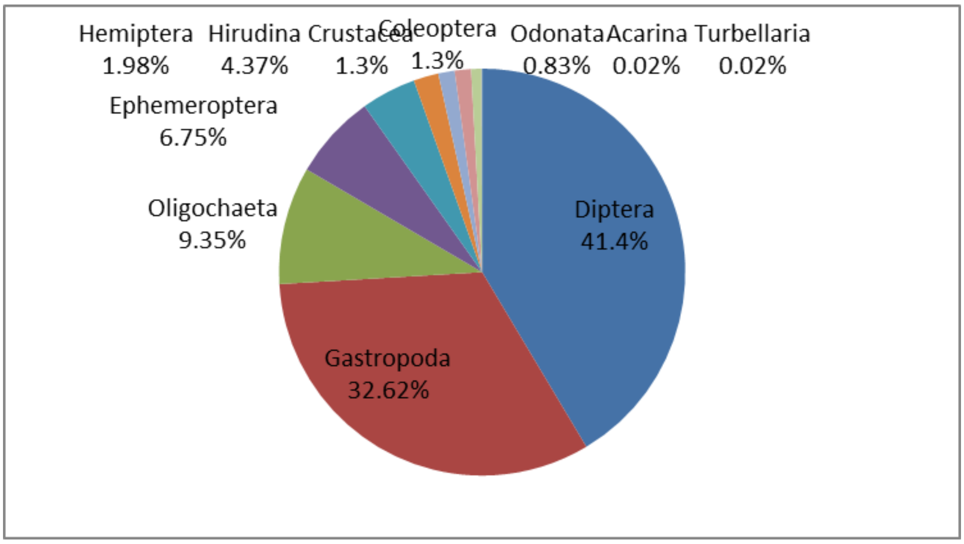

Figure 4.20 : Abundance of Macroinertebrate specimen per order of the Upper Kuils River

4.2.1.5 Spatial distribution of macroinvertebrate abundance between sites

The abundance of macroinvertebrates in the main stream ranged between 2067 specimens at site K5 and 1455 specimens at site K1. The site K4, a tributary of Kuils River showed the lowest (1380 specimens) abundance (Figure 4.21).

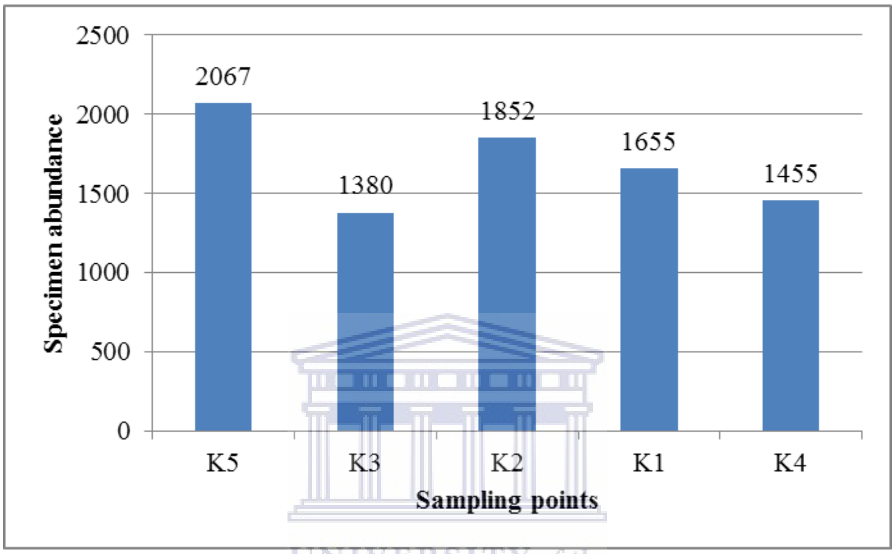

Figure 4.21: Spatial distribution of macroinvertebrate abundance between sites

4.2.1.6 Spatial distribution of macroinvertebrate abundance within systematic groups

The benthic macroinvertebrates varied between sites (**Figure 4.22**). Upstream sites (K3, K4, K5) were dominated by Diptera (1147 specimens at site K5; 1104 specimens at K3; 1083 specimens at site K4) and Ephemeroptera (336 specimens at site K5; 221 specimens at site K3), while downstream the study area was dominated by Gastropoda (1278 at site K2; 1029 at site K1), and Hirudina (190 specimens at site K1). Oligochaeta distributions were similar at different sites. Although less abundant, the order Odonata (70 specimens) and Coleoptera (110 specimens) were found at all sampling sites. Crustacae showed higher abundance at site K3 (43 individuals), followed by K5 (33 individuals), and K2 (24 specimens), whereas lower abundance was found at site K1 (7 specimens) and K4 (2 specimens). Overall, these taxa were recorded at all sampling sites; they are tolerant to water pollution. The order and abundance of macroinvertebrate at different sites were influenced by the nature and diversity of substrates.

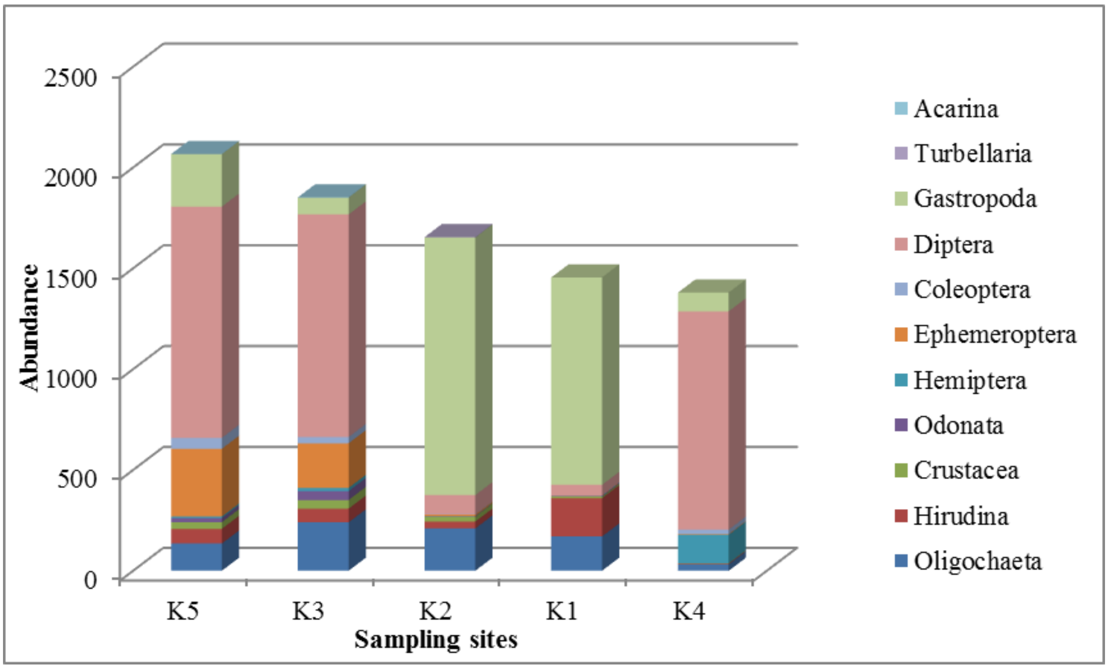

Figure 4.22: Spatial distribution of macroinvertebrate within systematic groups over the sampling period

4.2.1.7 Temporal distribution of systematic group abundances in the whole sampling sites

The weekly variations of abundances per systematic group in the whole sampling sites are in figures **4.23** and **4.24.** The abundances of Diptera and Gatropoda were very high during the sampling period.

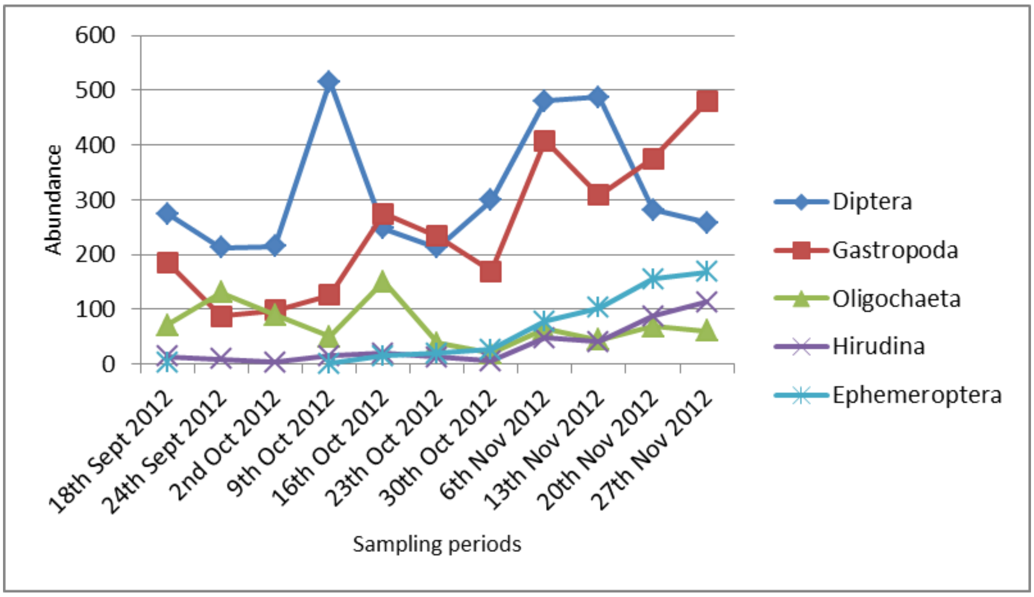

Figure 4.23: Temporal variation of systematic group abundances of the upper Kuils River

The least recorded orders were Hemiptera (167 individuals), Coleoptera (110 individuals), Crustacea (108 individuals) and Odonata (71 individuals).

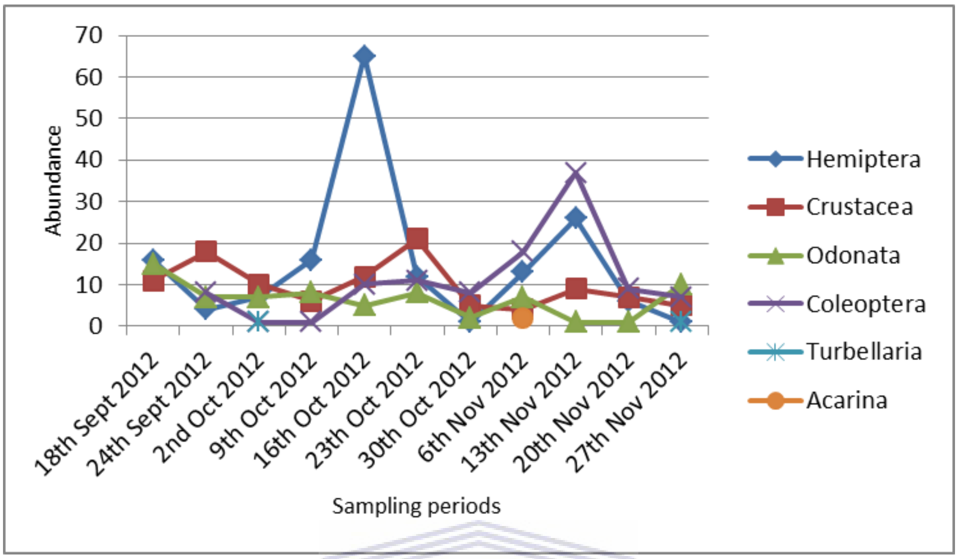

Figure 4.24: Temporal variation of systematic group abundances of the upper Kuils River

4.2.2 Taxa accumulation curves per site

4.2.2.1 Taxa accumulation curve at site K1

Taxa accumulation curve from 18th September to 27th November 2012 at site K1 is shown in **figure 4.25**. The accumulated number of taxa was constant with 5 taxa from 18th September to 2nd October 2012 before increasing with 2 taxa on the 9th October 2012. Another increase was observed on October 23th after which no new taxon was added. It is not likely that additional sampling could have added new taxa.

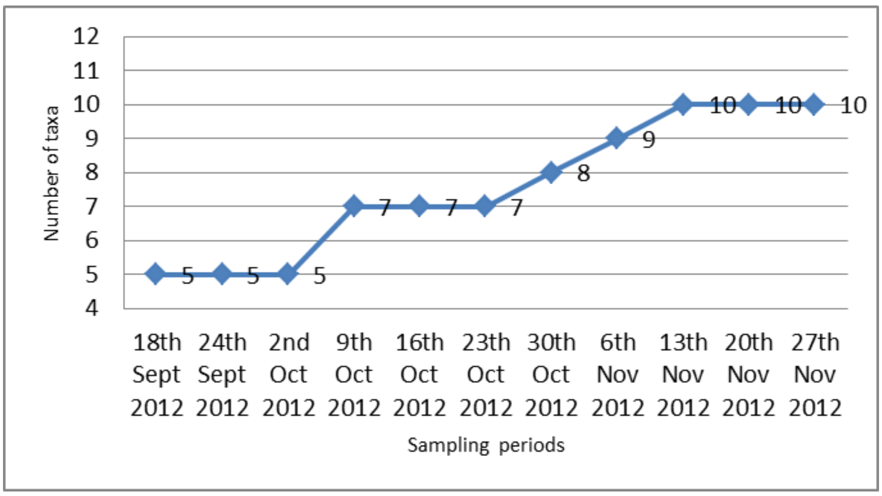

Figure 4.25: Taxa accumulation curve at site K1 from September to November 2012

4.2.2.2 Taxa accumulation curve at site K2

The cumulative number of macroinvertebrate taxa (families) sampled at site K2 illustrated in **figure 4.26** shows an increases since the beginning, and does not stabilize even at the end meaning that additional sampling could have added new taxa.

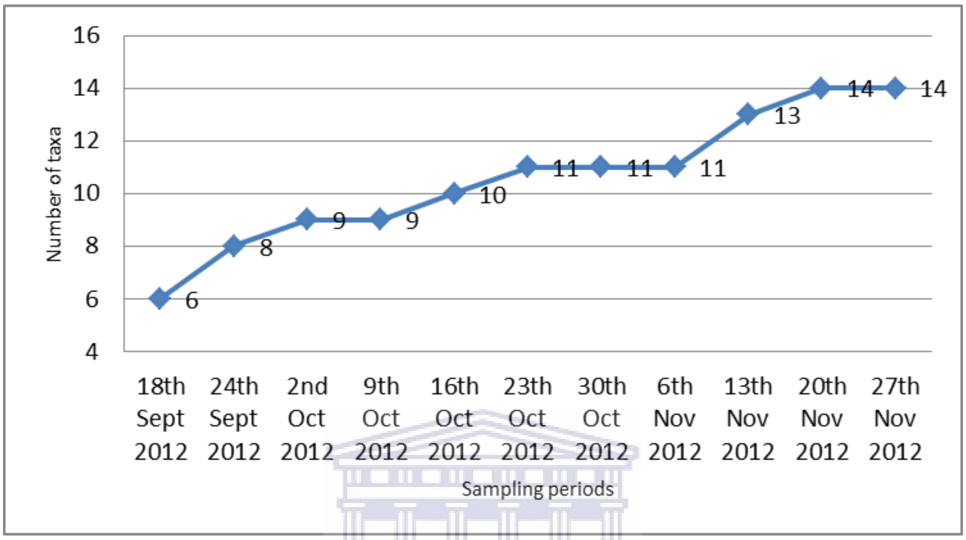

Figure 4.26:Taxa accumulation curve at site K2 from September to November 2012

4.2.2.3 Taxa accumulation curve at site K3

The ascending trend of the curve indicates that new taxa were added at all times. The accumulation curve stabilizes from 9th October to 23rd October 2012 before rising with 1 taxa on the 30th October 2012. The plateau seems not to be reached; meaning that additional samples might add new taxa (**Figure 4.27**).

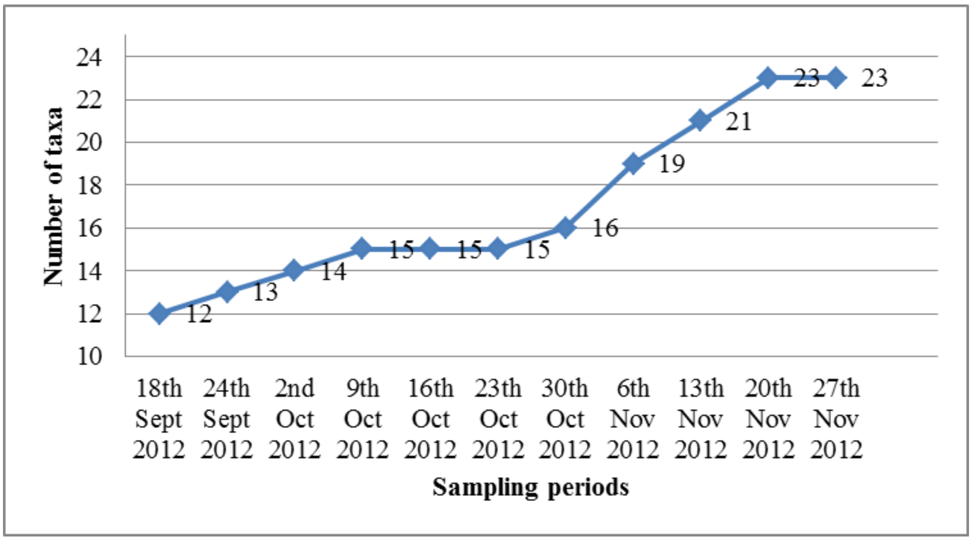

Figure 4.27: Taxa accumulation curve at site K3 from September to November 2012

4.2.2.4 Taxa accumulation curve at site K4

In site K4, the accumulation increases from the beginning and stabilizes at the end, reaching the plateau. It is possible that new samples might not add any additional taxon (**Figure 4.28**).

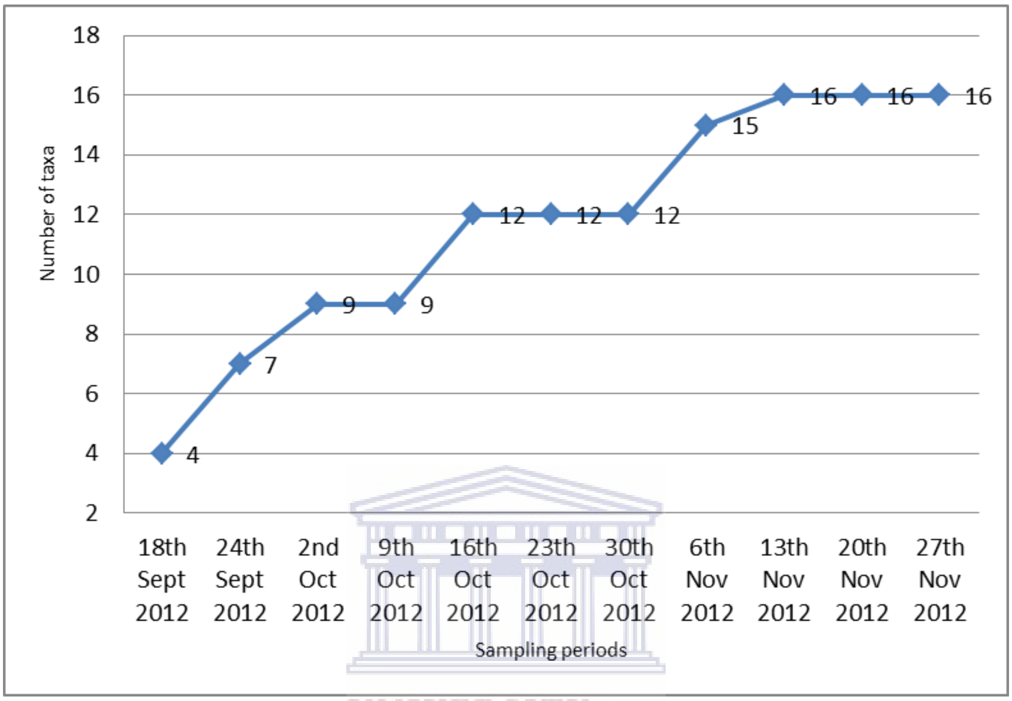

Figure 4.28: Taxa accumulation curve at site K4 from 18th September to 27th November 2012

4.2.2.5 Species accumulation curve at site K5

The accumulation curve in K5 shows an increase from the beginning and does not reach the plateau. New samples might add new taxa to the list (**Figure 4.29**).

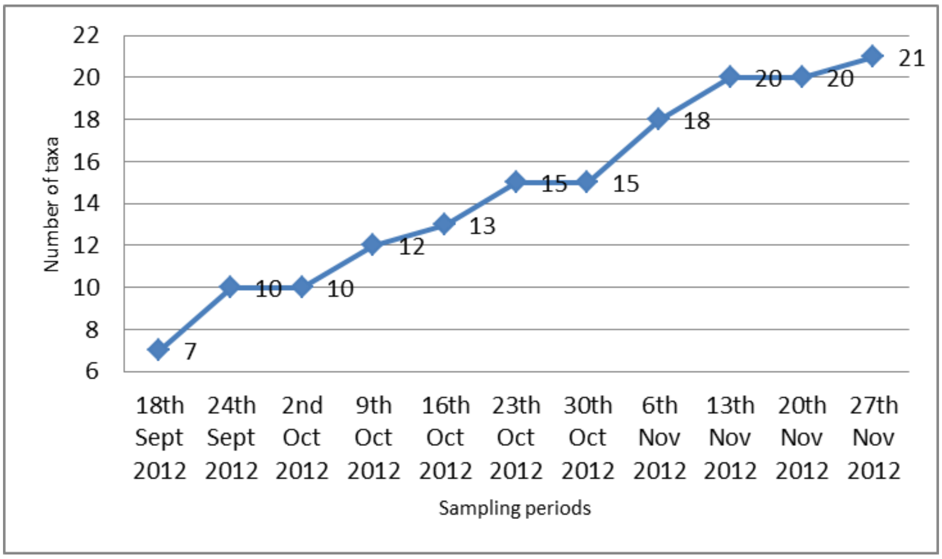

Figure 4.29: Taxa accumulation curve at site K5 from 18th September to 27th November 2012

4.2.4.6 Accumulation curve of the whole of site upstream of the river

The weekly accumulation curve of the sites shows an overall increase. The trend for all the sites combined is increasing but the plateau is not reached. This kind of curve suggests that additional sampling could have added new taxa to the list (**Figure 4.30**).

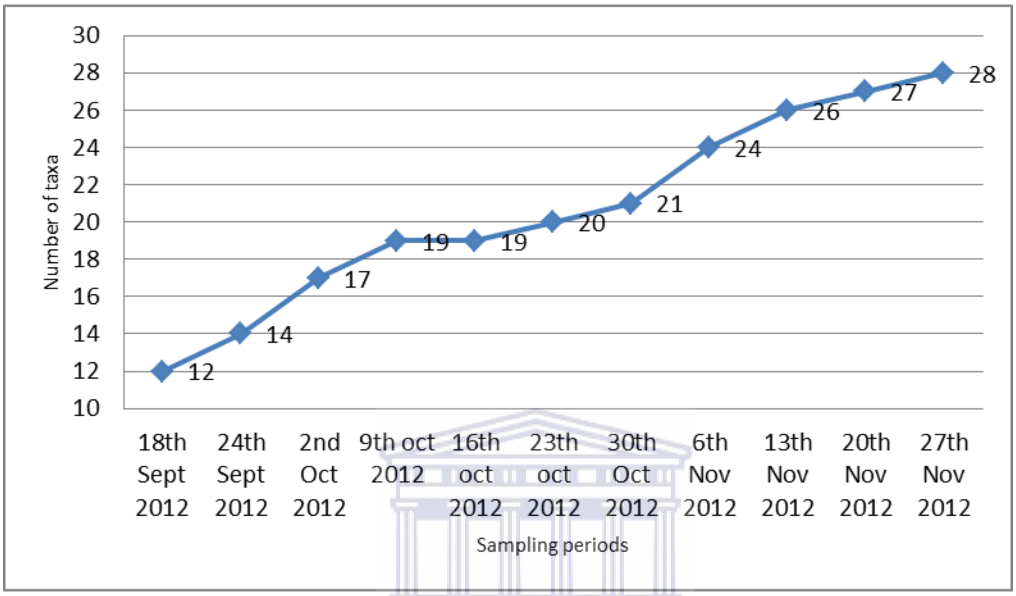

Figure 4.30: Taxa accumulation curve of the whole sampling site upstream of the river

4.2.3 Similarity between sites

Table **4.2** shows a strong similarity in macroinvertebrate community between sites. The highest similarity index was recorded between upstream sites (K5 and K3: 86.3%) while the lowest index was recorded between downstream sites (K2 and K1= 64.0 %).

Table 4.2 Macroinvertebrate similarity between sites upstream of the river

Sites	K5	K4	K3	K2	K1
K5		75.6	86.3	74.2	66.6
K4			66.6	66.6	74.0
K3				70.2	64.7
K2					64.0

4.2.4 Shannon diversity index upstream of the Kuils River

4.2.4.1 Weekly variations of Shannon diversity during the whole sampling period

In general Shannon-diversity index for the upper Kuils River was found to be weak. This weak index may probably be due to poor substrate habitat and water quality. The index diversity varied from 0.916 to 1.402 on the 30th October to 24th September 2012 respectively (**Figure 4.31**).

Figure 4.31: Temporal variation of Shannon diversity index of the upper Kuils River

4.2.4.3 Shannon-Weaver mean of diversity (đ)

The Shannon-Weaver mean of diversity index recorded at each sampling site during the whole sampling period varied between the sites (**Figure 4.32**). There is a decrease from upstream sites to downstream sites. The highest mean (2.759) of diversity was recorded at site K5 while lowest mean (1.226) value was recorded at K2 (1.226). The high Shannon's index at upstream sites may be attributed to the diversity of substratum. The site K3 and K5 consist of plant debris, marginal plants and gravel while downstream sites were dominated by sand and many filamentous algae.

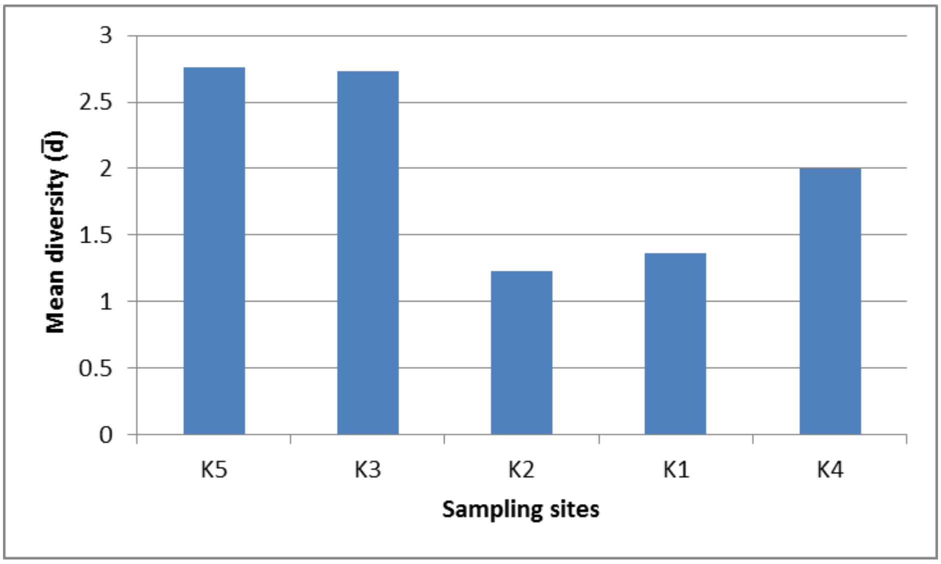

Figure 4.32: Shannon-Weaver mean of diversity index ((đ) at different sites

4.2.5 South African Scoring System (SASS5), number of taxa (NoT) and Average score per taxa (ASPT)

4.2.5.1 Temporal variation of South African Scoring System 5 (SASS5)

The mean weekly SASS varied from 20.80 on 2nd October to 34.4 on 6th November 2012 (Figure 4.33). The SASS5 values were less than the limit recommended by the South African Guidelines. The weak values obtained upstream of the river may be due to low scores attributed to tolerant taxa and poor diversity of benthic macroinvertebrate (BMI) fauna.

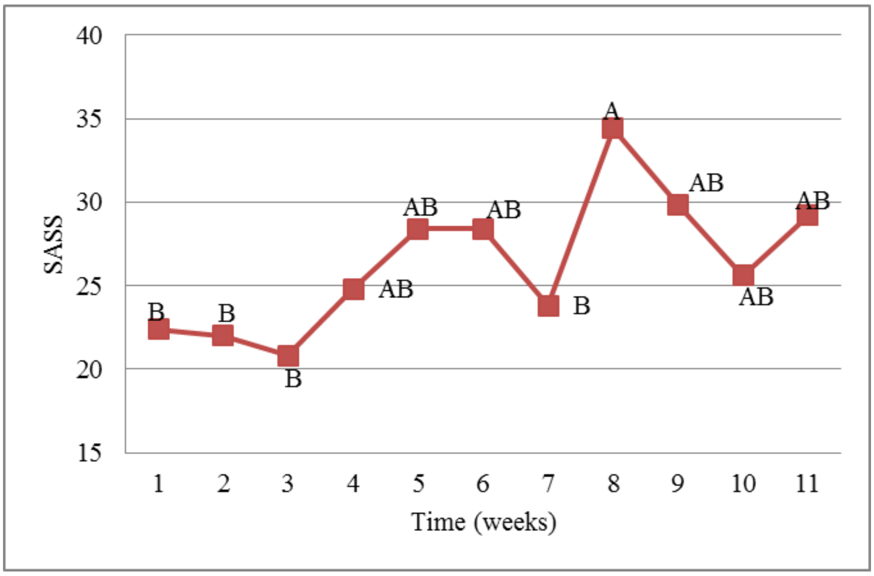

Figure 4.33: The Temporal variations of SASS5 of the upper Kuils River. Means with the same letter are not significantly different ($p \leq .05$).

4.2.5.1.2 Number of Taxa

The number of taxa shows a similar pattern of distribution as SASS. The mean weekly number of taxa ranged from (7) collected on the 18th and 24th September 2012 to (10) taxa recorded on 6th November 2012 (**Figure 4.34**). The low number of taxa recorded upstream of the Kuils River may be attributed to both poor substrate and poor water quality.

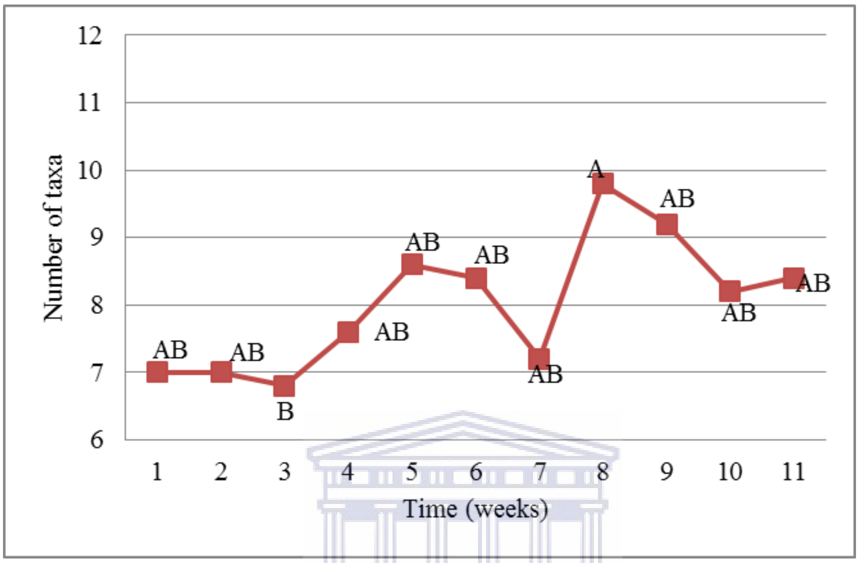

Figure 4.34 : The temporal variation of number of taxa of the upper Kuils River. Means with the same letter are not significantly different ($p \leq .05$).

4.2.5.1.3 Average score per taxa (ASPT)

The mean weekly ASPT scores ranged between 2.850 and 3.21 on the 27th November 2012 and 23th October 2012 respectively (Figure 4.35). ASPT over all sampling period were less than standard limit recommended by South African Guideline. These low ASPT scores may result from tolerant species.

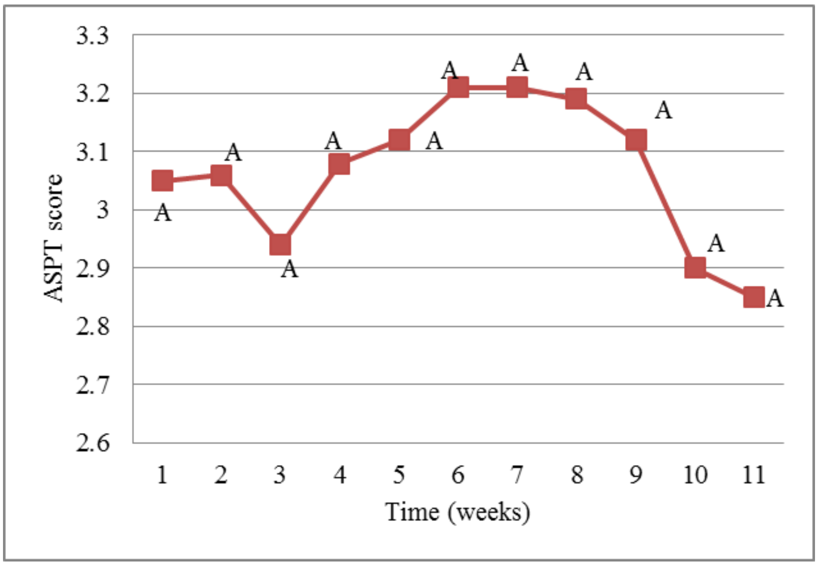

Figure 4.35 : Temporal variations of number of taxa of the upper Kuils River. Means with the same letter are not signicantly different.

4.2.5.2 SASS, Number of taxa ASPT average at different sampling sites

4.2.5.2.1 SASS average at different sites

Mean values of SASS5 for each sampling site as presented in **figure 4.36** are less than 50. In the main stream, the average values decreased from upstream to downstream sites. The highest mean score (42.54) was obtained at site K3 while the lowest (12.09) mean score was recorded at site K1. A decrease in water quality due probably to storm water from industrial and residences, organic matter from road-bridge and poor substrate diversity observed at downstream sites may have reduced the diversity in macroinvertebrate and therefore, the SASS score.

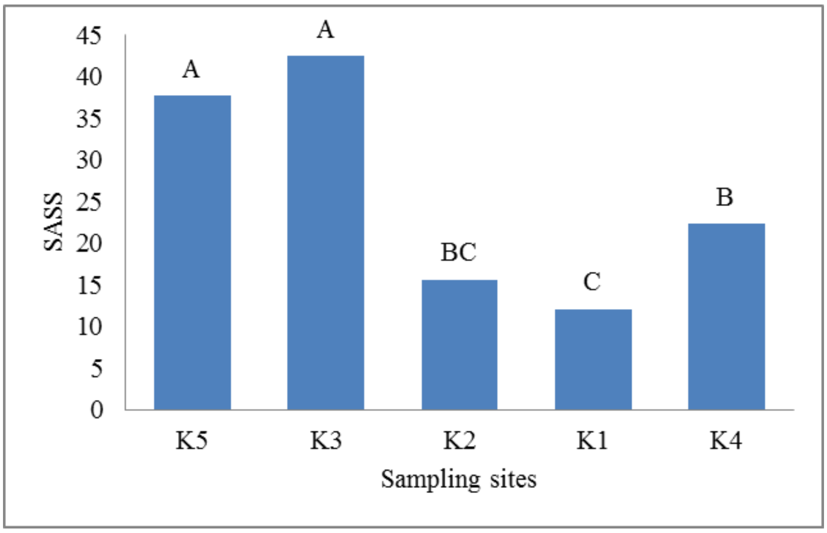

Figure 4.36: South Africa Scoring System (SASS5) at different sites of the upper Kuils River. Means with the same letter are not significant different (p≤.05)

4.2.5.2.2 Number of Taxa at different sites

The number of taxa at the different sampling sites indicate similar patterns of distribution as SASS. In the main stream the number of taxa decreased from upstream to downstream (Figure 4.37). The highest mean (12 taxa) number of taxa were observed at K3 while the lowest number (4 taxa) were recorded at K1. In addition to poor water quality, downstream sites are dominated by sand. At the tributary site (K4), the mean number of taxa was 6.81.

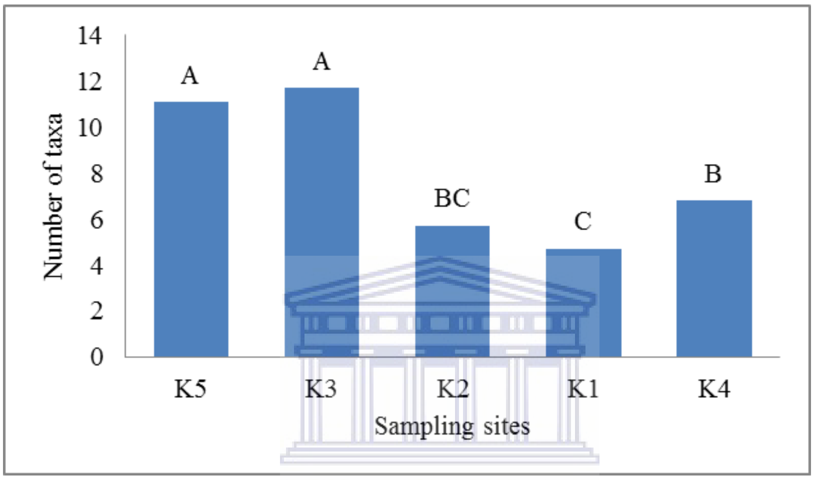

Figure 4.37: Number of Taxa at different sites of the upper Kuils River. Means with the same letter are not significant different (p≤.05)

4.2.5.2.3 Mean ASPT scores at different sites

The mean scores show a similar trend as the SASS with the highest mean score 3.61 at site K3 while the lowest mean score evaluated at site K1 was 2.55 (figure 4.38). At the tributary of site K4, the mean weekly ASPT score was 22.27. Low ASPT scores at downstream sites may be due to low scores attributed to non-sensitive species dominated by Gasteropoda (Physidae).

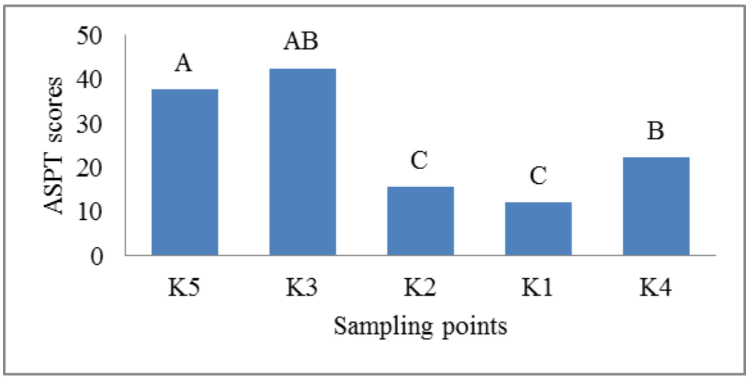

Figure 4.38 Average Score Per Taxa at different sites of the upper Kuils River. Means with the same letter are not significant different ($p \leq .05$)

4.2.6 Ecological quality of the river

Table **4.4** shows that the water quality in the upper sites of the Kuils River falls within unacceptable limits. The study results show that the sites K1 and K2 have been critically modified due to the following impacts; human settlement, road-bridge, golf course, and industrial activities. The site K4 has also been critically modified. This site is the stem of the Kuils River, and it is impacted by storm water from settlement areas, poison to kill plant roots, and dead plants. The upstream sites (K3, K4, K5) were also found to have been seriously modified because they have been impacted by surface runoff and stormwater from urban areas and soil erosion.

Table 4.4 Ecological state at each sampling point upper stream of the Kuils river per week

Site	Date	SASS	No of Taxa	ASPT	Ecological state
K1	18th September 2012	14	6	2,3	Critically modified
K1	24th September 2012	9	4	2,5	Critically modified
K1	2nd October 2012	9	4	2,5	Critically modified
K1	9th October	16	6	2,66	Critically modified
K1	16th October	12	5	2,4	Critically modified
K1	23th October	7	3	2,33	Critically modified
K1	30th October	15	5	3	Critically modified
K1	6th November	15	5	2,5	Critically modified
K1	13th November	19	6	3,1	Critically modified
K1	20th November	7	3	2,3	Critically modified
K1	27th November	10	4	2,5	Critically modified
K2	18th September	17	6	2,8	Critically modified
K2	24th September	17	6	2,8	Critically modified
K2	2nd October	12	5	2,4	Critically modified
K2	9th October	9	4	2,25	Critically modified
K2	16th October	15	6	2,5	Critically modified
K2	23th October	24	7	3,42	Critically modified
K2	30th October	14	5	2,8	Critically modified
K2	6th November	9	4	2,25	Critically modified
K2	13th November	14	6	2,33	Critically modified
K2	20th November	20	7	2,8	Critically modified
K2	27th November	21	7	3	Critically modified
K3	18th September	47	12	3,9	Seriously modified
K3	24th September	30	9	3,3	Seriously modified
K3	2nd October	32	9	3,5	Seriously modified
K3	9th October	45	12	3,75	Seriously modified
K3	16th October	48	12	4	Seriously modified
K3	23th October	39	11	3,54	Seriously modified
K3	30th October	40	11	3,63	Seriously modified
K3	6th November	49	13	3,76	Seriously modified
K3	13th November	49	14	3,5	Seriously modified
K3	20th November	47	14	3,35	Seriously modified
K3	27th November	42	12	3,5	Seriously modified
K4	18th September	13	4	3,25	Critically modified
K4	24th September	24	7	3,4	Critically modified
K4	2nd October	27	8	3,3	Critically modified
K4	9th October	24	7	3,42	Critically modified
K4	16th October	33	10	3,3	Seriously modified
K4	23th October	26	8	3,25	Critically modified
K4	30th October	22	7	3,14	Critically modified
K4	6th November	38	11	3,4	Seriously modified

K4	13th November	21	6	3,5	Critically modified
K4	20th November	11	4	2,75	Critically modified
K4	27th November	6	3	2	Critically modified
K5	18th September	21	7	3	Critically modified
K5	24th September	30	9	3,3	Seriously modified
K5	2nd October	24	8	3	Critically modified
K5	9th October	30	9	3,33	Seriously modified
K5	16th October	34	10	3,4	Seriously modified
K5	23th October	46	13	3,53	Seriously modified
K5	30th October	28	8	3,5	Critically modified
K5	6th November	61	15	4,06	Poor
K5	13th November	46	14	3,2	Seriously modified
K5	20th November	43	13	3,3	Seriously modified
K5	27th November	52	16	3,25	Poor

4.2.7 Comparison of ecological state between 2005 and 2012

Our results show that the current water quality of the upper Kuils River in 2012 is seriously impaired compared to 2005. The study revealed that the water quality at all the sampling locations is worst compared to 2005, as it varied from fair to poor health. Dissolved oxygen values diagnosed varied from good (wet period) to fair (early summer) water quality. The River's constituent nutrients exceed the limit recommended by ecosystem health criteria in South Africa to unacceptable levels. To date, benthic macroinvertebrates collected are non-sensitive to poor water quality and the SASS5 indicated that the water quality was seriously modified compared to 2005 when aquatic invertebrates were reported to be in fair condition (RHP, 2005).

4.3 Physical and chemical variations upper of the river from 1989 to 2012

The trends in the physical and chemical parameters evaluated in the upper Kuils River from 1989 to 2012 include the temperature, pH, dissolved oxygen, nitrate and phosphate. The statistical analyses are showed in appendix 3. No data were available from 2003- 2011.

4.3.1 Mean yearly water temperature

The mean yearly water temperature varied from one sampling point to another (Figure 4.39). The mean water temperature ranged between (20.2°C) recorded in 1992 and (15.21°C) recorded in 2002. The statistical analysis indicates no significant difference between the years ($F = 1.103$ and $p = .4073$).

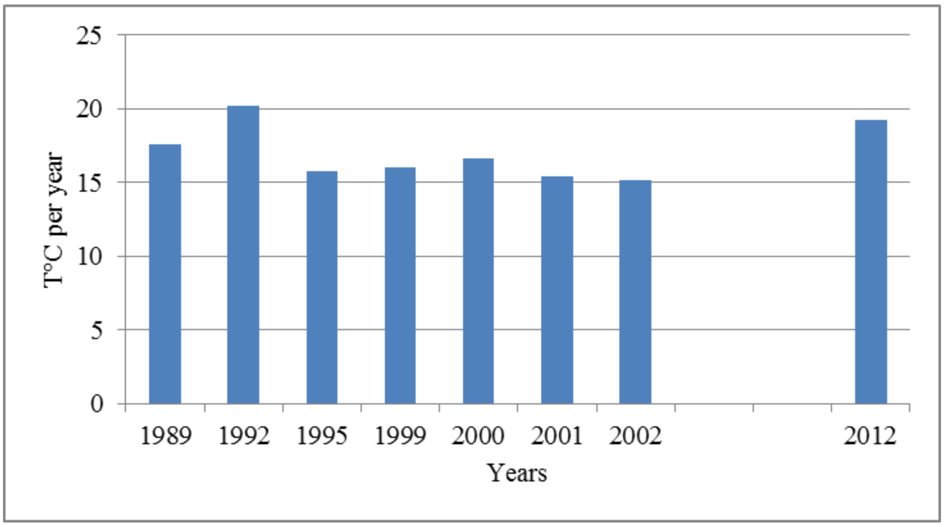

Figure 4.39 : Temporal water temperature variations of the Upper Kuils River from 1989 to 2012

4.3.2 Yearly change of pH variations

All pH means recorded were above 7, suggesting alkaline conditions (Figure 4.40). The highest mean recorded (7.903) was obtained in 2012, but overall, the trends of the curves overlap. The lowest mean pH (7.600) was registered in 1995 and 2002. Statisticaly, there is a significant difference between the years (F = 4.252 and p = .0079). At fine scale, a significant difference was found between 2002 and all the years (p-value: less than .05).

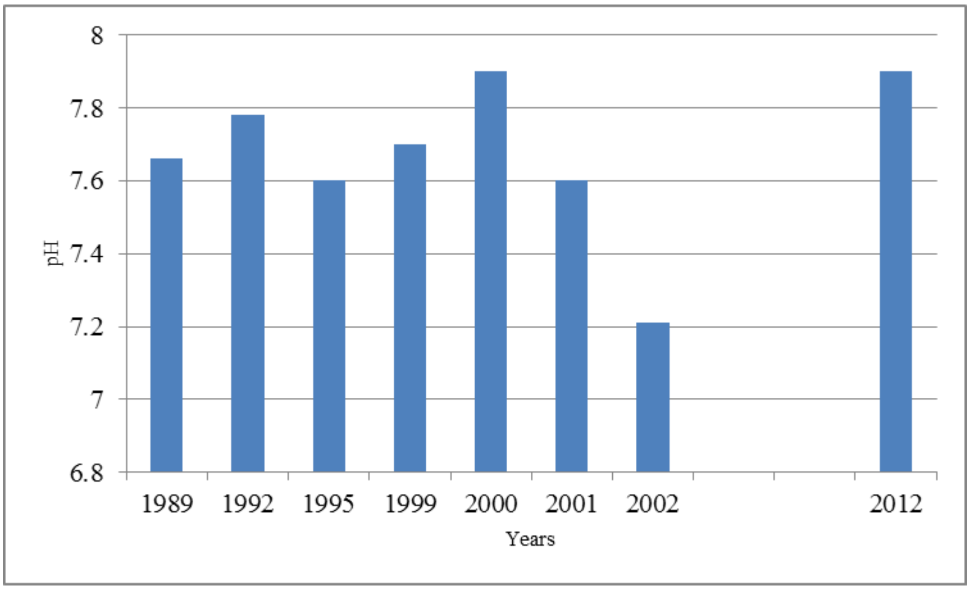

Figure 4.40 : Temporal pH variations of the Upper Kuils River from 1989 to 2012

4.3.3 Dissolved oxygen variations

Mean dissolved oxygen varied from one year to another (Figure 4.41**).** The highest dissolved oxygen means (8.233 mgL^{-1}) and (8.873 mgL^{-1}) were recorded in 1998 and 2012 respectively. The lowest mean (5.433 mgL^{-1}) was recorded in 2002. Generally, the ANOVA shows no significant difference between the years (F = 1.815 and p = .1531). Nevertheless, significant differences were observed between 2002 and certain years (2002 -1989: p = 0.0162; 2002 – 2001: p = 0.0075; 2002 – 2012: p = 0.0439).

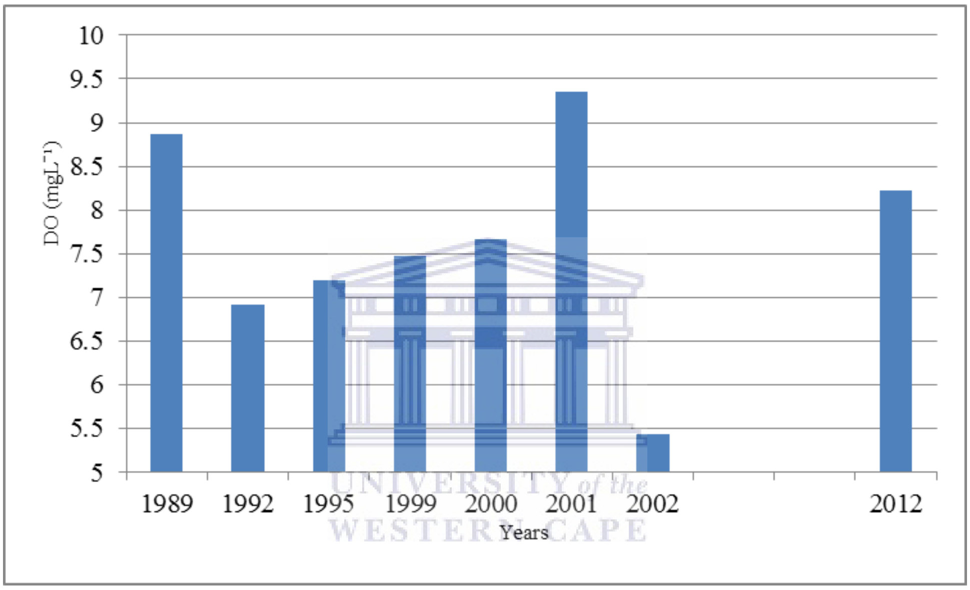

Figure 4.41 : Temporal dissolved oxygen variations of the Upper Kuils River from 1989 to 2012

4.3.4 Phosphate variations

The mean values of phosphate concentrations varied over the years (Figure **4.42**). The highest mean yearly phosphate concentration (1.00 mgL^{-1}) was recorded in 2012 while the lowest means (0.064 mgL^{-1} and 0.092 mgL^{-1}) concentrations were recorded in 2000 and 2002 respectively. We found a significant difference between years (F = 3.115 and p = 0.0283). A decrease in concentrations was observed from 1989 to 2002 before a sudden increase in 2012. The difference is significant between 2012 and all years (p <.05) except 1989 (p = 0.1171).

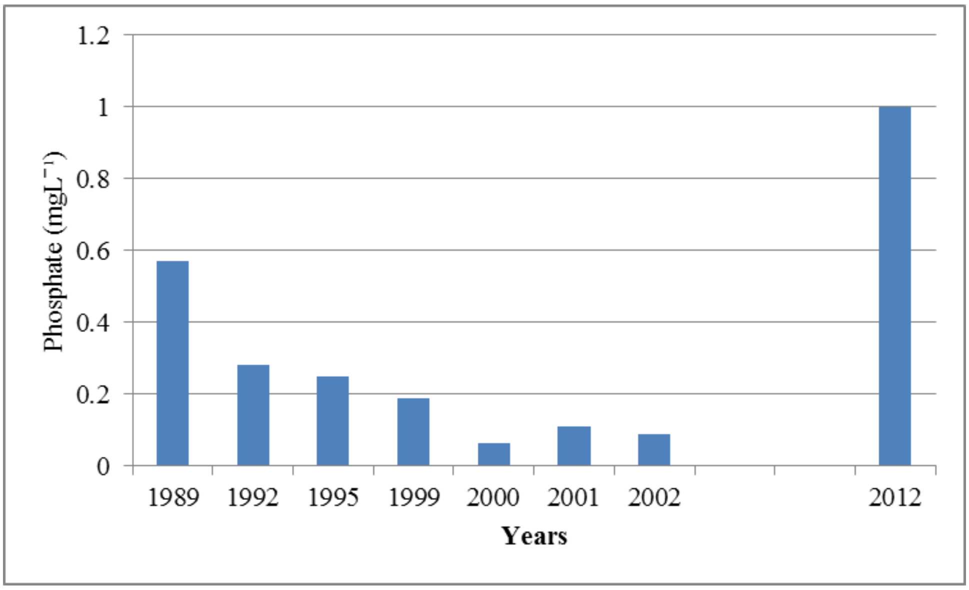

Figure 4.42 Temporal Phosphate variations of the upper Kuils River from 1989 to 2012

4.3.5 Nitrate variations

The mean yearly nitrate concentrations in the upper Kuils River varied from one year to another (Figure **4.43**). The highest yearly mean nitrate concentration (1.2 mgL^{-1}) was recorded in 1995 while the lowest mean (0.21 $mg\ L^{-1}$) concentrations were recorded in 2002 before rising to mesotrophic condition (1.007 $mg\ L^{-1}$) in 2012. There is no significant difference between years (ANOVA - $F = 2.611$ and $p = .0526$). Nevertheless, significant differences were observed between 2000 and all other years (p-value: less than .05), and between 2002 and all years except in 1999.

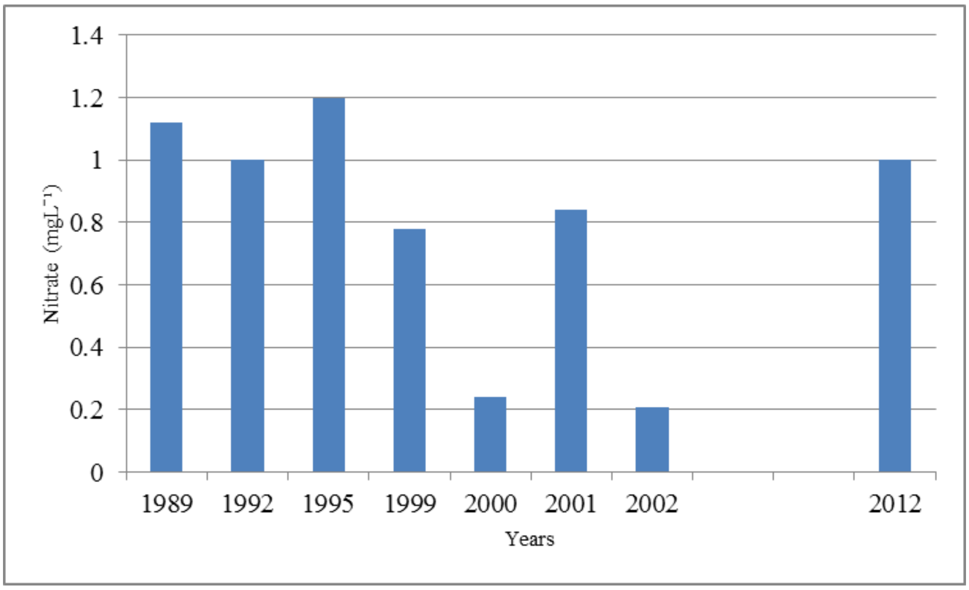

Figure 4.43 Temporal Nitrate variations of the upper Kuils River from 1989 to 2012

CHAPTER FIVE: DISCUSSION

The chapters discusses the results of the preceding chapter. Abiotic parameters and macroinvertebrate fauna were sampled to evaluate land use practices and their impacts on water quality in the upper Kuils River.

5.1 pH

The water pH is one of the important factors that determines water quality. In many natural freshwater, pH values that support a diverse aquatic fauna ranged from 5.0 to 9.0 (DWAF, 1996a). The pH values recorded here varied between 7.13 and 8.76. These values are indicative of alkaline conditions, and may be attributable to geological characteristics of the soil over which the river flows (Dallas and Day, 2004). With reference to South Africa's Water Quality Guidelines, these values are within the natural limits suggested to protect fish life (6.5-9.0), and in accordance with domestic use (6.0-9.0). To protect aquatic ecosystems, it has been suggested in the South Africa Water Quality Guidelines that pH values must not be greater than 5 % of the allowable limit to a specific station (DWAF, 1996a and c). Other studies in the upper Kuils River catchment found similar result (Itoba, 2010; Feng, 2005; Ninham, 1979). The alkalinity in the Kuils, Diep and Berg rivers is attributed to Malmesbury groups of rock which characterise the catchments (Adelana *et al.* 2010; Belcher, 2009; Nditwani, 2004). In the Luvuvhu river catchment, Makhera *et al.* (2011) attributed basic conditions to the geological formation of Sibasa basalt. In the Democratic Republic of Congo, natural alkalinity is observed in rivers that flow from Kahuzi Biega National Park. This natural alkalinity may be due to soils containing limestone mineral and igneous geology that originated from volcanic activity (Bagalwa *et al.* 2012; Ngera *et al.* 2009).

Abowei (2010) attribute pH variation to 3 factors namely: influx, debris decomposition and imbalance of hydrogen ion from surface runoff during the rain period. According to Dallas and Day (2004), biotic activities and human impacts may also have an influence on pH levels. Downstream sites (K1, K2) had lower pH **(Figure 4.8)**. This low pH may be due to high nutrients concentrations (nitrate and phosphate) from industrial and residential areas and road-bridge. Several authors support the view that low pH values may be attributed to nutrient enrichment favorable to growth of plants. Aquatic plants including algae, and organic matter decomposition by bacterial produce high level of carbon dioxide that reduces pH in water body (Klerk *et al.* 2012; Golder Associated, 2011; Abowei, 2010; Igbinosa and Okoh, 2009;

Ramollo, 2008). The tributary site K4 upstream was characterized by high phosphate concentrations and many macrophytes. This site had low when pH compared to other sites.

Other studies reported that a high TDS concentration is one of the major factors that influence pH fluctuations in aquatic ecosystems (Gueade et al. 2009; Ramollo, 2008; Hart and Campbell, 1994). These authors found that a decrease in pH values was proportional to low TDS concentrations in the Kuils River while higher pH related to an increased in TDS. Drawing on the ideas of these authors, there is a strong probability that higher pH values (K3 = 8.185, K5 = 8.300) recorded at upstream sites (K3, K5) may be justified by an increased in TDS (K3 = 733.423 mg L^{-1}, K5 = 747.692 mg L^{-1}) concentration whereas lower pH (K1 = 7.685, K2 = 7.84) at downstream sites may be attributed to lower TDS (K1 = 611.46 mg L^{-1}, K2 = 640 mg L^{-1}) at these sites.

In many natural freshwaters, low pH levels increase the toxicity of some substances. The combination of elevated hydrogen ion concentrations and heavy metals in solution can eliminate many types of aquatic life (Kimmel *et al.*, 1985). Both High pH values and low pH levels affect aquatic biota (Schofield and Trojnar, 1980).

5.2 WATER TEMPERATURE

In many freshwaters, temperature changes as a result of hydrological and climatological parameters of catchments and the region at spatial and temporal scales. Our results reveal that the water temperatures range between 12.3 °C and 27.1 °C. These values are within the margin for inland South Africa freshwater (5°- 30°C) DWAF, 1996a). Contrary to Vannote *et al.* (1980), the highest water temperatures **(Figure 4.9)** were obtained at upstream sites (K3, K4, K5). This result may be due to sampling time; we recorded the temperature in downstream sites (K1, K2) in the morning from 9:00 to 10:30 am while in upstream sites (K3, K4 and K5) temperature was recorded from 12:00 to 1:30 pm (**Figure 4.10**). Indeed, the changes in water temperature in many South African rivers were due to daily variation—with a decrease in water temperature during the night and early morning while an increase was observed from mid day to afternoon (Dallas and Rivers-Moore, 2011; Golder Associates, 2011; Dallas, 2008). Poole *et al.* (2001) observed that the stream was heated during the day due to sun radiation and cools down during the night. In addition, the changes in water temperature may also be attributed to daily fluctuation of cloud cover and relative air humidity. The clearing of riparian vegetation and the removal of canopy cover also expose small streams to significant temperature changes (Osibanjo *et al.* 2011; Dallas, 2008). In our study area, a near complete removal of riparian vegetation cover had disrupted the natural conditions of the river system. This has affected the ecological functions by increasing water temperature due to solar radiation which reaches the river (Fang, 2010; Dallas, 2008; Dallas and Day, 2004; Poole *et al.* 2001; Davies and Day, 1998; DWAF, 1996a).

Located in a region with Mediterranean climate, the water temperature of the Kuils River is subject to seasonal variations. The highest monthly mean values of water temperature were observed in November 2012 at all sampling sites and the lowest values, which occurred in September 2012, may be due to temporal variations (Figure **4.2** and Figure **3.1**). The seasonal temperature variations were similar to those reported by other authors (Ezekile *et al.* 2011; Abowei, 2010; Igbinosa and Okoh, 2009; Dallas, 2008; Ndiitwani, 2004).The algal proliferation coincided with the rise of water temperature at all sites in November 2012. High water temperature favors a rapid development of bacteria, phytoplankton and macrophytes and can cause algal bloom at high concentration of nutrients (Davies and Day, 1998; Chapman, 1996).

Many studiess maintain that an increase in water temperature degrades the physical environment due to a reduction in dissolved oxygen, decrease pH, and increase BOD due to high microorganism activities (CWT, 2010; Abel, 2002; Rivers-Moore *et al.*, 2008; Mason, 2002; Chapman, 1996). High water temperature also accelerates the rates of chemical reactions and increases toxicity, as well as aquatic organism vulnerabilities. It affects abundance, species richness, diversity and composition of macroinvertebrate community. Many sensitive species (Ephemeroptera, Trichoptera, Plecoptera) may disappear because of oxygen depletion and be replaced by more tolerant species (Simuliidae and Chironomidae) which increase in number and supplant the original species in the ecosystem (Rivers-Moore *et al.* 2008; Dallas, 2008).

5.3 TOTAL DISSOLVED SOLIDS

Total dissolved solids (TDS) concentrations recorded in the present study ranged between 416.0 mg L^{-1} and 916.5 mg L^{-1}. These concentrations fall within the limit suggested by South African guidelines. For instance, for domestic water use, the guidelines suggest a range of 450 to 1000 mgL^{-1} TDS (DWAF, 1996b) and from 200 to 1100 mgL^{-1} to protect aquatic ecosystems (Golder Associated, 2011; DWAF, 1996a). Naturally, total dissolved solids are influenced by soil and geological characteristics of the catchment (Adelana *et al.* 2010; Dougall, 2007; Chapman, 1996). In the present study, high TDS concentration values recorded at all sampling sites are attributed to Malmesbury shales that characterize the upper catchment. These rocks include high quantity of leachable ions able to conduct electricity (Brown and Magoba, 2009; Leske and Buckley, 2003). Studies conducted above the confluence with Bottelary River (Ninham, 1979) and in Berg river catchment (Belcher, 2009) revealed similar results. The Berg river tributaries revealed that shale Rivers had higher salinity in comparison with the TMS Rivers which present lower salinity (Clark and Ractliffe, 2007; Flugel, 1991). Dougall (2007) found that streams that flow over clay show higher conductivity because of the presence of materials that ionize when washed into the water. In the Democratic Republic of Congo, the high TDS concentrations recorded in Kahuzi-Biega National Park Rivers were attributed to the soil and igneous geology originating from volcanic action (Bagalwa *et al.,* 2012). In North America, high salinity concentrations are attributed to weathering and leaching from shale and glacial deposit (Flugel, 1991).

In addition to natural salinity attributable to geology, anthropogenic input in aquatic ecosystems and high water evaporation also lead to high total dissolved salts concentrations (Van der Laan, *et al.*, 2012; Rabies *et al.* 2011; Augustijn *et al.*, 2011; Hogan *et al.* 2007; Vhevha *et al.* 2000;). The clearing of Riparian vegetation and the removal of canopy cover observed from upstream to downstream is caused by urbanization, and agricultural activities. These may also have an influence on TDS concentrations change as reported by researchers such as (Kasangaki *et al.*, 2008; Ndiitwani, 2004). In semi-arid regions, an increase in salinity concentration is often attributed to evapotranspiration associated with irrigation and open water evaporation (Hogan et al. 2007; DWAF, 1996a).

In our study the spatial variation indicated lower TDS concentration at downstream sites (K1, K2) (**Figure 4.11**), which may be due to water dilution. The findings from this study are at variance with Dougall's (2007) study in United State of America. The increased flow rates (volume of water) may contribute to a decrease of TDS downstream (Marcellus, 2009; Deksissa, *et al.*, 2003). It has been shown that electrical conductivity decreases with distance because of H_2SO_4 concentration in the headwaters diluting with distance (Dougall, 2007; Chapman, 1996). Both sites K1 and K2 were not significantly affected by industrial zones and waste from bridges. At the upstream sites (K3, K4, K5) the rise in TDS concentration may be due low-flow (Marcellus, 2009; Deksissa, *et al,* 2003; U.S Department of the interior, 2005).

At site K4 high TDS levels may also be accelerated by decomposition of plant material (DWAF, 1996a). Several authors suggest that an increase in flow leads to a decline of TDS because of dilution rates while higher salts concentration may be attributed to low-flow and high water evaporation (Van der Laan, *et al.*, 2012; Klerk et al. 2012; Abowei, 2010; Clark and Ractliffe, 2007; Marofi and Maryanaji, 2007; Salinas *et al.*, 2000; Flugel, 1991).

The changes in TDS concentrations may affect individual species, community structure and nutrient cycling (DWAF, 1996a). The toxicity of salinity to macroinvertebrates often occurs at very high concentrations and varies from one species to another (O'Hayre and Amendola, 2010). High TDS may cause osmotic stress and affect osmoregulatory ability of aquatic fauna (Igbinosa and Okoh, 2009).

5.4 DISSOLVED OXYGEN

Dissolved oxygen is a fundamental factor used to determine aquatic ecosystem health (Nel *et al.* 2013). To protect fish and macroinvertebrates, the South African Guidelines suggest a Target Water Quality Range (TWQR) from 80 % to 120 % of oxygen saturation; saturation below 40 % may be lethal for aquatic life (DWAF, 1996a). In the present study, dissolved oxygen concentrations from 65 water samples collected in 5 sites ranged between 4.26 mgL^{-1} (46.3 %) and 14.68 mgL^{-1} (154 %) in the main stream, and between 0.07 mgL^{-1} (0.8 %) and 9.9 mgL^{-1} (100.5 %) in the tributary. Compared to TWQR, 86.15 % of our samples fall within limits suggested by South Africa guideline, whereas 13. 85 % of the samples are unacceptable to support many life stages. Nel *et al.* (2013) found similar results which suggest that the Kuils river water quality fell from natural in 2005 to unacceptable categories in 2008 for DO.

As regards spatial scale, the mean dissolved oxygen varied between 8.85 mg L^{-1} at K1 and 9.48 mg L^{-1} at K5 (**Figure 4.13**). This variation indicates that the Kuils River water quality fell within the natural category (Nel *et al.* 2013). However, poor dissolved oxygen levels recorded at site K4 during the majority of sampling periods may be attributed to slow or stagnant water, high phosphate concentrations, high phosphate concentrations, decaying vegetable matter, storm water and surface runoff from residential area and golf course. Dowling and Wiley (1986) reported that a decline in oxygen in slow flowing streams is attributed to the action of spring floods in removing vegetation or to high decay of organic matter followed by an increasing water temperature. The discharge of effluents from residential zone and decaying plant in receiving water bodies also reduce dissolved oxygen concentrations as a result of the increased microbial activities occurring during the degradation of organic matter (Nel *et al.* 2013; CSIR, 2010; Osibanjo *et al.* 2010; Oberholster and Ashton, 2008; Dallas and Day, 2004; Mason, 2002; DWAF, 1996a). Oxygen depletion depends on the total amount and nature of the organic material load in the rivers, and the numbers and types of bacteria which degrade waste discharged into the river (Mason, 2002). High biological oxygen demand (BOD) depletes oxygen in natural aquatic ecosystems because microorganisms are using up the dissolved oxygen (Canadian council of Ministers of the Environment, 1999).

High oxygen concentrations occurred during the major part of our sampling period may be attributed to the wet period. During the rainy season dissolved oxygen concentration is often higher because the rain interacts with oxygen in the air as it falls (Mason, 2002). The temporal fluctuations showed higher mean dissolved oxygen in September and October 2012 due to lower temperatures while a decrease in dissolved oxygen was observed in November 2012 when temperature increased (**Figure 4.5**). In addition, during dry and hot periods, water movement is slow because it mixes less with the air which leads to a decline in dissolved oxygen concentration. Water quality degradation in DO (5.404 mgL^{-1}) in November 2012 coincided with algae proliferation which may be due to high phosphate concentrations. Many studies revealed that high nitrate and phosphate concentrations results in eutrophic conditions and algal blooms (primary productivity) that cause oxygen depletion (e.g Nyenje *et al.* 2010; Chapman, 1996; Perry and Vanderklein, 1996; NRC, 1978). Similar results were obtained on studies carried out in Bottelary River, the main tributary of the Kuils River (Itoba, 2010; Ma, 2005) and in Nigeria (Ezekiel *et al.* 2011). Several studies confirm that seasonal fluctuations of dissolved oxygen are higher in wet winter and lower during the dry season (Annalakshmi, and Amsath, 2012; Rahman *et al.* 2012; Panigrahi and Patra, 2011; Manikannan *et al.* 2011; Nkwoji *et al.* 2010; Pejman *et al.* 2009; DWAF, 1996a).

The depletion of dissolved oxygen is usually linked to accumulation and decomposition of dead organic matter which consumes oxygen and generates harmful gases such as methane and hydrogen sulfide (Frost and Sullivan, 2010; Nyenje *et al.* 2010). In aquatic ecosystem, oxygen is a fundamental factor that influences fauna composition. The oxygen requirements of benthic macroinvertebrates vary with species, life stages and different life processes (Alabaster and Lloyd 1982 cited by NWQMS, 2000), and size. The more sensitive benthic macroinvertebrates including Ephemeroptera (mayflies), Trichoptera (caddisflies), and Plecoptera (stoneflies) which respire with gills or by direct cuticular exchange decline and may be entirely eliminated with oxygen depletion (Abel, 2002; Dallas and Day, 2004). The tolerant taxa are Oligochaeta (worms), Hirudina (leeches), and Chironomidae (chironomids) which usually dominate over other benthic macroinvertebrates in the more altered water (Couceiro *et al.*, 2007; Abel, 2002).

5.5 PHOSPHATE CONCENTRATIONS

In many natural freshwaters, phosphate appears as dissolved orthophosphate where their concentrations vary between 0.005 mg L^{-1} and 0.020 mg L^{-1} (Mason, 2002; Chapman, 1996). To protect aquatic ecosystems, South African Guidelines record that phosphorus concentrations < 0.005 mg L^{-1} are associated with oligotrophic conditions, whereas 0.005 – 0.025 mg L^{-1} indicates mesotrophic conditions, concentrations from 0.025 to 0.250 mg L^{-1} indicate eutrophic conditions and concentrations greater than >0.250 mg L^{-1} lead to hypertrophic conditions (Nel *et al.* 2013; Van Ginkel, 2011; DWAF, 1996a). In many South Africa natural water resources, the mean value of phosphorus concentrations is 0.73 mg L^{-1}. This value shows that many South Africa Rivers are excessively enriched and indicate hypertrophic conditions (Oberhoster and Ashton, 2008). In the present study, phosphate concentrations (0.28 mgL^{-1} to 5.27 mg L^{-1}) exceed the recommended limits by ecosystem health criteria in South Africa showing *hypertrophic* condition.

Higher mean value of phosphate concentration at downstream site (K1) (**Figure 4.15**) is attributed to overloading of discharge from hospital, industrial and residential areas, and discharge from road-bridge. Recently, Nel *et al.* (2013) found that Kuils River presents unacceptable water quality in terms of phosphate concentrations. The previous studies carried out in Kuils and Bottelary rivers catchment present similar results (Feng, 2005; Ma, 2005). Several studies carried out in many South Africa rivers including Cape Town rivers and streams revealed that higher phosphate concentrations were due to waste water treatment works, detergent, industrial and informal settlement effluents, pump stations, recreational grass, golf courses and agricultural irrigation (Nel *et al.* 2013; Van Ginkel, 2011; CSIR, 2010; Belcher 2009; Oberholster and Ashton, 2008; Clark and Ractliffe, 2007; De Villiers, 2007). In Nigeria, several studies revealed that the discharge of effluents from industrial areas are responsible for high phosphate concentrations which favor eutrophic conditions (Osibanjo *et* *al.* 2010; Igbinosa and Okoh, 2009). Other authors found similar results (Annalakshmi and Amsath 2012; Bisimwa, 2009).

Compared to K3 and K5, the high level of phosphate observed at site K4 (**Figure 4.15**) may be attributed to dead vegetation litter, storm water from residential and golf courses areas, and to slow and shallow water which characterize the stream. Because there is a significant difference between K3 (downstream) and K4, one might reasonably suppose that high concentration observed at site K4 did not affect significantly the river in terms of phosphate. The tributary site K4 is a temporary small stream susceptible to increase in phosphate concentrations in the main river during the rainy period. Davis and Koop (2006) reported that in shallow aquatic ecosystem, the subsurface sediments constitute a primary source of phosphate concentrations. To avoid flooding during winter, all aquatic vegetation in site K4 was cleared and left in the stream with the aim of increasing nutrients. In Forest Rivers and streams, decomposition of organic matter from the riverbank vegetation is the main source of nutrients (Bagalwa *et al.* 2012).

The temporal variation shows higher monthly means of phosphate concentrations in November 2012 (**Figure 4.6**) which may be due to rain reduction. Recently, Klerk *et al.* (2012) and Clark and Ractliffe (2007) associated high flow and discharge with low phosphate concentrations due to dilution, while dry period with low flow showed highest phosphate concentrations. In many Nigeria water bodies, higher concentrations of pollutants are observed in dry season due to low dilution of effluent (Kanu *et al.* 2011; Igbinosa and Okoh, 2009). The increase in phosphate concentrations in November 2012 coincides with algae proliferation at all sampling sites. There is evidence that in many freshwaters, high levels of phosphate concentrations lead to eutrophication that stimulates the growth of blooms of cyanobacteria, and excessive growth of macrophytes (Van Ginkel, 2011; Karels and Petnkeu, 2010; Belcher, 2009; Allan, 2004; Fried *et al*, 2003; Mason, 2002; Chapman, 1996; Jones and Lee, 1984). The excessive alga blooms alter water quality in term of oxygen depletion which leads to a loss of sensitive biodiversity in term of abundance, composition and species richness. In the Upper Kuils River, aquatic biota were dominated by tolerant species such as Chironomid, Gastropod, Hirudina and Simulid which may be linked to their ability to tolerate the lower oxygen concentrations (Frost and Sullivan, 2010; Nyenje *et al.* 2010; Dodds, 2006; Rast and Thornton, 1996).

5.6 Nitrate concentrations

Nitrate is usually more abundant than phosphorus in aquatic ecosystems (Scindler, 1974 and 1977). In natural South Africa freshwater, inorganic nitrogen concentrations are less than 0.5 mg L^{-1} and may range between 5 and 10 mgL^{-1} when water is highly impacted (DWAF, 1996a). To protect human health, South Africa Guidelines suggest a range of 0 – 6 mg L^{-1} as nitrate concentrations without adverse health effects, and from 0 – 100 mg L^{-1} with no adverse effects for livestock watering (DWAF, 1996b and 1996c). To protect aquatic ecosystems the South Africa guidelines propose the following ranges of nitrate concentrations: <0.5 mg L^{-1} as oligotrophic conditions and from 0.5 – 2.5 mg L^{-1} as mesotrophic conditions (DWAF, 1996a). In the present study weekly means of nitrate concentrations at all sampling sites show mesotrophic conditions ranging between 0.680 and 1.340 mg L^{-1}.

Spatial distributions of nitrate concentrations reflect the nature and type of pollutant sources from upstream to downstream sites. At downstream sites (K1 and K2), higher nitrate concentrations (**Figure 4.14**) are attributed to wastewater discharge and runoff from industrial areas, decomposition of organic matter under the bridge, and storm water and runoff from residence and hospital areas. Nel *et al.* (2013) found that unacceptable water quality category in Kuils River was due to waste water treatment work, leaking sewers and storm water ingress or infiltration. Several other studies support that high nitrate concentrations are due to discharges from industrial effluents and urban runoff into the rivers (Kanu *et al.* 2011; Osibanjo *et al.* 2010; Oberholster and Ashton, 2008; De Villiers and Thiart, 2007; Environment Canada, 2003; Mason, 2002; Novotny, 2002; Fatoki *et al.* 2001; Fleming and Fraser, 1999). Annalakshmi and Amsath, (2012) cited manures, inorganic fertilizer, sewage disposal and ground water as principal sources of nitrate concentrations in surface water. In comparison with other sites, K4 had low nitrate concentration (0.09 mgL^{-1}) may be due to rapid growth of aquatic plants, and shallow and slow water which often dries in summer. Several studies support that growth of macrophytes and algae consume nitrate and lead to an elevated evapotranspiration rates (Mason, 2002; Johnes and Burt, 1993).

Although temporal variation (**Figure 4.7**) was not observed in the present study, other studies revealed the highest nitrate concentration in winter and spring while a decrease was observed in summer due to greater biological productivity and high rates of evapotranspiration (Environmental Canada, 2003). Many researchers found that denitrification rates increase with increasing temperature (Cavari and Phelps, 1977; Holmes *et al.* 1996). Generally, nitrate is non toxic to aquatic organisms, however, higher concentrations can be harmful to juvenile organisms (Annalakshmi and Amsath, 2012).

5.7 HISTORICAL DATA

This section discusses historical water quality data of the upper Kuils River over twenty years (1992-2012). The yearly variation in water temperature shows no major difference. Poole *et al.* (2001) reported that it is difficult to predict inter-annual variation in water temperature. However, many researches link the inter-annual thermal change in water temperature with human influences associated with increasing population growth and climate change (Dallas, 2008). The mean water temperature which ranged from 15.21°C in 2002 to 20.3°C in 1992 fell within the acceptable limit to protect aquatic ecosystem. Many South African inland waters indicate water temperature range from 5°C to 30°C (DWAF, 1996a).

The pH values indicate alkaline conditions due to the influence of soil geology. The differences observed after each ten years may be attributed to human activities. In 1992 the pH was 7.783 then decrease deeply in 2002 to 7.217 before increasing to 7.903 in 2012. Dissolved oxygen shows the same trend between 2002 and 2012 ($p = .0439$). The mean dissolved oxygen recorded in 1992 was (6.917 mgL^{-1}) then decrease slightly in 2002 to (5.433 mgL^{-1}) before improving to (8.233 mgL^{-1}) in 2012. According to South Africa guidelines, those values varied from good to fair category in terms of water quality (Nel *et al.* 2013).

In 1992 nitrate concentrations were high (1.00 mgL^{-1}) then decline to (0.218 mgL^{-1}) in 2002 before increasing to (1.007 mgL^{-1}) in 2012. A significant difference was observed after each ten year period (1992-2002: $p = .0307$ and 2002-2012: $p = .0295$). With regards to phosphate, in 1992 the concentration value was 0.289 mgL^{-1} followed by a drop of 0.092 (mgL^{-1}). It then rose agian in 2012 to (1.000 mgL^{-1}). A significant difference was observed between each ten year period (1992 – 2012: $p = .0127$ and 2002-2012: $p = .0027$) may be due to population growth and extension of the urban area. According to South Africa guidelines the trophic conditions ranged from mesotrophic to hypertrophic conditions (Nel *et al.* 2013; Van Ginkel, 2011). The main causes of water quality degradation include population growth, industrial activities, urban development, and an increasing demand for agricultural irrigation and stock-water, and combustion of fossil fuels (Voelz *et. al*, 2005; Davies *et al.*, 2010; Boesch, 2002). Galloway and Cowling (2002) revealed that during the last century, anthropogenic Nitrogen exceeded natural sources owing to rising demand for food due to growing human population. An increasing trend in nitrogen or phosphorus was noticed in many seas after 1950s and primary production by phytoplankton doubled from 1960s to the 1990s (Boesch, 2002). In Cape Town, the population growth (2,563,095 in 1996 to 3,740,026 in 2011), rapid industrial and residential development, and increased road infrastructure observed from 2001 to 2011 seem to go together with the rise of nutrient in the river (City of Cape Town, 2011b; City of Cape Town, 2012; River Health Programme, 2005). In many Cape Town rivers the phosphorus concentration increased from 0.125 mg L^{-1} in 2000 to >0.25 mg L^{-1} in 2011 (City of Cape Town, 2011c). Studies conducted by De Villier and Thiart (2007) and De Villier (2007) revealed that phosphate and nitrate concentrations had doubled after ten years (1995 – 2004) due to anthropogenic input such as fertilizer and sewage effluent. The most pronounced concentration levels of 10.05 μgP L^{-1} per year were observed in Swartkops River (Eastern Cape) from 1970 to 2005.

In Southern Ontario, Fleming and Fraser (1999) showed trends in nitrate concentrations over time from 1960s to 1990s. In urban sites, nitrate concentrations was 1.03 mgL^{-1} between 1964 and 1969, then 1.77 mgL^{-1} from 1980-1989 and 3.35 mgL^{-1} from 1990 -1994. In contrast a decrease in phosphate concentrations was observed from 2.17 mgL^{-1} in 1960s to 0.09 mgL^{-1} in 1990s. The clearing of riparian vegetation due to extending of urban area also reduced the capacity of river catchment to retain nutrient resulting in the elevated nutrient loads, flow modification, and low dissolved oxygen concentrations (River Health Programme, 2006, 2005 and 2003; Boesch, 2002).

5.8 MACROINVERTEBRATES AND WATER QUALITY

At all sampling sites, a total of 8409 specimens representing 28 families and 11 orders were sampled. The most abundant orders included the Gastropoda, Diptera, Ephemeroptera, Oligochaeta, and Hirudina. The macroinvertebrate composition indicates weak diversity in macroinvertebrates fauna probably due to physical and chemical degradation coupled with low substrate diversity. There is much evidence which indicates that the nature and diversity of substrates influence the abundance, composition and distribution of macroinvertebrate at different sites. For each specific group there is a habitat preference which influences the benthic macroinvertebrate distribution in the river system (Klemm *et al.* 1990; Silveira *et al.* 2006). Dallas, (2007) asserted that stone and vegetation lodge the higher number species of benthic macroinvertebrate, whereas sandy biotopes are characterized by a lower number of species.

At downstream sites (K1 and K2) low diversity (**Figure 4.30**) may be attributed to a sand biotope and a growth of algae bound on marginal and aquatic vegetation. In addition, these sites receive wastewater from urban and industrial areas and debris from road-bridges resulting in many hazardous chemical substances which affect aquatic biota. These sites were dominated by Physidae (Gastropoda). According to Davies *et al.* (2010) the predominance of Gastropoda in urban stream, a decrease in biodiversity, and lower EPT score are due to habitat degradation. Fisher (2003) reported that the poor diversity in Kuils River is caused by the predominance of sand which is not the ideal habitat for benthic macro invertebrates.

At the upstream sites (K3 to K5) the increase in diversity (**Figure 4.30**) may be due to litter and marginal vegetation, stones, solid blocks, and gravel. These substrates are favorable for settlement of benthic macroinvertebrates. Those sites were populated by Simuliidae (Diptera), Chironomidae (Diptera), and Baetidae (Ephemeroptera). Some families belonging to Odonata (Coenagrionidae, Libellulidae and Aeschnidae) having vegetation as a preferential biotope (Dallas, 2007) were also recorded in small numbers. The small tributary site (K4), characterized by decaying and aquatic vegetation showed the predominance of Corixidae (Hemiptera), Chironomidae, and Simuliidae. The Oligochaeta and Hirudina showed a regular pattern at all sampling sites.

In the present study, all benthic macroinvertebrates found at all sampling sites are widespread in many Cape Town Rivers (Smith-Adao, 2004; Fisher, 2003). Many studies have shown that urban impacts reduce macroinvertebrates diversity resulting in a community dominated by tolerant taxa such as Oligochaeta (worms), Simulidae (black flies), Chironomidae (bloodworms), Hirudina (leeches) and Gastropoda (snails) while the sensitive taxa belonging to Ephemeroptera, Plecoptera, and Trichoptera decline in abundance and diversity (Souto *et al.* 2011; Al-Shami *et al.* 2010; Davies *et al.* 2010; Makoba *et al.* 2008; Miserendino, *et al.*, 2008; Roberston, 2006; Silveria *et al.*, 2006; Abel, 2002; Paul and Meyer, 2001; Davies and Day, 1998; Winter and Duthie, 1998). Overall, these taxa were recorded at all sampling sites; they are tolerant to water pollution. It has been found that changes in environmental conditions in aquatic ecosystems affect abundance and composition of benthic macroinvertebrates (Klemm, *et al.*, 1990). According to Knoben *et al.* (1995) human actions at the landscape scale are a principal threat to the ecological integrity of river ecosystems, impacting habitat water quality and biota. The high nitrate and phosphate concentrations led to hypertrophic conditions that stimulate the growth of algae. The excessiveness of algae decreased the amount of oxygen in the water causing a change in macroinvertebrates composition, abundance and diversity (Kolar and Rahel, 1993).

5.9 SOUTH AFRICAN SCORING SYSTEM (SASS)

Untreated stormwater runoff from urban areas includes a variety of pollutants which are harmful to benthic macroinvertebrates and alter water quality (Voelz *et al.*, 2005). In South Africa, to assess land use impacts on water quality the South African Scoring System index was initiated using benthic macroinvertebrates (Dicken and Graham, 2002; Dallas, 2007). With respect to water quality, there are sensitive species with higher scores and tolerant species with lower scores (Dallas, 1997).

According to the RHP the mean SASS and ASPT scores evaluated in the 5 sites were below 50 and 4 respectively. These values were less than the limit recommended by the South African Guidelines (Golder Associates, 2009; Dallas, 2007b; Dicken and Graham, 2002). The low values obtained up river may be due to the low score attributed to tolerant taxa and poor diversity of benthic macroinvertebrate fauna found. The majority of taxa found during our sampling period are tolerant to water pollution and have a score that ranged between 1 and 5. From our findings, lower SASS scores at downstream sites (K1, K2) (**Figure 4.34**) may result partly from poor habitat suitability for benthic macroinvertebrates due to homogeneity of sand biotope. In addition hypertrophic conditions due to pollutants from industries, road-bridges, storm water and surface runoff from industrial and residential areas may also affect the macroinvertebrate fauna composition. The predominant taxa and their respective score are Physidae (Pouch snail: 3), Oligochaeta (worms: 1), Hirudina (leeches: 3), Chironomidae (Midges: 2). The low score attributed to each taxon justifies the lower SASS and ASPT scores and therefore shows water quality degradation. These taxa are tolerant of low dissolved oxygen and high nutrients loads (Armah *et al.* 2012; Al-Shami *et al.* 2011).

About one decade ago, Fisher (2003) examined the impact of channelization on the geomorphology and ecology of the Kuils River and found that lower SASS (less than 20) and ASPT (below 5) scores were due to both bad water quality and sand biotope. Several authors have showed that the lack of diversity and abundance of biotopes on the one hand, and poor physical and chemical characteristics on the other hand had an influence on macroinvertebrate diversity; and therefore on SASS5, NoT and ASPT score (Couceiro *et al.* 2007; Dallas, 2007, Thirion, 2007; Dallas, 2005; CES, 2004; Dicken and Graham, 2002; Dallas, 1997; Klemm *et al.* 1990).

At the upstream sites (K3, K5) a slightly higher SASS and ASPT scores (**Figures 4.34** and **4.36**) could be explained by aquatic vegetation, gravel and solid blocks or stones observed in the river. Studies conducted by Dallas (1997) and Coastal and Environmental Service (2004) revealed that stones habitats have high SASS and ASPT scores due to elevated number of sensitive species, followed by vegetation biotopes while lowest SASS and ASPT scores occur in sand biotopes. Similar results were also obtained in the Western Cape and Mpumalanga (Dallas, 2007) and also in Lourens River respectively (Smith-Adao, 2004).

Several studies support the view that the temporal change in environmental characteristics may affect benthic macroinvertebrates favoring an increase in non-sensitive species (Dallas, 2004; Reece and Richardson, 2000). In the present study, the monthly variation in SASS and ASPT score was not observed during all sampling periods. However, a progressive change in abundance of tolerant taxa was noticed from October to November. In October the predominant macroinvertebrates included Diptera (Simuliidae and Chironomidae: 1489 specimens), Gastropoda (Physidae: 901 specimens), Oligochaeta (348 specimens), Hirudina (56 specimens), and Baetidae (62 specimens). In November an increase in abundance for the same taxa was observed: Diptera (Simuliidae and Chironomidae: 1506 individuals), Gastropoda (Physidae: 1570 individuals), Hirudina (289 specimens), and Ephemeroptera (Baetidae: 504 specimens) except Oligochaeta which decrease with 237 specimens. These taxa dominate other benthic macroinvertebrates, and therefore they lead to lower SASS and ASPT scores (Zweig and Rabeni, 2001; Davies and Day, 1998; Dallas, 1997; Lenat *et al.* 1979).

5.10 THE CURRENT ECOLOGICAL STATE COMPARED TO THE RIVER's HEALTH IN 2005

In reference to the South African RHP, the upper reach of the Kuils River presents unacceptable water quality due to low scores attributed to non-sensitive taxa (Nel *et al.* 2013; Golder Associates, 2009). Most of the macroinvertebrates collected at the different sites have low score except some taxa such as Chlorocyphidae (score = 10), Aeschnidae (score = 8), and Naucoridae (score = 7).

At downstream sites (K1, K2), with SASS (7 to 24) and ASPT (2.3 to 3.42), the water quality is classified in category **F** i.e critically impaired (**Table 4.3**). In critically impaired water, there are only very few tolerant pollution taxa that subsist (Golder Associates, 2009). The ecological category of water at upstream sites (K3, K4 and K5) is **E** i.e seriously modified and only tolerant taxa exist in the water body except some taxa cited above (Golder Associates, 2009). Currently, high values of nitrate and phosphate concentrations were recorded with higher values at downstream sites (K1 and K2) than upstream sites (K3, K5). According to the South African guidelines, the water quality in terms of nutrient varies from eutrophic to hypertrophic conditions. It is clear that industrial areas are the principal cause of water degradation in the river in terms of nutrient enrichment. Many studies indicated that high nutrient concentrations recorded in rivers and streams originating from human activities differring with land use (De Villeirs and Thiart, 2007; Environment Canada, 2003; Mason, 2002; Fatoki *et al.* 2001).

In 2005, the aquatic ecosystem health of the upper reach of the Kuils River was in category **D** i.e fair condition (RHP, 2005). The river was characterized by a loss of pollution-sensitive benthic macroinvertebrate taxa. The sensitive macroinvertebrates are replaced by non-sensitive species (City of Cape Town, 2011c; Belcher, 2009; River Health Programme, 2005). According to RHP (2005) stormwater runoff waste and detritus from urban areas were major causes of water quality degradation which may also pose a problem to human health. In terms of nutrient enrichment, the Kuils River water quality ranged from fair to unacceptable while dissolved oxygen varied from natural to good (Nel *et al.* 2013). Comparing the historic data to the present study, we notice that nutrient concentrations (nitrate and phosphate) in Kuils River have increased from 2002 to 2012, probably due to growth of population and rapid urban development. Brown and Magoba (2009) and City of Cape Town (2011c) have maintained that water quality degradation in terms of nutrient enrichment (nitrate and phosphate) may be due to human activities. Information from monitoring points in Cape Town rivers have shown that possible sources of pollution include informal settlements, industries, wastewater treatment works, pump stations, golf course runoff, urban runoff and leaking sewers (Nel *et al.* 2013; Ninham, 1979).

In addition to nutrient enrichment, the River Health Programme (2005) maintains that the poor habitats noticed in many Cape Town Rivers are due to channelization which resulted in loss of benthic macroinvertebrate. Despite efforts made since 2002 by the City of Cape Town to protect water quality and ecological resources, and to ensure the environmental health

(Brown and Magoba, 2009) many Cape Town Rivers including Kuils River water quality remained impaired due to human impacts (River Health Programme, 2005). Studies conducted by Fisher (2003) indicated that the SASS and ASPT scores averages were less than 20 and 4 respectively which indicate that the ecological health of the river is critically modified. In Fisher's perspective, the large number of Chironomidae and Simulidae was associated with water quality degradation.

CHAPTER SIX: CONCLUSION AND RECOMMENDATIONS

This chaper presents the conclusions and some reommendations in light of the preceding discussions. Given the location of the Kuils River coupled with the mediterranean climate it experiences, the Kuils River is under pressure from urban extension from the City of Cape Town due to rapid population growth. Characterized by various wetland ecosystems that play important ecological and economic roles, the Kuils River requires a good management scheme to ensure its conservation and protection. These wetlands are important for the maintenance of the Cape Flats aquifer, for aquatic life, to attenuate floods and improve water quality by the uptake of nutrients, as well as filtering pollutants and sediments (Heydorn and Grindley, 1982).

The study was conducted in the upper reach of the Kuils River and compared the state of the river in 2012 to that of 2005 using Water Quality parameters and the South African Scoring System. Our results show that the nature and quantity of pollutants vary according to land use practice which impacts the river. The major sources of pollution include storm water and surface runoff from urban and industrial area and golf courses and organic matter from litter under the road-bridges. However, the industrial processes can have far-reaching implications for water quality in the study area.

Nutrient enrichment (phosphate and nitrate mainly) was higher at all sampling sites than the limit set in the South African guidelines to protect aquatic ecosystems. Phosphate and nitrate concentrations classified the river into hypertrophic and mesotrophic conditions respectively. The phosphate and nitrate concentrations during the study period combined with a low level of dissolved oxygen in November 2012, and abundant growth of algae at downstream sites are issue of major concern. The high nutrient load associated with rapid growth of algae constitutes a precursor to general eutrophic conditions in the river, a situation that could potentially lead to anoxic conditions with severe consequences for the fauna of the river.

Because of the nutrient overload observed, it is undoubtedly true that the high dissolved oxygen levels recorded in this study were due to the rain. Thus, there is strong evidence that the decrease in dissolved oxygen recorded in November 2012 parallels the high nutrient (nitrate, phosphate) in the study area. In addition to the Malmesbury shale conditions of the catchment area, high total dissolved salts (TDS) and basic conditions (high pH) observed at all sampling sites were influenced by human impacts.

Biological monitoring through benthic macroinvertebrates confirmed the bad ecological state of the river. From the SASS5 scores recorded at all sampling sites, it may be concluded that the upper reach of the river has unacceptable water quality. Although the SASS5 and ASPT score at upstream sites appeared to be high than at downstream sites, these values were below 50 and less 3.5 respectively. This indicated that water quality in this study area is critically modified.

The upper reach of the river is critically modified and almost all of the original biotope and macroinvertebrate species have been lost. Poor water quality and biotope availability were the main cause for poor biodiversity in the river. As a matter of fact, the predominance of non-sensitive taxa on the one hand, and the absence of pollution sensitive taxa on the other hand at upstream sites and downstream sites were the major causes for low SASS5 and ASPT scores and therefore indicate poor water quality. Certain BMI belong to Plecoptera, Trichoptera, and Ephemeroptera are known as sensitive to water pollution and respond rapidly to change in water quality. These indicators of good water quality were totally absent during our study period.

The taxa accumulation curve (ascending curve) showed that if the sampling would continue, additional taxa might be recorded. However, the high similarity observed between sites showed that the absence of the non-tolerant species for one, and low macroinvertebrates diversity for another resulted from poor water quality and low habitat diversity in the river.

From evidences above, there is a strong possibility that the upper reach of the Kuils River is more altered than in 2005. In 2005 the upstream Kuils River water quality was found to be in fair conditions. However, the duration of our study (three months) was too short and limited to certain parameters and they cannot confirm that the river is polluted from upstream to downstream.

To other researchers and Department of Water Affairs, we recommend:

To extend this study at all seasons from upstream to downstream to include other parameters namely heavy metals, index of habitat integrity, and microbial studies to have a comprehensive view of the river functioning. Due to inadequate functioning, the Kuils River requires restoration to avoid the risk of losing its economic and ecological values.

Regular bioassessment and monitoring is necessary to evaluate human influences on water quality to ensure good water management and to propose measures of mitigation by DWAF to minimize pollution effects. Macroinvertebrates, microbiology and physical and chemical parameters should be used to understand the functioning of river.

REFERENCES:

Abel, P. D., 2002. Water Pollution Biology. Second Edition. TAYLOR & FRANCIS Ltd. 286pp.

Abowei, J.F.N 2010. Salinity, Dissolved oxygen, pH and Surface water Temperature conditions in Nkoro River, Niger Delta, Nigeria. Advance Journal of Food Science and Technology 2(1): 36-40

Adelana, S., Xu, Y., and Vrbka, P., 2010. A conceptual model for the development and management of the Cape Flats aquifer, South Africa. Water SA Vol. 36 No. 4. pp 461-474

Ahuja, S., 2009. Handbook of Water Purity and Quality. Elsevier. First Edition. New York. 435 pp.

Allan, J.D., 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35:257–84

Al-Shami, S.A., Rawi, C.S.M., Hassan Ahmad, A., Hamid, S.A., Nor, S.A.M., 2011. Influence of agricultural, industrial, and anthropogenic stresses on the distribution and diversity of macroinvertebrates in Juru River Basin, Penang, Malaysia. Ecotoxicology and Environmental Safety 74: 1195-1202.

Al-Shami, S.A., Rawi, C.S.M., Hassan Ahmad, A., Nor, S.A.M., 2010. Distribution of Chironomidae (Insecta: Diptera) in polluted rivers of the Juru River Basin, Penang, Malay. Journal of Environmental Sciences. 22(11) 1718–1727.

Annalakshmi, G., and Amsath, A., 2012. Nutrient Status of Arasalar River, a Tributary of Cauvery River at Tanjore District of Tamilnadu, India. International Journal of Plant, Animal and Environmental Sciences. Volume-2: 214-222

Arimoro, F.O., Ikomi, R.B and Efemuna, E. 2007. Macroinvertebrate Community Patterns and Diversity in Relation to Water Quality Status of River ASE, Niger Delta, Nigeria. Journal of Fisheries and Aquatic Science 2(5):337 - 344

Armah, F.A, Ason, B, Luginaah, I. and Essandoh, P.K, 2012. Characterization of macro-benthic fauna for ecological health status of the Fosu and Benya lagoons in coastal Ghana. J. Ecol. Field Biol. 35(4): 279-289.

Arthur, J.B., 2010. The influence of upstream forest on macroinvertebrate communities in pastoral landscapes. A thesis degree of Master of Science in Ecology at the University of Canterbury.

Augustijn, D., Fares, A., and Tran, D.N., 2011. Temporal and spatial variations in total suspended and dissolved solids in the upper part of Manoa stream Hawaii. Journal of Sustainable Watershed Science & Management 1(1): 1-9.

Bagalwa, M., Zirirane, N., Pauls, S., Karume, K., Ngera, M., Bisimwa, M., and Mushagalusa, N.G. 2012. Aspects of the physic-chemical characteristics of rivers in Kahuzi-Biega National Park, Democratic Republic of Congo. Journal of Environmental Protection, 3, 1590-1595.

Belcher, T., 2009. Freshwater Assessment: Proposed Belle Riva Development Mosselbank River Near Fisantekraal. Report Prepared for Withers Environmental Consultants.

Benstead, J.P and Pringle, C.M. 2004. Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. Freshwater Biology 49, 490–501

Benstead J.P, De Rham P.H, Gattolliat J-L, Gibon F-M, Loiselle P.V, Sartori M, Sparks J.S, and Stiassny, M.L. 2003. Conserving Madagascar's Freshwater Biodiversity. Vol. 53 No. 11. BioScience 1101

Bessong, P.O, Odiyo, O.J, Musekene, J.N and Tessema, A., 2009. Spatial distribution of diarrhoea and microbial quality of domestic water during an outbreak of diarrhea in the Tshikuwi Community in Venda, South Africa. JHEALTH POPUL NUTR Vol. 27(5): 652-659

Bilotta and Brazier, 2008. Understanding the influence of suspended solids on water quality and aquatic biota. Water research. 42. 2849-2861.

Bisimwa, A.M., 2009. Water quality management in Lake Kivu Basin: A case study of Bukavu sub-basin, DR Congo. Thesis submitted for the degree of Masters in Integrated Water Resources Management. University of Dar es Salaam.

Biswas, A.K., 2006. Water Management for Major Urban Centres. Water Resources Development. Vol. 22 No2, 183-197.

Biswas, A.K 2004. Integrated Water Resources Management: A Reassessment a Water Forum contribution. International Water Resources Association, Water International, Volume 29, Number 2, Pages 248-256.

Boesch, D.F., 2002. Causes and consequences of nutrient overenrichment of coastal waters. International seminar on nuclear war and planetary emergencies. 26th Session. World Scientific Publishing, Singapore.

Bredenhand, E., 2005. Evaluation of macroinvertebrates as bio-indicators of water quality and assessment of the impact of the Klein Plaas dam of the Eerste River. Thesis presented for degree of Master. University of Stellenbosch.

Brown, C. and Magoba, R 2009. Rivers and wetlands of Cape Town. Caring for our rich aquatic heritage. Water Research Commission. Report No TT 376/08. 377p

Camargo, A., Alonso, A. and Salamanca, A., 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. Chemosphere 58: 1255-1267.

Canadian Council of Ministers of the Environment. 1999. Canadian water quality guidelines for the protection of aquatic life: Dissolved oxygen (freshwater). In: Canadian environmental quality guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg.

Carpenter, S.R., Caraco, N.F., Corell, D.L., Howarth, R.W., Sharpley, A.N., and Smith, V.H., 1998. Non point Pollution of Surface Water with Phosphorus and Nitrogen. Ecological Applications, Vol. 8 No 3(559-568).

Cavari, B.Z and Phelps, 1977. Denitrification in Lake Kinneret in the presence of oxygen. Freshwater Biology. Volume 7.

Chambers, D.B. and Messinger. T., 2001. Benthic Invertebrate Communities and their Responses to Selected Environmental Factors in the Kanawha River Basin in West Virginia, Virginia, and North Carolina. Water Resources Investigation Report 01-4021. NAWQA , US Dept of Interior and USGS.

Chapman, D., 1996. Water Quality Assessments - A Guide to Use of Biota, Sediments and Water in Environmental Monitoring. Second Edition. UNESCO/WHO/UNEP. University Press, Cambridge. 626p

Chapman L.J. and Chapman C.A. 2002. Tropical forest degradation and aquatic ecosystems: our current state and knowledge. Conservation of Freshwater Fishes, pp 235 – 249.

Chapman, C.A and Chapman, L.J., 2003. Deforestation in Tropical Africa. Impacts on aquatic ecosystems.

Chutter, F.M. 1972. An empirical biotic index of the quality of water in South African Streams and Rivers. Water Research Pergamon. Vol. 6pp 19-30.

City of Cape Town (CoCT), 2011a. Northern District Plan: Spatial development plan & environmental management framework. Volume 1. Baseline Information and Analysis Report. City Space. Planning Cape Town. 64p

City of Cape Town (CoCT), 2011b. Tygerberg District Plan: Spatial development plan & environmental management framework. Volume 1. Baseline Information and Analysis Report. City Space. Planning Cape Town. 64p

City of Cape Town (CoCT), 2011c. Catchment, stormwater and river management: inland and coastal water quality for the 12 month period ending June 2011. Report to Transport, Roads and Major Projects Portfolio Committee.

City of Cape Town (CoCT), 2012. City of Cape Town – 2011 Census – Cape Town Trends and Change - 10 years: Census 2001 – Census 2011.

Clark, B., and Ractliffe, G., 2007. Berg River baseline monitoring programme. Final Report. Volume 5: synthesis DWAF Report No. P WMA 19/G10/00/2107. Anchor Environmental consultants & freshwater consulting group.

Cleaver, G., Brown, L.R., Bredenkamp, G.J., Smart, M.C., and De Rautenbach, C.J 2003. Assessment of Environmental impacts of Groundwater abstraction from Table Moutain Group (TMG) Aquifers on Ecosystems in Kammanassie Nature Reserve and Environs. WRC Report No 1115/1/03

Coastal and Environmental Services (CES), 2004. Earstern Cape River Health Programme. Draft Technical Report: Buffalo River Monitoring, 2002-2003. DWAF: Southern Cluster.

Codd, G.A., Morrison, L.F., and Metcalf, J., 2005. Cyanobacterial toxins: risk management for health protection. Toxicology and Applied Pharmacology. 203: 264-272.

Coe, M.T, Latrubesse, E.M, Ferreira, M.E and Amsler, M.L., 2011. The effects of deforestation and climate variability on the streamflow of the Araguaia River, Brazil. Biogeochemistry. DOI 10.1007/s10533-011-9582-2.

Coetzee, M.A.S (undated). Water pollution in South Africa: its impact on the wetland biota. Center for Water in the Environment. Pp 247-262

Collier K.J, Cooper A.B, Davies-Colley R.J, Rutherford J.C, Smith C.M and Williamson R.B., 1995. Managing riparian zones: A contribution to protecting New Zealand's rivers and streams. Volume 1: Concepts. Department of Conservation. Wellington, New Zealand.

Connolly, N. M., Crossland, M. R., Pearson R.G., 2004. Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. J. N. Am. Benthol. Soc., 2004, 23(2):251–270.

Conradie, W.J., Carey, V.A., Bonnardot, V., Saayman, D., and Van Schoor, L.H. 2002. Effect of different environmental factors on the performance of sauvignon blanc grapevines in the Stellenbosch/Durbanville District of South Africa. I Geology, Soil, Climate, Phenology and Grape Composition. S. Afr. J. Enol. Vitic. Vol. 23, No2.

Conseil canadien des ministres de l'environnement. 1999. Recommandations canadiennes pour la qualité des eaux : protection de la vie aquatique-oxygène dissous (eau douce), dans Recommandations canadiennes pour la qualité de l'environnement, 1999, Winnipeg, le Conseil.

Couceiro, S.R.M, Hamada, N, Luz, S.L.B, Forsberg, B.R., and Pimentel, T.P., 2007. Deforestation and sewage effects on aquatic macroinvertebrates in urban streams in Manaus, Amazonas, Brazil. Hydrobiologia. Hydrobiologia, vol. 575, n.1, p. 271-284.

CSIR, 2010. A CSIR Perspective on Water in South Africa-2010. CSIR Report No. CSIR/NRE/PW/IR/2011/0012/A. ISBN: 978-0-7988-5595-2.

CWT (Clean Water Team), 2010. The clean water team guidance compendium for watershed monitoring and assessment. Citizen Monitoring Program of the State Water Resources Control Board.

Dallas, H.F 2008. Water temperature and riverine ecosystems: An overview of knowledge and approaches for assessing biotic responses, with special reference to South Africa. Water SA Vol.34 No.3 (393-404).

Dallas, H.F., 2007. The effect of biotope-specific sampling for aquatic macroinvertebrates on reference site classification and the identification of environmental predictors in Mpumalanga, South Africa. African Journal of Aquatic Science, 32(2): 165–173

Dallas, H.F., 2007. River Health Programme: South African Scoring System (SASS) Data Interpretation Guidelines. Institute of Natural Resources and Department of Water Affairs and Forestry

Dallas, H.F., 2005. River health programme: site characterisation field-manual and field-data sheets. Resource Quality Services Department of Water Affairs and Forestry.

Dallas, H.F., 2004. Seasonal variability of macroinvertebrate assemblages in two regions of South Africa: implications for aquatic bioassessment. African Journal of Aquatic Science, 29(2): 173-184

Dallas, H.F., 2000. Ecological reference conditions for riverine macroinvertebrates and the River Health Programme, South Africa. 1st WARFSA/WaterNet Symposium: Sustainable use of Water Resources.

Dallas, H.F., 1997. A preliminary evaluation of aspects of SASS (South Africa Scoring System) for the rapid bioassessment of water quality in rivers, with particular reference to the incorporation of SASS in a National Biomonitoring Programme. Sth. Afr. Sc. 23(1) 79-94

Dallas, H.F., and Day, J.A 2004. The Effect of Water Quality Variables on Aquatic Ecosystems. WRC Report No 224/04. 222pp.

Dallas, H. F., and Rivers-Moore, N. A., 2011. Micro-scale heterogeneity in water temperature. Water SA Vol. 37 No. 4. Pp505-512

Davies, B., and Day, J.A., 1998. Vanishing Waters. University of Cape Town Press. 487pp.

Davies, P.J., Wright, I.A., Findlay, S.J., Jonasson, O.J. and Burgin, S. 2010. Impact of Urban Development on Aquatic Macroinvertebrates in South Eastern Australia: degradation of in-stream habitats and comparison with non-urban streams. Aquat. Ecol. 44: 685-700.

Davis, J.R and Koop, K. 2006. Eutrophication in Australia Rivers, reservoirs and estuaries- a southern hemisphere perspective on the science and its implications. Hydrobiologia. 559: 23-76.

Deksissa, T., Ashton, P.J., and Vanrolleghem, P.A. 2003. Control options for river water quality improvement: A case study of TDS and inorganic nitrogen in the Crocodile River (South Africa). Water SA Vol. 29 No2. Pp 209-217

Derleth, P. 2003. Benthic Macroinvertebrates and logging activities: A case study in a lowland tropical forest in East Katimantan (Borneo, Indonesia). Thèse No 2836 Pour l'obtention du Grade de Docteur ès Sciences. Faculté Environment Naturel, Architectural et construit. Université de Lausane.

De Villiers, S. 2007. The deteriorating nutrient status of the Berg River, South Africa. Water SA Vol. 33, No5: 659-664.

De Villiers, S., and Thiart, C., 2007. The nutrient status of South African Rivers: concentrations, trends and fluxes from the 1970s to 2005. South African Journal of Science 103 (343 – 350).

Dicken, CWS and Graham, P.M., 2002. The South African Scoring System (SASS) version 5 rapid bioassessment method for rivers. African Journal of Aquatic Science, 27: 1-10.

Dodds, W.K., 2006. Eutrophication and trophic state in rivers and streams. Limnol. Oceanogr. 51 (1, part 2), 671-680.

Dong, Y. Zhang, Z., Jin, Y., Zhirang Li and Jian Lu., 2011. Nitrification performance of nitrifying bacteria immobilized in waterborne polyurethane at low ammonia nitrogen concentrations. Journal Environmental Sciences, 23 (3) 366-371.

Dougall, J.A., 2007. Downstream Effects of Glaciers on Stream Water Quality. Thesis for Degree of Master of Science in Geography. Portland State University. 130pp.

Dowling, D.C., and Wiley, M.J., 1986. The effects of dissolved oxygen, temperature, and low stream flow on fishes: A literature review. Illinois Natural History Survey. Aquatic Biology Section Technical Report.

Downes, B.J., Barmuta, L. A., Fairweather, P. G., Faith, D. P., Keough, M. J., Lake, P. S., Mapstone, B. D., Quinn, G. P., 2002. Monitoring Ecological Impacts. Concepts and practice in flowing waters. Cambridge University Press. 434 pages.

Du Preez, C.C., Strydom, M.G., Le Roux, P.A.L., Pretorius, J.P., Van Rensburg, L.D and Bennie, A.T.P., 2000. Effect on water quality on irrigation farming along the lower Vaal River: the influence on soils and crops. WRC Report No. 740/1/00. Water Research Commission, Pretoria, SA.

Duran, M. and Suicmez, M. 2007. Utilization of both benthic macroinvertebrates and physicochemical parameters for evaluating water quality of the stream Cekerek (Tokat, Turkey). J. Environ. Biol. 28(2), 231-236.

Duran, M. 2006. Monitoring water quality using benthic macroinvertebrates and physicochemical parameters of Behzat Stream in Turkey. Polish J. of Environ. Stud. Vol. 15, No5: 709-719

DWAF, 1996a. South African Water Quality Guidelines. Volume 7. Aquatic Ecosystems. Department of Water Affairs and Forestry, Pretoria, South Africa.

DWAF, 1996b. South African Water Quality Guidelines (second edition). Volume 1. Domestic use. Department of Water Affairs and Forestry, Pretoria, South Africa.

DWAF, 1996c. South African water quality guidelines (first edition). Volume 8: Field Guide

DWAF, 1996d. South African Water Quality Guidelines. Vol. 4: Agricultural Water Use: Irrigation. Second Edition. Department of Water Affairs and Forestry, Pretoria, South Africa.

Environment Canada, 2003. Canada Water Quality Guidelines for the Protection of Aquatic life: Nitrate ion. Ecosystem Health: Science-based Solutions Report No 1-6. National Guidelines and Standards Office, Water Policy and Coordination Directorate, Environment Canada 115 pp.

Ewart-Smith, J. and Ractliffe, G. 2002. Assessment of the potential impacts of the proposed N1N2 winelands toll highway project on aquatic ecosystems. Report submitted to Crowther Campbell & Associates on behalf of National Road Agency.

Ezekiel, E.N., Hart, A.I., and Abowei, J.F.N., 2011. The physical and chemical condition of Sombreiro River, Niger Delta, Nigeria. Research Journal of Environmental and Earth Sciences 3(4): 327-340.

Fang, W-T., 2010. River continuum and flood pulse: Exploring ecological and hydrologic concepts in riparian wetland. National Taiwan Museum Special Publication No. 14: 101-111

Fatoki, O.S, Muyima, N., and Lujiga, N., 2001. Situation analysis of water quality in the Umtata River catchment. Water South Africa. Vol. 27 No 4(467 – 474).

Feng Xiao Yi, 2005. A survey of inorganic chemical pollution in the Bottelary River, Cape Town. MSc, Thesis, University of Western Cape.

Fisher, RM. C., 2003. The Impacts of channelization on the geomorphology and ecology of the Kuils River, Western Cape, South Africa. Master's thesis. Department of Earth Sciences, UWC, 146p

Fleming, P.E.R and Fraser. 1999. Nitrate and Phosphorus Levels in Selected Surface Water Sites in Southern Ontario-1964-1994. Surface Water Quality.

Flugel, W.A., 1991. River salination due to dryland agriculture in the Western Cape Province, Republic of South Africa. IAHS Publ. no. 203.

Friberg, F., Otto, C., and Svensson, B.S., 1980. Effects of acidification on the dynamics of allochthonous leaf material and benthic invertebrate communities in running waters, p. 304-305. In D. Drab16s and A. Tollan (e d.) Ecological impact of acid precipitation. SNSF Project, Oslo, Norway.

Fried, S., Mackie, B and Nothwehr, E., 2003. Nitrate and phosphate levels positively affect the growth of algal species found in Perry Pond. Tillers 4. 21-24.

Frost and Sullivan, 2010. Eutrophication Research Impact Assessment. WRC Report No. TT 461/10

Galloway, J.N and Cowling, E.B., 2002. Reactive nitrogen and the world: 200 years of change, Ambio, 31 (2), 64-71.

Gaston, K.J., 1996. Biodiversity: A biology of numbers and difference. Blackwell Science. 396p.

GEI Consultants, Inc., 2009. Update on Issues for Further Investigation in Regard to the "Pond et al. Study" on Effects of Mountaintop Mining and Valley Fill on Benthic Invertebrate Communities. Technical Memorandum, GEI Consultants, Inc., Littleton, CO.

Gerber, A., and Gabriel, M.J.M., 2002. Aquatic invertebrates of South African Rivers. Version 2. Institute for Water Quality Studies. Department of Water Affairs and Forestry.

Germs, W., Coetzee, MS., Van Rensburg, L. and Maboeta, M.S., 2004. A prelimary assessment of the chemical and microbial water quality of the Chunies River-Limpopo. Water SA Vol.30 No2: 267-272

Golder Associates, 2011. Aquatic ecosystem study associated with the proposed NDC weirs in the Leeuspruit. Report Number 12786-10084-8.

Golder Associates, 2009. Ecological Baseline and Impact Assessment associated with the expansion of ERWT Welgedacht Wastewater Treatment Works. Report Number: 12070-8522-2. ERWAT WELGEDACHT EIA.

Gooday, A.J., Jorissen, F., Jorisen, F., Levin, L.A., Middelburg, J.J., Naqvi, S.W.A., Rabalais, N.N., Scranton, M., and Zhang, J. 2009. Historical records of coastal eutrophication induced hypoxia. Biogeosciences, 6, 1-39.

Gueade, G.A., Ouattara, A., Edia, O.E., and Gourene, G. 2009. Influence de la Geologie du Milieu Sur les Caracteristiques Physico-chimiques de Quelques Petits Cours d'eau

Naturels du sud de la côte d'ivoire (Afrique de l'ouest). European Journal of Scientific Research. Vol.34 No.3, pp.384-394

Haines, T.A., 1981. Acidic precipitation and its consequences for aquatic ecosystems: A review. Trans. Am. Fish. Soc. 110: 669 - 707

Hart B.T. and Campbell I.C. 1994. Assessment of river ''health in Australia. In: Uys M.C. (Editor) Classification of rivers, and environmental health indicators. Proceedings of a joint South African/Australian workshop. February 7-14 1994, Cape Town, South Africa. Water Research Commission Report No. TT 63/94.

Hartman, K. J., M. D. Kaller, J. W. Howell, and J. A. Sweka. 2005. How much do valley fills influence headwater streams? Hydrobiologia 532: 91-102.

Hauer, F.R., and Lamberti, G.A., 2006. Methods in Stream Ecology. Second Edition. Elsevier. 877pp.

Helmens, B.S., 2008. Response of Aquatic Biota to Changing Land Use Pattern in Streams of West Georgia, USA. Thesis Degree of Doctor of Phylosophy. Auburn University

Heydorn, A.E.F and Grindley, J.R 1982. Estuaries of the Cape. Part II Synopses of available information on individual Systems. Report No 16 Eerste (CSW6). CSIR Estuary Index Number.

Hindahl, M.S., Miltner, E.D., Cook, T.W and Stahnke, G.K 2009. Surface water quality impacts from golf course fertilizer and pesticide applications. International Turfgrass Society, Research Journal Volume 11.

Hogan, J.F, Phillips, F.M, Mills, S.K, Hendrickx, J.M.H., Ruiz, J., Chesley, J.T and Asmeron, Y. 2007. Geologic origins of salinization in a semi-arid river: the role of sedimentary basin brines. Geological Society of America, Inc. Vol. 35 No12, p.1063 – 1066.

Holmes, R.M., Jones, J.B., Fisher, S.G and Grimm, N.B. 1996. Denitrification in a nitrogen-limited stream ecosystem. Biogeochemistry 33: 125-146

Howard, H.S., Berrang, B., Flexner, M., Pond, G., and Call, S 2000. Kentucky Mountaintop Mining Benthic Macroinvertebrate Survey, Central Appalachian Ecoregion, Kentucky. Science and Ecosystem Support Division, Ecological Assessment Branch, Athens, GA.

Igbinosa, E.O and Okoh, A.I., 2009. Impact of discharge wastewater effluents on the physico-chemical qualities of a receiving watershed in a typical rural community. Int. J. Environ. Sci. Tech., 6 (2), 175-182

Itoba, T.E.F., 2010. Landuse impacts on water quality of the Bottelary River in Cape Town, Western Cape. CPUT. Theses & Dissertations. Paper 342.

Jacobsen, D., 2000. Gill size of trichopteran larvae and oxygen supply in streams along a 4000-m gradient of altitude. J.N. Am. Benthal. Soc. 2000. 19(2): 329-343

Jacobsen, D., 2008. Low oxygen pressure as a driving factor for the altitudinal decline in taxon richness of stream macroinvertebrates. Oecologia. Vol. 154 No 4 pp. 795-807.

Jacobsen, D., and Marin, R., 2008. Bolivian Altiplano streams with low richness of macroinvertebrates and large diel fluctuations in temperature and dissolved oxygen. Aquat Ecol 42:643–656

Jones, A.R., and Lee, F.G., 1984. Impact of phosphorus load reductions on eutrophication-related water quality of Roodeplaat Dam (Reservoir), Republic of South Africa. Water SA, Vol. 10 No 3(115-120).

Kaller, M. D., and Kelso, W. E., 2007. Association of macroinvertebrate assemblages with dissolved oxygen concentration and wood surface area in selected subtropical streams of the southeastern USA. Aquat Ecol 41:95–110.

Kanu, Ijeoma and Achi, O.K., 2011. Industrial effluents and their impact on water quality of receiving rivers in Nigeria. Journal of Applied Technology in Environmental Sanitation, 1(1): 75-86

Karels, J and Petnkeu, S., 2010. Determination of phosphate, nitrate, and sulfate in the red River by ion chromatography. Concordia College Journal of Analytical Chemistry 1: 24-28

Kasangaki, A., Babaasa, D., Efitre, J., Alastair McNeilage, A and Bitariho, R., 2006. Links between anthropogenic perturbations and benthic macroinvertebrate assemblages in Afromontane forest streams in Uganda. Hydrobiologia 563:231–245

Kasangaki, A, Chapman, L.J and Balirwa, J., 2008. Land use and the ecology of benthic macroinvertebrate assemblages of high-altitude rainforest streams in Uganda. Freshwater Biology.

Kenney, A. M., Sutton-Grier, E. A., Smith, F. R., and Gresens, E. S., 2009. Benthic macroinvertebrates as indicators of water quality: The intersection of science and policy. Terrestrial Arthropod Reviews 2 p99–128

Kimmel, W.G., Murphey, J.D., Sharpe, W.E., and DeWalle, D.R., 1985. Macroinvertebrate community structure and detritus processing rates in two southwestern Pennsylvania streams acidified by atmospheric deposition. Hydrobiologia 124, 97-102.

KIMO (Kommunenes Internasjonale MiliØorganisasjon) 2011. Good practice guide series: Mitigating eutrophication a manual for municipalities. Local Authorities International Environmental Organization pp 42.

Klemm, D.J., Lewis, P.A., Fulk, F., and Lazorchak, J.M., 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. United States. Environmental protection Agency. EPA/600/4-90/030.

Klerk, A.R, De Klerk, L.P, Chamier, J. and Wepener, V., 2012. Seasonal variations of water and sediment quality parameters in endorheic reed pans on the Mpumalanga Highveld. Water SA Vol. 38 No. 5

Knoben, R.A.E., Ross, C. and van Oirschot, M.C.M., 1995. Biological Assessment Methods for Watercourses. Volume 3. UN/ECE Task Force on Monitoring & Assessment. 86p

Kolar, C.S and Rahel, F.J., 1993. Interaction of a biotic factor (predator presence) and an abiotic factor (low oxygen) as an influence on benthic invertebrate communities. Oecologia 95:210-219

Le Roux, P., du Preez, CC., Strydom, M.G., van Rensburg, L.D., and Bennie, A.T.P, 2007. Effect of irrigation on soil salinity profiles along the Lower Vaal River, South Africa. Water SA Vol. 33 No. 4

Leske, T. and Buckley, C., 2003. Towards the development of a salinity impact category for South African environmental life-cycle assessments: Part 1 – A new impact category. Water SA Vol. 29 No. 3: 289 - 296

Lévêque, C. 1996. Ecosystèmes aquatiques. HACHETTE, 43, quai de Grenelle, 75905 Paris Cedex 15.

Li Rui, 2005. The use of treated effluent for agricultural irrigation in the Bottelary River area: Effluent quality farmers' perception and potential extent. Minithesis submitted for the degree of Masters of Science. Department of Earth Science, University of the Western Cape p 87.

Lorion, C. M and Kennedy, B.P., 2009. Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in Neotropical headwater streams. Freshwater Biology 54, 165–180

Lorion, C.M., 2007. Effects of Deforestation and Riparian Buffers on Lotic Communities in Southeastern Costa Rica: Implications for Biodiversity Conservation in Tropical Streams. A Dissertation Degree of Doctor of Philosophy with a Major in Natural Resources in the College of Graduate Studies University of Idaho.

Luger, M and Brown, C. (no dated). The impact of Treated Sewage Effluent on Urban Rivers. An Ecological, Social and Economic perspective.

Ma, Y., 2005. Monitoring of heavy metals in the Bottelary River using Typha capensis and Phragmites australis. A mini-thesis for the degree of Magister Scientiae. Department of Biodiversity and Conservation Biology, University of Western Cape.

Maiolini, B and Bruno, C.M., 2007. The River Continuum Concept revisited: Lessons from the Alps. Alpine space-man & environment, Vol. 3: The water balance of the Alps 67-76.

Makhera, M., Gumbo, J.R. and Chigayo, K. 2011. Monitoring of microcystin-LR in Luvuvhu River catchment: implications for human health. African Journal of Biotechnology. Vol. 10(3), pp. 405-412.

Makoba, H., Shivoga, W., Muchiri, M., and Miller, S.N., 2008. Use of Benthic Macroinvertebrates as indicators of Water Quality in River Njoro, Kenya. The 12th World Lake Conference: 2161-2168.

Manikannan, R., Asokan, S., and Samsoor Ali, A.H.M., 2011. Seasonal variations of physic-chemical properties of the Great Vedaranyan Swamp, Point Calimere wildlife Sanctuary, South-east coast of India. African Journal of Environmental Science and Technology. Vol. 5(9), pp. 673-681.

Marcellus Shale Committee, 2009. Evaluation of high TDS concentrations in the Monongahela River. Tetra Techn NUS, Inc.

Marofi, S., and Maryanaji, Z., 2007. Stream water quality in the Western regions of Iran. African Journal Biotechnology. Vol. 6(14), pp. 1728-1731.

Mason, C., 2002. Biology of Freshwater Pollution. 4th Edition. Pearson Education. 387pp.

McCabe, D. J. 2010. Rivers and Streams: Life in Flowing Water. Nature Education Knowledge 1(12):4

Merricks, T.C., D.S. Cherry, C.E. Zipper, R.J. Currie, and T.W. Valenti., 2007. Coal-mine hollow fill and settling pond influences on headwater streams in southern West Virginia, USA. Environmental Monitoring Assessment 129: 359-378.

Miserendo, M.L., Brand, C. and Di Prinzio, C.Y., 2008. Assessing urban impacts on water quality, benthic communities and fish in streams of Andes Mountains, Patagonia (Argentina). Water Air soil Pollut. 194: 91-110.

Moon, T.C., and Lucostic, C.M., 1979. Effects of Acid Mine Drainage on a South Western Pennsylvania Stream. Water, Air, and Soil Pollution 11: 377 – 390.

Moyo, P. and Mapira, J. 2012. Bioremediation with the water hyacinth (Eichhornia crasspes): a Panacea for river pollution in the city of Masvingo (Zimbabwe)?

National Water Quality Management Strategy, 2000. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Volume 1. The Guidelines. ISBN 09578245 05 (set).

Ndiitwani, T.B., 2004. The Water Quality and Ecological Status of the Diep River Catchment, Western Cape, South Africa. MSc in Integrated Water Resource Management. Faculty of Biology Conservation and Biodiversity. University of Western Cape.

Neda, K., Parisa, N., Sara, K., Mohammad Taghi, N., 2011. Variations in nitrate and phosphate contents of waters in the Southwest Caspian Sea. Journal of the Persian Gulf (Marine Science). Vol.2/No5/27-34.

Nel, N., Parker, A., Silbernagl, P., 2013. Improving water quality in stormwater & river systems: an approach for determining resources. Journal of the South African Institution of Civil Engineering. Vol 55 No 1, Pages 22–35, Paper 797

Ngera M.F, Baluku B, Cammaerts, D., et Bisimwa, M., 2009. Evaluation biologique de la pollution de la rivière Kalengo par les invertébrés aquatiques. Cahier du CERUKI, Numéro special CRSN-Lwiro p p 89-94

Nijboer, R.C and Verdonschot, P.F.M, 2004. Variable selection for modeling effects of eutrophication on stream and river ecosystems. Ecological Modelling 177: 17-39

Ninham Shand. 1979. Flow in the Kuils River and Drainage of the Catchment. Report No. 543/3200. Ninham Shand. Cape Town.

Nkwoji, J.A., Yakub, A., Ajani, G.E., Balogun, K.J., Renner, K.O., Igbo, J.K., Ariyo, A.A., and Bello, B.O., 2010. Seasonal variations in the water chemistry and benthic macroinvertebrates of a South Western Lagoon, Lagos, Nigeria. Journal of American Science Vol. 6(3): 85-92.

Novotny, V., 2003. Water Quality. Diffuse Pollution and Management. Second Edition. JOHN WILEY & SONS, INC. 864pp.

Novotny and Bendoricchio, 1989. Water Quality: linking nonpoint pollution and deterioration, Water Environ. Technol., Nov., pp 400-407.

NRC, 1978. Nitrates: An Environmental Assessment. Scientific and Technical Assessments of Environmental Pollutants. National Academy of Sciences. Washington, D.C. 723 pp.

Nyenje, P.M., Foppen, J.W., Uhlenbrook, S., Kulabako, R., and Muwanga, A. 2010. Eutrophication and nutrient release in urban areas of sub-saharan Africa- A review. Science of the Total Environment 408: 447-455.

Oberholster, P.J and Ashton, P.J., 2008. An overview of the current status of water quality and eutrophication in South African Rivers and Reservoirs. State of the Nation Report. Parliamentary Grant Deliverable.

Ochieng, G.M, Seanego, E.S and Nkwonta, O.I, 2010. Impacts of mining on water resources in South Africa: A review. Scientific Research and Essays Vol. 5(22), pp. 3351-3357

Ogbeibu, A.E and Oribhabor, B.J. 2001. The Ecological impact of stream regulation using benthic macroinvertebrates as indicators. Journal of Aquatic Sciences 16(2): 132 - 138

O'Hayre, A., and Amendola, F., 2010. Comments on ''A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams'' EPA/600/R-10/023A. A White Paper. Norwest Corporation. 22pp.

O'Keeffe, J.H. 1986. Ecological research on South Africans rivers- a preliminary synthesis. South African National Scientific Programmes Report No 121.

Olomukoro, J.A and Ezemonye, L.I.N 2007. Assessment of macroinvertebrate fauna of rivers in Southern Nigeria. African zoology Vol. 42 No1: 1-11

Panigrahi, S. and Patra, A.K., 2011. Studies on physic-chemical properties of River Mahanadi in the District of Cuttack, Odisha. E-planet 9(2): 50-55.

Parson, R., (2002) Impact of wastewater Treatment works on Groundwater. Water Research Commission (WRC).

Paul, M.J., and Meyer, J.L., 2001.Streams in the Urban Landscape. Annu. Rev. Ecol. Syst. 32: 333-65.

Pejman, A.H., Bidhendi, N.G.R., Karbassi, A.R., Mehrdadi, N. and Bidhendi, E.M., 2009. Evaluation of spatial and seasonal variations in surface water quality using multivariate statistical techniques. Inst. J. Environ. Sc. Tech., 6(3), 467–476.

Perry, J., and Vanderklein, E., 1996. Water Quality Management of a Natural Resource. Blackwell Science, Inc. 639pp.

Pool, E., 2008. The estrogenicity of sewage effluent entering the Eerste-kuils river Catchment system. WRC Report No 1590/1/08.

Poole, G., Risley, J. and Hicks, M. 2001. Spatial and temporal patterns of stream temperature. Prepared as Part of EPA Region 10 Temperature Water Quality Criteria Guidance Development Project. United States Envirionmental Protection Agency. EPA-910. 33p.

Rabee, A.M., Abdul-Kareem, B.M., Al-Dhamin, A.S., 2011. Seasonal variations of some Ecological parameters in Tigris River Water at Baghdad, Region, Iraq. Journal of Water Resource and Protection, 3, 262-267.

Rahman, L.A.K.M., Islam, M., Hossain, M.Z., and Ahsan, M.A., 2012. Study of the seasonal variations in Turag River water quality parameters. African Journal of Pure and Applied Chemistry. Vol. 6(10), pp. 144-148.

Ramollo, P.P., 2008. Bioassessing the impact of water quality on the health and parasite composition of Oreochromis mussambicus at the Phalaborwa industrial complex (PIC) and the barrage (Olifants River) in the Limpopo Province, South Africa. Submitted for the degree of Master of Science in Zoology. Department of Biodiversity School of Molecular and Life Sciences. Faculty of Science and Agriculture, University of Limpopo.

Rast, W., and Thornton, J., 1996. Trends in eutrophication research and control. Hydrological processes, Vol. 10. 295-313.

Reece, P.F., and Richardson, J.S., 2000. Benthic macroinvertebrate assemblages of coastal and continental streams and large rivers of southwestern British Columbia, Canada. Hydrobiologia **439:** 77–89

River Health Programme, 2003. State-of-Rivers Report: Diep, Hout Bay, Lourens and Palmiet River Systems. Department of Water Affairs and Forestry. Pretoria. ISBN No: 0-620-30757-9

River Health Programme 2005. State of Rivers Report: Greater Cape Town's Rivers. Department of Water Affairs and Forestry. Pretoria. ISBN No: 0-620-34026-6

River Health Programme. 2006. State of Rivers Report: Olifants/Doring and Sandveld Rivers. Department of Water Affairs and Forestry. Pretoria. ISBN No: 0-620-36021-6

Rivers-Moore, N.A., Hughes, D. A. & S Mantel, S., 2008. Links between water temperatures, ecological responses and flow rates: a framework for establishing water temperature guidelines for the ecological reserve. WRC Report No KV 214/08. 67pp.

Robertson, P., 2006. The Influence of Agricultural Land Use and the Mediating Effect of Riparian Vegetation on Water Quality in Jones Creek, Alberta, Canada. Enquiries Journal of Interdisciplinary Studies for High School Students. Vol. 2, No. 1. pp. 13 – 24

Ross, R.M., Long, E.S, and Dropkin, D.S., 2007. Response of macroinvertebrate communities to remediation-simulating conditions in Pennsylvania Streams influenced by acid mine drainage. Environ Monit Assess 145: 323 – 338.

Salinas, M.J., Blanca, G. and Romero, A.T 2000. Riparian vegetation and water chemistry in a basin under semiarid Mediterranean Climate, Andarax River Spain. Environ Manage. 26(5): 539 – 552

Schindler, D.W. 1974. Eutrophication and recovery in experimental Lakes: implications for Lake Management. Science, New Series, Vol.184, No.4139, pp.897-899

Schindler, D.W. 1977. Evolution of phosphorus limitation in lakes: Natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. Science, Vol.195, pp.260-262

Silveira, M. P, Buss, D. F., Nessimian, J. L. and Baptista, D. F. 2006. Spatial and temporal distribution of benthic macroinvertebrates in a South Eastern Brazilian River. Braz. J. Biol., 66(2B): 623-632.

Simmons J.A, Lawrence, E.R and Jones, T.G., (undated). Treated and Untreated Acid Mine Drainage Effects on Stream Periphyton Biomass, Leaf Decomposition, and Macroinvertebrate Diversity.

Smith-Adao, L.B. 2004. The relationship between channel discharge, hydraulic biotopes and channel morphology in the Lourens River, Western Cape, South Africa. MSc Department of Earth Sciences, University of the Western Cape

Soulsby, C., Turnbull, D., Hirst, D., Langan, S.J and Owen, R., 1997. Reversibility of stream acidification in the cairngorm region of Scotland. Journal Hydrology 195: 291-311

Souto, R.M.G., Facure, K.G., Pavanin, L.A and Jacobucci, G.B., 2011. Influence of environmental factors on benthic macroinvertebrate communities on urban streams in Vereda habitats, Central Brazil. Acta Limnologica Brasiliensia, Vol. 23, No3, p. 293-306.

Spellerberg, I.F and Fedor, P.J., 2003. A tribute to Claude Shannon (1901-2001) and a plea for more rigorous use of species richness, species diversity and the Shannon-Wiener Index. Global Ecology & Biogeography, 12: 177-179.

Statistics South Africa, 2005. Natural resource accounts: Water quality accounts for South Africa, 2000. Statistics South Africa, Private Bag X44, Pretoria 0001.

Stief, P., 2001. Influence of sediment and pore-water composition on nitrite accumulation in a nitrate-perfused freshwater sediment. Wat. Res. Vol. 35, No. 12, pp. 2811–2818.

Stoyanova, T., Traykov, I., Yaneva, I., and Bogoev, V., 2010. Ecological quality assessment of Luda River, Bulgaria. Natura Montenegrina, Podgorica, 9(3): 341-348.

Storey, R.G and Cowley, D.R, 1997. Recovery of three New Zealand rural streams as they pass through native forest remnants. Hydrobiologia **353:** 63–76.

Struyf, E., D Bal, K., Backx, H., Vrebos, D., Casteleyn, A., De Deckere1, E., Schoelynck, J., Brendonck, L., M Raitt, L., and Meire, P., 2012. Nitrogen, phosphorus and silicon in riparian ecosystems along the Berg River (South Africa): The effect of increasing human land use. Water SA Vol. 38: 597 – 606

Sundermann, A., Paul, S.U., Clarke, R.T and Haase, P. 2008. Within-stream variability of benthic macroinvertebrate samples and EU water framework directive assessment results. Fundamental and Applied Limnology Archiv für Hydrobologie. Vol. 173/1: 21 – 34.

Tachet, H; Bournaud, M et **Usseglio-Polatera, P., 2003:** Invertébrés d'eau douce. Systématique, Biologie, Ecologie. C.N.R.S Editions. 15, rue Malebranche-75005 Paris. 587p.

Theron, J and Cloete, T.E., 2002. Emerging waterborne infections: Contributing Factors, Agents, and detection Tools. Critical Reviews in Microbiology, 28(1):1-26

Thirion, C., 2007. Module E: Macroinvertebrate Response Assessment Index in River EcoClassification: Manual for EcoStatus Determination (version 2). Joint Water Research Commission and Department of Water Affairs and Forestry report. WRC Report No.

Thomas A. S., and Tris H.W., 1996. Ecological responses to reductions in freshwater supply and quality in South Africa's estuaries: lessons for management and conservation. *Journal of Coastal Conservation 2: 115-130.*

U.S Department of the interior, 2005. Quality of water Colorado River Basin. Reclamation Managing water in the West. Progress Report No22.

Van der Geest, H.H., 2007. Behavioural responses of caddisfl y larvae (Hydropsyche angustipennis) to hypoxia. Contributions to Zoology, 76 (4) 255-260 (2007).

Van der Laan, M., Van Anttwerpen, R., and Bristow, K.L., 2012. River water quality in the northern sugarcane-producing regions of South Africa and implication for irrigation: A scoping study. Water SA Vol. 38 No1.

Van Ginkel, C.E., 2011. Eutrophication: Present reality and future challenges for South Africa. Water SA Vol. 37 No. 5: 693 - 702

Vannote, R.L., Minshall, G.W., Cummins K.W., Sedell J.R. and Cushing C.E. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences, 37: 130 – 137.

Vannote, R.L and Sweeney, B.W., 1980. Georgraphic analysis of thermal equilibria: A conceptual model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. The American Naturalist. Vol.115, No.5: 667 – 695

Vhevha, I., Ndamba, J., and Mtetwa, S. 2000. Changes in river water quality with increasing distance from site of wastewater use. 1st WARFSA/WaterNet Symposium: sustainable use of Water Resources.

Virbickas, T., Pliuraite, V. and Kesminas, V. 2011. Agricultural Land use on Macroinvertebrate Fauna in Lithuani. Pol. J. Environ. Stud. Vol. 20, No 5, 1327-1334.

Voelz N. J., Zuellig R.E, Shieh, S-H, and Ward, J.V. 2005. The effects on urban areas on Benthic Macroinvertebrates in Two Colorado Rivers. Environmental Monitoring and Assessment.101: 175-202

Wade, K.R., 0rmerod, S.J. and Gee, A.S., 1989. Classification and ordination of macroinvertebrate assemblages to predict stream acidity in upland Wales. Hydrobiologia 171: 59-78

Wang, L. and Kanehl, P. 2003. Influences of watershed urbanization and instream habitat on macroinvertebrates in cold water streams. Journal of the American Water Resources Association, 39: 1181 – 1196.

Water and River Commission (WRC), 2000. Water notes. The values of Riparian. Water and Rivers Commission River Restoration Report No. RR 2.

Weatherleyl, N.S., Ormerod, S.J., Thomas, S.P. and Edwards, R.W. 1988. The response of macroinvertebrates to experimental episodes of low pH with different forms of aluminium, during a natural spate. Hydrobiologia 169: 225-232

Winter, J.G and Duthie, H.C. 1998. Effects of urbanization on water quality, periphyton and invertebrates communities in Southern Ontario Stream. Canadian Water Resources Journal Vol. 23, No3.

Zamxaka, M., Pironcheva, G and Muyima Nyo 2004. Microbiological and Physico-chemical assessment of the quality of domestic water sources in selected rural communities of the Eastern Cape Province, South Africa. Water SA Vol. 30 No3: 333-340

Ziglio, G., Siligardi, M. and Flaim, G., 2006. Biological Monitoring of Rivers. Applications and persepectives. John Wiley & Sons, Ltd.

Zweig, L. D., and Rabeni, C. F., 2001. Biomonitoring for deposited sediment using benthic invertebrates: a test on four Missouri Streams. J. N. Am. Benthol. Soc. 20(4): 643-657

Appendices 1. Physical and chemical parameters recorded weekly at different sites

1st Week: 4th September 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat (%)	TDS (mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	17.5	8.15	14.68	154.3	864	0.67	ND	ND
K4	15.5	7.68	9.9	100.5	825.5	0.64	ND	ND
K3	16.5	8.41	14.66	151	916.5	0.71	ND	ND
K2	13.5	7.81	11.66	112.1	780	0.61	ND	ND
K1	12.3	7.56	11.46	106.7	786.5	0.61	ND	ND

2nd Week: 11th September 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	16.2	8.17	10.22	103.3	767	0.59	ND	ND
K4	15.4	7.66	8.63	86.8	680.5	0.48	ND	ND
K3	14.7	7.94	9.74	96.5	747	0.58	ND	ND
K2	14	7.75	9.82	95.6	604.5	0.46	ND	ND
K1	13.9	7.63	9.55	92.9	500.98	0.46	ND	ND

3rd Week: 18th September 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	20.4	8.14	8.25	91.5	617	0.47	0.5	0.24
K4	19.2	7.42	3.46	37.5	715	0.55	0.6	1.33
K3	18.8	7.94	8.33	89.5	520	0.40	0.6	0.24
K2	17.1	7.52	7.67	79.7	520	0.39	2.1	0.48
K1	16.1	7.58	8.74	88.4	728	0.56	1.5	0.28

4th Week: 24th September 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	19.4	8.1	8.98	98.3	734	0.56	1	0.31
K4	16.7	7.76	8.09	83.7	676	0.52	0.6	1.55
K3	16.9	7.95	8.81	91.3	702	0.54	0.8	1.01
K2	15	7.76	9.3	91.6	500.5	0.38	1.1	0.33
K1	14.9	7.60	9.1	90.6	494	0.38	1.4	0.39

5th Week: 2nd October 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	21.1	8.14	9.4	106	780	0.61	1.4	0.2
K4	18.6	7.92	9.6	106.2	591.5	0.45	0.6	1.48
K3	19.4	8.02	8.6	94.2	754	0.58	1	0.51
K2	16.6	7.81	8.7	92.2	689	0.53	1.9	0.49
K1	15.9	7.66	8.8	89.4	682.5	0.52	1.8	0.40

6th Week: 9th October 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	18.38	8.48	9.63	103	825	0.64	1.3	0.11
K4	15.6	8.08	8.55	86.1	572	0.44	0.9	1.32
K3	18.1	8.36	9.27	99.6	806	0.62	1.3	1.34
K2	17	8.18	9.96	103.6	663	0.51	1.4	0.39
K1	16.8	8.01	9.69	99.6	656.5	0.50	1.6	0.33

7th Week: 16th October 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	25.1	8.76	10.69	128.5	799	0.61	0.1	0.24
K4	24.6	7.77	4.31	51.1	754	0.57	0.6	1.4
K3	22.1	8.54	11.12	127.7	812.5	0.62	0.8	0.37
K2	18.6	8.13	9.73	104.2	695.5	0.53	2.2	0.9
K1	19.1	7.95	9.2	98.5	578	0.44	1.2	1.23

8th Week: 23rd October 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	24.4	8.76	10.2	122.6	786	0.6	0.7	0.25
K4	24.3	7.61	1.94	23.4	660	0.49	0.5	2.36
K3	22.2	8.48	9.34	107.7	760	0.58	0.5	0.3
K2	17.8	7.95	8.73	92.4	513	0.39	2.1	0.67
K1	17.1	7.85	8.66	89.9	416	0.31	1.4	0.61

9th Week: 30th October 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	26.9	8.55	8.65	110.2	747	0.57	0.4	0.28
K4	22.7	7.49	2.02	23.2	903	0.70	0.00	2.4
K3	25.7	8.46	10.89	125	767	0.59	0.9	0.28
K2	20	7.95	11.72	129.3	676	0.52	1.9	0.14
K1	19.8	7.62	12.84	140	552.5	0.42	3.1	2.28

10th Week: 6th November 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	27.1	8.66	10.66	133.6	780	0.59	0.7	0.99
K4	25.7	7.54	1.63	20.4	676	0.51	0.1	2.6
K3	26	8.61	9.8	123.4	773.5	0.59	0.7	0.62
K2	20.5	8.02	9.73	108.9	703.5	0.54	1.8	0.78
K1	20.5	7.82	8.77	96.5	559	0.42	1.9	3.52

11th Week: 13th November 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	22.2	7.73	7.01	81.2	669	0.51	0.8	0.35
K4	20.2	7.3	0.28	3.2	764	0.58	0.8	3.63
K3	21.6	7.73	7.33	84.15	663	0.51	1	0.51
K2	19.4	7.74	7.81	84.7	669.5	0.51	1.5	0.9
K1	19.4	7.56	6.77	73.8	604.5	0.46	2.3	2.49

12th Week: 20th November 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	21.9	8.1	7.66	88.7	702	0.53	0.8	0.45
K4	20.7	7.55	0.07	0.8	910	0.7	0.0	3.5
K3	21.6	8.04	7.76	88.8	708.5	0.54	0.8	0.62
K2	20.2	7.7	7.16	79.7	676	0.51	1.5	0.88
K1	20.5	7.55	7.19	80.4	747.5	0.57	1.5	5.27

13th Week: 27th November 2012

Sites	T(°C)	pH	DO(mgL^{-})	Oxysat(%)	TDS(mgL^{-})	Salinity (mgL^{-1})	NO_3^- (mgL^{-1})	PO_4^{3-} (mgL^{-1})
K5	23	8.16	7.3	86	650	0.5	0.5	0.43
K4	18.3	7.13	0.31	34	832	0.64	0.0	2.26
K3	20.9	7.92	7.47	84.6	604.5	0.46	0.7	0.44
K2	18.6	7.7	7.68	82.6	633.5	0.48	1.3	0.60
K1	19.1	7.52	4.26	46.3	643.5	0.49	0.9	0.98

Appendices 2 List of macroinvertebrate collected per week at different sites

18th September 2012

Taxa	K1	K2	K3	K4	K5
Oligochaeta	29	18	22		3
Hirudina	10		4		
CRUSTACEA					
Potamonautidae		2	5		4
EPHEMEROPTERA					
Baetidae		1	1		
ODONATA					
Coenagrionidae			5		3
Aeshnidae			3		
Libellulidae	1	1	2		
HEMIPTERA					
Corixidae	1		5	9	1
DIPTERA					
Ceratopogonidae			2		
Chironomidae	100	31	29	28	32
Simuliidae			11	41	1
GASTROPODA					
Physidae	82	80	5	4	15
Abundance	**223**	**133**	**94**	**82**	**59**
Specific richness	6	6	12	4	7
Diversity	1,18	1,05	2,032	1,1	1,3

24th September 2012

	K1	K2	K3	K4	K5
Oligochaeta	25	58	31	3	14
Hirudina	2	1	3		3
CRUSTACEA					
Potamonautidae		3	11		4
ODONATA					
Coenagrionidae			1		2
Libellulidae			2		2
HEMIPTERA					
Corixidae				4	
COLEOPTERA					
Dytiscidae			2	2	4
DIPTERA					
Ceratopogonidae		3			
Chironomidae	5	27	34	23	14
Simuliidae			42	53	10
Tipulidae				1	
GASTROPODA					
Physidae	10	61	9	2	5
Abundance	**42**	**153**	**135**	**88**	**58**
Specific richness	**4**	**6**	**9**	**7**	**9**
Diversity	1.04	1.22	1.67	1.13	1.95

2nd October 2012

Taxa	K1	K2	K3	K4	K5
Turbellaria		1			
Oligochaeta	27	36	17	3	7
Hirudina	1				2
CRUSTACEA					
Potamonautidae		1	5	1	3
ODONATA					
Coenagrionidae			2		1
Libellulidae			2		
HEMIPTERA					
Naucoridae			1		
Corixidae			2	2	2
COLEOPTERA					
Dytiscidae				1	
DIPTERA					
Chironomidae	20	2	17	21	8
Simuliidae			36	87	23
Stratiomidae				1	
GASTROPODA					
Physidae	33	38	7	11	9
Abundance	**81**	**78**	**89**	**127**	**55**
Specific richness	4	5	9	8	8
Diversity	1.13	0.90	1.65	1.02	1.66

9th October 2012

	K1	K2	K3	K4	K5
Oligochaeta	12	9	17	2	10
Hirudina	8		6		1
CRUSTACEA					
Potamonautidae	1	1	4		
ODONATA					
Coenagrionidae	1		3		1
Chlorocophydae			1		
Libellulidae			2		1
HEMIPTERA					
Corixidae			1	15	
EPHEMEROPTERA					
Baetidae			1		
COLEOPTERA					
Dytiscidae				1	
DIPTERA					
Ceratopogonidae			3		3
Chironomidae	3	7	52	83	49
Simuliidae			117	177	20
Tipulidae				1	
GASTROPODA					
Physidae	91	5	2	21	7
Limnaeidae					1
Abundance	**116**	**22**	**209**	**300**	**93**
Specific richness	6	4	12	7	9
Diversity	0.77	1.20	1.31	1.07	1.39

16th October 2012

	K1	K2	K3	K4	K5
Oligochaeta	18	12	84	6	31
Hirudina	2	1	10	2	5
CRUSTACEA					
Potamonautidae	2	1	5	1	3
ODONATA					
Coenagrionidae			3		
Aeschnidae			1		
Libellulidae				1	
HEMIPTERA					
Corixidae			1	64	
EPHEMEROPTERA					
Baetidae		5	4	1	5
COLEOPTERA					
Dytiscidae			1	8	1
DIPTERA					
Ceratopogonidae			1		6
Chironomidae	3		27	101	46
Simuliidae		1	28	24	11
GASTROPODA					
Physidae	132	115	5	11	12
Limnaeidae					5
Abundance	**157**	**135**	**170**	**219**	**125**
Specific richness	5	6	12	10	**10**
Diversity	0.54	0.57	1.56	1.43	1.80

23rd October 2012

	K1	K2	K3	K4	K5
Oligochaeta	5	12	6	3	12
Hirudina	3	2	4		4
CRUSTACEA					
Potamonautidae		10	4		7
ODONATA					
Coenagrionidae			1		1
Libellulidae			6		
HEMIPTERA					
Corixidae				11	
Notonectidae					1
EPHEMEROPTERA					
Baetidae		1	3	1	18
COLEOPTERA					
Dytiscidae		1	6	1	3
DIPTERA					
Ceratopogonidae		4	3		2
Chironomidae			21	64	15
Simuliidae			24	54	23
Tipuliidae				1	1
GASTROPODA					
Physidae	102	104	4	4	12
Limnaeidae					1
Abundance	**110**	**134**	**79**	**139**	**100**
Specific richness	3	7	11	8	**13**
Diversity	0.30	0.83	1.91	1.2	2.11

30th October 2013

	K1	K2	K3	K4	K5
Oligochaeta	5	1	3	1	9
Hirudina	2		3		
CRUSTACEA					
Potamonautidae		2	1		3
ODONATA					
Coenagrionidae	1		1		
Libellulidae					
HEMIPTERA					
Corixidae				1	
EPHEMEROPTERA					
Baetidae	1		9	1	15
COLEOPTERA					
Dytiscidae			3	2	2
Hydrophilidae			1		
DIPTERA					
Ceratopogonidae			3		14
Chironomidae		4	32	100	16
Simuliidae		2	62	4	62
GASTROPODA					
Physidae	100	48	2	4	15
Abundance	**109**	**57**	**120**	**113**	**136**
Specific richness	5	5	11	7	**8**
Diversity	0.37	0.63	1.44	0.49	1.65

6th November 2012

	K1	K2	K3	K4	K5
Hydrachnella			1		1
Oligochaeta	10	21	12	2	19
Hirudina	19	3	7	1	18
CRUSTACEA					
Potamonautidae	2		1		1
ODONATA					
Aeshnidae					1
Coenagrionidae			2		3
Libellulidae					1
HEMIPTERA					
Corixidae				12	
Notonectidae			1		
EPHEMEROPTERA					
Baetidae			41		37
COLEOPTERA					
Dytiscidae			4	2	9
Hydrophilidae			2	1	
DIPTERA					
Ceratopogonidae					5
Chironomidae	2	4	71	46	109
Ephydridae				1	2
Simuliidae			43	41	155
Tipuliidae				1	
GASTROPODA					
Physidae	103	200	7	12	78
Limnaeidae	1		2		4
Planorbidae				1	
Abundance	**137**	**228**	**194**	**120**	**443**
Specific richness	6	4	13	11	**15**
Diversity	0.83	0.45	1.73	1.52	1.75

13th November 2012

	K1	K2	K3	K4	K5
Oligochaeta	3	27	12		2
Hirudina	9	2	13		17
CRUSTACEA					
Potamonautidae	1	2	2		4
ODONATA					
Coenagrionidae	1				
HEMIPTERA					
Corixidae			1	23	
Notonectidae					1
Veliidae			1		
EPHEMEROPTERA					
Baetidae			63		40
COLEOPTERA					
Dytiscidae	1		3	3	25
Hydrophilidae			4		1
DIPTERA					
Ceratopogonidae			2		
Chironomidae			92	13	34
Ephydridae		1	6		2
Simuliidae			128		205
Syrphidae		1			1
Tipuliidae				1	1
GASTROPODA					
Physidae	116	117	18	12	39
Limnaeidae			2	1	3
Abundance	**131**	**150**	**347**	**53**	**375**
Specific richness	6	6	14	6	**14**
Diversity	**0.48**	**0.67**	**1.67**	**1.35**	**1.52**

20th November 2012

	K1	K2	K3	K4	K5
Oligochaeta	23	7	27		12
Hirudina	44	21	11		11
CRUSTACEA					
Potamonautidae		2	3		2
ODONATA					
Coenagrionidae			1		
HEMIPTERA					
Corixidae			1		2
Gerridae			1		
Notonectidae		1	1		
EPHEMEROPTERA					
Baetidae			20		135
COLEOPTERA					
Dytiscidae			1	1	5
Hydrophilidae			1		1
DIPTERA					
Ceratopogonidae					3
Chironomidae		2	56	25	30
Ephydridae					1
Simuliidae		1	59		102
Syrphidae			1		
Tipuliidae				1	
GASTROPODA					
Physidae	103	240	3	4	22
Limnaeidae					3
Abundance	**170**	**274**	**186**	**31**	**329**
Specific richness	3	7	14	4	**13**
Diversity	**0.92**	**0.50**	**1.72**	**0.66**	**1.60**

27th November 2012

	K1	K2	K3	K4	K5
Oligochaeta	13	9	10	12	16
Hirudina	90	4	7		12
CRUSTACEA					
Potamonautidae	1		2		2
ODONATA					
Coenagrionidae			4		2
Libellulidae		1	1		2
HEMIPTERA					
Belostomatidae					1
EPHEMEROPTERA					
Baetidae			82		86
COLEOPTERA					
Dytiscidae		1	2		2
Hydrophilidae			1		1
DIPTERA					
Chironomidae		5	25	10	26
Ephydridae			1		1
Simuliidae			76		111
Syrphidae					1
Tipuliidae					2
GASTROPODA					
Physidae	156	270	18	6	28
Limnaeidae					1
TURBELLARIA					
Dugesiidae		1			
Abundance	**260**	**291**	**229**	**28**	**294**
Specific richness	4	7	12	3	**16**
Diversity	0.84	0.35	1.62	1.06	1.68

Appendix 3: Statistic test

ANOVA Table for Temperature

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Sites	4	188.688	47.172	4.837	.0019	19.347	.950
T°C	Residual	60	585.157	9.753				
	Sites	4	4.831	1.208	19.482	<.0001	77.926	1.000
pH	Residual	60	3.720	.062				
	Sites	4	236.252	59.063	10.138	<.0001	40.553	1.000
DO	Residual	60	349.546	5.826				
	Sites	4	205941	51485.316	5.652	.0006	22.609	.977
TDS	Residual	60	546531	9108.855				
	Sites	4	.114	.029	5.004	.0015	20.015	.957
Salinity	Residual	60	.342	.006				
	Sites	4	20.703	5.176	14.396	<.0001	57.586	1.000
NO_3	Residual	49	17.616	.360				
	Sites	4	28.796	7.199	10.503	<.0001	42.011	1.000
PO_4	Residual	49	33.586	.685				
	Sites	4	7986.291	1996.573	30.028	<.0001	120.111	1.000
SASS	Residual	50	3324.545	66.491				
	Sites	4	454.545	113.636	26.371	<.0001	105.485	1.000
NoT	Residual	50	215.455	4.309				
	Sites	4	9.315	2.329	21.500	<.0001	85.998	1.000
ASPT	Residual	50	5.416	.108				
	Sites	4	9.186	2.296	31.207	<.0001	124.829	1.000
H'	Residual	50	3.679	.074				

Difference between weeks

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Date	12	511.377	42.615	8.443	<.0001	101.314	1.000
T°C	Residual	52	262.468	5.047				
	Date	12	2.533	.211	1.824	.0683	21.890	.823
pH	Residual	52	6.017	.116				
	Date	12	218.646	18.220	2.581	.0091	30.967	.951
DO	Residual	52	367.152	7.061				
	Date	12	212665.207	17722.101	1.707	.0921	20.486	.789
TDS	Residual	52	539807.360	10380.911				
	Date	12	.136	.011	1.833	.0668	21.990	.825
Salinity	Residual	52	.321	.006				
	Date	10	13.570	1.357	1.208	.3124	12.079	.536
PO_4	Residual	44	49.430	1.123				
	Date	10	1.988	.199	.415	.9318	4.153	.185
NO_3	Residual	44	21.060	.479				
	Date	10	794.836	79.484	.333	.9675	3.326	.153
SASS	Residual	44	10516.000	239.000				
	Date	10	48.182	4.818	.344	.9635	3.437	.157
NoT	Residual	44	616.800	14.018				
	Date	10	.786	.079	.260	.9868	2.595	.127
ASPT	Residual	44	13.333	.303				
	Date	10	.905	.090	.333	.9674	3.328	.153
H'	Residual	44	11.963	.272				

Difference between sites

	T°C P-value	pH P-value	DO P-value	TDS P-value	salinity P-value	NO P-val	PO P-value	SASS P-value	NoT P-val	ASPT P-value	H' P-value
K1-K2	.8561	.1017	.7093	.4440	.5700	.9972 S	.0025 S	.3128	.2235	.2874	.9875
K1-K3	.0170 S	<.0001 S	.5150	.0018 S	.0033 S	.0015 S	.0020 S	<.0001S	<.0001S	<.0001S	<.0001S
K1-K4	.0483 S	.4340	<.0001 S	.0016 S	.0047 S	<.0001 S	.2553	.0051 S	.0172 S	<.0001S	.0062 S
K1-K5	.0005 S	<.0001 S	.5042	.0006 S	.0012 S	.0006 S	.0003 S	<.0001S	<.0001S	<.0001S	<.0001S
K2-K3	.0266 S	.0010 S	.7802	.0157 S	.0155 S	.0012 S	.9347	<.0001S	<.0001S	<.0001S	<.0001S
K2-K4	.0717	.0172 S	<.0001 S	.0137 S	.0214 S	<.0001 S	<.0001S	.0621	.2235	.0011 S	.0059 S
K2-K5	.0010 S	<.0001S	.7672	.0057 S	.0063 S	.0004 S	.4886	<.0001S	<.0001S	<.0001S	<.0001S
K3-K4	.6618	<.0001S	<.0001 S	.9592	.8972	.0059 S	<.0001S	<.0001S	<.0001S	.0021 S	<.0001S
K3-K5	.2355	.2421	.9864	.7044	.7369	.7503	.5411	.1720	.4755	.0628	.9377
K4-K5	.1067	<.0001 S	<.0001 S	.7421	.6420	.0136 S	<.0001S	<.0001S	<.0001S	.1679	<.0001S

Difference between months

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Months	2	303.312	151.656	19.983	<.0001	39.966	1.000
T°C	Residual	62	470.533	7.589				
	Months	2	1.263	.632	5.373	.0071	10.745	.834
pH	Residual	62	7.288	.118				
	Months	2	117.519	58.759	7.780	.0010	15.559	.954
DO	Residual	62	468.279	7.553				
	Months	2	2726.102	1363.051	.113	.8936	.225	.066
TDS	Residual	62	749746.466	12092.685				
	Months	2	3.594	1.797	.024	.9759	.049	.053
Salinity	Residual	62	.456	.007				
	Months	2	9.158	4.579	4.422	.0168	8.845	.740
PO_4	Residual	52	53.842	1.035				
	Months	2	.506	.253	.584	.5612	1.168	.138
NO_3	Residual	52	22.542	.433				
	Months	2	338.676	169.338	.803	.4537	1.605	.174
SASS	Residual	52	10972.160	211.003				
	Months	2	28.142	14.071	1.149	.3249	2.298	.233
NoT	Residual	52	636.840	12.247				
	Months	2	.106	.053	.196	.8227	.392	.078
ASPT	Residual	52	14.014	.270				
	Months	2	.381	.190	.792	.4582	1.585	.172
H'	Residual	52	12.488	.240				

Difference between months

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Months	2	303.312	151.656	19.983	<.0001	39.966	1.000
T°C	Residual	62	470.533	7.589				
	Months	2	1.263	.632	5.373	.0071	10.745	.834
pH	Residual	62	7.288	.118				
	Months	2	117.519	58.759	7.780	.0010	15.559	.954
DO	Residual	62	468.279	7.553				
	Months	2	2726.102	1363.051	.113	.8936	.225	.066
TDS	Residual	62	749746.466	12092.685				
	Months	2	3.594	1.797	.024	.9759	.049	.053
Salinity	Residual	62	.456	.007				
	Months	2	9.158	4.579	4.422	.0168	8.845	.740
PO_4	Residual	52	53.842	1.035				
	Months	2	.506	.253	.584	.5612	1.168	.138
NO_3	Residual	52	22.542	.433				
	Months	2	338.676	169.338	.803	.4537	1.605	.174
SASS	Residual	52	10972.160	211.003				
	Months	2	28.142	14.071	1.149	.3249	2.298	.233
NoT	Residual	52	636.840	12.247				
	Months	2	.106	.053	.196	.8227	.392	.078
ASPT	Residual	52	14.014	.270				
	Months	2	.381	.190	.792	.4582	1.585	.172
H'	Residual	52	12.488	.240				

Difference between years (1989 to 2012)

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Years	7	73.466	10.495	1.103	.4073	7.719	.327
T°C	Residual	16	152.286	9.518				
	Years	7	1.009	.144	4.252	.0079	29.766	.926
pH	Residual	16	.542	.034				
	Years	7	31.226	4.461	1.815	.1531	12.704	.531
DO	Residual	16	39.327	2.458				
	Years	7	2.988	.427	2.611	.0531	18.275	.715
PO_4	Residual	16	2.616	.164				

Difference between months from 1989 to 2012

		DF	Sum of square	Mean of square	F-value	P-value	Lambda	Power
	Months	2	91.543	45.772	7.162	.0043	14.324	.903
T°C	Residual	21	134.208	6.391				
	Months	2	.002	.001	.014	.9859	.028	.052
pH	Residual	21	1.550	.074				
	Months	2	16.561	8.280	3.221	.0603	6.441	.544
DO	Residual	21	53.992	2.571				
	Months	2	.191	.095	.571	.5737	1.141	.129
PO	Residual	21	3.513	.167				
	Months	2	.417	.208	.843	.4445	1.686	.170
NO	Residual	21	5.188	.247				

Difference between one year and another

Difference between Years	Temperature			pH			DO			Nitrate			Phosphate		
	Mean diff.	Crit. Diff.	P-value	Mean diff.	Crit. Diff.	P-value	Mean diff.	Crit. Diff.	P-value	Mean diff.	Crit. Diff.	P-value	Mean diff.	Crit. Diff.	P-value
1989-1992	-2.683	5.340	.3026	-.117	.319	.4491	1.951	2.714	.1459	.123	.700	.7138	.293	.542	.2679
1989-1995	1.850	5.340	.4733	.067	.319	.6634	1.673	2.714	.2096	-.077	.700	.8194	.327	.542	.2194
1989-1999	1.583	5.340	.5385	-.033	.319	.8273	1.390	2.714	.2936	.338	.700	.3210	.386	.542	.1504
1989-2000	.983	5.340	.7014	-.233	.319	.1402	1.207	2.714	.3599	.880	.700	.0170 S	.513	.542	.0619
1989-2001	2.150	5.340	.4060	.067	.319	.6634	-.477	2.714	.7145	.287	.700	.3983	.465	.542	.0876
1989-2002	2.400	5.340	.3549	.450	.319	.0086 S	3.440	2.714	.0162 S	.907	.700	.0143 S	.484	.542	.0763
1989-2012	-1.643	5.340	.5234	-.237	.319	.1350	.640	2.714	.6239	.117	.700	.7286	-.423	.542	.1171
1992-1995	4.533	5.340	.0908	.183	.319	.2403	-.283	2.714	.8276	-.200	.700	.5534	.033	.542	.8979
1992-1999	4.267	5.340	.1097	.083	.319	.5870	-.567	2.714	.6639	.215	.700	.5244	.093	.542	.7217
1992-2000	3.667	5.340	.1648	-.117	.319	.4491	-.750	2.714	.5661	.757	.700	.0359 S	.220	.542	.4027
1992-2001	4.833	5.340	.0730	.183	.319	.2403	-2.433	2.714	.0755	.163	.700	.6277	.172	.542	.5113
1992-2002	5.083	5.340	.0607	.567	.319	.0017 S	1.483	2.714	.2636	.784	.700	.0305 S	.191	.542	.4657
1992-2012	1.040	5.340	.6852	-.120	.319	.4365	-1.317	2.714	.3190	-.007	.700	.9841	-.717	.542	.0127 S
1995-1999	-.267	5.340	.9170	-.100	.319	.5154	-.283	2.714	.8276	.415	.700	.2270	.059	.542	.8194
1995-2000	-867	5.340	.7353	-.300	.319	.0633	-.467	2.714	.7202	.957	.700	.0105 S	.186	.542	.4765
1995-2001	.300	5.340	.9067	0.000	.319	.	-2.150	2.714	.1125	.363	.700	.2876	.138	.542	.5958
1995-2002	.550	5.340	.8299	.383	.319	.0214 S	1.767	2.714	.1865	.984	.700	.0089 S	.158	.542	.5460
1995-2012	-3.493	5.340	.1845	-.303	.319	.0607	-1.033	2.714	.4314	.193	.700	.5665	-.750	.542	.0097 S
1999-2000	-.600	5.340	.8148	-.200	.319	.2021	-.183	2.714	.8879	.542	.700	.1206	.127	.542	.6260
1999-2001	.567	5.340	.8249	.100	.319	.5154	-1.867	2.714	.1641	-.052	.700	.8777	.079	.542	.7612
1999-2002	.817	5.340	.7500	.483	.319	.0054 S	2.050	2.714	.1288	.569	.700	.1042	.098	.542	.7055
1999-2012	-3.227	5.340	.2185	-.203	.319	.1950	-.750	2.714	.5661	-.222	.700	.5118	-.809	.542	.0060 S
2000-2001	1.167	5.340	.6495	.300	.319	.0633	-1.683	2.714	.2070	-.593	.700	.0914	-.048	.542	.8534
2000-2002	1.417	5.340	.5816	.683	.319	.0003 S	2.233	2.714	.1002	.027	.700	.9351	0.029	.542	.9121
2000-2012	-2.627	5.340	.3126	-.003	.319	.9826	-.567	2.714	.6639	-.763	.700	.0345 S	-.936	.542	.0021 S
2001-2002	.250	5.340	.9222	.383	.319	.0214 S	3.917	2.714	.0075 S	.621	.700	.0786	.019	.542	.9406
2001-2012	-3.793	5.340	.1516	-.303	.319	.0607	1.117	2.714	.3959	-.170	.700	.6138	-.888	.542	.0031 S
2002-2012	-4.043	5.340	.1280	-.687	.319	.0003 S	-2.800	2.714	.0439 S	-.791	.700	.0293 S	-.908	.542	.0027 S