Effects of catchment management on physical river condition, chemistry, hydrogeomorphology and ecosystem service provision in small coastal rivers of the Western Cape

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Abstract

River systems are by nature complex and dynamic systems, which vary in structure and therefore function, and are closely connected to their landscapes. The primary aim of this thesis was to develop a systems operational understanding of how river patterns and processes (geomorphology and hydrology) link to aquatic and riparian systems and biodiversity (ecology) in a framework of evolving land cover/use and management. This illustrated the hydrogeomorphic controls regulating the structure and functioning of rivers in the provision of goods and services that vegetation, especially riparian vegetation, perform as ecological infrastructure, with a focus on the Duiwe River catchment.

This study used a combination of desktop and field analysis. The desktop analysis followed the spatial and temporal historical land use change detection of river sub-catchments to assess the influence on water quality and river flow. It included historical water quality, flow records, rainfall data and aerial photograph time series analysis for trend detection, which were linked to changes in land use activities. The field surveys included cross-section surveys, physical and chemical sediment analysis, vegetation distribution, ground-water depth surveys and instream biological surveys of aquatic bioindicators.

The study illustrated a correlation between land cover/use, water quality and river ecological integrity. When spatial heterogeneity of the catchments was altered by human or natural events, it was reflected by changes in the water quality. The linkages between the land cover/use and ecological integrity were examined using macroinvertebrates and algae. Macroinvertebrates were indicative of habitat integrity and river condition, while the benthic filamentous algae were indicative of increased nutrients and alkalinity. Results indicated that the full consortium of algae and macroinvertebrates be used as bioindicators for ecological integrity assessments in these short, coastal rivers.

The influence of riparian vegetation and its effectiveness in providing regulating (retaining sediment and nutrients) and provisioning (good water quality for humans and the aquatic environment) services was examined by relating contrasting land uses, riparian vegetation, nutrient dynamics and water quality. The land covers generated different runoff volumes, water quality parameter concentrations and associated nutrient loads. Agriculture and alien *Acacia mearnsii* trees had the greatest impact on nutrient loads. However, a decreasing trend in nutrient concentrations was observed in the cross-section from the pastures to the riparian zones to the river at all sites.

The key findings from this study were formulated into a conceptual framework flow-chain model demonstrating the linkages between river pattern, processes and ecology in the provision of ecosystem services. This interdisciplinary investigation demonstrated strong links between climate, topography, hydrogeomorphology, land cover/use, human activities and their influence on ecological river integrity. The developed framework provides a hierarchical model to link the different disciplines. It illustrates the top-down constraints provided by the system

controllers and habitat drivers, coupled with the anthropogenic impacts as controllers to determine the response of biological entities (riparian vegetation and aquatic biota) at different scales, to ultimately provide ecosystem services. It provides the basis for an understanding of the linkages, processes and interactions that allows, prevents or alters ecosystem service provision by river ecosystems and in the study context, by riparian buffer zones.

Keywords

Land cover

Land use

Water quality

Riparian morphodynamics

Spatial analysis

Ecological integrity

Benthic algae

Macroinvertebrates

Surface runoff

Nutrient loads

Declaration

I declare that the thesis, entitled ***'Effects of catchment management on physical river condition, chemistry, hydrogeomorphology and ecosystem service provision in small coastal rivers of the Western Cape'*** is my own work, that it has not been submitted for any degree or examination in any other university, and that all the sources I have used or quoted have been indicated and acknowledged by complete references.

Chantel R. Petersen March 2019

Signature: … ……………………

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List of Acronyms

ANOSIM	Analysis of Similarities
ANOVA	Analysis of Variance
AQ/MV	Marginal and Aquatic Vegetation
ARC	Agricultural Research Council
ASPT	Average Score Per Taxon
CGA	Centre for Geographical Analysis
CID	Charge Injection Device
CHD	Chemodetector
CSIR	Council for Scientific and Industrial Research
COD	Chemical Oxygen Demand
dbRDA	Distance-based Redundancy Analysis
Diss/SD	Dissimilarity Coefficient/Standard Deviation
DISTLM	Distance-based Linear Modelling
DLA-CDSM	Department of Land Affairs Chief Directorate: Surveys and Mapping
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
ERT	Electrical Resistivity Tomography
FFGs	Functional Feeding Groups
FIA	Flow Injection Analysis
GEF	Global Environment Fund
GPS	Global Positioning System
GRI	Garden Route Imitative
GSM	Gavel-Sand-Mud
ICP OES	Inductively Coupled Plasma Optical Emission Spectrometry
IHAS	Invertebrate Habitat Assessment System
mamsl	meters above mean sea level
MAR	Mean annual runoff

MDS	Multi-Dimensional Scaling
NLC	National Land Cover
PCA	Principal Component Analysis
ProEcoServ-SA	Project for Ecosystem Services- South Africa
SASS	South African Scoring System
SANBI	South African National Biodiversity Institute
SAWS	South African Weather Service
SIC	Stones-In-Current,
Sim/SD	Similarity Coefficient/Standard Deviation
SIMPER	Similarity Percentages
SIMPROF	Similarity Profile
SOOC	Stones-Out-of-Current
SUDEM	Stellenbosch University Digital Elevation Model
TMG	Table Mountain Sandstone
TWQR	Target Water Quality Ranges

Chapter 1: Introduction

1.1 Background

River systems are by nature complex and dynamic systems, which vary in structure (pattern) and therefore function (process), spatially and temporally. Since catchment controls such as geology, soils, climate, hydrology, land cover and land use management control the processes of erosion, deposition and storage of sediment in various areas in a catchment, as well as runoff and sediment supply regimes in rivers, the river structure and function are intimately linked to the catchment of which they form a part (Rowntree and Wadeson, 1999, Thorp et al., 2006). Harvey (2002) refers to this as the coupling concept which can occur at different scales; the local scale where within-hillslope coupling occurs, hillslope to channel coupling, within-channel coupling, tributary junction and reach to reach coupling. The larger scale coupling occurs between major zones of a system (zonal coupling) or at regions with relation to entire catchments (regional coupling) (Harvey, 2002). River systems are therefore connected to their landscapes laterally (e.g. overbank flows on floodplains), longitudinally (e.g. inputs from riparian zones or upstream-downstream material transport), vertically (e.g. hypophoreic zone or surface-groundwater interactions) or in variation over multiple time scales (Ward, 1989). In order to understand the structure and function of natural or altered rivers a holistic ecosystem interdisciplinary understanding is required of system components and the interactions between them (Ward and Tockner, 2001).

This requires an operational understanding of how river pattern and process (geomorphology and hydrology) link to aquatic ecosystems, riparian vegetation and biodiversity (ecology) in a framework of evolving land cover/use and management. Such an interdisciplinary approach will provide insight into and allow predictions of the response of patterns to process and the influences of patterns on process (Dollar et al., 2007), which will further knowledge on appropriate river management, restoration measures and on the provision of services provided by river ecosystems. Dollar et al. (2007) describes such an interdisciplinary approach in a multi-level (hierarchical) flow-chain model, which consists of the driver (abiotic/biotic agent of change), the template or substrate which the driver acts on, controllers of the driver or agent of change and a process or entity that responds to the driver. In their model water and sediment act as agents of change and the river template represents the initial state for process and pattern. Controllers can include geomorphological or biological components such as vegetation and the responders (biological entities) provide the links to the ecological subsystem. The scale in the interaction process was identified as an important specification since a responder at one scale may become a controller at another (Dollar et al., 2007).

This thesis contributes to a systems understanding, based on interdisciplinary river data and other knowledge gaps identified in section 1.3. This was achieved by investigating the relations between hydrogeomorphic controls on river and riparian ecosystems, the biophysical interactions, linkages and drivers of change to water quality and quantity and ecological river integrity in the Touws and Duiwe River catchments, South Africa.

This chapter provides an overview of the elements regulating the structure and functioning of rivers, which allows the provision of ecosystem services. It then outlines some knowledge gaps that were used to inform the research aim and objectives of this thesis. The chapter ends by describing the intended research approach to address the aim and objectives, and the thesis structure.

1.2 Key concepts

1.2.1 River classification: Physical and ecological template

The rivers in this study were examined at various scales including catchment, sub-catchment, riparian corridor and local scale. It is accepted in geomorphology that river form implies process which resulted in various classification systems for hillslopes, landscapes and rivers (Buffington and Montgomery, 2013). Most of the geomorphological river classification systems are hierarchical, which provide a means of interpretation for the complex nature of rivers and provides insights into linkages and feedbacks between components that may play out across one or more scales (Dollar et al., 2007). A range of hierarchical classification systems were developed (Frissell et al., 1986, Rosgen, 1994, Van Niekerk et al., 1995, Rowntree and Wadeson, 1999, Brierley and Fryirs, 2000, Brierley et al., 2002). In spatially-nested hierarchical systems the higher levels of the hierarchy constrain the lower levels in river systems, and different geomorphological processes will assume a greater relative importance as the scale changes (Rowntree et al., 2000).

In South Africa two hierarchical river classifications were developed including Van Niekerk et al. (1995), which is a bottom-up approach based on an assemblage of morphological units into channel types. The other, which is applied most often, was developed by Rowntree and Wadeson (1999) and provides a scale-based framework that links the components of a river system consisting of the catchment, the river segment, the river zone, the river reach, the morphological unit and hydraulic biotope. In the framework, rivers are classified according to geomorphological zones based on river discharge, sediment load and regional slope. The longitudinal zonation identifies geomorphologically similar river reaches occurring in uniform geology, associated with particular morphological units each having their own flow characteristics but still retaining the concept of downstream changes longitudinally (Rowntree et al., 2000).

The longitudinal changes along a river were formulated with the River Continuum Concept (RCC) where all rivers have continuous gradients of physical and chemical conditions such as width, depth, velocity, flow, volume, temperature and eutrophy that are modified continuously and progressively from the source to sea (Vannote et al., 1980). This is linked to shifts in organic matter supply, river processing rates, and the relationship between primary production and respiration (Harding et al., 1999). However, the four dimensions of connectivity as proposed by Ward (1989), will influence river ecology and other concepts have since recognized the lateral connectivity rivers have with their floodplains such as the flood pulse concept (Junk et al., 1989), the river productivity model (Thorp and Delong, 2002) that focused on instream (autochthonous) production and the vertical and temporal dimensions of the natural flow regime paradigm (Ward, 1989, Poff et al., 1997). Species assemblages can vary with the physical habitat template and the flow regime so that discontinuities can occur along a river gradient (Thorp et al., 2006). For example, changes in river flow can result in changes in food resources such as algae and organic matter and as a consequence influence macroinvertebrate species assemblages (Dewson et al., 2007). Bunn et al. (2006) showed that in ephemeral dryland rivers, water holes provide refuge to aquatic biota and although in-channel flood flows are essential to their persistence/connectivity, it created periods of food limitation and stress to fish and other consumers. Changes can occur in the physical river template or the riparian vegetation providing lateral input can be altered continuing down a river gradient by natural or anthropogenic impacts. Thorp et al. (2006, 2010) proposed that rivers consist of hydrogeomorphic patches, formed by catchment geomorphology and flow characteristics, each with their associated chemical and physical conditions controlling ecological structure and function. Hydrogeomorphic patches were also found to occur in large floodplain rivers (Blettler et al., 2016). These patches can be referred to functional process zones at a valley or reach scale (Thoms and Parsons, 2002, Thorp et al., 2010).

Although the river reach is primarily the focus of many studies, the hydrogeomorphological character at a reach is the culmination of hydrological, geomorphological and ecological processes that operates on much larger catchment and time scales. Anthropogenic impacts to rivers alters form-process links along a longitudinal gradient changing the interactions between floodplain, riparian and aquatic environments (Grabowski and Gurnell, 2016) so that pattern will not always imply process (Thorp et al., 2006). For successful process-based restoration efforts and effective river management, a catchment scale perspective is essential (Beechie et al., 2010, Grabowski and Gurnell, 2016). The current study used different river longitudinal zones along a river impact gradient. The purpose was to relate the rivers to their catchments and to relate the instream processes to biological processes to examine the ecological river integrity (Chapter 4).

1.2.2 Links between hydrogeomorphology, riparian vegetation and aquatic ecology

Hydrogeomorphic processes can be viewed as providing the physical template of rivers, which constrains the biological and chemical processes occurring within (Tabacchi et al., 1998). The inter-relationships between hydrogeomorphic processes occur at different spatial and temporal scales and are responsible for river morphodynamics such as channel adjustment, and river habitat and ecology (Jacobson, 2013, Solari et al., 2016). Hydrogeomorphic parameters such as river discharge regime, water table fluctuations, sediment transport regimes, erosion, deposition, sediment texture and topography controls plant succession and eventually the development of the riparian zone (Steiger et al., 2005). Riparian zones form a boundary between the terrestrial and aquatic ecosystems and as such they are a unique ecotone where hydrologic, geomorphic, vegetative and nutrient biogeochemical processes interact (Noe, 2013, Connolly et al., 2015). The riparian vegetation is an integral part of the fluvial system since it influences and responds to hydrogeomorphic processes in such way that it is often referred to as a bioengineer or system engineer of floodplains and rivers (Noe, 2013, Gurnell, 2014, Solari et al., 2016).

Riparian vegetation affects hydromorphology with species specific above-ground and below-ground traits which influence flows and sediment deposition (Grabowski and Gurnell, 2016). The above-ground biomass results in flow resistance that reduces velocities and bed shear stress and with that erosion, sediment transport and deposition thereby determining channel morphology. Vegetation contributes to channel roughness and resistance by factors such as height of the plants, form, morphology, flexibility of the stems, plant leaf mass, density and spacing of vegetation as well as the seasonal and successional plant dynamics (Tabacchi et al., 2000, Sandercock and Hooke, 2010).

Rivers consist of various morphological units or landforms so the underlying geomorphology can determine the species, growth form and position of vegetation on these landforms, influencing channel formation (Rowntree, 1991, Hupp, 1996). The vegetation pattern observed on various fluvial landforms such as the river banks and floodplains, channel bars, benches/shelves and channel bed will be dependent on environmental conditions, site suitability and therefore the hydrogeomorphological processes that determine each landform (Bendix and Hupp, 2000). River dynamics affects fluvial landforms and as such they may be modified over time or even the substrate of which they are composed may change. This will determine the type of vegetation (age, composition, traits, and morphology) colonization. For example, a channel shelf will undergo change more frequently and may only be colonized by vegetation able to complete their life cycles between floods such as annuals (Merritt, 2013). Hupp (1996) showed that patterns of vegetation were related to the distribution of fluvial landforms created by flood disturbance or flow duration, where new sites for establishment can form. In semi-arid and arid sites the vegetation establishment was linked to surface flow or floods and persistence was linked to groundwater levels (Lite et al., 2005, Richardson et al., 2007). Others such as Van Coller et al. (1997, 2000) presented similar findings in semi-arid South African settings. Vegetation patterns often follow vertical (height above the channel), lateral (horizontal distance away from the channel) and longitudinal (distance down river) gradients. Van Coller et al. (2000) showed that the vegetation pattern in certain areas was influenced by the height of the macro-channel bank as well as substratum type, morphological units and channel type. So although fluvial processes and hydrology influences the patterns of riparian vegetation, valley form and processes including rates of runoff and erosion and deposition, channel planform, and cross-sectional form and slope are also influenced by the presence, form, structure, distribution, and abundance of vegetation (Osterkamp and Hupp, 2010, Merritt, 2013).

The below ground biomass influences hydraulic and mechanical properties with root reinforcement, which ultimately influences soil moisture and erodibility (Abernethy and Rutherfurd, 2001, Solari et al., 2016). Bank vegetation and bank material co-exist and have feedback loops between them. On banks that have lower clay and silt percentages vegetation can have a compensating role for stability. Rowntree and Dollar (1999) estimated that high vegetation densities on banks provided increased resistance to erosion to that of banks composed of 70% silt and clay. Others also found that narrower channels resulted where vegetation covered a greater extent of the bank even though lower silt and clay percentages occurred in the channel margins (Anderson et al., 2004). Soils with increased sand contents are likely to erode at lower shear stresses (Wynn and Mostaghimi, 2006) but lower erosion rates can occur on sandy soils due to higher root densities (Dunaway et al., 1994). However, others showed that more erosion occurred on banks that had higher clay content due to lower root densities, where root growth was sometimes inhibited. The interactions between channel width, bank strength due to soil cohesion and root effects, volume of root density and silt and clay contents are complex and will vary between different environmental settings (Anderson et al., 2004, Pollen-Bankhead and Simon, 2010).

Riparian vegetation is also significant to the ecology of river ecosystems. Riparian zones play an important role in the quantity, quality and seasonality of external resource inputs into streams by altering sunlight regimes and allochthonous organic matter dynamics as well as affecting the regime of environmental stressors such as sediment inputs and loading, temperature and chemistry concentrations (Márquez et al., 2015). Woody riparian vegetation provides more shading, which can reduce the growth of algae and macrophytes when compared to grassy riparian zones, which are open to sunlight. Woody banks are prone to large woody debris and debris dams in channel, which retains organic matter in the stream. Woody riparian zones have a lack of autochthonous (within stream) production and high levels of allochthonous (outside stream) organic input. The size of the channel will also play a role (Gurnell, 2013). In smaller rivers the amount of coarse particulate organic matter can be high but will be removed by floods, and decomposition occuring instream. Dissolved organic matter provided by riparian vegetation will also modify water quality in different ways depending if the source is from saturated or unsaturated riparian soils (Tabacchi et al., 2000). Different river styles and riparian vegetation type affects both the habitat and therefore also the biological communities (Lyons et al., 2000).

Anthropogenic or natural changes to the riparian zones will have an important impact on river integrity. Steiger and Gurnell (2002a) showed the importance of the geomorphological structure of the riparian zone in sedimentation deposited with floods of varying magnitudes. More sedimentation occurred in natural vegetation riparian zones when compared to alien vegetation and the concentrations of total phosphorus, total organic nitrogen and organic carbon differed with the quantity and calibre of sedimentation and varied between the different fluvial landforms. Carpenter et al. (2012) reported that in events where channel widening or channel migration occurred due to excess sediment provided from upstream areas, less shaded areas resulted due to removal of riparian vegetation, which increased sunlight penetration, water temperatures and promoted increased bacterial growth and lowered dissolved oxygen concentrations. The increased sunlight also causes the development of periphyton, especially nuisance types such as filamentous green algae and macrophytes (Oberholster et al., 2013c). Similar observations were made by Gilvear et al. (2002) where channel changes resulted due to changes in land use in the catchment, which altered runoff and sediment loads. With the removal of riparian vegetation water temperatures increased and leaf litter input (food source for macroinvertebrates) to the stream was reduced. Beside the addition of sediment to rivers caused by bank erosion, nutrient concentration, especially phosphorus, could also increase, depending on the sediment composition (e.g. cohesive banks of fines with phosphorus absorbed) (Miller et al., 2014).

Certain species of alien vegetation in the riparian zone are responsible for the leaching of allelopathic substances. Allelopathy is the process through which invasive plants produce biochemicals that influence the growth, survival and reproduction of indigenous species. This was found to occur with species such as eucalyptus (Sasikumar et al., 2001), pines (Nektarios et al., 2005) and mesquite (Al-Humaid and Warrag, 1998). Studies from South Africa showed that alien shrubs and trees have increase evaporation rates as compared to indigenous species, which results in reduced groundwater reserves and river flows thereby reducing the river's dilution capacity and increasing concentrations of nutrients, salinity and other pollutants (Le Maitre et al., 1999, Dye and Jarmain, 2004, Le Maitre et al., 2013, Le Maitre et al., 2016). In the South African setting most tree invaders create dense stands and excessive litter, which causes increased nutrients and ultimately impacts on stream water quality. When fires occur within these dense stands they burn at a greater intensity than indigenous vegetation and water repellency of the soil can occur (Scott, 1997, Scott et al., 1998a). Since alien vegetation limits or prevents undergrowth such as smaller shrubs required for greater binding of soil by shallow roots, soil erosion is increased due to increased overland flow post-fire (Van Wilgen et al., 2010). This influences water quality by increasing sediments, water temperature and nutrients (Chamier et al., 2012).

Where riparian zones are modified by agricultural land use, not only is water quality affected but also the feeding rates, growth, densities and survival of aquatic populations resulting in changes in the structural and functional attributes of stream communities (Dolédec et al., 2006, Márquez et al., 2015). Using biological indicators in addition to chemical parameters to assess water quality and overall river integrity has proven extremely valuable (Márquez et al., 2015, Gökçe, 2016). Water quality fluctuates over time and space and the evaluation thereof can be inconsistent. When including fauna and flora such as algae and macroinvertebrates, when assessing ecological conditions of a river, a better indication of the overall river integrity and water quality is provided. For example, alterations to a riparian zone may modify the functional feeding group composition of macroinvertebrate communities by altering the food availability

(Compin and Céréghino, 2007), which in turn may modify algal communities. Using multiple groups of organisms from different trophic levels in an aquatic ecosystem to assess impacts on water quality, further improves the value of such information (Lammert and Allan, 1999, Callisto et al., 2004, Harding et al., 2005, Smith et al., 2007, Li et al., 2010, Qu et al., 2016).

In South Africa macroinvertebrates assessment of river health has been routinely applied as part of assessing river health (Palmer and O'Keeffe, 1992, Dallas, 1997, De la Rey et al., 2004, Munyika et al., 2014). Macroinvertebrates and periphyton algae and especially diatoms are ideal to use as indicators of river health and water quality since they are ubiquitous and ecologically important. Algae and diatoms are at the base of the food web as primary producers, and both macroinvertebrates and algae are species rich adding to biodiversity in rivers and streams (Graça et al., 2004, Oberholster, 2011). These species are also sensitive to a broad range of environmental stressors and respond rapidly to changes in water chemistry due to anthropogenic pollution (Dallas, 2002, Oberholster, 2011). Diatoms have been used as indicators of specific water quality problems such as heavy metal and organic pollution as well as eutrophication as they consist of a wide group of epilithic algal communities with species indicative of specific water quality issues (Walsh and Wepener, 2009). Nutrient enrichment in streams are also a fundamental determinant of benthic algal biomass, which can determine benthic algal taxonomic biomass and richness (Biggs and Smith, 2002).

Although epilithic diatom community indices have been applied in numerous studies in Australia, Europe and the United States (Descy and Coste, 1991, Dela-Cruz et al., 2006, Porter et al., 2008), algae and diatoms have only recently been used in studies in South Africa assessing stream ecosystem health (Oberholster, 2011), as a screening tool for assessing water quality (Taylor et al., 2007b, Oberholster et al., 2013) or to illustrate the link to environmental variables (Ewart-Smith, 2012, Oberholster et al., 2016). This study examined aquatic macroinvertebrates and the full consortium of algae in relation to the physical river template, riparian vegetation and land cover/use down a longitudinal river impact gradient in an agriculturally dominated catchment, to determine ecological integrity and bioindicator suitability.

1.2.3 Ecosystem services linked to riparian vegetation

Ecological infrastructure is gaining momentum in water and land use management. The term ecological infrastructure can be defined as naturally functioning ecosystems that produce and deliver valuable services to people (Mander et al., 2017). Riparian vegetation is an important form of ecological infrastructure and it is well known that intact riparian zones of natural vegetation mitigate the impact of land-based activities, which helps maintain healthy freshwater ecosystems able to support resilience and adaptation to climate change. In order for ecosystems to persist, resilience is required but human impact is compromising their structural and functional integrity thereby jeopardising ecosystem resilience (Rebelo, 2012, Capon et al., 2013). This is especially true of riparian zones. Riparian zones perform valuable direct and indirect ecosystem services to benefit society and the environment such as performing roles of flood attenuation, aquifer recharge, stabilizing streambanks and reducing channel erosion, trapping or removing phosphorus, nitrogen, and other nutrients and contaminants, trapping sediment, maintaining habitats (terrestrial and aquatic) and moderating water temperatures and providing recreation and aesthetics (Wenger, 1999, Steiger et al., 2005, Chase et al., 2016, Tanaka et al., 2016b). These services are either of a supporting, provisioning, regulating or cultural nature (Vidal-Abarca et al., 2016, Sieben et al., 2018) and have importance at local scales as well as at a landscape scale. The degree of lateral, vertical or longitudinal connectivity between the terrestrial and aquatic environments govern much of the regulating services, which will be influenced by changing hydrology and precipitation (Capon et al., 2013).

Numerous studies (Förstner and Solomons, 1980, Hasholt, 1991, Atalay, 2001, Klatt et al., 2003, Rodríguez-Blanco et al., 2010) have shown that nutrients and other contaminants readily adsorb to sediment and with erosion, runoff may reach surface water bodies. It is well documented that riparian vegetation is able to retain suspended sediment and its associated nutrient content derived either from flood waters or from surface runoff associated with agricultural landscapes. However, the ability to remove nutrients from land by the riparian zone can be variable and will depend on factors such hydrology (e.g. water residence times), soil (especially organic carbon status and reducing conditions in the sub-soil) and vegetation (Connolly et al., 2015). The vegetation slows the velocity of runoff and facilitate the removal of sediment and sediment-bound pollutants, while promoting infiltration of soluble pollutants into the soil. This was found for nitrogen and phosphorus bound to sediment eroded from croplands (Lee et al., 2000). For example, Lee et al. (2000, 2003) reported that dissolved nutrients and clays were more effectively attenuated in riparian buffers of native vegetation with a surface litter cover to slow water velocities and where the soils had a high porosity to increase infiltration capacity.

The riparian zone was shown to be efficient in phosphorus removal where agricultural areas were the source (Daniels and Gilliam, 1996, Mander et al., 2005). Phosphorus can be transported directly to runoff waters from sediments in different forms and usually is processed by shallow groundwater (Tabacchi et al., 2000). Diffuse subsurface pollutants such as nitrogen can also be effectively buffered by riparian vegetation by plant uptake and microbial denitrification. The microbial communities found on above-ground plants, leaf litter, soil and fine roots are also able to assimilate dissolved nutrients from surface water (Tabacchi et al., 1998). Riparian zones can however, be a source (release of nutrients form decomposing vegetation) or a sink for pollutants depending on the hydrologic conditions and flow paths of water draining to rivers (Capon et al., 2013). Nitrogen, for example, is water soluble and can be delivered via groundwater so the efficiency of the riparian zone in regulating it can depend on geomorphic features, which will determine flow paths. The riparian zone could consist of a range of morphological units, which may or may not be hydrologically connected to the subsurface flow (Ranalli and Donald, 2010). The efficiency of attenuation of nitrogen and other pollutants by riparian zones will not only depend on the amount of surface area covered by vegetation but also by the length of the hydrological contact with the riparian zone as pollutants move from the source of pollution to the stream (Tabacchi et al., 1998). While riparian buffers of varying vegetation types can increase the trapping efficiency of sediment and the total nutrients, higher rainfall intensities can increase the total transport of nutrients (Lee et al., 2000). The current study examined the effectiveness of riparian buffer zones differing in vegetation type and width at attenuating nutrient fluxes from agricultural and natural land use along a longitudinal impact gradient.

1.3 Existing knowledge and gaps

Ecological infrastructure and natural capital is now a priority to socio-economic development in South Africa (SANBI, 2014) and riparian vegetation as ecological infrastructure is increasingly being considered in water and land use planning. Riparian buffers are an accepted form of best management practice worldwide in the improvement of water quality in rivers draining agricultural areas. Agricultural land use is associated with increased nutrients entering streams and the effectiveness of riparian buffers in attenuating sediment and associated nutrients to improving water quality are dependent on factors such as soil, buffer width, and flow rate, rainfall intensity, slope and area ratio of buffer to source field, vegetation type and relative height of water to plants (Liu et al., 2008). The combination of plant species of trees, shrubs and grasses have been shown to maximize the adsorption and retention of pesticides, nutrients and other non-point source agricultural pollutants (Schulze, 1995, Hawes and Smith, 2005).

Numerous studies exist on appropriate riparian buffer widths and vegetation types that are effective in the mitigation of agricultural impacts to surface and groundwater quality. However, most studies focussed on pollutant removal efficacy with very specific environmental conditions and the relationships identified between buffer efficacy and the associated factors can be inconsistent (Zhang, 2010). Very few of these studies occurred in a South African context with a Mediterranean climate and rainfall throughout the year. Guidelines for the development of buffer zones for rivers, wetlands and estuaries were recently developed for South Africa (Macfarlane et al., 2014, Macfarlane and Bredin, 2017) but these guidelines were primarily based on international studies with limited field-based information from South Africa.

It is important to define the factors and variables conducive to local South African conditions and to include the linkages between land cover and use, riparian morphodynamics and river-morphodynamics, which are found to be lacking in river management and restoration (Camporeale et al., 2013). Understanding the hydrogeomorphic river structure and the ecological linkages are essential when assessing the ecosystem services, especially in predicting the impact and change in services when channel alteration occurs or in river rehabilitation efforts (Thorp et al., 2010). Studies focussed on a multidisciplinary understanding of whole river system dynamics between the different ecosystems (terrestrial, riparian, aquatic) in the provision of ecosystem services, especially in a South African context are scant. There is also a lack of studies focussed on linking the physical river structure to the provision of ecosystem services and land use effects on water resources in the context of ecosystem service provision (Thorp et al., 2010, Doody et al., 2016).

This study will allow the opportunity to assess the link between hydrogeomorphology, land cover/use and riparian vegetation and its effectiveness as a mitigation measure on water quality and ecological integrity using biological indicators, in an agricultural river ecosystem. It will also provide scientific knowledge to substantiate the guidelines proposed for riparian buffers to river systems draining agricultural areas in nutrient mitigation. This research proposes to examine these linkages and processes and aims at addressing the knowledge gaps above.

1.4 Research aim, objectives and approach

The primary aim of this thesis was to develop a systems understanding of the interactions between different ecosystems, land cover and the influence of past and current land use management practices in the Touws and Duiwe River catchments. The Touws River drain into the Touws estuary and the Duiwe River feeds the Wilderness Lake System, which is a Ramsar site and is protected in the Garden Route National Park. Poor upstream land management in these catchments have been implicated as a primary driver in changing water quality which affected the biophysical and chemical properties of these water bodies (Russell, 2013). The Lake System provides essential ecosystem services to the area and its economy, such as ecotourism and recreation. Further impact on river water quality will ultimately impact on these benefits.

The focus of this study is on the biophysical interactions, linkages and drivers of change on water quality and quantity and ecological river integrity. This will form the process-based understanding of the hydrogeomorphic controls on the provision of goods and services that vegetation, and especially riparian vegetation, perform as ecological infrastructure in the mitigation of agricultural impact on water quality and ecological river integrity. A synthesis framework will be informed by the field research data, which will provide an understanding of the effects on and impacts to the provision of ecosystem services with changes to land cover/use and riparian zone structure and composition. Given the significant influence riparian zones have on fluvial systems, its ecological importance and its ability to provide numerous goods and services, it becomes necessary to understand the linkages between these ecotones, rivers and their terrestrial environments. In doing so the research will provide knowledge to inform appropriate river management for ecological integrity and characteristics for desirable riparian buffer zones to provide maximum ecosystem services.

The objectives of this study were therefore to: -

1. To examine historical influences of land cover and land use changes on water quality (Chapter 3)
2. To evaluate linkages between riparian morphodynamics, the physical river template and aquatic organisms and their use as indicators of water quality and river ecological integrity (Chapter 4)
3. To investigate controls on the morphodynamics of riparian zones (this incorporates vegetation effects) and to assess the linkages between agricultural land use and water quality (Chapter 5)
4. To synthesise the information from these objectives to develop an ecological infrastructure framework assessing ecosystem service provision of riparian zones (Chapter 6 with inputs from Chapters 3, 4 and 5)

1.5 Structure of the thesis and overview

This thesis comprises 7 Chapters (Figure 1.1). Chapters 3-6 will address the objectives and all limitations/uncertainties and comprehensive literature reviews were captured in each chapter. A combination of desktop analysis and field surveys at study sites was used in the research approach. Chapter 6 serves as a synthesis framework chapter while Chapters 3, 4 and 5 are data analysis chapters. Each of the data chapters are divided into sections that include an introduction, methods to present data and analysis, results from analysis and a synthesis discussion and conclusion.

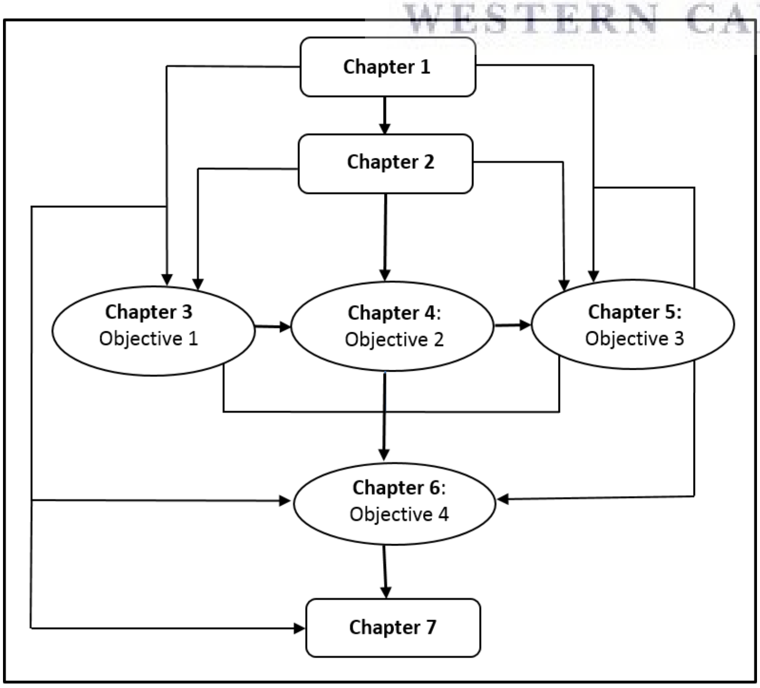


Figure 1.1 Visual outline of the thesis with chapter linkages

Due to the integrative nature of the study, multidiciplinary data sets were necessitated and assistance from experienced professionals were required. These professionals and the extent of their invlovement is stated and acknowledged in the relevant chapter descriptions below. However the chapter development and writing thereof as well as the data anlysis, graphic production and writing and development of the subsequent papers was the principle responsibility of the author of this thesis.

Chapter 1: Introduction: Chapter 1 provides a general overview of the research topic and provides background and context for the research undertaken while reviewing the broader literature base.

Chapter 2: Study area and site selection: This chapter introduces the study area and provides information related to the biophysical environment and the anthropogenic impacts. The beginning of Chapter 2 in section 2.1 is related to the regional setting (Breede-Gouritz WMA) of the two coastal catchments on which the research questions are focussed on. The sections to follow specifically deals with the Touws and Duiwe river catchments. This chapter also provides a description of the sampling sites within the Duiwe River study area. Field surveys were conducted at the sampling sites, which involved river cross-sections and resistivity surveys. A total of 12 cross-sections were completed at four sites, which were linked to 72 quadrats in which the physical and chemical sediment properties and vegetation were sampled. Along each cross-section biological indicators of water quality, which included macroinvertebrates and algae, were recorded along with chemical water quality parameters, water levels and river discharge. Field sampling occurred seasonally over a two and half year period (2014 to mid-2017). The quadrats represented the position on the river bank (upper, middle, lower bank or instream).

Chapter 3: Links between catchment land cover and water quality: This chapter was a desktop study and examined the historical land cover and uses at a catchment scale, namely the Touws and Duiwe River catchments, to provide a context for the present water quality state in the study area. The approach followed a spatial and temporal analysis of historical catchment land use change using historical aerial photography and topographical maps in a Geographical Information System (GIS) to visualise and quantify changes to land cover/use in the study catchments. It focussed on trends in historical surface water quality, river flows and rainfall patterns, which provided the links to changes in the physical land cover and environment. The chapter was presented at the International Association of Sediments & Water Science (IASWS), Rhodes University, Grahamstown during 2014. It was also presented and formed part of the Conference Proceedings of the 11th International Symposium on Ecohydraulics in Melbourne, Australia during 2016: This chapter was later formulated as a paper and was published by Water SA during 2017. Professor N. Jovanovic, Dr D. Le Maitre and Dr M. Grenfell were co-authors and contributed to the concepts therein and the review of the manuscript.

Chapter 4: Linkages between the physical river template and biological water quality indicators: This chapter examined how the physical river template, macroinvertebrates and algae are associated with the template and their link on water quality at 4 sampling sites in the Duiwe River catchment. Data were collected over a two year period (2014-2016) and included river cross-sections, sediment sampling from river banks and bed and algae and macroinvertebrate sampling. Researchers from the CSIR (Stellenbosch) who contributed to data collection included a surveyor for river cross-sections, R. Vonk, a hydrogeologist, R. Bugan, who assisted with resistivity data collection, P. Oberholster, who identified algae species sampled and reviewed the chapter and P. Cheng, who processed water and algal samples in the laboratory for chlorophyll *a* and assisted in the sampling of algae in the field. This chapter was formulated as a paper and was published by Hydrobiologia during 2018. Professor N. Jovanovic, Dr M. Grenfell, Dr P. Oberholster and Dr P. Cheng and were co-authors and contributed to the concepts therein and the review of the manuscript. The chapter was also presented as a paper at the Southern African Association of Geomorphology (SAAG), University of Swaziland, Kwaluseni (Manzini), Swaziland during 2017.

Chapter 5: Linkages between riparian morphodynamics, land cover and water quality: This chapter examines riparian vegetation patterns, hydrogeomorphic controls thereon and provides a quantitative assessment to determine the effectiveness of riparian zones as a mediating measure of nutrients from the agricultural land on water quality. The study sites consisted of riparian buffer zones differing in vegetation types and land use impact (no impact, indigenous vegetation to alien invaded riparian vegetation, agricultural impact), along a longitudinal impact gradient. Runoff plots were installed in three different land covers: indigenous riparian forest, riparian buffer zone partially invaded with alien vegetation, a riparian buffer zone almost completely invaded with alien vegetation and dairy pastures adjacent to the riparian zones. Runoff water and sediment were collected from the different land covers for 2.5 years (2014 to mid-2017) and analysed for quality. This chapter was formulated as a paper and was submitted during 2018 to the African Journal of Aquatic Science and is currently in review. Professor N. Jovanovic and Dr M. Grenfell are co-authors and contributed to the concepts therein and the review of the manuscript.

Chapter 6: Discussion: Inter-relationships between hydrogeomorphology, ecology, anthropogenic impacts, and ecosystem service provision: This chapter provides a synthesis of the overall findings from the data chapters 3-5 with input from Chapters 2, 3 and 4. The field research data informed the development of a hierarchical conceptual framework flow-chain model in linking hydrogeomorphology and ecology in assessing ecosystem service provision of riparian zones.

Chapter 7: General conclusions and recommendations: The key messages, findings and recommendations are outlined regarding the implications for river ecological integrity and conservation in South Africa. Recommendations to inform desirable riparian buffer zone characteristics for South African conditions and possible restoration are outlined.

Chapter 2: Study area and site selection

2.1 Regional setting

The Breede-Gouritz Water Management Area (WMA), which is approximately 53 139 km^2 in surface area, is situated along the southern coast of South Africa almost entirely within the Western Cape Province (Figure 2.1). It extends from the coast inland and the topography is characterised by flat open plains of the Great and Little Karoo. This results in distinct zones of water resources where the Gamka River catchment occurs in the semi-arid Great Karoo, to the north of the Swartberg Mountains and the Groot River catchment, to the west is the Klein Swartberg Mountains, while the southern boundary of the Groot River catchment occurs along the Langeberg Mountains (Figure 2.2). The arid zone is fed by the Olifants River with mountain streams rising in the Swartberg Mountains to the north, the central Kammanassie Mountains and the coastal Outeniqua Mountains in the south. The Coastal Belt rivers drain from the Langeberg and Outeniqua Mountains along the eastern boundary of the WMA, which is referred to as the coastal sub-catchment area. The other numerous short reach coastal rivers drain an area of 7 437 km^2 (DWAF, 2004). The coastal sub-catchment area, which consists of secondary catchments K1 to K7, contains a total of 27 quaternary catchments (Figure 2.1). Physiographically, the sub-catchment area consists of the steep mountain ranges, the mountain foothills, the coastal platform, and the coastal embayment (Figure 2.2)

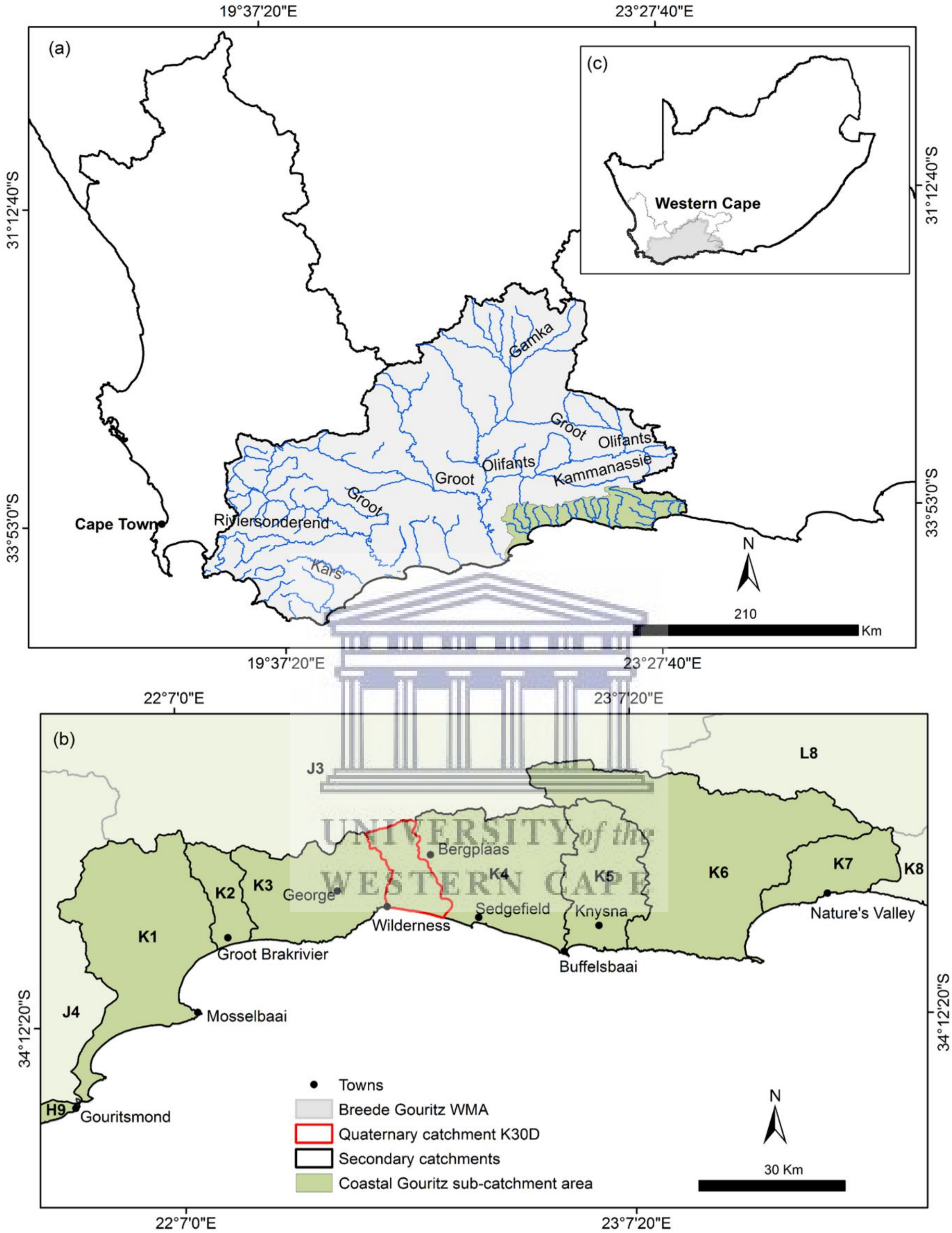


Figure 2.1 The regional setting of the (a) Breede-Gouritz Water Management Area (WMA) and mainstem rivers, (b) the coastal Gouritz sub-catchment area with secondary catchments and (c) the location of the Breede-Gouritz Water Management Area (WMA) within the Western Cape

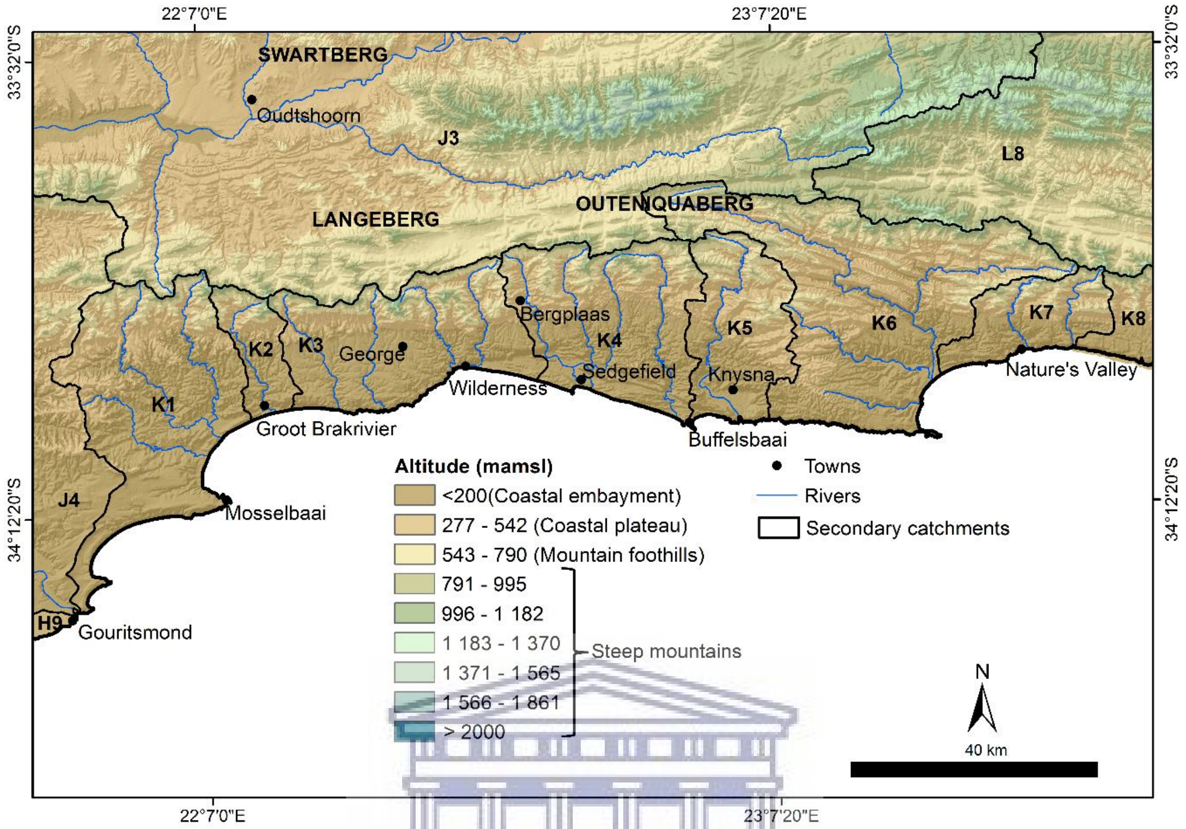


Figure 2.2 Physiographic regions in the Gouritz coastal sub-area (secondary catchments K1-K7)

2.2 Catchment setting

2.2.1 Location and topography

The area of focus occurs in quaternary catchment K30D (Figure 2.1b) in two coastal catchments, the Touws River (95.81 km^2) and Duiwe River (33.82 km^2) (Figure 2.3). The middle and upper catchment of the Touws River occurs between the coastal ridge and the crest of Outeniqua Mountains (±1100 mamsl). The upper catchment is characterised by steep-sided valleys and fast flowing streams which supply the Touws River. The Klein Keurbooms River forms one of the tributaries of the mainstem Duiwe River, which together with the Langvleispruit feeds the Wilderness Lakes, consisting of Rondevlei, Langvlei and Eilandvlei. The Lakes with their rivers and the Serpentine channel (±7 306 ha) are all recognized Ramsar sites. The Touws River feeds an estuary. The Duiwe River feeds into the north-eastern side of Eilandvlei Lake, while Langvleispruit feeds Langvlei Lake. Rondevlei is groundwater fed. The Wilderness estuary is connected to Eilandvlei via the Serpentine River, which in turn is connected by a channel to Langvlei, with Rondevlei being the most easterly of the three lakes

(Russell, 2013). The Serpentine channel connects the lake system and the Touws River estuary. There are no major dams on the Wilderness Rivers and water supply is by direct abstraction on a run-of-river basis, which supplies towns with water. Wilderness Town is supplied from the Touws River and also the Garden Route Dam on the Swart River (beyond catchment boundary). The coastal sub-catchment area accounts for approximately 56% of the total population in the Breede-Gouritz WMA but the population has increased in recent years. Most coastal towns are also popular holiday destinations and accommodates an influx of people during the summer season (DWAF, 2004). The area is ecologically sensitive and has a rich species biodiversity with a variety of ecosystems, with vegetation biomes forming part of the Cape Floral Kingdom, which is one of six in the world (Pauw, 2009). Cathment morphometry are described in Table 2.1. The Touws and Duiwe River systems are essential water supply rivers in the catchments and the natural biodiversity of the area is important to the economy of the area (DWAF, 2004). Catchment characteristics are summarised in Table 2.2.

Table 2.1 Morphometric parameters for the Touws and Duiwe River catchments (Gordon et al., 2004, Kuchay and Bhat, 2013)

	Touws	**Duiwe**
Catchment area (km^2)	95.81	33.82
Catchment slope (mean) (%)	30.4	12.1
Form factor	0.38	0.32
Stream order (at gauging weirs)	5	4
Main stream length (km)	28.4	19.46
Main stream slope (%)	3.4	2.6
Drainage density (km/km^2)	2.51	3.59
Length of overland flow (km)	0.19	0.139

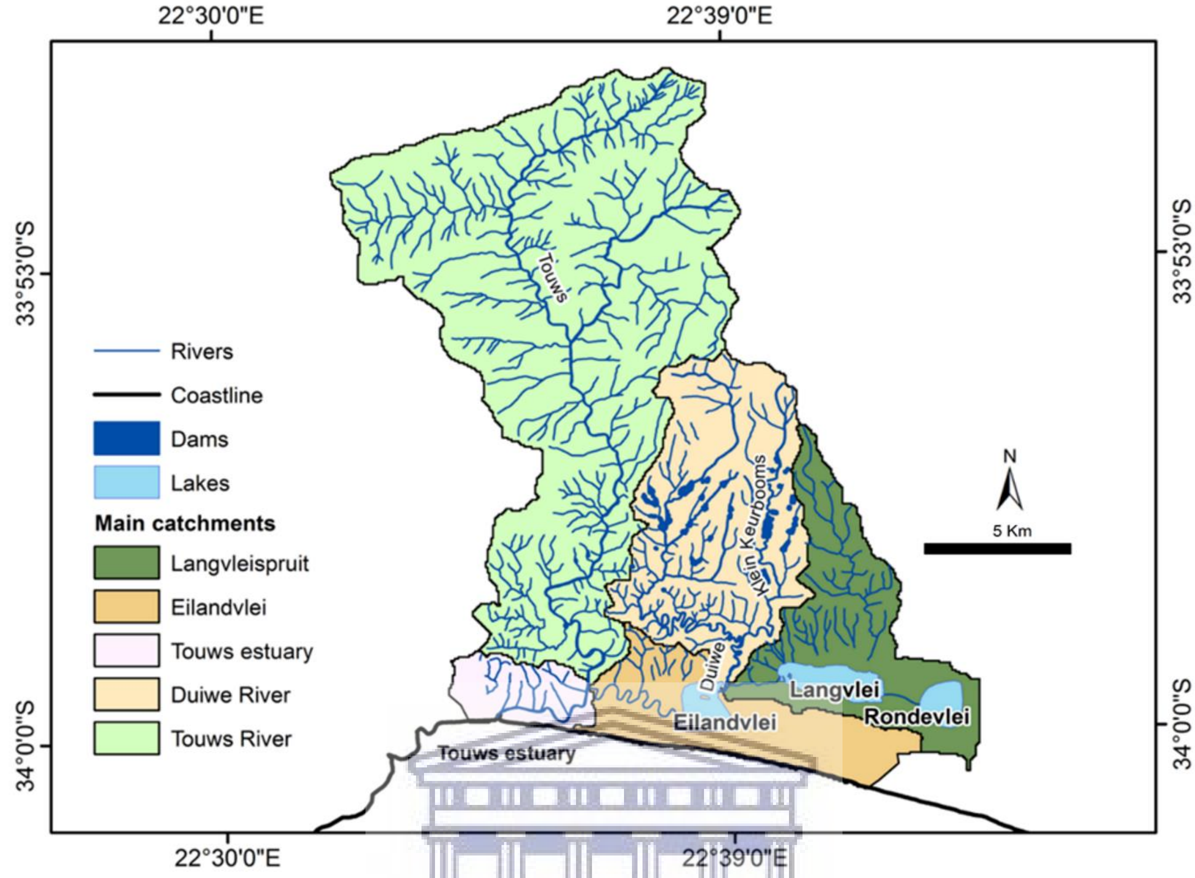


Figure 2.3 The main river catchments of quaternary K30D

The Touws estuary is estimated to be closed 40% of the year, changing it from a naturally temporary open/closed system and causing a change in the biophysical and chemical properties of the lakes (Russell, 2013). More recent data shows the estuary to be open only 28% of the time due to reduced freshwater inflows and artificial breaching (Russell, 2013). To prevent flooding of adjacent property to the lakes the estuary mouth is breached at 2.2 to 2.4 mamsl. The management technique is to maintain the estuary berm at 2.4 m so that optimal hydraulic conditions are maintained to allow scouring of the sand out of the mouth during floods (Allanson and Whifield, 1983, Russell, 2013). Breaching does sometimes occur at times not optimal for marine fish recruitment and/or when the water levels are lower than the recommended 2.2 to 2.4 mamsl. Low water levels may also cause the estuary to close prematurely due to non-scouring of accumulated marine sediments (Russell, 1996). The Touws and lower Duiwe Rivers occur within the Garden Route National Park. The indigenous state forests and mountain catchment areas in the Outeniqua and Tsitsikamma Mountains and the Wilderness National Park are managed by the South African National Parks (SANParks) (SANParks, 2010).

Table 2.2 Catchment characteristics of the Touws and Duiwe Rivers. Land type coding according to ARC (2005)

Land type code	% (catchment)	Soil depth (mm)	Clay top soil (%)	Texture	Geology	Characteristics	Land cover	
							Touws	Duiwe
Ib2	64 (Touws) 10 (Duiwe)	357	5.5	Sandy	Quartzitic sandstone and subordinate shale of the Table Mountain Group (TMG), Cape Supergroup	Rock outcrops comprise >60% of land type	Indigenous natural area	Thicket, dense bush
Gb2	21 (Touws) 17 (Duiwe)	396	6.4	Loamy sand	Quartzitic sandstone, subordinate shale of the TMG; schist and hornfels (Kaaimans Group), gneissic granite and granodiorite	Podzols comprise >10% of land type; dominantly shallow	Plantations, Thicket, dense bush	Thicket, dense bush, Natural forests, fynbos
Db33	6 (Touws) 53 (Duiwe)	443	8.9	Loamy sand	Gneissic granite, granodiorite, phyllite, schist, grit, hornfels and quartzite (Kaaimans Group)	Sandier topsoil abruptly overlying more clayey subsoil, >50% of land type; <50% of duplex soils have non-red B horizons	Smallholdings urban, degraded fynbos	Agriculture
Fa39	9 (Touws) 20 (Duiwe)	403	11	Loamy sand	Phyllite, grit, quartzite, schist and hornfels (Kaaimans Group), with locally quartzitic sandstone of the TMG	Shallow soils; little or no lime in landscape	Natural forests, fynbos, smallholdings	Natural forests, fynbos

2.2.2 Drainage

There are two main rivers in the quaternary catchment K30D: (1) the Touws River and its tributaries and (2) the Duiwe River and its tributaries (Figure 2.4). The hydrological characteristics and the reference water quality are determined, in part, by the geology of the two catchments. This is achieved by the ability of the rock and weathered material to store and transmit water, by the soils that are generated through interactions between the geology, geomorphology, weathering processes and by the vegetation. Most the Touws and the upper catchment of the Duiwe occur in the quartzitic sandstones of the TMG. These rocks are highly fractured and faulted which enables limited quantities of groundwater storage (Lin et al., 2014)

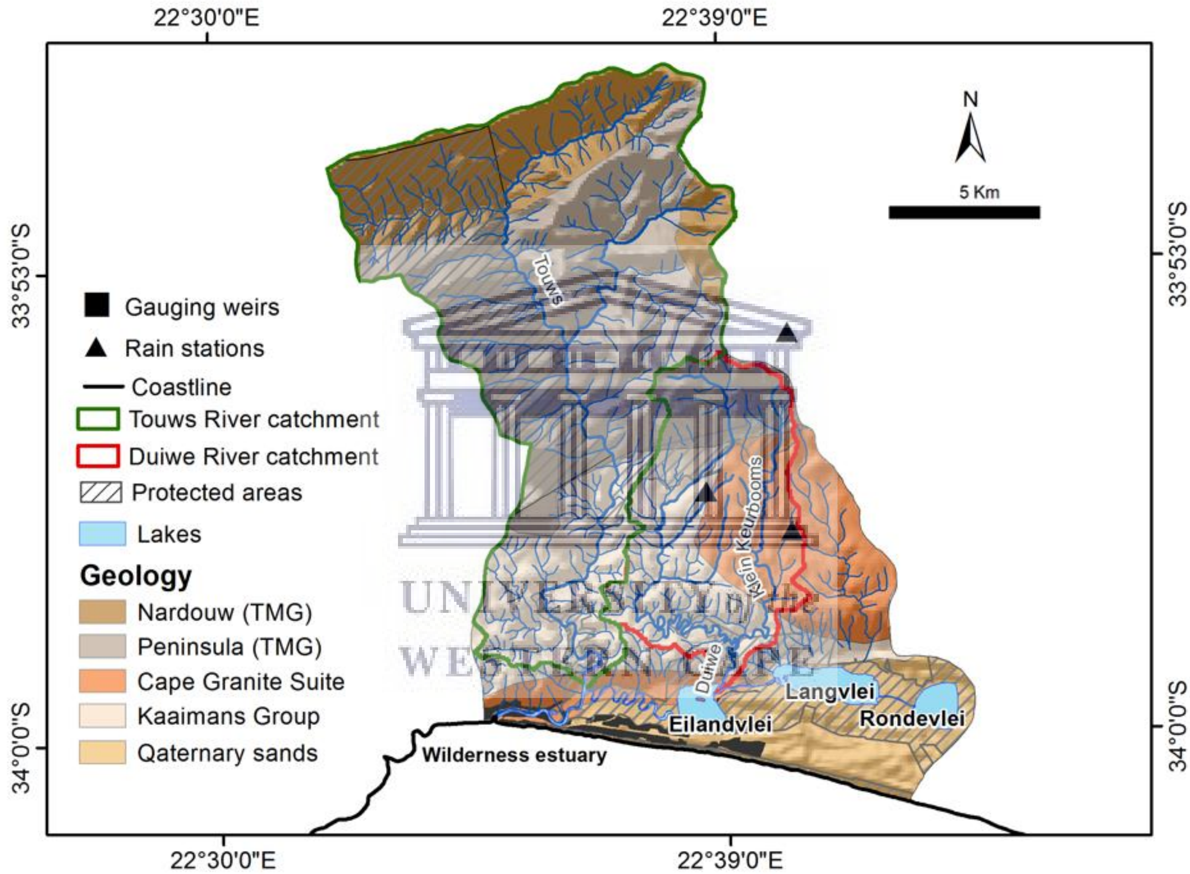


Figure 2.4 Drainage and geology of the Touws and Duiwe River catchments

The stream network in both catchments displays a rectangular trellis drainage pattern controlled by the underlying geology (Gordon et al., 2004). The Touws and Duiwe catchments are characterized by high drainage densities. Low stream orders frequently occur in the Touws and upper Duiwe catchment ranging from 1st to 6th orders (Table 2.1). The Touws River is undammed and due to the incised nature of the catchment remained mostly natural. Limited impoundments occur in the catchment. The natural flow is perennial in both systems where periodic high-rainfall events results in short-lived peak flow periods. The Duiwe River, have several impoundments due to agricultural activities and as a result lower flows to zero flows are common (Russell, 2013).

The geological differences in the two catchments are an important determining factor in the amount of runoff and infiltration that occurs. The mean steep slope of the Touws catchment increases the stream slope and decreases the infiltration rates allowing more runoff. The Duiwe catchment is mostly underlain by granites resulting in gentle slopes favouring infiltration and less runoff. This coupled with the impacts from land use and land cover further reduces the amount of rainfall to runoff. The flows and water quality in both the Duiwe and Touws Rivers as impacted by catchment and land use which is explored in Chapter 3.

2.2.3 Geology, geomorphology and soils

The three main geomorphological formations in the Touws and Duiwe River catchments include the Outeniqua mountain range (Cape Fold Mountains Geomorphic Province), the coastal platform (Southern Coastal Platform Geomorphic Province), and the coastal embayment (Southern Coastal Lowlands Geomorphic Province) (Marker and Holmes, 2010, Partridge et al., 2010, Reinwarth et al., 2013) (Figure 2.2). River catchments in the Outeniqua Mountains (250-750 mamsl) consists mostly of nutrient-poor sandstones and quartzites of the Table Mountain Group (TMG) (Figure 2.3). Predominantly occurring erosion resistant quartzites and sandstones of the Skurweberg (Kouga) formation occurs and in the inter-mountain valleys softer shales of the Goudini (Tchando), Baviaanskloof and Cedarberg formations occur. Iron and manganese contained in the Goudini formation often weathers brown on the surface. The coastal plateau (150-250 mamsl) is an old sea floor dating back to the Tertiary period, which have been deeply incised by rivers, which include the Touws and Duiwe. It is underlain by pre-Cape granite, Kaaiman Group metal sediments (phyllites, schists, shales) and TMG sandstones, mantled in places by alluvium and aeolian deposits. High level terraces from the original erosion surface capped by silcrete and fellicrete occur and remnants remain in places (Marker and Holmes, 2010). The coastal embayment (< 150 mamsl) dates to predominantly the Pleistocene period and some areas of the landform is still in the process of formation. Dune deposits mostly comprises the landform with Quaternary sands in which dune rock or aeolianite has been formed from the cementing of sandy ridges by calcium carbonate. The inland coalesced dune deposits can be traced back to the Middle Pleistocene (± 1 million years ago). When the rivers were cut off by dune cordons the Wilderness lakes formed as it was prevented from flowing directly into the sea (Russell et al., 2012).

The soils of the Touws and Duiwe Rivers upper catchments draining from the Outeniqua Mountains are generally, acidic, leached, low in nutrients, and poor in buffering capacity. The shallow soils derived from the TMG are pale-coloured and peaty and quartzites have low clay contents and few sorption sites which results in decomposition processes that produce clear water but dark brown in colour, stained by humic and fulvic acids (Midgley and Schafer, 1992).

The soils are therefore unsuitable for cultivated agriculture but are suited for afforestation using pines. The Tchando Formation produces soils that are deeper enabling the pines to develop deeper root systems and grow faster. Due to this a portion of the eastern part of the upper Touws catchment and parts of the coastal plateau have been retained as pine plantations while the rest of the plantations have been removed with the aim to restore these areas to fynbos (Pauw, 2009). Deficiencies in phosphate and maganese in pine trees resulted in extensive fertilisation programmes in the upper catchment plantations. Sandstones produce soils in the upper river catchments, which are light textured, acidic and podzolised fine sandy loams. The north-facing slopes have rocky and well-drained soils but poorly drained or peaty soils occur on southern slopes. In the wetter areas dark acidic topsoils with high organic matter content occur at high altitudes under indigenous forests as the humic topsoils are high in organic matter and biological activity, while in drier areas topsoils are usually ash-grey in colour with low nutrients due to podzolisation processes where iron, aluminium and organic matter are stripped from the topsoil and deposited lower down in the profile (Schafer, 1991).

Soils originating from the Kaaiman Group and granites produce deeper, loamy, more fertile soils which are suitable for cultivation. The middle reaches of the Duiwe River catchment occur in this formation. The Kaaimans Formation soils, give rise to red/yellow or red soils, and the soils derived from the granites, appear to absorb the dissolved organic carbon, which results in pale-coloured or colourless water with a higher pH (6-7), and higher conductivities (Midgley and Schafer, 1992). The Kaaimans Group formation is characterised by gently sloping ridge-tops and shallow river valleys which become steep-sided, incised river valleys and then deep gorges as they cut through the southern edge of the coastal platform. The areas underlain by granites are characterised by gentle ridges and valleys that become steeper where gorges develop. Soils of both the Kaaimans Group and granites support the irrigated orchard and irrigated pastures but vegetable farming is only practised in an area underlain by granites (Petersen et al., 2017).

The coastal plateau soils have a clay subsoil at between 300 and 500 mm depth, overlain by a thin concretionary gravel horizon. The soils are poorly drained, acidic and have low biological activity. Crusting and compaction of the topsoil occurs easily due to the fine texturing of the soil (Schafer, 1991, Russell et al., 2012). At the coast the coastal embayment soils are derived mostly from the Pleistocene and the recent Coastal Sands rock (Allanson and Whifield, 1983). The younger dune areas display inceptisols to finely textured, poorly drained podzols and in the older dunes duplex soils occur. The topsoils are high in silt and fine clay while the underlying layer is impervious clay or rock layers and thin ironpans, which restrict drainage. The Lakes Systems floodplain consist of dark alluvium rich in organic matter (Allanson and Whifield, 1983, Schafer, 1991, Russell et al., 2012). The Lakes have developed in the barrier dunes, which spanned at least the last two glacial-interglacial cycles. The phases of aelolian deposition seems to be controlled by interglacial and subsequent interstadial sea-level high stands (Russell et al., 2012). Soil classification was based on the South African Soil Classification (Soil Classification Working Group, 1991) and ARC (2005).

2.2.4 Groundwater

Most of the groundwater monitoring occurs in the greater Gouritz catchment area in the Karoo and the regional scale monitoring is inadequate in terms of density and representivity of the data for interpretation of regional groundwater patterns. Various studies related to groundwater occurred in the catchment such as the impact of groundwater abstraction on ecosystems in the Kammanassie Nature Reserve and surrounding areas. Generally the groundwater from the TMG rocks are of good quality while the quality from the Little Karoo associated with the Bokkeveld and Cretaceous (Uitenhage Group) can be brackish. Along the coastal belt the groundwater use is mainly for stock watering and as a supplement for some urban supplies where good quality is associated with the TMG outcrop areas (DWAF, 2004).

2.2.6 Climate

The weather influencing the Touws and Duiwe River catchments is mainly shaped by a succession of east moving subtropical low-pressure cyclones interacting with subtropical high-pressure anti-cyclones lying over the oceans. The climate can be described as moderate with rainfall throughout the year (aseasonal) with maximum rainfall occurring in September to March (wet season) and the late autumn to winter months of April-August being relatively drier (dry season) (Figure 2.5). The rainfall is orographically influenced where the mountains separate the moist coastal regions from the arid inland Little Karoo (Geldenhuys, 1993). The annual rainfall ranges between 900 to 1400 mm in the upper catchments and decreasing away from the mountains (Görgens and Hughes, 1981). The summers are warm (22-25^{o}C) and winters mild (18-21^{o}C), with occasional bergwinds that will raise the temperatures to the upper 30^{o}C, mainly in spring as a result of strong sub-continental anti-cyclone (low pressure) systems moving from the west past the Southern Cape. The desiccating bergwinds are caused by dry subsiding air moving off the interior plateau in response to strong coastward pressure gradients, which is an important element in the fire patterns in the fynbos and therefore the distribution of natural vegetation in both forests and fynbos over the landscape. Bergwind conditions are often followed by rainfall (Geldenhuys, 1994, Van Wilgen et al., 2010). Rainfall in the catchments are associated with south-westerly winds while fair weather is associated with south-easterly winds. High wind speeds are rare in the area with 97% below 30 km^{-1}.Onshore easterly winds are prevalent during summer (Russell et al., 2012). During winter westerly winds dominate with northerly bergwinds. Mist and cloud cover frequently occurs and results in lower potential annual evaporation rates of about 916 mm per annum (Schulze et al., 2008).

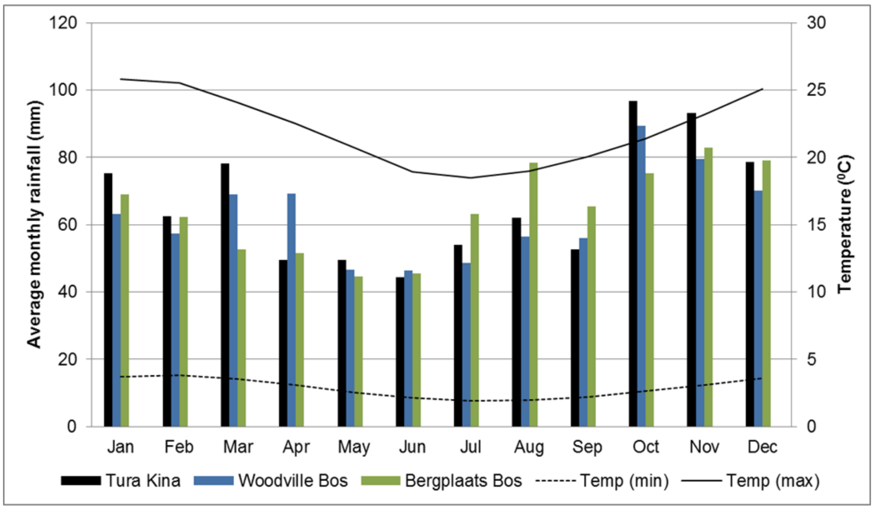


Figure 2.5 Meteorological data for the catchment area.

Data included from rain stations located in the catchments. Data source: South African Weather Service and private landowners

2.2.7 Vegetation

The terrestrial vegetation of the Touws and Duiwe River catchments can be described as either being indigenous forests, plantations or coastal fynbos. Afforestation in the form of plantations occurs in the upper Touws and Duiwe River catchments. The vegetation in the area was well descibed by Mucina and Rutherford (2006) and Vlok et al. (2008a) also compiled a fine-scale vegetation classification, which was done at a 1:50 000 scale and modified by the South African National Parks to be compatible with the land-cover mapping. The main natural vegetation types that occur in the catchments are shown in Figure 2.6. The high forests occur on the coastal plateau and mountain foothills and the variation in topography, soils and the microclimate produces diversity in the species composition and forest structure. A contributing factor to the distribution of the forests are the spatial distribution of fires as determined by weather conditions (Kraaij et al., 2011). Berg winds are associated with a larger incidence of fires or greater size and severity (Kraaij et al., 2011). The indigenous forests are confined in gullies and bergwind shadow areas where they are protected against fynbos fires (Geldenhuys, 1993). The vegetation in the area consists of fire-prone and fire-dependant fynbos areas and fire-resistant or fire-free forests (Kraaij et al., 2011). Although fire is important in maintaining species and endemic richness of fynbos species, the management thereof was influenced by the need to protect plantations so that fynbos in the vacinity of plantations was compromised by short-rotation and low intensity fires (Van Wilgen et al., 1992, Kraaij et al., 2011). The fires in the Outeniqua Mountains appreared to be seasonal occurring mostly during summer months, however, winter posed a fire danger due to bergwinds and low rainfall. Further studies did show that fire in the eastern coastal fynbos was not limited to any season in particular (SANParks, 2014). Fire return periods in the remote catchment fynbos occur every 10-25 years and in fynbos with plantations every 12-15 years. Prescribed burning takes place by plantation managers during October-November and February-March (outside Bergwind seasons) while in the Garden Route National Park managers implements prescribed burning November-April (Kraaij et al., 2011).

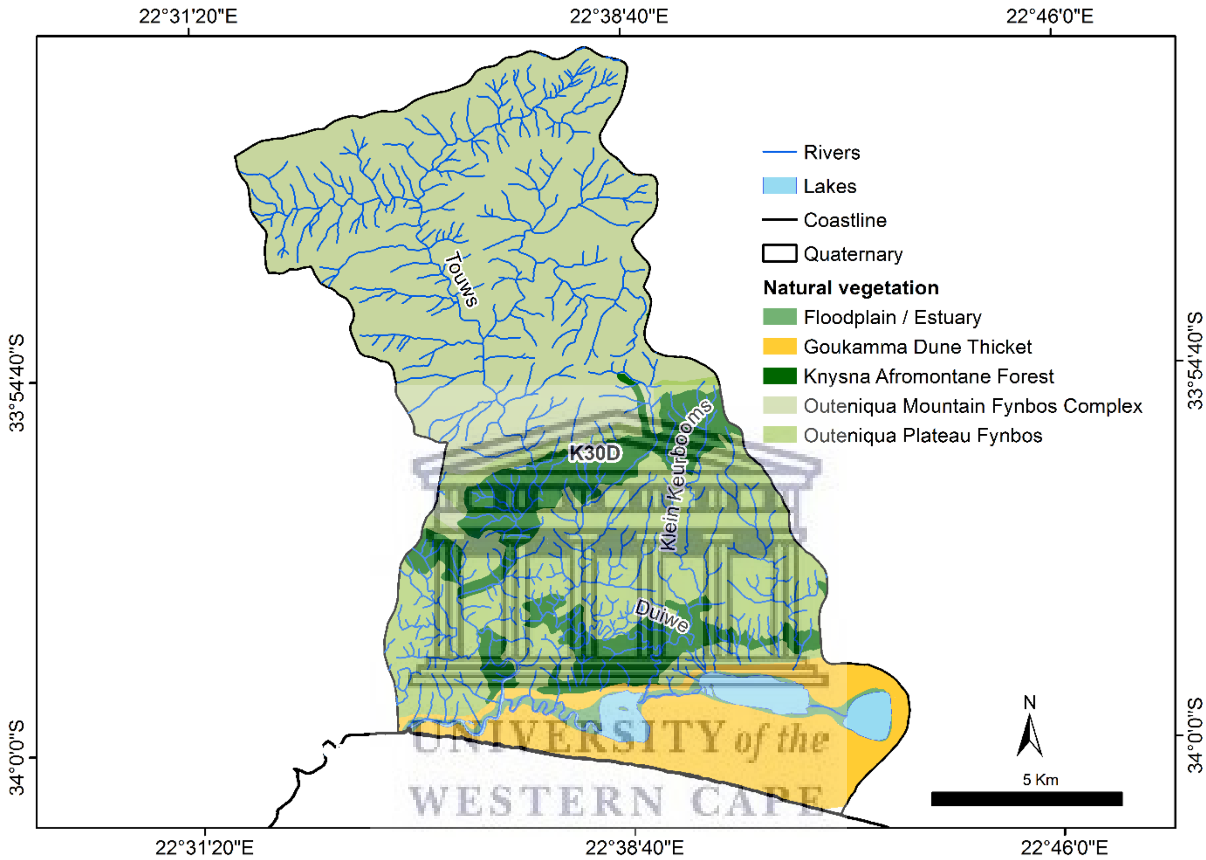


Figure 2.6 The natural vegetation types, Touws and Duiwe River catchments

The Knysna Afromontane Forest is similar to the Southern Cape Afrotemperate forest that stretches along a wide altitudinal range over a very short distance (Geldenhuys, 1993, Mucina and Rutherford, 2006). The forest has a well developed tree layer (> 75% canopy cover) with the understorey consisting of shrubs, ferns and herbaceous species and a ground layer of herbs, sedges and grasses. Common tree species include: *Podocarpus falcatus* (Outeniqua yellow wood), *Apodytes dimidiata*, *Canthium inerme*, *Cassine peragua*, *Curtisia dentata*, *Olea capensis sp*., *Olinia ventosa* and *Trichocladus crinitus* as a common understorey shrub (Geldenhuys, 1993). In the Outeniqua Mountain and Plateau Fynbos tall shrubs from the Proteaceae families form the overstorey, with a middle layer of fine-leaved shrubs (such as Fabaceae, Ericaceae) and a groundlayer of short shrubs, reeds, sedges, geophytes and other herbaceous species. Characteristic species occurring in the Plateau Fynbos are grasses, sedges and restios on northern slopes, and relatively sparse Proteaceae (such as *Leucadendron* *eucalyptifolium*, *Protea neriifolia*) that occur mainly found on the southern slopes (Vlok et al., 2008c). The differences in the vegetation species between the Moutain Fynbos and the Plateau Fynbos is generally due to the well-developed, finer textured soils derived from the Kaaimans Group and the granites favour a different suite of species. Large portions of natural forests still occur in the Duiwe River and Touws River and some lies within the boundaries of the Garden Route National Park (Russell et al., 2012). Fragmented pockets or islands of lowland Fynbos occur throughout the Garden Route National Park area and the landscape, isolated from larger fynbos areas by either expanding indigenous forests, by plantation forestry, agriculture or the development of infrastructure (SANParks, 2014).

The Goukamma Dune Thicket is a mixture of sandplain fynbos and thicket clumps which develop into a low forest in protected sites. The thicket clumps include species such as *Carissa bispinosa*, *Cussonia thyrsiflora*, *Euclea racemosa*, *Olea exasperata*, *Sideroxylon inerme* and *Tarchonanthus camphoratus* (Vlok et al., 2008c). The vegetation on the floodplain that is rarely inundated and dominated by grass and shrubs, while the areas closer to the lakes are frequently inundated and dominated by *Phragmites australis* (reeds) and *Typha capensis* (bulrushes). Species such as *Juncus kraussii* and *Schoenoplectus scirpoideus* occur on the floodplains along with *Phragmites australis*. Vegetation mapping showed that *Phragmites australis* has increased substantially along with grass and tall scrub/trees, with a decline in *Juncus kraussii* and *Schoenoplectus scirpoideus* (Russell and Kraaij, 2008). According to Mucina and Rutherford (2006) azonal coastal vegetation also occur in the area, known as Cape Seashore vegetation, where short Asteraceae fynbos occur mostly on the seaward slopes of primary dunes and other slopes adjacent to the sea. The areas adjacent to the lakes and the Serpentine channel consist of *Passerina* species and annual fynbos herbs while Restoid and grassy dunes are largely restricted to the Rondevlei area consisting of true fynbos elements (Vlok et al., 2008c). A range of dry and mesic and mesic types also occur with dune fybos and Kaffrarian thicket occurring along the coastal dunes (Schafer, 1991).

The estuary shoreline consists of species typical of the saltmarshes such as *Chenolea diffusa*, *Exomis microphylla*, *Salicornia meyeriana* and *Suaeda fruticosa*) and in certain areas along the Wilderness Lakes, which grades into terrestrial communities dominated by species such as *Cynodon dactylon* (Le Maitre et al., 2015a).

Invasive alien plants

Large portions of indigenous vegetation of the Touws and especially the Duiwe River catchments has become invaded by exotic species while other portions of land are utilized for plantations of pines (*Pinus pinaster* and *Pinus radiata*) or Eucalyptus species and agricultural land (Figure 2.7) (DWAF, 2004). The invasive alien plants were mapped for the Garden Route Initiative (GRI) and further refined by SANParks. Figure 2.7 shows the density of alien vegetation invasions together with the land cover in the Touws and Duiwe River catchments. Alien vegetation was introduced to the coastal catchments as early as the 1780's (Phillips, 1963). Afforestation in the catchments dates back to before 1884 (Geldenhuys, 1994, Le Maitre, 2000) and species such as *Hakea* and *Pinus* as well as *Acacia mearnsii* were all introduced for the production of timber, fuelwood or tan-bark due to a lack of fast growing trees in South Africa in the form of plantations. Species such as *Populus* (poplars), *Eucalyptus globulus* (bluegum), *Acacia melanoxylon* (blackwood) and several species of pine trees were introduced to the forest areas (Phillips, 1963). When areas were burnt they were planted with *Acacia dealbata* (silver wattle), *Acacia mearnsii* (black wattle), *Eucalyptus camadulensis* (red river gum), and *Pinus pinaster* (cluster pine) among others (Le Maitre et al., 2015a). Unfortunately, these species have spread beyond afforested areas invading large areas of these catchments (Le Maitre et al., 2002).

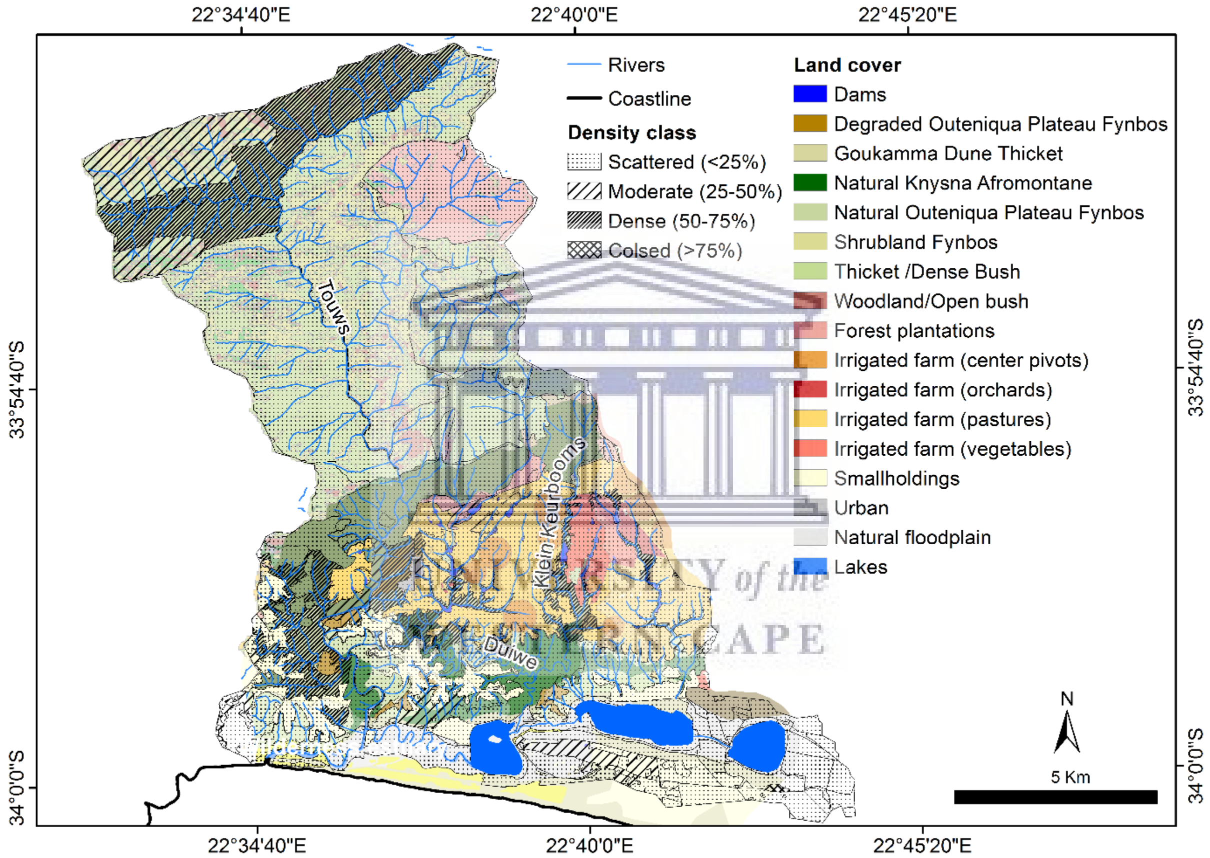


Figure 2.7 Alien vegetation density in the Touws and Duiwe River catchments with present day land cover. Data from the GRI and SANParks

The invasive species *Hakea*, *Pinus* as well as *Acacia mearnsii* are also the major invaders in the Outeniqua Mountain Fynbos Complex areas. Some of the other invading species include *Leptospermum laevigatum* (myrtle), *Rubus* species (brambles), *Solanum mauritianum* (bugweed), *Sesbania punicea* and a variety of smaller shrub, herbs and grasses (Le Maitre et al., 2015a). Invasive species such as pines are known to reduce water flows in rivers by using more water than indigenous vegetation (Dye and Jarmain, 2004, Le Maitre et al., 2004, Le Maitre et al., 2013). Le Maitre et al. (2004) showed that pine plantations reduced river flows in South Africa and Kapp et al. (1995) estimated that the mean annual virgin run-off at 18 million m^3 per year in the Touws catchment was reduced by approximately 21.7% due to mainly pine plantations in the upper catchment.

Acacias such as black wattle (*Acacia mearnsii*) and *Hakea* invades the riparian zones of rivers on the plateau in wetter areas, especially where fynbos originally grew (Henderson, 1998, Dudenski, 2007). Rivers such as the Klein Keurbooms River has dense invasions of black wattles, blackwoods and Eucaluptus species (GRI, 2008). The Afromontane Forest has low density invasions of particularly *Acacia melanoxylon*, while the Outeniqua Plateau Fynbos remnants are invaded by a variety of species, which alters the ecological integrity either by fire protection or exclusion (Kraaij et al., 2011, Le Maitre et al., 2015a). Large percentages of the coastal dunes are covered by alien shrubs and trees especially *Acacia cyclops and Acacia saligna* (used for drift sand reclamation). Alien vegetation clearing programmes are on-going in the study catchment by Working for Water, SANParks and CoastCare (Russell et al., 2012). Table 2.3 summarises the main invading species and the total area covered in the Touws and Duiwe River catchments from the GRI mapping and SANParks updates.

Table 2.3 The main invading species in the Touws and Duiwe River catchments and condensed area (after Le Maitre et al. 2015). Based on GRI mapping and SANParks updates. (Condensed invaded area calculated by multiplying the area of invasion by the percentage invasion cover)

Species	**Common name**	**Condensed invaded area (ha)**
Acacia cyclops	Rooikrans	42.65
Acacia mearnsii	Black wattle	371.69
Acacia melanoxylon	Blackwood	259.34
Acacia saligna	Port Jackson Willow	7.84
Anredera cordifolia	Madeira vine	0.01
Cestrum laevigatum	Ink berry	0.01
Cortaderia selloana	Pampas grass	0.80
Eucalyptus spp	Eucalypts, gums	87.44
Hakea sericea	Silky hakea	604.23
Lantana camara	Lantana	1.52
Leptospermum laevigatum	Australian myrtle	1.16
Opuntia ficus-indica	Prickly pear	2.64
Paraserianthes lophantha	Stink bean	0.04
Pinus spp	Pines	962.20
Ricinus communis	Castor oil bean	0.91
Rubus spp	Bramble	4.27
Sesbania punicea	Red sesbania	0.23
Solanum mauritianum	Bugweed	2.33
Nephrolepis exaltata	Sword Fern	0.05
Grand Total		2352.99

2.2.8 Land use

Land use in the Touws and Duiwe River catchments can be divided into forestry, agriculture natural areas (conservation) and urban areas (Figure 2.4). Detailed analysis of land cover/use in the catchments are explored Chapter 3.

Forestry

Forestry became an important economic activity in these catchments with the first woodcutters post established at Hoogekraal in 1777. The exploitation of the indigenous forests began when timber was required by people settling in the area. Forest destruction occurred over a short time period as the exploitation was unmanaged. Degraded forest areas were sold as farm land. After a wild fire occurred in 1869 burning large tracts of land, a need for protection and better land management arose. Later a systematic forest management system was developed leading to demarcation of the forests and eventual closure of indigenous forests to all wood cutters by 1939. Timber, however, was still in demand and this led to the introduction of alien tree species in the form of plantations. The plantations were largely managed by the national forestry department and private afforestation was encouraged by both companies and private land owners (Le Maitre et al., 2015a). The plantations were controlled by the South African Forestry Company Limited (SAFCOL) and by the 1980’s the mountain fynbos areas were transferred to provincial conservation agencies (Kraaij et al., 2011). By 1994 the forestry industry was privatised by regions and the land was managed by the Mountain to Ocean Forestry (MTO) Company.

Agriculture

Agriculture dominates the land use on the coastal platform, being either plantations, irrigated and non-irrigated pastures for commercial dairy farming and vegetable farming. The majority of the agricultural activity occurs in the Duiwe River catchment with agriculture occurring on a small scale in the Touws River catchment. Farmers have been practicing intensive dairy and vegetable farming in the catchment for more than 60 years, forming part of the major milk producing regions in South Africa. The dairy farming is pasture based (Gertenbach, 2007). Pastures in the area are managed by a once-off ploughing and sowing with kikuyu grass (*Pennisetum clandestinum*) and once established minimum tillage practises are used after which perennial rye grass is sown. On some pastures both grasses are then over-sown with clover annually for nitrogen fixation. This was one adaption that was necessary in order to save on cost as more recently farmers in the region have experienced increases in the costs of energy and inputs such as fertilisers and pesticides (De Lange and Mahumani, 2013). Another change that most farmers implemented, with the exception of the vegetable farming area, has been the switch to centre pivots to maximise irrigation efficiency and monitoring irrigation requirements using tension meters. The vegetable farming requires a different form of irrigation and quick coupling pipes with draglines are used. Vegetables grown include artichokes, brussel sprouts, celery and horseradish, while the rest of the land is rotated with dry wheat, barley, maize or left fallow. Most farmers have also diversified their interests into tourism offering accommodation on farms as an additional income (De Lange and Mahumani, 2013).

Conservation

The exploitation of indigenous forests led to the establishment of an indigenous research station at Saasveld near the town of George in 1964. The system developed made use of a multiple-use conservation management system, which formed the basis of the management in place presently. Multiple-use includes the production of timber from both indigenous and plantations, fire management, alien plant invasion control, soil/water conservation and recreation (Kraaij et al., 2011). In 1964 the Tsitsikamma Coastal and Forest National Parks were proclaimed to protect and conserve the coastal forests of the region and the first marine protected area in South Africa was established. In 1983 the Wilderness National Park was proclaimed to conserve the Lakes system, which includes the Swartvlei system (from 1986), state land in the Wilderness National Lake Area (from 1987), Rondevlei and lands between Rondevlei and Swartvlei Lake (from 1991), and the lower Duiwe River (from 1991) (SANParks, 2010). The park was proclaimed with the aim of protecting the Touws and Swartvlei Systems, collectively known as the Wilderness Lakes, as well as the historic, cultural and natural landscapes. The Rondevlei, Langvlei, Eilandvlei and Serpentine channel were designated as Ramsar sites, according to the Ramsar Convention, as Wetlands of International Importance in the terms of the Convention on Wetlands in 1991 (SANParks, 2014). By 2009 the Garden Route National Park was proclaimed which includes all the above mentioned parks, the indigenous state forests and the mountain catchment areas of the Outeniqua and Tsitsikamma Mountains. These areas are managed by the SANParks (SANParks, 2010). The protected areas in the Touws and Duiwe River catchment are shown in Figure 2.4.

Urban development

Most of the urban development occurred along the coast which is the most densely populated. The town of Wilderness was founded in 1877 near the estuary mouth and it became established as populations grew (Figure 2.4). The economy increased steadily after the 2nd World War and tourism increased from the late 1950s to the 1970s and gave rise to the older housing developments in Wilderness, Wilderness East, Hoekwil and other locations around the coastal lakes. Wilderness became a tourist destination and boarding houses and hotels became established around 1921. Many of the smallholdings (e.g. Wilderness Heights, Hoekwil) became established often by the sub-division of the original farms (Kapp et al., 1995). The population in the Wilderness areas is expanding placing increasing pressures on resources. This also increased the developments of low cost housing settlements such as Touwsranten and Kleinkrantz, which continues to grow (WLSDF, 2015)

2.3 Sampling site selection and description

To achieve the objectives of this study the Southern Cape region in the Western Cape Province was selected as the study area. The area was selected as several studies indicated that the coastal regions were experiencing increasing disasters resulting from floods, droughts and associated water supply problems, fires and sea storms (Nel et al., 2014). The Southern Cape region is particularly vulnerable to climate change as it is in a climatic transition zone between winter and summer rainfall. It is predicted that due to temperature warming and seasonal rainfall changes (more intense storms) due to climate change the natural disasters will increase in frequency. Alternating floods and droughts will impact water security further affecting the people living in the area (Nel et al., 2011b). Agriculture is currently the main water user and coupled with the increasing urbanisation as reported by municipalities (DWAF, 2004), there is a risk of demand exceeding water availability. There have also been reports of deteriorating river water quality impacting the lakes and estuaries in the region threatening the ecotourism of the coastal region, which remains an important generator of income (Nel et al., 2011b). Due to the broad based nature of the environmental issues, the Touws and Duiwe River catchments provided the opportunity to evaluate linkages between catchment management, biophysical interactions and riparian zone morphodynamics as drivers of change on water quality and ecological integrity at a catchment and site scale, together with the key controls on the provision of ecosystem goods and services.

The general study design was to spatially and temporally analyse the historical catchment land use change in the Touws and Duiwe River catchments (at the catchment scale) to assess how land cover and land use changes influenced water quality and river flow. Each sub-catchment has unique characteristics which influence land use and water quality and the purpose was to analyse each one separately (Chapter 3). To illustate the linkages between catchment management, biophysical interactions (Chapter 4) and riparian zone morphodynamics as drivers of change on water quality (Chapter 5) and ecological integrity at a site scale, sampling sites were selected in the Duiwe River catchment. The sampling sites selected would differ in land cover/use with varying impact on water quality along a longitudinal impact gradient. The sites would differ in riparian buffer vegetation characteristics to evaluate the controls on the morphodynamics of riparian zones and so assess the linkages with water quality (ecosystem service provision). A desktop analysis employing Google Earth and digital 1:50 000 topographical maps were used in the selection of sites. Once potential sites were identified, the accessibility and suitability was assessed in a scoping visit to verify in the field. The criteria for site selection were:

1. Sites representative of natural vegetation and alien vegetation invasion. This included partial invasion and complete invasion along a longitudinal gradient.
2. Sites in an agricultural landscape and in a landscape with a low degree of disturbance for a reference measure.
3. Site suitability or representivity. This involved capturing the diverse geomorphic features (instream features e.g. riffles), bank condition (i.e. vegetated banks, bare river banks and degraded banks).
4. Accessibility (e.g. land owner permission and site access) and safety for monitoring equipment.

Site locations with the above criteria was challenging in the study catchment as no spatial replication of the sites occurred with similar geomorphological and hydrological features. There was a lack of suitable foothill sites as most were impacted to varying extents by some form of anthropogenic activity. The coastal rivers in this area are steep and short with a sharp gradient from the foothills before flowing into the sea or coastal lakes. Due to this regional physiographic setting most rivers along the southern Cape coastline have very short foothill zones and no lowland geomorphological zones before entering the sea. This results in flashy river flows associated with periods of increased rainfall (Lubke and de Moor, 1998). All sites selected were classified as perennial flow and were representative of environmental characteristics such as flow, bottom substrate, and land use activities.

At a site scale the Duiwe River catchment provided more suitability and 4 sampling sites (K1-K4) were selected and sampled seasonally over a 2.5 year period (2014-mid-2017); 3 on the Klein Keurbooms River (a tributary of the Duiwe River) and one on the Duiwe River (Figure 2.8b and Figure 2.9). Sampling site characteristics are summarised in Table 2.4. The study sites selected were representative of the various land covers (pastures, contrasting riparian vegetation types) and land uses (agriculture and natural) on water quality. The sampling sites selected ranged from a minimally impacted reach to a cumulative reach impacted by agricultural activities. The sampling sites therefore represented minimal impact (reference conditions at site K1), further downstream a site was located in a riparian zone partially invaded by alien vegetation (semi-indigenous) (site K2) and the third site furthest downstream was a riparian zone almost completely invaded by alien vegetation (degraded) (site K3) and cumulative impact conditions (site K4). Runoff plots were located in two separate areas in the pasture field, one adjacent to the semi-indigenous riparian zone and one adjacent to the degraded riparian zone site (Figure 2.8c). The same sampling sites were used to illustrate the biophysical interactions in the Duiwe River catchment in Chapter 4 and the riparian zone morphodynamics as drivers of change on water quality in Chapter 5. Although the riparian vegetation analysis included the cumulative impact site K4 on the Duiwe River, the runoff plots were only located along the Klein Keurbooms River for the analysis in Chapter 5. Riparian zone vegetation and runoff plot characteristics are summarised in Table 2.5 and shown in Figure 2.10. Sites on the upper Klein Keurbooms River were selected due to accessibility but were representative of the upper Duiwe River.

All sampling occurred on a seasonal basis during autumn, winter, summer and spring. Although rainfall occurred throughout the year as shown in section 2.2.6 (Figure 2.5), wetter periods occurred during September to March with drier periods occurring during April to August. In the analysis of macroinvertebrate and algae data the wet and dry seasons are referred to.

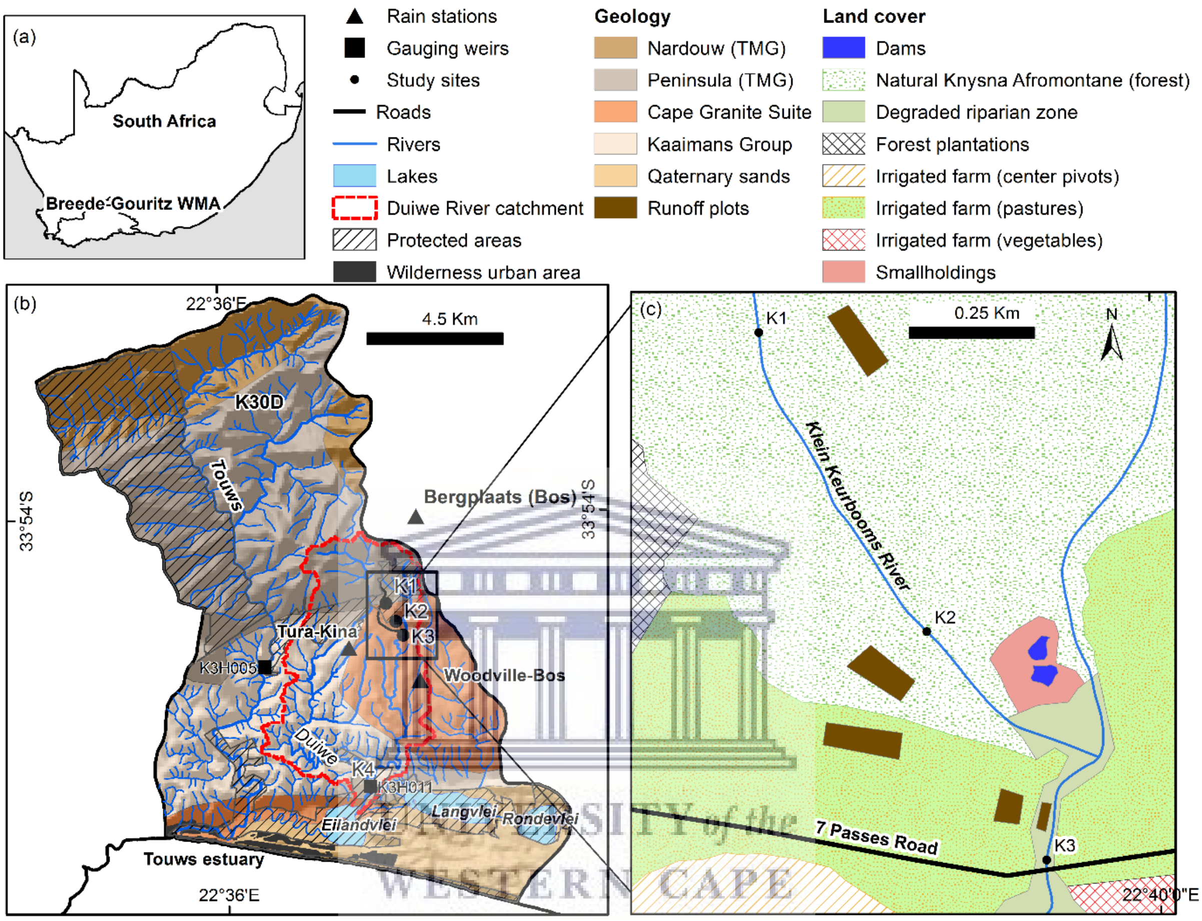


Figure 2.8 (a) The location of the study area within the Southern Cape region of South Arica, (b) the Klein Keurbooms and Duiwe River catchments within the quaternary catchment K30D and sampling sites K1, K2 and K3 on the Klein Keurbooms River and site K4 on the Duiwe River, (c) sites K1, K2 and K3 with runoff plots and land cover/use in black square

Figure 2.9 **The four study sites selected on the Klein Keurbooms and Duiwe Rivers. Site K1on the Klein Keurbooms River during (a) dry season and (b) wet season; Site K2 on the Klein Keurbooms River during (c) dry season and (d) wet season; Site K3 the Klein Keurbooms River during (e) dry season and (f) wet season and Site K4 on the Duiwe River during (g) dry season and (h) wet season**

Table 2.4 Sampling site descriptions

Sites	Klein Keurbooms River			Duiwe River
	K1	K2	K3	K4
Coordinates	-33.927866° 22.659388°	-33.932357° 22.661725°	-33.935326° 22.664873°	-33.981744° 22.651842°
Geomorphic zone	Mountain stream	Upper foothills	Upper foothills	Transitional
Geomorphic characteristics	Step-pool morphology, single channel	Pool riffle morphology; single channel	Pool-riffle morphology; single channel	Pool-riffle morphology, high sinuosity; upstream channel confined in gorge; right bank steep mountainous hillslope
Stream order	1	2	3	4
Wetted channel width (m) dry – wet season	6-8	2.5-5	5.7-7	7-9
Dominant substrate	Cobble (81%) Gravel (19%)	Gravel (65%) Cobble (35%)	Gravel (65%) Cobble (35%)	Gravel (82%) Cobble (6%)
Vegetation characteristics	Indigenous forests	Indigenous vegetation (trees) on the left bank; indigenous vegetation on the right bank invaded by alien trees	Right bank almost completely invaded by alien trees; left bank some indigenous vegetation remains interspersed with alien vegetation	Indigenous vegetation (mostly trees)
Disturbance characteristics	Instream woody debris present	Instream aquatic vegetation and instream woody debris upstream of site	Degraded, downstream channel much narrower due to vegetation overgrowth and fallen trees in channel (large woody debris)	Site in a protected area; causeway in channel; gauging weir downstream; some vegetation clearing
Canopy cover %	80	60	50	0
Average channel depth (m)	<1	0.5	1.3	1.4
River bank stability	Good, localised erosion	Poor	Poor	Stable
Land use impacts	Natural forest	Agriculture (dairy pastures); self-catering cottages (Tourism)	Agriculture (dairy pastures); self-catering cottages Tourism	Agriculture/Natural

Table 2.5 Riparian and runoff plot characteristics (upstream to downstream) along the Klein Keurbooms River

Sites	K1 (Reference)	K2 (Partial invasion)	K3 (Dense invasion)	Agricultural site (adjacent to sites K2 and K3)
Land cover/vegetation	Indigenous forest, riparian zone Undergrowth of grasses and shrubs	Riparian buffer strip along pasture consisting of indigenous and alien trees. Undergrowth of grasses and shrubs, No undergrowth beneath alien trees.	Riparian buffer strip along pastures, densely invaded with alien trees. No undergrowth beneath alien trees.	Grazing pasture 100% grass covered.
Number of runoff plots	3	3	3	6
Average slope angle for runoff plots (degrees)	2 - 3	3.6 - 4	4 - 6	3.4 - 6.4
Runoff plot dimension	1 m x 1 m	1 m x 1 m	1 m x 1 m	1 m x 1 m
% soil composition site and runoff plots	Sand (95%) Gravel (2%) Silt (3%)	Sand (100%)	Sand (97%) Silt (3%)	Sand (65%) Gravel (35%)
Canopy cover % according to Stevenson and Bahls (1999)	80	60	50	No tree canopy
Average slope % for river reach location	10-15	5-10	2-5	2-5
Riparian zone width (m)	25	17-26	6-12	-

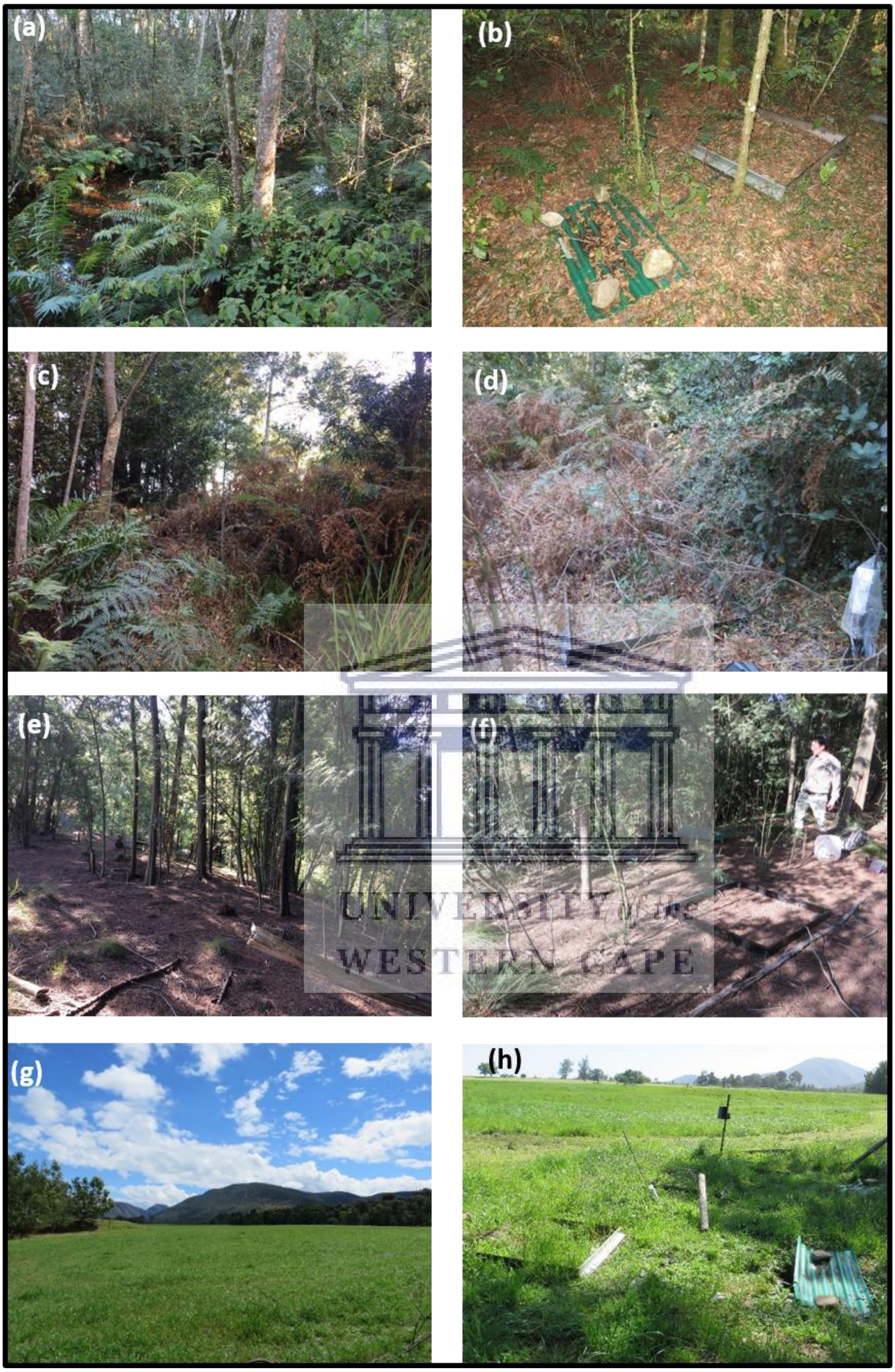


Figure 2.10 The riparian zones and associated runoff plots along the Klein Keurbooms River. (a) Indigenous forest riparian zone at site K1 and (b) runoff plot; (c) Semi-indigenous riparian zone at site K2 and (d) runoff plot; (e) Alien invaded riparian zone at site K3 and (f) runoff plot; (g) Pasture adjacent to site K2 and K3 and (h) runoff plot

2.3.1 Physical and chemical factors of selected sites

Surveyed cross-sections/transects

The surveyed cross-section data was used in the analysis in Chapter 4 and Chapter 5. The physical habitat was assessed at the stream reach scale (150 m), for each site, (Figure 2.8), which was defined as a repeat of geomorphological sequences of either step-pools or pool-riffles depending on the geomorphological zone the site was located in. Cross-section form as outlined in (Gordon et al., 2004) was required to obtain information on changes of river form (bed and banks). Repeat cross-sections were required to ascertain river channel changes over the study period during the low flow period. Three cross-sections were aligned perpendicular to the river channel or direction of streamflow and were spaced approximately 50 m apart at each site. The spacing and number of cross-sections were considered as sufficient resolution to represent the variability that occurred. The TCRP 1205 total station was used, which combines the electronic theodolite and electronic distance measuring device in one unit. A high degree of accuracy is obtained (± 2 cm). Permanent base stations were also set up at each site using a Leica 1200 RTK GPS base station. A CNAV 3000 GPS unit was also utilised to set up a control network. Steel pegs were inserted into the ground and cemented in place as permanent control points that marked the end points of each cross-section. This was necessary to ensure accurate repeat surveys. The water depth (m), temperature (°C) and conductivity (mS m^{-1}) were recorded using Solinst Levelogger Junior LTC (Model F30/M10) water level loggers as well as discharge (m^3 s^{-1}) and wetted channel width (m) obtained from the surveyed cross-sections. An OTT MF pro handheld electromagnetic flow meter was used for discharge measurement on the Klein Keurbooms River and discharge for the Duiwe River was obtained from the Department of Water and Sanitation (DWS) gauging weir present at site K4 (Figure 2.11). The riparian canopy cover, substrate embeddedness and armouring was determined visually according to Stevenson and Bahls (1999).

Figure 2.11 Instream obstructions at site K4 on the Duiwe River (a) shallow causeway for river crossing, looking downstream and (b) DWS gauging weir and water quality monitoring point, looking upstream

Site K1

Site K1 was located in an indigenous forested area with well-developed banks but localised erosion occurred in the form of undercut and scoured banks. Site K1 was characterised by a single thread channel (Figure 2.12a). The active channel width at all cross-sections was < 10 m and the wetted channel widths were between 6 - 7 m with water levels < 1 m. The wetted channel width increased during the survey in March 2014 except at cross-section 1.3 where it increased during September 2014 from 6.47 m to 6.82 m. The reach was classified as a mountain stream channel where the reach type consisted of step-pool morphology at the upstream cross-sections (1.1 and 1.2) and further downstream as pool-riffle (1.3). The channel width was narrower and very similar at cross-section 1.1 and 1.2 with a slightly wider channel occurring at cross-section 1.3. A section of the bed at transect 1.2 was bedrock. The bed material in the pool at cross-section 1.3 was slightly embedded by finer material and the channel bed was gravel dominated. A much steeper left bank occurred (1 - 1.5 m) at all cross-sections with a bench forming at cross-section 1.2 due to bank collapse from previous high flows. The channel cross-section showed very little change throughout the surveyed period and remained stable. A wide forested riparian zone and floodplain occurred on the left bank of the channel and dense vegetation (shrubs and trees) occurred on the right bank limiting accessibility, which resulted in a shorter cross-section at 1.1 than at 1.2 and 1.3.

Site K2

Site K2 was located in an agricultural area and the site occurred at a lower elevation than site K1. The channel remained single thread and the morphology type changed to pool-riffle with pool-riffle-run hydraulic biotopes (Figure 2.12b). An extensive floodplain occurred beyond the right bank, consisting of agricultural pasture areas, while on the left bank an indigenous forest strip occurred. The wetted channel widths were much narrower confined by a steeper vegetated right bank than left bank. Channel depth remained shallow throughout the survey period seldom reaching > 0.5 m in pools during the sampling events. The wetted channel at these site transects were all at about 2.5 m - 4.3 m during each survey and the width increased during the March 2014 surveys. A walking path occurred on the left bank which is reflected by cross-section 2.1 and 2.2 (Figure 2.12b). The left bank was well developed and well vegetated up to the water's edge with no signs of erosion. The right bank, although well vegetated, had undercut banks at cross-section 2.1 and further downstream toward cross-section 2.2 and 2.3 the steep bank had increased bank scour and slumping, which was dominated by alien trees (*Acacia mearnsii*). The vegetation on the right bank at transect 2.1 were indigenous forest trees and shrubs. A lateral bar was present at cross-section 2.2 during all surveys along the lower left bank. At cross-section 2.3 the lateral bar along the lower right bank channel was only present during the survey in 2013 and not during 2014/2015.

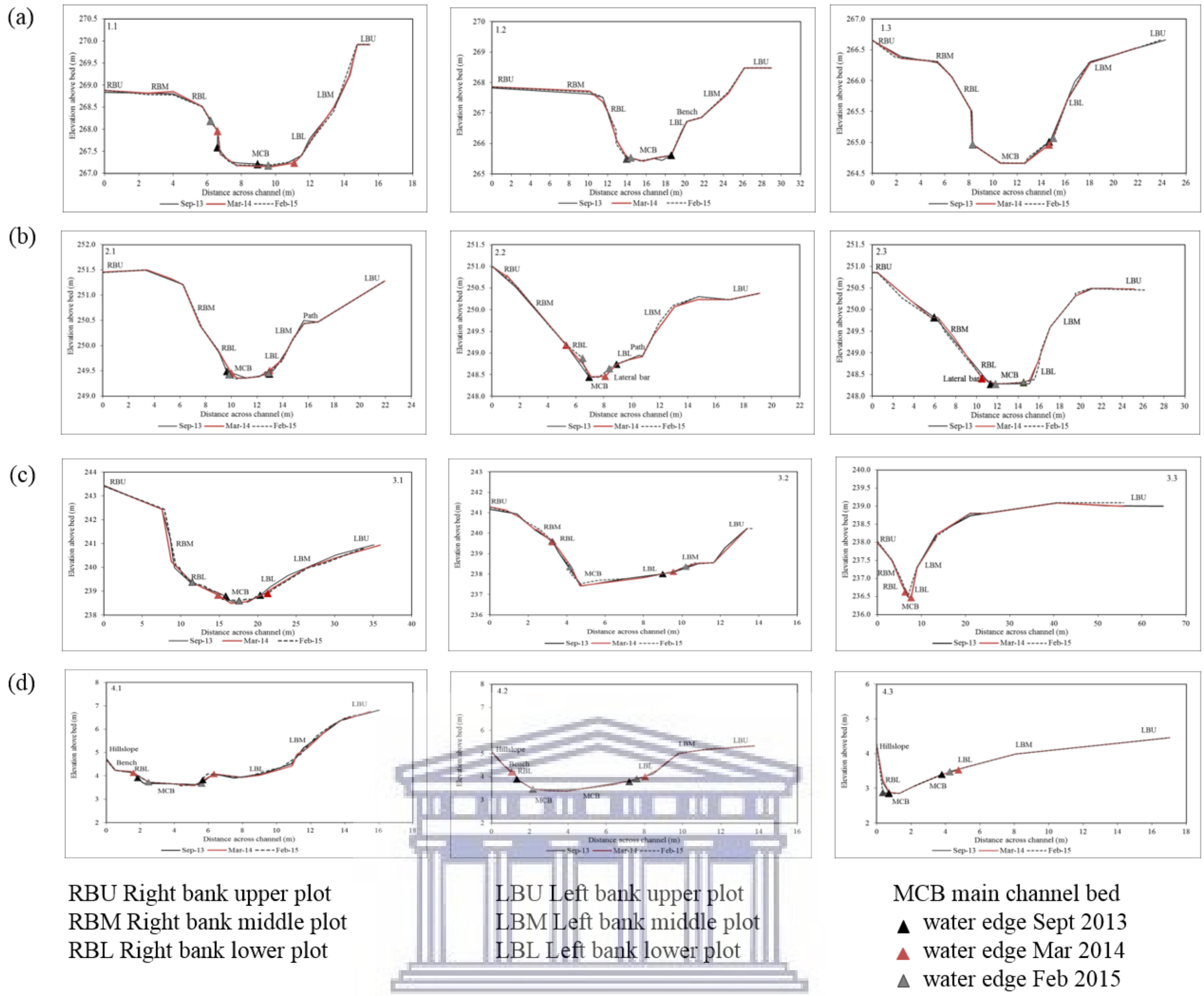


Figure 2.12 Cross-sections and sediment sampling plots (2013-2014) at (a) site K1, (b) site K2, (c) site K3 and (d) site K4

Site K3

Site K3 was located in an agricultural area. The banks at this site were almost completely invaded by alien vegetation, predominantly *Acacia mearnsii* and the entire length of reach at this site had extensive erosion on the right bank in the form of undercut banks and scouring with slumped bank material forming at the base of the bank at cross-section 3.1.(Figure 2.12c). The right bank at cross-section 3.1 and 3.2 was very steep compared to the cross-section upstream and a height of 3 m was measured from the top of the bank to the bottom. The site was located on a private property where land use on the right bank side was agricultural pastures and on the left bank side self-catering accommodation cottages occurred. As a result the riparian vegetation on the left and right bank was often disturbed or even removed on occasion to increase access to the river. Trampling by human interference often occurred and the steep right bank at this site was degraded as a result. Despite erosion occurring during previous high flows, the bank and channel at the cross-section location changed little during the survey period. The channel remained single thread with the morphology type being pool-riffle with pool-riffle-run hydraulic biotopes. Pool depths at this site were much deeper than upstream sites ranging from 1 m to 1.3 m at the deepest section. The wetted channel widths at transect 3.1 and 3.2 ranged from 5.75 m to 6.71 m, the widths increasing during the March 2014 surveys. Upstream of transect 3.3 the channel was completely inaccessible due to woody debris from fallen *Acacia* trees and encroaching vegetation such as shrubs and climbers from both banks. This resulted in a very narrow wetted channel of only 1.29 m, which increased to 1.40 m during the March 2014 survey (Figure 2.12c). The debris dam created resulted in the deeper pool depth at the upstream cross-sections at 3.2 and 3.1.

Site K4

Site K4 was located in a protected area in the Garden Route National Park. An extensive floodplain occurred on the left bank and a steep hillslope on the right bank side (Figure 2.11d). The river in this reach was sinuous and the cross-section 4.2 was located upstream of a meander bend and 4.1 upstream of a shallow causeway (Figure 2.11a), which resulted in a much deeper pool upstream (between 1 and 1.4 m). A gauging weir occurred downstream of cross-section 4.3 (Figure 2.11b). The morphology at this reach was pool-rapid becoming pool-riffle with pool-riffle-run hydraulic biotopes. Large cobble that occurred instream at this site was often embedded by finer material and the bed was gravel dominated. The three transects surveyed were much shorter due to inaccessibility of the hillslope. There were times during low flows (e.g. February 2016) when the riffle area of this site was a dry channel bed and water was confined to the pools. The wetted channel width was narrowest at cross-section 4.3 (3.4 m to 4.1 m) located downstream of the meander bend, compared to the upstream cross-sections (5.06 m to 8.7 m). A bench occurred at cross-section 4.1 and 4.2 on the right bank (Figure 2.12d). The banks were well developed and vegetated with mostly indigenous vegetation. The channel was stable and remained unchanged during all survey periods.

Grain size distribution and soil chemistry

River bank sediments and sediment in the runoff plots were sampled during 2014, 2015 and 2016. This data set was used in the analysis and discussion of Chapter 4 and 5. Samples were recorded at the near surface (0.1 m depth) and upper root zone at 0.3 m depth using a sediment corer from all sites to assess physical and chemical properties in relation to runoff water quality. River banks were sampled from the upper, middle and lower bank positions along transects established for the vegetation survey. Samples were analyzed for particle size, total nitrogen (TN), total phosphorus (TP) and phosphorus as ortho-phosphate (PO_4^{3-}-P), pH, and organic carbon and organic matter (Loss on Ignition %). Organic carbon was determined using the Walkley-Black method and pH was determined by the potassium chloride method. This data was also used in the analysis of environmental variables driving vegetation distribution. This data as well as the sediment grain size data is available in Appendix 2.1 and Appendix 2.2.

Percent substrate type was recorded using the Wolman (1954) method for instream sampling. The particle sizes of bed material were computed by analysing the grain size distribution frequencies of particle sizes within a particular sample. Larger particles that could not be lifted were measured using a tape measure and recorded while finer particles were recorded as finer than the smallest phi/mm size, until 100 - 150 pebbles were measured and tallied (Wolman, 1954, Bunte, 2001). Grain size classes (Table 2.6), based on Bunte and Abt (2001) fines (sand, silt and clay), gravels, cobbles and boulders were included.

Table 2.6 Sediment size classes (Bunte and Abt, 2001)

Class name (Wentworth scale)	Size range, mm (b-axis)	Phi-units (ø)
Clay	0.00006 to 0.0039	14 to 9
Silt	0.0039 to 0.0625	8 to 4
Sand	0.0625 to 2	4 to -1
Gravel	2 to 64	-1 to -6
Cobble	64 to 256	-6 to -9
Boulder	> 256	> -9

Site K1

Instream plots at all transects revealed that grain sizes were very coarse with most consisting of gravel, cobble and boulder (Appendix 2.1) (Figure 2.13). The reach was dominated by cobble, very coarse gravel and boulders. Bed material was poorly sorted and skewness values were between 0.27 and 0.22, which indicated that the material was skewed towards the coarse side. The river banks, irrespective of plot position, consisted of predominantly sand with no cobble or boulders (Appendix 2.1). The middle plots on both the left and right banks at site K1 (1.1 and 1.2) consisted of the highest gravel percentages with the exception of the third transect 1.3, which had no gravel present in the mid-bank plots. Very low percentages of silt occurred at site K1 (< 4% in all plots on both banks).

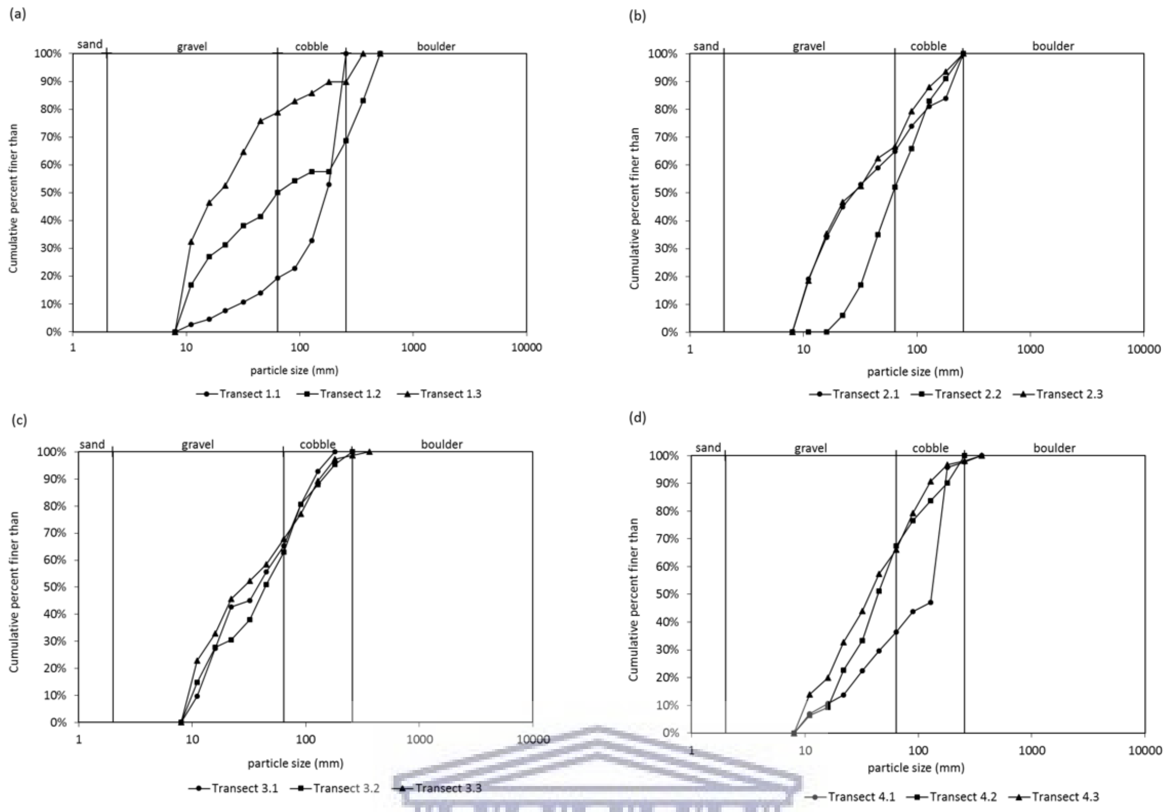


Figure 2.13 Cumulative percentages of grain sizes for instream plots for transects at all sites (all sampled periods); (a) site K1, (b) site K2, (c) site K3 and (d) site K4

Organic carbon percentages were variable throughout all sites and plots. At site K1, cross-section 1.2 had the highest percentages (28%) in the upper right bank plot (Figure 2.14a), while the middle and lower plots on the left bank had 17% and 16% respectively (Figure 2.14b, c). Nitrogen percentages were below 1% at all bank plots. Correlation tests showed a significant positive correlation between nitrogen and organic carbon for all upper bank plots across the sites (Table 2.7). Phosphate (PO_4-P) followed the trend of the total phosphorus (TP). Correlation tests showed that TP and PO_4-P were positively correlated at all bank plot positions. The highest recorded PO_4-P was at cross-section 1.2 with 260.5 mg kg^{-1} recorded at the upper and middle left bank plots and 208 mg kg^{-1} recorded at the lower left bank plot. In the mid-bank plots TP and PO_4-P were significantly positively correlated to gravel and negatively correlated to sand. The soils on all plots on both banks were acidic with pH values 3.6 to 5.9.

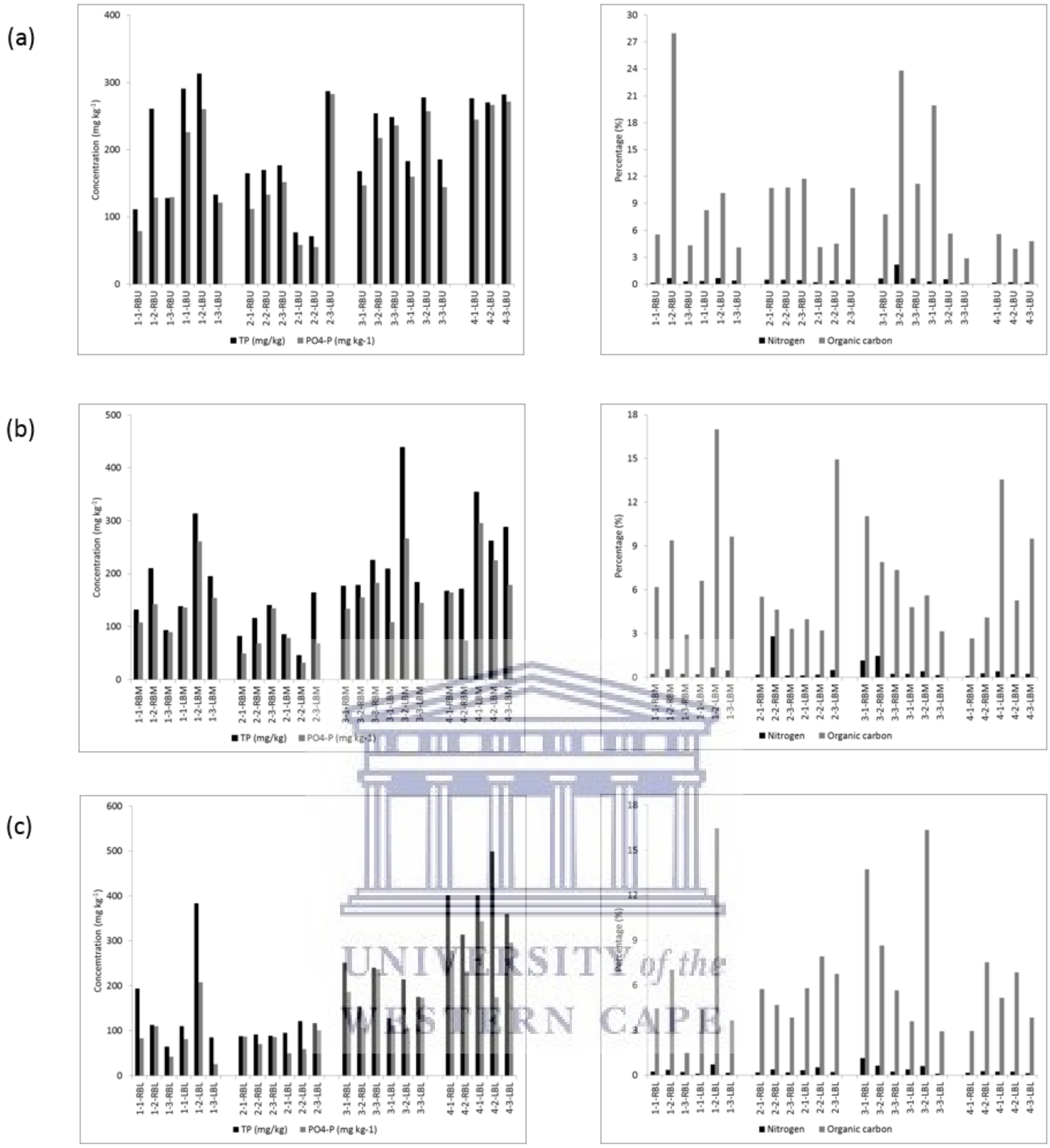


Figure 2.14 Percentage soil nitrogen, total phosphorus, phosphate and organic carbon for all sites (all sampled periods), separated by (a) upper, (b) middle and (c) lower plot positions per bank. RB-right bank, LB-left bank, U-upper, M-middle, L-lower plots

Table 2.7 Correlation results of the Spearman rank correlation (r_s) between physical and chemical soil parameters for the upper, middle and lower bank plots at all sites (all sampled periods). Values in bold are significant at $p \leq 0.05$

Variables	N (%)	P (mg/kg)	PO_4 (mg/kg)	C (%)	pH (KCI)	Mud (%)	Sand (%)	Gravel (%)
Upper bank								
N (%)	**1**	0.225	0.065	**0.674****	-0.240	-0.264	0.167	-0.198
TP (mg kg^{-1})	0.225	**1**	**0.856*****	0.340	0.291	-0.098	-0.080	0.086
PO_4-P (mg kg^{-1})	0.065	**0.856*****	**1**	0.164	0.297	-0.061	0.017	0.008
C (%)	**0.674****	0.340	0.164	**1**	-0.131	-0.277	-0.328	0.307
pH (KCI)	-0.240	0.291	0.297	-0.131	**1**	-0.120	0.071	-0.020
Mud	-0.264	-0.098	-0.061	-0.277	-0.120	**1**	0.057	-0.089
Sand	0.167	-0.080	0.017	-0.328	0.071	0.057	**1**	**-0.994*****
Gravel	-0.198	0.086	0.008	0.307	-0.020	-0.089	**-0.994*****	**1**
Middle bank								
N (%)	**1**	0.406	0.125	**0.673****	-0.081	-0.148	-0.309	0.323
TP (mg kg^{-1})	0.406	**1**	**0.813*****	**0.526***	0.001	0.082	**-0.523**	**0.492***
PO_4-P (mg kg^{-1})	0.125	**0.813*****	**1**	0.361	0.029	0.081	**-0.482**	**0.461***
C (%)	**0.673****	**0.526***	0.361	**1**	-0.092	-0.143	-0.177	0.181
pH (KCI)	-0.081	0.001	0.029	-0.092	**1**	-0.326	-0.210	0.278
Mud	-0.148	0.082	0.081	-0.143	-0.326	**1**	0.340	**-0.488***
Sand	-0.309	**-0.523**	**-0.482**	-0.177	-0.210	0.340	**1**	**-0.972*****
Gravel	0.323	**0.492**	**0.461**	0.181	0.278	**-0.488***	**-0.972*****	**1**
Lower bank								
N (%)	**1**	0.206	0.038	**0.775*****	0.062	0.244	0.137	-0.146
TP (mg kg^{-1})	0.206	**1**	**0.837*****	0.323	0.221	0.292	-0.269	0.237
PO_4-P (mg kg^{-1})	0.038	**0.837*****	**1**	0.148	0.172	0.398	**-0.507***	**0.484***
C (%)	**0.775*****	0.323	0.148	**1**	-0.036	0.195	0.186	-0.198
pH (KCI)	0.062	0.221	0.172	-0.036	**1**	0.094	-0.053	0.045
Mud	0.244	0.292	0.398	0.195	0.094	**1**	-0.158	0.111
Sand	0.137	-0.269	**-0.507***	0.186	-0.053	-0.158	**1**	**-0.997*****
Gravel	-0.146	0.237	**0.484***	-0.198	0.045	0.111	**-0.997*****	**1**

Significance: *** < 0.0001; ** < 0.001; * < 0.05

Site K2

Instream material was poorly sorted and skewness ranged between -0.02 and 0.11 indicating that the bed material was nearly symmetrical, towards coarse material. The banks at this site were more or less uniform in sediment distribution since they were also sand dominated at all plots. The lower left bank plots had the highest percentages of gravel (3.8% to 9.3%) except at transect 2.3 where the upper plot (3.7%) had increased gravel compared to the lower (2.9%) plot. The lower plots on the right bank also had increased gravel percentages (except 2.1) and with the highest percentage occurring at transect 2.3 at 25.7%, which was due to the accumulation of the lateral bar.

The upper right bank plots showed similar organic carbon content (10.73 to 11.74%) with lower percentages on the left bank (< 5%) with the exception of plot 2.3 on the left bank (10.75%).

On the middle left banks organic carbon further increased at transect 2.3, up to 14.94%. The levels in the lower plots on both banks were below 8%. Nitrogen was less than 1% except at the middle plot at transect 2.2 where the recorded level was 2.82%. Nitrogen and organic carbon was significantly positively correlated at all bank plot positions (Table 2.7). The highest phosphorus and PO_4-P levels were in the upper bank plots on the right bank at all cross-sections at site K2 (plot closest to the agricultural land), except at transect 2.3 on the left upper bank plot. The soils on all plots on both banks were acidic with pH values 3.9 to 4.9.

Site K3

The instream sediment was similar to transects at site K2, was poorly sorted and the skewness was nearly symmetrical towards coarse material. The plots on both banks were dominated by sand with hardly any silt present, which was < 4% at all plots, except transect 3.3 with the most present at 8% on the upper left bank plot (Appendix 2.1). Gravel percentages increased toward the lower plots on both banks at transects 3.2 and 3.3 but the right bank lower plot at transect 3.3 had the highest percentage at 18.7%.

Organic carbon was highest at transect 3.2 on the upper right bank (23.8%) followed by 3.1 on the upper left bank (19.4%) (Figure 2.14). The organic carbon content increased at transect 3.1 on the right middle (11.05%) and lower (13.7%) bank compared to the upper bank. A substantial increase also occurred on the lower left bank at transect 3.2 to 16.3% compared to the lower percentages in the upper and middle plots. The nitrogen content was higher on the right bank than the left at site K3 at all bank positions. The nitrogen levels increased at transect 3.2, right upper (2.15%) and middle (1.47%) bank plots compared to the upper bank at transect 3.1. Phosphates were highest at the upper left banks at 3.2 (257.2 mg kg^{-1}), and 3.3 (144.3 mg kg^{-1}) and on the right upper bank at 3.2 (217.42 mg kg^{-1}). The middle right bank at 3.2 decreased to 154.41 mg kg^{-1} compared to the upper plot and levels further decreased on the lower right bank at transect 3.2. At transect 3.3 middle right bank phosphate levels were decreased at 181 mg kg^{-1} compared to the upper and lower right bank plots at 236 mg kg^{-1}. The soils on all plots on both banks were acidic with pH values 3.5 to 5.8.

Site K4

At site K4 the right bank was a hillslope so sample plots were only accessible until mid-bank and were inaccessible at transect 4.3. Instream sediment at this site was poorly sorted and the majority of material consisted of coarse gravel with smaller percentages of cobble and boulders. A low causeway occurred between transect 4.1 and 4.2 and as a result the upstream pool at transect 4.1 was much deeper and the coarse material were always embedded in fine sediment due to deposition from upstream sediment transport. The instream material at transect 4.1 was also skewed towards coarse material. Skewness values for transects 4.2 and 4.3 were

0.06 and -0.01 respectively, indicating material were nearly symmetrical. Both banks were sand dominated at all plot positions (Appendix 2.1). The lower plots of the right bank at transect 4.2 and lower left bank at 4.3 consisted of increased percentages of silt at 16.1% and 19.8% respectively. The left bank occurred at a lower elevation resulting in deposition during increased flows at transect 4.3. The transect was also located downstream of a meander bend where deposition typically occurs on the inner bend (lower left bank). At transect 4.2 the increased silt was also due to deposition as the right bank had a lower elevation bench present (Figure 2.12d) on which fine material was deposited during higher flow events.

Organic carbon content on the left bank upper plots was very similar (3.9% to 5.58%) (Figure 2.14a). At transect 4.1 and 4.2 the right bank middle plots contained less organic carbon content than on the left bank middle plots. On the lower bank plots the organic content increased at cross-section 4.2 on both banks compared to the middle plot. The highest organic carbon content occurred in the middle left bank at transect 4.1 (13.6%). Nitrogen levels were very low across all plots at all transects (< 1%). Phosphorus content of the middle and lower plots on the left bank were generally increased compared to the right bank plots. The soils on all plots on the both banks were acidic except at transect 4.2 and 4.3 where slightly higher values occurred (5.6 to 6.3).

Groundwater site characterisation

Electrical Resistivity Tomography (ERT) was used to determine groundwater table depth and possible interactions thereof with vegetation at the study sites. The resistivity tomography technique provides a rapid method to acquire a pseudo-section or picture of the subsurface according to varying resistivities. Variation in resistivity in the subsurface units mainly occur due to changes in salinity of the groundwater, changes in porosity or changes in water saturation (Telford et al., 1990, Loke and Barker, 1996), which is then related to the expected changes in the properties of the subsurface. The method is therefore effective in the delineation of water tables due to changes in water saturation. There were no boreholes or piezometers present in the study area to use to calibrate the groundwater level data as determined by resistivity surveys but the method did provide a rapid means to obtain such data. Two surveys were completed during September 2013 and February 2015 (Figure 2.15). The profiles were located parallel to the river and adjacent to the degraded riparian strip, on the right bank side of the river. The profiles had the same location on both sampling occasions. The method used is described in Loke and Barker (1996) and Telford et al. (1990).

The electrical resistivity tomography (ERT) interpretation was carried out by Soltau and Peek (2015). According to DWAF (1999) the underlying aquifer is classified as intergranular and fractured with a yield potential of 0.1 – 0.5 L s^{-1} with good average groundwater quality indicated by an average electrical conductivity of 70 – 300 mS m^{-1}. The profile acquired during

September 2014 (Figure 2.15a) shows a higher resistivity (> 380 ohm.m) layer in the shallow subsurface (up to 10 m depth below surface) which is likely indicative of unsaturated sandy material. The sediment samples collected from this area indicated that the composition was 80 - 90% sand. Below this is a lower resistivity layer (90 – 480 ohm.m), which is indicative of the underlying saturated sand. The profile showed a mostly level groundwater table, mimicking the topography, which can be expected from an unconfined aquifer. The deepest water level (approximately 10 m below the surface) occurred towards the northern end of the profile and the groundwater level occurred at a shallower (approximately 4 m below the surface) depth towards the south. A similar but drier profile was acquired during February 2015 (Figure 2.15b) (Soltau and Peek, 2015). A higher resistivity was observed indicating a dry shallow subsurface sandy layer, underlain by a saturated sand layer with lower resistivity. The groundwater table was deeper (approximately 12 m) towards the north and shallower (approximately 5.5 m) towards the south (Soltau and Peek, 2015). The groundwater table level was approximated and delineated for both surveys, which is indicated by the dashed line in Figure 2.15a and b.

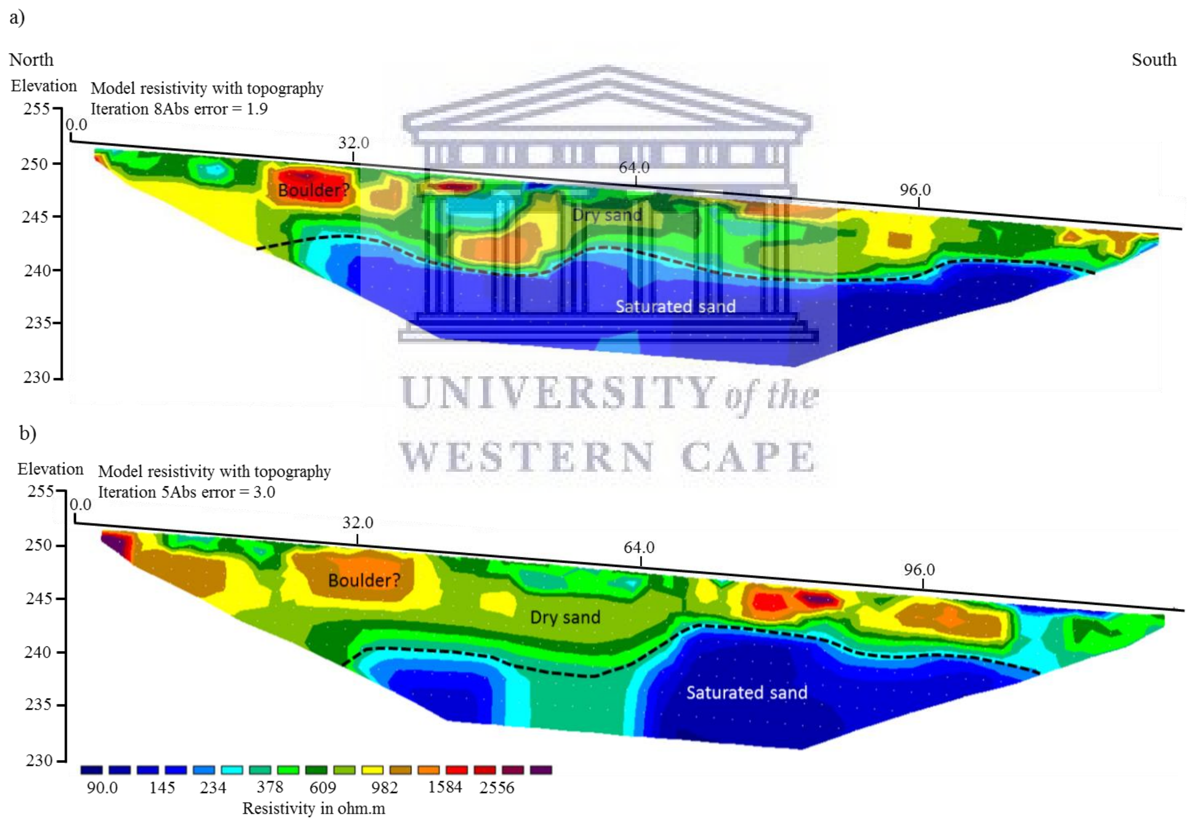


Figure 2.15 Electrical resistivity profiles. Modified from Soltau and Peek (2015)

Vegetation sampling

The study area is in a region where river/riparian knowledge is limited. Detailed vegetation analysis was necessary to explain the type and species composition of the indigenous, semi-indigenous and degraded riparian zones. The riparian vegetation type in each of these descriptions was used to explain instream macroinvertebrate biota and algal species composition and distribution in Chapter 4. The knowledge of vegetation distribution and taxonomic type along the river banks was used in the synthesis Chapter 6 and to make recommendations regarding buffer zone vegetation in Chapter 7.

In each of the three riparian zone areas assessed (indigenous, semi-indigenous, degraded), the riparian vegetation was characterised along three transects oriented perpendicular to the valley thalweg, covering both the right and left channel margins (Table 2.5). The same cross-sections surveyed for the physical geomorphological characterisation was used as vegetation transects. Nine transects in total were established at each of the three riparian zone areas. The vegetation at each site was quantified using an area-based vegetation sampling method by means of quadrats over a two year sampling period (2014 and 2016) (Kent and Coker, 1992). The locations of the quadrats were located on the upper, middle and lower banks (including the water's edge) to represent the different bank positions in the riparian zones and the size was based on the area available at each position, the species richness and growth forms. A standard size of 2 X 1 m sample quadrats was used. In each quadrat the following variables were recorded: cover abundance, health (alive or dead), type (e.g. herb, shrub or tree) (Table 2.8), species (indigenous or invasive), height (short, medium or tall) and position (upper, middle or lower bank, or bar). For each quadrat the cover of each species was visually estimated as the proportion of the ground occupied by a perpendicular projection of the aerial parts of each species expressed as a percentage. The maximum height of the tallest individual of each species, composition and the percentage of live plants (e.g. live plants – high percentage, 100%; dead plants – low percentage, 0%) was estimated. The cover and abundance values were estimated for species using the Braun-Blanquet cover-abundance scale methodology (Gordon et al., 2004). Visual estimates of the substrate cover were also made e.g. where bare ground was present or leaf litter occurred and disturbances were also noted. The riparian zone widths varied at each transect but were most extensive in the forest with no impacts (a width of 25 m was assessed). At site K2 the riparian widths at transects 2.1 and 2.2 and 2.3 were 17 m, 26 m and 26.2 m respectively while at the degraded site K3 the widths were 6 m, 12.7 m and 12 m for transects 3.1, 3.2 and 3.3 respectively.

Plants were identified to species where possible or else the genus was recorded. All vegetation species as well as plant age estimates were recorded in the field by an experienced botanist familiar with the species of the study area. Where species could not be positively identified in the field they were collected, pressed and identified by the South African National Parks (SANParks) and information on plant species descriptions were also obtained from the South African National Biodiversity Institute (SANBI) from digitized collections accessible at the website: http://www.sanbi.org/information (accessed July 2016). Further descriptions were also obtained from Vlok et al. (2008c), Goldblatt and Manning (2000), Manning and Goldblatt (2012) , Bromilow (2010) and Snijman (2013).

Table 2.8 Growth form definitions. Adapted from Mucina and Rutherford (2006) and Smith-Adao (2016). Information was also obtained from the SANBI using the website: http://posa.sanbi.org/intro_posa.php (accessed July 2016).

Growth form category	Definition
Climber	A weak-stemmed plant that derives its support from climbing, twining or creeping along a surface.
Geophyte	An herbaceous plant that responds to an unfavourable season with underground storage structures such as rhizomes, bulbs, tubers and corms.
Grass (Graminoid)	An herbaceous plant in the family Poaceae.
Helophyte	An herbaceous plant resting in marshy ground.
Herb	A plant that is not woody but soft and leafy (a few centimetres to > 1 m). May be annual, biennial or perennial.
Hydrophyte	An herbaceous plant with morphological adaptations to spend at least part of its life cycle submerged under water.
Scrambler	An herbaceous plant that produces long weak shoots by which it grows over other plants.
Shrub	A woody perennial plant often with multiple stems (<1 to 2 m).
Succulent	A plant with fleshy and juicy stems and leaves that contain reserves of moisture.
Tree	A woody perennial plant usually with main truck and multiple stems or branches (2 to > 10 m). If taller than 10 m, a distinct elevated crown is present.

2.3.2 Physico-chemistry characterisation of selected sites

River physico-chemistry

Samples for water chemistry were collected on a seasonal basis at all sites. This data set was used for analysis and discussion in Chapter 4 and Chapter 5. All water samples were collected in pre-rinsed, 1 L polyethylene bottles and placed on ice in the dark, with no addition of preservatives. In addition to seasonal samples, monthly water chemistry and flow data were also recorded at the downstream cumulative site K4 where more impact was expected, which were downloaded from the South African Department of Water and Sanitation (DWS) database (DWS, 2015a, b). This data set was used in Chapter 3 and Chapter 4. Analysis was conducted by the DWS laboratories according to the methods outlined in DWA (2009). These databases had a longer time series recorded from July 1998 - February 2016. The water sample analysis was based on methods applied by the CSIR Stellenbosch Analytical Laboratory test method manuals. Water samples were analysed for electrical conductivity (EC), pH, nitrate (NO_3^--N) and nitrite (NO_2^--N) (from here on referred to as NO_x), total inorganic nitrogen (TN, which is the total NH_4^+-N, NO_3^--N and NO_2^--N,), total phosphorus (TP) and phosphate (PO_4^{3-}-P), alkalinity, sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), chemical oxygen demand (COD), silica (Si) and chlorophyll a (Chl-*a*) (benthic and suspended in water column) per sampling season. Nitrate was quantitatively reduced to nitrite by passage of the sample through a copperized cadmium column. The nitrite (reduced nitrite plus original nitrite) was determined by diazotizing with sulphanilamide followed by coupling with N-(1-naphthyl) ethylenedaimine dihydrochloride (SALM, 2018c). To determine TN samples were thermocatalytically converted to nitrogen monoxide, which is determined with the aid of electrochemical detectors/sensor or chemodetector (CHD). Digestion was performed in the multi N/C 3100 by thermocatalytic high-temperature oxidation in the presence of platinum catalyst, enabling a quantitative digestion for complex nitrogen compounds (SALM, 2017). The TP in samples were determined by hot plate digestion where samples were then filtered through 0.45µm filter and aspirated with a concentric nebuliser and cyclonic spray chamber into the Inductively Coupled Plasma (ICP). The Thermo iCAP 6500 simultaneous ICP with Charge Injection Device (CID) detector was for detection with iTEVA software for quantification (SALM, 2018d).

The COD in samples was based on the chemical decomposition of organic and inorganic contaminants, dissolved or suspended in water. The COD test indicates the amount of water-dissolved oxygen consumed by the reducing agents or contaminants, during two hours of decomposition from a solution of boiling potassium dichromate (SALM, 2018a). The alkalinity was calculated by titration of the known volume of sample to a predetermined endpoint, using an auto-titrator (SALM, 2019). The cations, sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}) were determined using Inductively Coupled Plasma optical emission spectrometry (ICP OES). Samples were filtered through a 0.45µm syringe filter and then aspirated with a concentric nebuliser and cyclonic spray chamber into the Inductively Coupled Plasma. The Thermo iCAP 6500 simultaneous ICP with Charge Injection Device (CID) detector was utilised for detection with iTEVA software for quantification (SALM, 2018b).

River samples collected were refrigerated and analysed within a maximum of two days. The nutrients phosphorus and nitrogen are essential contributors to the growth of algae and therefore changes in these variables may influence the concentrations during sampling events. Silica is also important to diatom development thereby influencing the composition and biomass (Oberholster, 2011, Ewart-Smith, 2012). Water quality results were compared to the Target Water Quality Requirements (TWQR) guidelines according to DWAF (1996c).

Runoff quality

Runoff water samples were collected from all runoff plot containers to assess quality from the land covers used in the study. A composite sample of runoff was sampled quarterly (2014-2017), at the end of rainfall events, after thorough mixing of the sample to suspend the contents. This data set was used for analysis and discussion in Chapter 5. All water samples were collected in pre-rinsed, 1 L polyethylene bottles and placed on ice in the dark, with no addition of preservatives. Samples for runoff water chemistry analysis were collected coinciding with river water sampling. The water sample analysis was based on methods applied by the CSIR Stellenbosch Analytical Laboratory test method manuals. Samples were analysed for electrical conductivity (EC), pH, total alkalinity ($CaCO_3$) and the dissolved inorganic ions sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}) as well as the nutrients phosphorus as ortho-phosphate (PO_4^{3-}-P), nitrate (NO_3^--N) and nitrite (NO_2^--N) (hereafter referred to as NO_x-N), nitrogen as ammonium (NH_4^+-N), total nitrogen (TN) and total phosphorus (TP). Additional analyses undertaken for the runoff samples were dissolved organic carbon (DOC), turbidity and suspended solids (SS).

Chapter 3: Links between catchment land cover and physico-chemistry

Two publications originated from Chapter 3; one was published in Water SA in 2017 and another presented and published as a conference proceedings at the 11th International Symposium on Ecohydraulics in Melbourne, Australia as indicated below:

Petersen, C.R., Jovanovic, N.Z., Le Maitre, D.C., Grenfell, M.C., 2017. Effects of land use change on streamflow and stream water quality of a coastal catchment. *Water SA*, 43, 551-564. Available online: http://dx.doi.org/10.4314/wsa.v43i1.16.

Petersen, C.R., Jovanovic, N.Z., 2016. Historical land use change and its influence on stream ecosystems and water quality. Paper 25998 in: Webb J.A. C.J.F., Casas-Mulet R., Lyon J.P., Stewardson M.J. (Ed.), 11th International Symposium on Ecohydraulics, Melbourne, Australia. Available online: https://researchspace.csir.co.za/dspace/handle/10204/9114?show=full.

3.1 Introduction

Water is a limited resource and water quality deterioration is one of the main threats to water resources in Southern Africa given the prevalence of the natural climatically driven water scarcity. Recent national scale assessments of water resources showed that freshwater and estuarine ecosystems are highly threatened (Nel et al., 2011c). Moreover the cumulative impacts from freshwater systems are received by coastal ecosystems and the effects of land cover change can negatively impact productivity, biodiversity and ecological functioning of these ecosystems (Lemley et al., 2014). As coastal ecosystems are often of economic and ecological significance to a country (Barbier et al., 2011) estuarine management can no longer be limited to the coastline. Managers are realising that their strategies need to extend further inland to include the impacts on water quality and flow in rivers feeding these systems (Schlacher and Wooldridge, 1996, Lemley et al., 2014).

An understanding of the influence of land management activities and their uses within a catchment is required, to assess the impact of non-point source pollution on water quality. This, in turn, is related to spatial and temporal scales and catchment variability (Chu et al., 2013, Ding et al., 2015). Land use and land cover are equally important, where land use refers to the management practices influencing water quality associated with the activities of humans and land cover refers to the biophysical or physical cover observed (e.g. crop types) (Dabrowski et al., 2013). Much of what we observe in present day landscapes and its associated ecosystems is a product of the past. Therefore it is critical to study past events when trying to holistically interpret the present. One way of interpreting the response of rivers to land use change is by historical analysis (Spink et al., 2010). Aerial photography provides important insights into the history of landscape development and river responses. For example, Parsons and Gilvear (2002) illustrated the re-establishment of nature conservation corridors as landforms evolved with changes in river management. Hoffman and Rohde (2007) used repeat landscape photography to illustrate the environmental history of an area. As land cover/use and rivers are intimately linked the opportunity is provided to assess how changes to this can impact rivers and their water quality, whether quality is determined by factors such as climate, landscape and geology or whether land use and management plays a role (Mehdi et al., 2015).

Hydrological processes such as evapotranspiration, interception, infiltration and percolation may change due to modifications of land cover type and can alter the water balance of a catchment (Nosetto et al., 2012). The management of land use activities can play a role, e.g. altering river channels or changing the imperviousness of a surface (Schulze, 2000, Brummer et al., 2016) so surface water discharge increases or decreases (Royall, 2013) altering the flow in rivers and groundwater and affecting the physical, chemical and biological processes in the receiving water bodies (Tong and Chen, 2002).

Studying the past allows us to place recent and ongoing changes on water quality into perspective, and adopt a proactive rather than reactive approach to river and land management so that adaptive management can be implemented (Spink et al., 2010). Although there have been several studies elsewhere (e.g. Ierodiaconou et al. (2005), Hardison et al. (2009) and Yu et al. (2013)), there have been few studies of African coastal catchments. This is especially true where the rivers are an important water supply source for the human population, economically and for the provision of ecosystem goods and services, such as along the Southern Cape coast.

The Touws and Duiwe Rivers are an important water source for domestic, agricultural and the eco-tourism sectors. Deteriorating water quality originating from poor land management in the upstream catchment areas, especially of the Duiwe River catchment, has resulted in biophysical changes to the downstream Ramsar site, the Wilderness Lake System. This chapter focused on land cover and use and how it changed on a catchment-wide scale and within 100 m buffer zone (including the riparian and adjacent area) of two mainstem rivers, the Touws and Duiwe, thereby influencing water quality and flows. The chapter is primarily designed to meet objective 1 of the study (Chapter 1, section 1.4). Historical analysis was used to investigate the causal linkages between land cover/use change, management activities and water quality over time and space. Each of the sub-catchments studied have unique characteristics which influence water quality and the aim was to analyse each one separately. This chapter was aimed at deducing; 1) how has land cover/use changed over time and 2) how has the land use (directly and indirectly) influenced water quality and flows in the past and present. The chapter is divided into a methods (Section 3.2), results (Section 3.3), and discussion and conclusion (Sections 3.4 and 3.5). The methods deals with the historical aerial photograph interpretation, land cover change analysis and historical water quality chemistry and surface flows and historical rainfall records. The key chapter findings are interpreted in the discussion using all results.

3.2 Methods: Data collection and analysis

Two types of data sets were used. Secondary data sources were used for the hydrological information for the Touws and Duiwe River catchments, which included surface water quality, surface river flow and rainfall records. The historical land cover and use were derived from aerial photographs and analysis were conducted at two scales; a catchment scale and within a 100 m buffer from river edge on the left and right bank side. The buffer included the mainstem Touws River and in the Duiwe River drainage catchment of the mainstem Duiwe River (with main tributaries) and the Klein Keurbooms River.

3.2.1 Mapping

To gain an understanding of land cover change and use over time and space, aerial photographs were mapped from 1980 to 2013 for the Touws and Duiwe River sub-catchments as well as the buffered areas. The historical analysis provided an initial context for water quality comparison in each catchment since water quality response to the historical pattern of land use change and catchment impacts can be immediate or have a lag effect. The aerial photographs were obtained from the South African Department of Rural Development and Land Reform: Chief Directorate DLA-CDSM, National Geo-Spatial Information. Digital, orthorectified aerial photographs were obtained for the years 2006, 2010 and 2013. Each set of photographs was orthorectified to a common scale and later mosaicked. Images were orthorectified by the Centre for Geographical Analysis (CGA) at Stellenbosch University, South Africa. Topographical data was provided by the 5m Stellenbosch University Digital Elevation Model (SUDEM) and used as a base layer. Overlapping aerials were further matched to each other using 10-12 tie points per overlap, and were orthorectified using cubic convolution as a resampling method using the PCI Geomatics OrthoEngine 2013 software. The same area was assessed and digitised on-screen using ArcMap 10.2 (ESRI) and the resulting maps were overlain to observe changes in land cover/use and give a total estimate of the amount of change for each of the years for the mapped categories. The mapping boundary used were sub-catchments delineated by Maherry et al. (2013). The major types and land cover mapped included categories for natural forests, areas of natural and degraded vegetation, irrigated pastures, vegetables and orchards, urban and rural developments, rivers, dams, natural lakes and undeveloped where land development was minimal. Some of the classes were derived from the National Landcover map (NLC 2000) (Van den Berg et al., 2008) for South Africa. The Garden Route Initiative (GRI) mapping (Vlok et al., 2008b) assisted with the delineation of the vegetation types since this was done at a finer scale. Field surveys (e.g. field observations, vegetation sampling and recent photographs) were used to cross-check delineations made from the historical aerial photographs. The percentage for each land cover category was calculated for the sub-catchment and 100 m buffer using ArcMap 10.3 for all mapped years.

3.2.2 Physico-chemistry, rainfall and flow data

Historical surface water quality was obtained from DWS where the Resource Quality Services maintains a database (DWS, 2015b). There were two gauging stations where the river water quality has been sampled on a bi-monthly basis; one on the Touws River (K3H005) and another on the Duiwe River (K3H011) (Figure 3.1). There are limitations using such databases as the data were collected for monitoring purposes without a focused systematic experimental design. Even with limitations, it was still the best and most comprehensive historical dataset available for the Touws and Duiwe Rivers (Mantel et al., 2010). A gauging weir was also located further downstream in the Touws catchment but data were inconsistent and not long-term so only river sampling stations with long-term data were utilised. The two gauging weirs used were the only river sampling stations with long-term data (> 10 years). The available water quality data were downloaded from the database for the period 1980-2013 (33 years) for the Touws River and 1998-2013 (15 years) for the Duiwe River. Records were more consistent for the Duiwe River than the Touws River where more missing values occurred in the 36 year period and data were selected from the date where the first complete set of analysis occurred (from 1977). Major water quality parameters measured at both the Touws and Duiwe River stations, which were used in this study were: electrical conductivity (EC), pH, total alkalinity and the dissolved inorganic ions measured: potassium (K^+), sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), chloride (Cl^-), sulphate (SO_4^{2-}), fluoride (F^-), silica (Si) as well as the nutrients phosphate (PO_4-P) as phosphorus, nitrate (NO_3^--N) and nitrite (NO_2^--N) (NO_x) and ammonium (NH_4^+-N) as nitrogen. The analysis of nutrients was conducted by the laboratories of DWS according to the methods outlined in DWA (2009). Flow data for the same sites were obtained from DWS (DWS, 2015a).

Statistical analysis was applied on water quality and flow data. This included the non-parametric Spearmans rank correlation (r_s) test and the seasonal Mann-Kendall trend test in XLSTAT (2015 Addinsoft). The Spearman rank correlation is a non-parametric measure of statistical dependence between two variables, which assesses how well the relationship between two variables can be described using a monotonic function (Helsel and Hirsch, 1992). This described the relationship between flow and the water quality variables. The trend tests were used to determine if the water quality parameters measured generally increased (or decreased) over some period of time in statistical terms (Helsel and Hirsch, 1992). All analyses were assumed statistically significant at $p \leq 0.05$.

Daily rainfall data were obtained from the South African Weather Service (SAWS) database for two rainfall stations and a third was obtained from a private landowner. Rainfall and streamflow data were analysed to detect trends as well as to clarify wet and dry years and used to determine if there were any links to water quality. A hydrological year was classified as dry (25th percentile or less), wet (75th percentile or greater) or normal (between the 25th and 75th percentiles) (du Toit and O'Connor, 2014). The simple daily intensity index, which is the annual total precipitation divided by the number of rainy days (precipitation ≥ 1 mm) in the year, were calculated in RClimDex for all stations (Zhang and Yang, 2004).

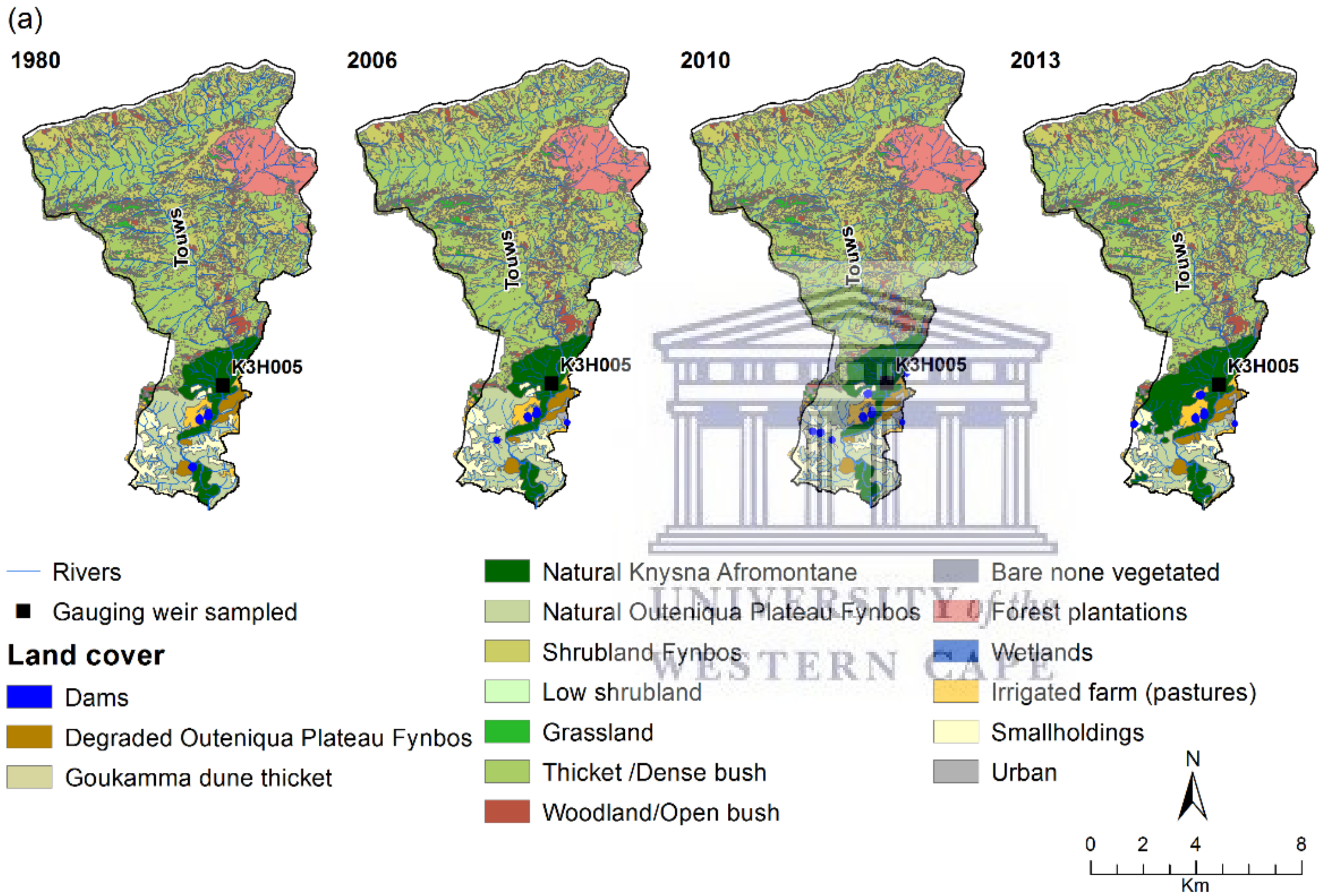
3.2.3 Links between land cover and surface physico-chemistry

To determine if there were any significant correlations (spatially) between water quality and land cover, Principal Component Analysis (PCA) was performed using XLSTAT (2015 Addinsoft) so that the influence of land cover data on water quality within the buffer area and sub-catchments could be examined. The land cover compositions (in km^2) were extracted for each of the sub-catchments and a buffered area of 100 m along each of the rivers (using the buffer facility provided in the ArcMap 10.3 GIS software) was used to determine any spatial changes in land cover/use. In this way any influence of land cover data on water quality could be detected. The buffer was examined as the sub-catchment heterogeneity may mask any signal from land cover/use impact and the impacts may be more pronounced from the area closest to the rivers. Spearman rank correlation was used and results were considered statistically significant at $p \leq 0.05$. The mean values for all water quality variables mentioned in section 3.2.2 were used in the PCA analysis. Spearman rank correlation was used and results were considered statistically significant at $\alpha = 0.05$.

3.3 Results

3.3.1 Mapping

The Touws River sub-catchment remained largely natural in its upper catchment throughout the 33 year period (Figure 3.1a, Table 3.1) as the natural forests areas were formally protected from the late 1800's. The buffer area land cover is mostly natural as well (Table 3.2). Exotic pine plantations cover 672 ha in this catchment. The most notable change occurred in the lower Touws River catchment on the coastal plateau and coastal plain areas with the establishment of irrigated pastures and smallholdings. Smallholdings increased between 1959 and 1980 (2 to 4%) but decreased again from 318 ha in 1980 to 266 ha in 2013 (Table 3.1). None of the agricultural land use impact is reflected in the buffer area (Table 3.2). In the buffer area smallholding remained consistent throughout the mapping period. Irrigated pastures in the Touws catchment increased from 130 ha in 1980 to 137 ha in 2013. Some dams became established with the increase in agriculture but the percentage remained small. Degradation and/or clearing of the natural Outeniqua Plateau fynbos occurred to make way for smallholdings and pastures, shown by the decrease in area from 1980-2013 (Table 3.1). This same trend is evident in the buffer area for the Natural Outeniqua Plateau fynbos.


(a)
1980
2006
2010
2013
Touws
K3H005
Rivers
Gauging weir sampled
Land cover
Dams
Degraded Outeniqua Plateau Fynbos
Goukamma dune thicket
Natural Knysna Afromontane
Natural Outeniqua Plateau Fynbos
Shrubland Fynbos
Low shrubland
Grassland
Thicket /Dense bush
Woodland/Open bush
Bare none vegetated
Forest plantations
Wetlands
Irrigated farm (pastures)
Smallholdings
Urban
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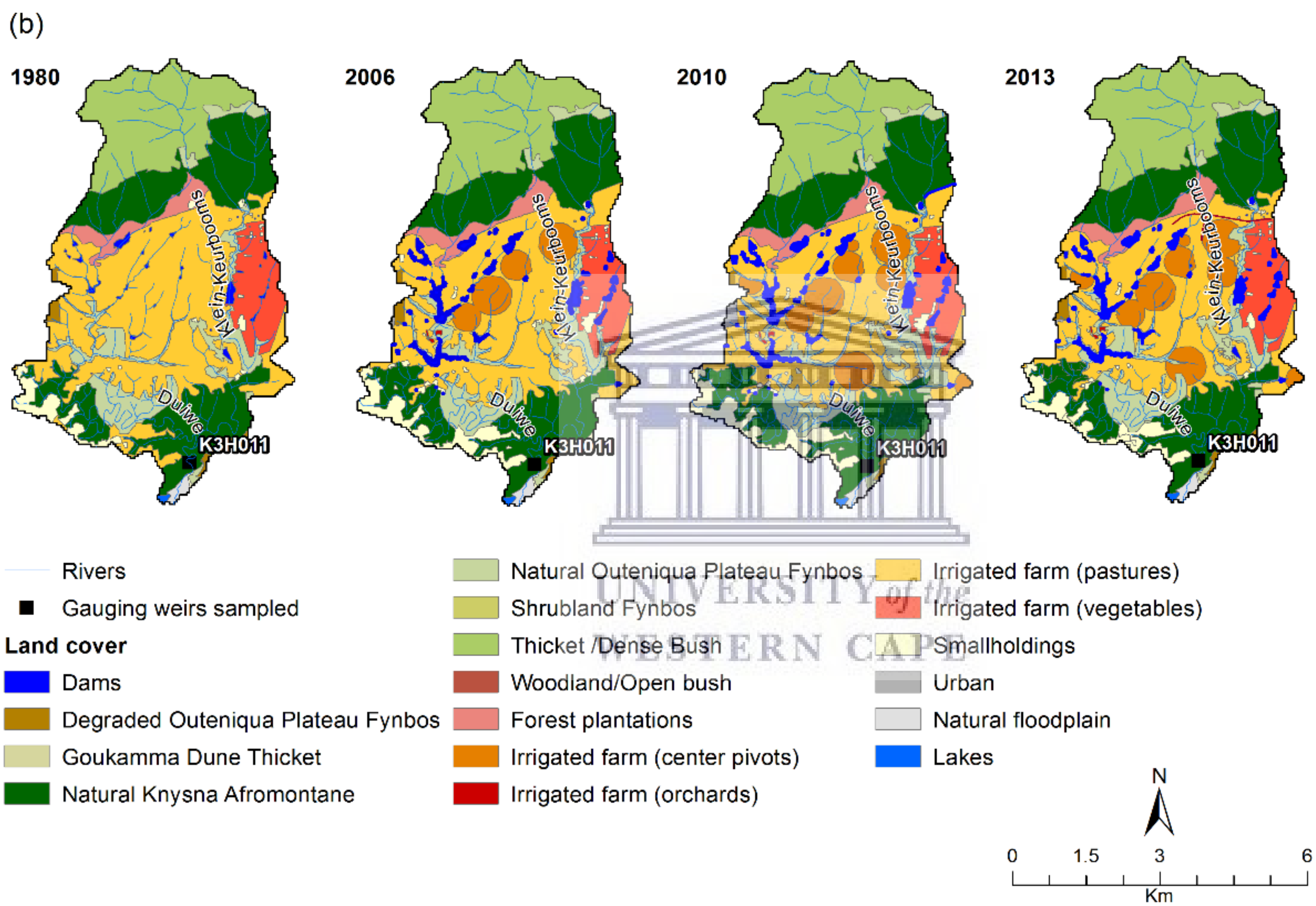


Figure 3.1 Land cover change maps for the a) Touws and b) Duiwe River catchments (1980-2013)

Table 3.1 Summary table of the area of land cover change (ha) from 1980-2013 in the Touws and Duiwe River catchments for the mapped categories

	Years							
	Touws				**Duiwe**			
Land cover category	**1980**	**2006**	**2010**	**2013**	**1980**	**2006**	**2010**	**2013**
Dams	2	1	2	2	32	53	53	50
Degraded Outeniqua Plateau Fynbos	149	110	103	128	25	17	17	26
Natural floodplain	0.1	0	0	0	14	14	14	16
Natural Knysna Afromontane	586	586	586	871	837	844	844	834
Natural Outeniqua Plateau Fynbos	670	670	673	444	335	344	344	335
Goukamma Dune Thicket	1	8	8	--	--	--	--	--
Bare none vegetated	33	33	33	31	--	--	--	--
Grassland	194	194	194	194	--	--	--	--
Low shrubland	1	1	1	1	--	--	--	--
Shrubland Fynbos	2538	2553	2553	2540	1	1	1	1
Thicket /Dense bush	3809	3814	3814	3790	538	537	537	537
Wetlands	7	7	7	7	--	--	--	--
Woodland/Open bush	432	442	442	438	0.2	0.2	0.2	0.2
Plantations	672	672	672	673	95	96	96	99
Irrigated pastures	130	115	114	137	1202	953	953	919
Irrigated vegetables	--	--	--	--	200	184	184	173
Irrigated orchards	--	--	--	--	--	2	2	3
Irrigated center pivots	--	--	--	--	--	161	161	237
Smallholdings	318	321	319	266	92	165	165	130
Urban	--	15	21	20	--	--	2	11
Total	9542	9542	9542	9542	3371	3371	3371	3371

-- Category not observed

The Outeniqua Plateau fynbos occurred along a portion of the lower Duiwe River catchment (Figure 3.1b) and intact pockets remained between agricultural and indigenous forest areas. Much of the coastal plateau landscape was dominated by agricultural areas with 1202 ha (36%) covered by irrigated pastures in 1980 and 200 ha (6%) covered by vegetable farming in the catchment. The same trend was observed in the buffered area (Table 3.2). As agriculture intensified, more water was required to support agricultural activities so the number of farm dams increased, from 32 ha to 53 ha (0.9% to 1.6%) by 2006 (Table 3.1). In the buffer area the farm dams increased from 3% in 1980 to 5% by 2013 in the Duiwe catchment. The increase in farming prompted an increase in human settlement, which, equated to the establishment of smallholdings from 3% in 1980 to 4% in 2013. The Knysna Afromontane forests increased in the Touws by 2013 and remained approximately the same in the Duiwe catchment (Table 3.1). However, some of this increase in the Touws catchment was also due to invasions by exotic black wattle.

A substantial shift in agriculture occurred after 1980 with the introduction of centre pivot irrigation systems in the Duiwe River catchment. By 2010 the amount of area under centre pivots had increased (5% to 7%) but there was little change in the agricultural areas (Table 3.1). The buffer area of the Duiwe River showed similar increases from 5% in 2006, 7% in 2010 and approximately 6% in 2013 (Table 3.2). In the Touws catchment, the Outeniqua Plateau fynbos made way for a formal low cost housing project, Touwsranten, which was initiated in 1996. The housing project expanded to reach its current extent (from 15 ha in 2006 to 31 ha in 2013) mainly occurring in the Touws River catchment (Table 3.1). The urban land cover category was not present in the buffer area of the Touws or Duiwe Rivers.

Table 3.2 Summary table of the area of land cover change (ha) from 1980-2013 in the Touws and Duiwe River buffer area for the mapped categories

	Years							
	Touws				Duiwe			
Land cover category	**1980**	**2006**	**2010**	**2013**	**1980**	**2006**	**2010**	**2013**
Dams	0	0	0	0	18	36	36	35
Degraded Outeniqua Plateau Fynbos	14	15	14	20	0	0	0	3
Natural floodplain	0	0	0	0	10	10	10	12
Natural Knysna Afromontane	104	102	103	102	222	222	222	220
Natural Outeniqua Plateau Fynbos	39	40	39	34	141	149	150	143
Goukamma Dune Thicket	0	1	1	1	--	--	--	--
Bare none vegetated	3	3	3	3	--	--	--	--
Grassland	3	3	3	3	--	--	--	--
Shrubland Fynbos	120	120	120	120	--	--	--	--
Thicket /Dense bush	257	257	257	257	55	55	55	55
Wetlands	2	2	2	2	--	--	--	--
Woodland/Open bush	17	17	17	17	--	--	--	--
Plantations	61	61	61	61	26	26	25	26
Irrigated pastures	130	115	114	137	229	165	150	163
Irrigated vegetables	--	--	--	--	11	9	8	8
Irrigated orchards	--	--	--	--	0	1	1	1
Irrigated centre pivots	--	--	--	--	0	34	52	43
Smallholdings	2	2	2	2	6	9	8	7
Total	623	623	623	623	718	718	718	718

-- Category not observed

3.3.2 Surface physico-chemistry, flows and rainfall

The three rain stations showed the same trends with very wet and very dry periods occurring in the same years, which were linked to the occurrences of floods and droughts in the catchment. The coefficient of variation (Table 3.2) between the rain stations was very similar and analysis of the number of rain days showed a decreasing trend for all stations but the rain intensity at each station overall was increasing (Figure 3.2). The daily flow data for the Duiwe River indicated that periods of zero flows often occur. Overall flow records for the Duiwe River showed a decreasing trend (Tau = -0.095; p = < 0.0001) while the trend increased in the Touws River (Tau = 0.079; p =< 0.0001). Rainfall showed a better correlation to the flow data in the Touws River (R^2 = 0.565) than in the Duiwe (R^2 = 0.175) but in both catchments river flows increased when rainfall increased (Figure 3.3) so that both were responsive during particularly wet years (Table 3.3). However, the rainfall effects on the river discharge in the Duiwe River are reduced due to increased anthropogenic activities in the catchment related to dams and run-of-river abstractions. The location of the rain station that was used was also further upstream in the catchment compared to where flows were recorded in the Duiwe River.

Table 3.3 Rain gauges [station code] showing the coefficient of variation percentage (CV), dry (25th percentile or less) and wet years (75th percentile or greater) periods. Values in brackets represent rainfall (mm)

Station name	CV %	≤ 25 percentile	≥ 75 percentile
Bergplaats-Bos [29294]	29	1984(669), 1986(665), 1987(657), 1990(657) 1998(493), 2004(648), 2005(409), 2008(572), 2009(462)	1981(1428), 1983 (870), 1989(871), 1994 (904), 1996 (1007), 2006 (980), 2007(1214), 2011 (984)
Tura-Kina [Private farm]	28	1998 (569), 2001(577), 2005 (474), 2009(497)	1981(1508), 1982(1013), 1983 (863), 1992(917), 1993(991), 1994 (877), 1995 (953), 1996 (948), 2003 (863), 2006(1114), 2007(1259), 2011(965)
Woodville-Bos [29297]	32	1980(643), 1984(578), 1986(651), 1988 (605), 1991(644), 1998(515)	1981(1388), 1993(858), 1995(853), 1996(896)

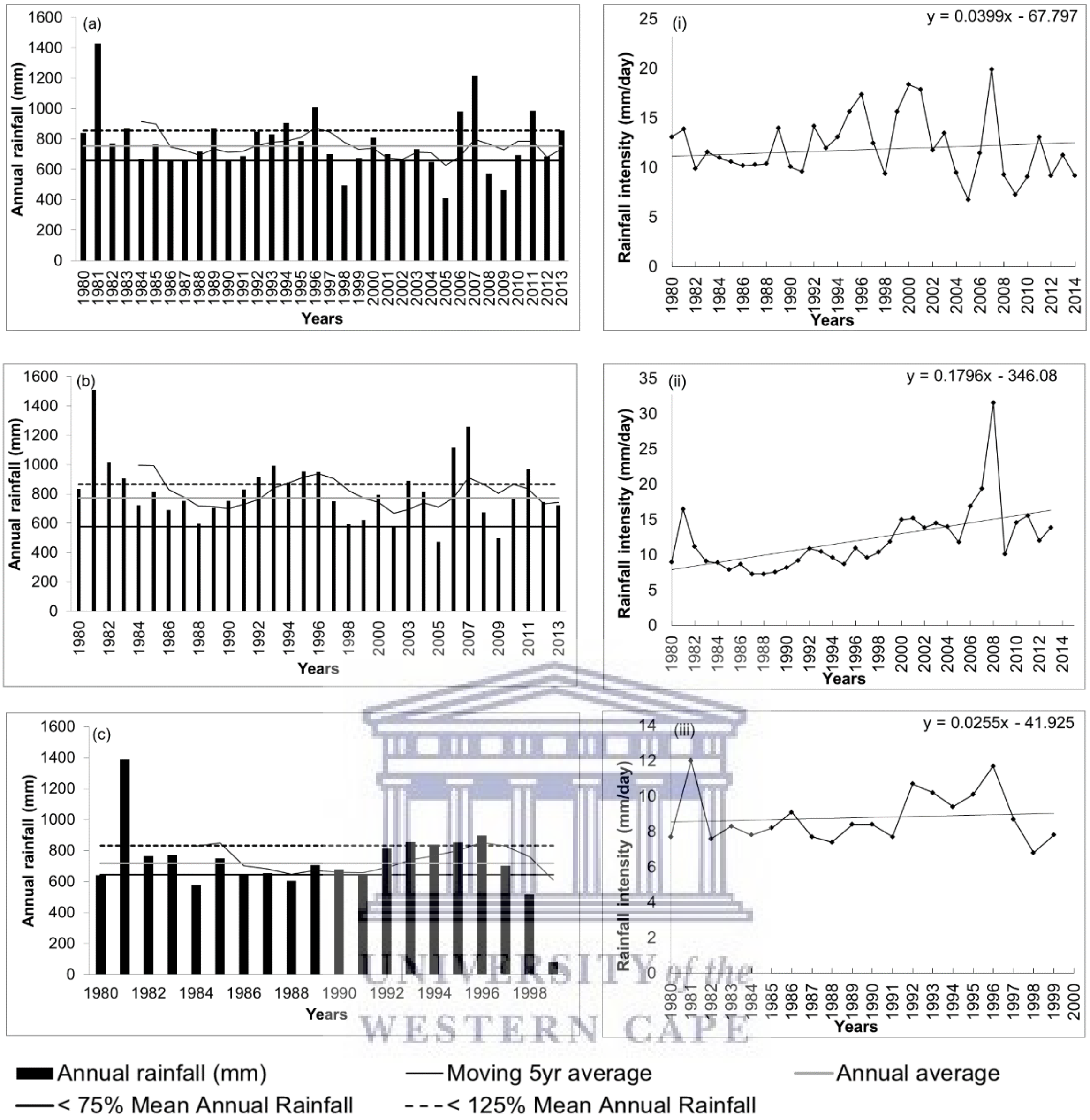


Figure 3.2 Annual rainfall data and daily rainfall intensity index (mm/day) measured at: (a) (i) Bergplaats-Bos; (b) (ii) Tura-Kina (1980-2013); (c) (iii) Woodville-Bos (1980-1999)

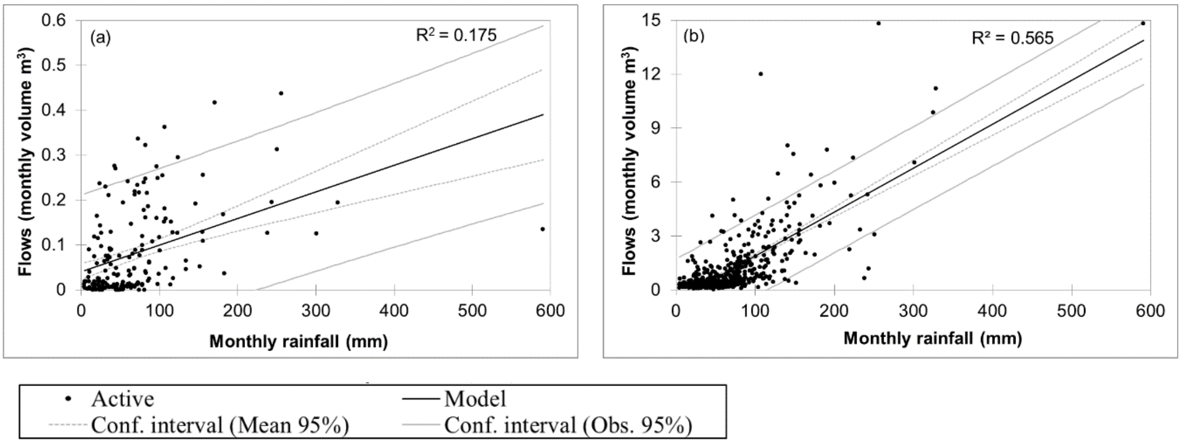


Figure 3.3 Monthly rainfall for the (a) Duiwe and (b) Touws Rivers plotted against monthly river flows. (data from Tura-kina for period 1980-2013)

The annual runoff coefficients showed high variability throughout the years but generally increased for the Touws River and decreased for the Duiwe River (Figure 3.4)

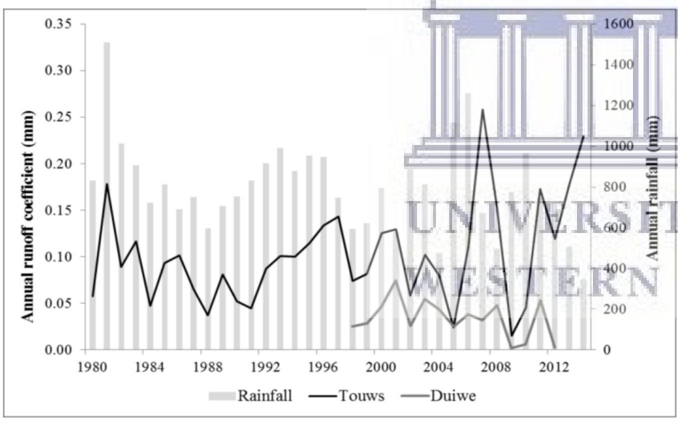


Figure 3.4 Annual runoff coefficients for the Touws and Duiwe River catchments

By the time flow and water quality monitoring commenced (1998) in the Duiwe catchment, agriculture was well established with grass pastures dominating the land cover. No pre-development runoff data are available for these catchments, so a dataset of mean annual runoff, which was estimated from mean annual rainfall using regional rainfall-runoff relationships developed for water resource studies (see (Scott et al., 1998a) and updated by Nel et al. (2013), was used to estimate the pre-development mean annual runoff for each study catchment (Table 3.3). The Touws River accounted for more than half the runoff volume. The gauged mean annual runoff (MAR) from the Touws catchment is approximately 14.1 10^6 m^3 per year, which was based on the flow record for 1980-2013 from DWS, which was relatively close to the estimated pre-development mean annual runoff. For runoff from the Duiwe catchment it was only 0.96 10^6 m^3 per year compared with an estimated 6.6 10^6 m^3 per year prior to development (pre-1950's).

Table 3.4 Estimates of the pre-development mean annual runoff from sub-catchments with relative contributions based on rainfall-runoff relationships

Catchment	Area (ha)	MAR (million m^3)	% total area
Touws	9581	16.32	52.70
Duiwe	3382	6.6	21.9

Water quality of the Touws River can be characterised as acidic (mean pH: 4.3) and low in dissolved substances shown by EC concentrations with a mean of 17 mS m^{-1} (Figure 3.5). The Duiwe River is more alkaline (mean pH: 7.5) and had a mean EC concentration of 134 mS m^{-1}. In the Duiwe River EC followed the trends of the rainfall and flows (Figure 3.6) recorded in the catchment, resulting in a negative correlation with river flows (r_s = -0.46, $p < 0.05$) (Table 3.4).

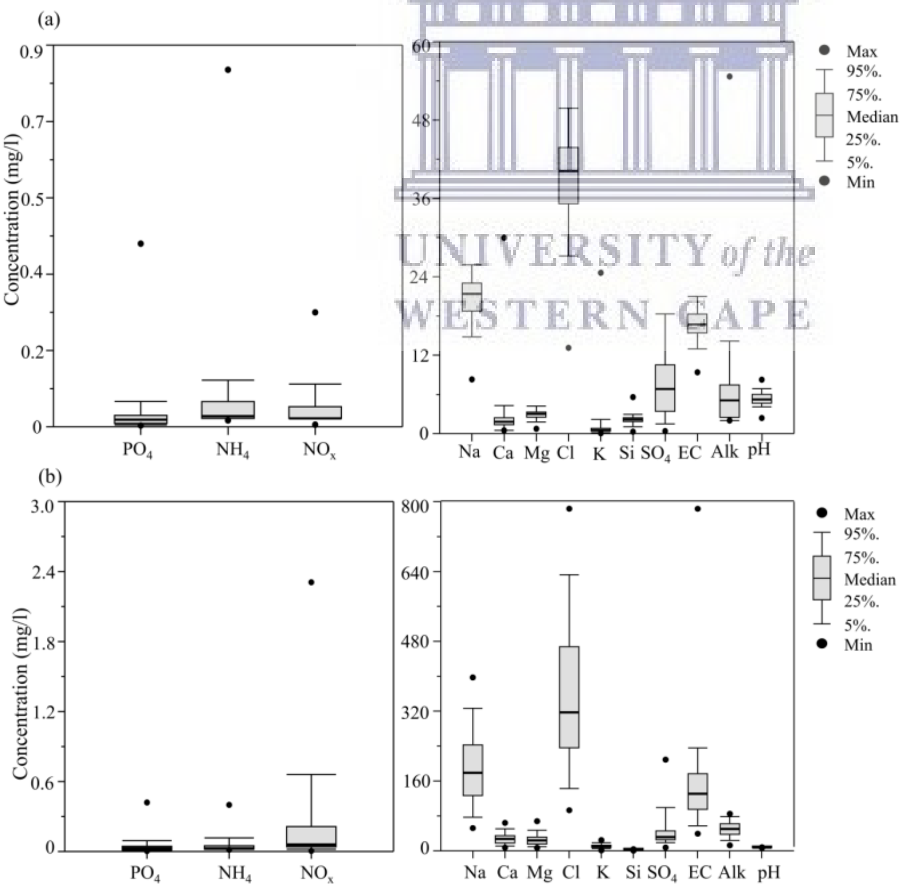


Figure 3.5 Summarised water quality parameters measured for the a) Touws (1980-2013) and b) Duiwe River (1998-2013) (mg L^{-1}, EC in mS m^{-1}, Alk: Alkalinity). Bar indicates the standard deviation

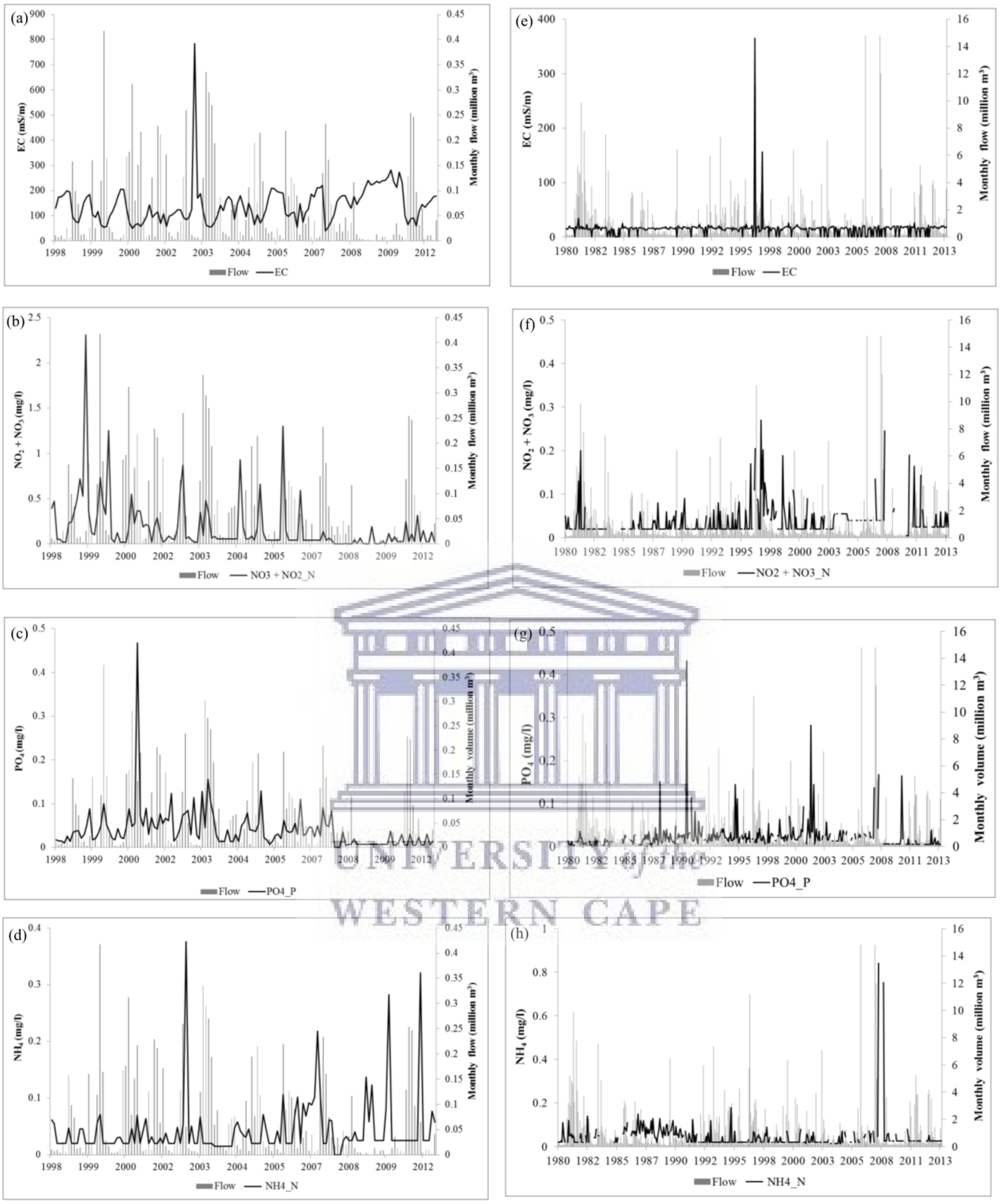


Figure 3.6 Monthly water quality and flow data for the (a-d) Duiwe River (1998-2013) and (e-h) Touws Rivers (1980-2013)

The NO_x-N concentrations in the Duiwe River showed increased levels during or just after flood events, which were especially evident during 1999-2000 (0.87 to 2.3 mg L^{-1}) and 2003-2006 (0.59 to 1.29 mg L^{-1}). Runoff also increased during these years. The NO_x-N concentration showed a decreasing trend over time (Tau = -0.187) (Table 3.4). Ammonium (NH_4^+-N) levels spiked after increased river flows, with an increasing trend in concentrations especially after 2007. The PO_4-P concentrations were strongly positively correlated to flows (rs = 0.521) with a significant decreasing trend evident (Tau = -0.142; p =< 0.05). In the Duiwe PO_4-P levels were below 0.15 mg L^{-1} during the recorded period, with the exception of 2001 to 2003 and 2005, which coincided with increased river flows (Figure 3.6). All major ions were significantly negatively correlated with river flows (i.e. as river flow increases, variable concentration decreases) (Table 3.4). Drought periods also showed increases in solutes illustrated by EC concentrations in Figure 3.6. The NO_x-N, PO_4-P, and Si were the exception as concentrations increased with river inflows and were positively correlated to flows.

Table 3.5 Seasonal Mann-Kendall trend analysis with the Spearman rank correlation (r_s) between river flow and water quality parameters (mg L^{-1}, EC in mS m^{-1})

Parameter	Spearmans r_s	Trend	Kendall's Tau
Touws (1980 – 2013)			
Total Alkalinity	0.040	-	-0.036
Calcium Ca^{2+}	-0.002	+	0.158***
Chloride Cl^-	-0.229***	0	0.006
Electrical conductivity (EC)	-0.25**	+	0.120*
Fluoride F^-	0.054	+	0.168***
Potassium K^+	0.210***	+	0.239***
Sodium Na^+	-0.206***	0	-0.079*
Magnesium Mg^{2+}	-0.200***	0	-0.002
pH	-0.190**	+	0.287***
Phosphate PO_4-P	0.177**	+	0.116*
Nitrite and nitrate NO_2^- and NO_3^- (N)	0.137*	+	0.248***
Ammonium NH_4^+(N)	-0.028	-	-0.193***
Sulphate SO_4^{2-}	0.287***	+	0.101*
Silica Si	0.021	0	-0.044
Duiwe (1998 - 2013)			
Total Alkalinity	-0.630***	+	0.173*
Calcium Ca^{2+}	-0.727***	0	0.012
Chloride Cl^-	-0.720***	0	0.018
Electrical conductivity (EC)	-0.46*	+	0.132*
Fluoride F^-	-0.175	+	0.230*
Potassium K^+	-0.638***	0	-0.036
Sodium Na^+	-0.721***	0	0.048
Magnesium Mg^{2+}	-0.734***	0	0.024
pH	-0.389***	0	0.042
Phosphate PO_4-P	0.521***	-	-0.142*
Nitrite and nitrate NO_2^- and NO_3^- (N)	0.301*	-	-0.187**
Ammonium NH_4^+(N)	-0.088	+	0.258***
Sulphate SO_4^{2-}	-0.277*	0	0.077
Silica Si	0.553***	0	0.089

(+), upward trend; (-), downward trend; (0), no significant trend * Significance: *** < 0.0001; ** < 0.001; * < 0.05

In the Touws River EC concentrations were negatively correlated to flows (r_s = -0.25, p = 0.001). Small increases coincided with very low river flow years (monthly flows of 0 - 0.5 10^6 m^3) and exceptions to this occurred during the period 1996-1997 when the highest level reached was 365 mS m^{-1}. Increased levels of NO_x-N occurred during increased river flows (r_s = 0.137) and were lower than those recorded in the Duiwe River. The NH_4^+-N levels showed a decreasing trend but increases in concentrations occurred following high rainfall especially after 2007 when levels were the highest on record (0.84 mg L^{-1}). Table 3.5 shows the periods in the Touws and Duiwe Rivers when concentrations were increased and exceeded the Target Water Quality Ranges (TWQR) thresholds (DWAF, 1996a, b, c). Most solutes were negatively correlated to river flow with periods when concentrations for all solutes were above the TWQR thresholds. In most cases, with the exception of NO_x-N, the Touws concentrations also exceeded those recorded in the Duiwe River.

Table 3.6 Solute concentrations compared with DWS Target Water Quality Ranges (TWQR) for domestic (D), irrigation (I) and aquatic environment (AE). Years when fires occurred are indicated. Fire data source: (CAPE, 2011)

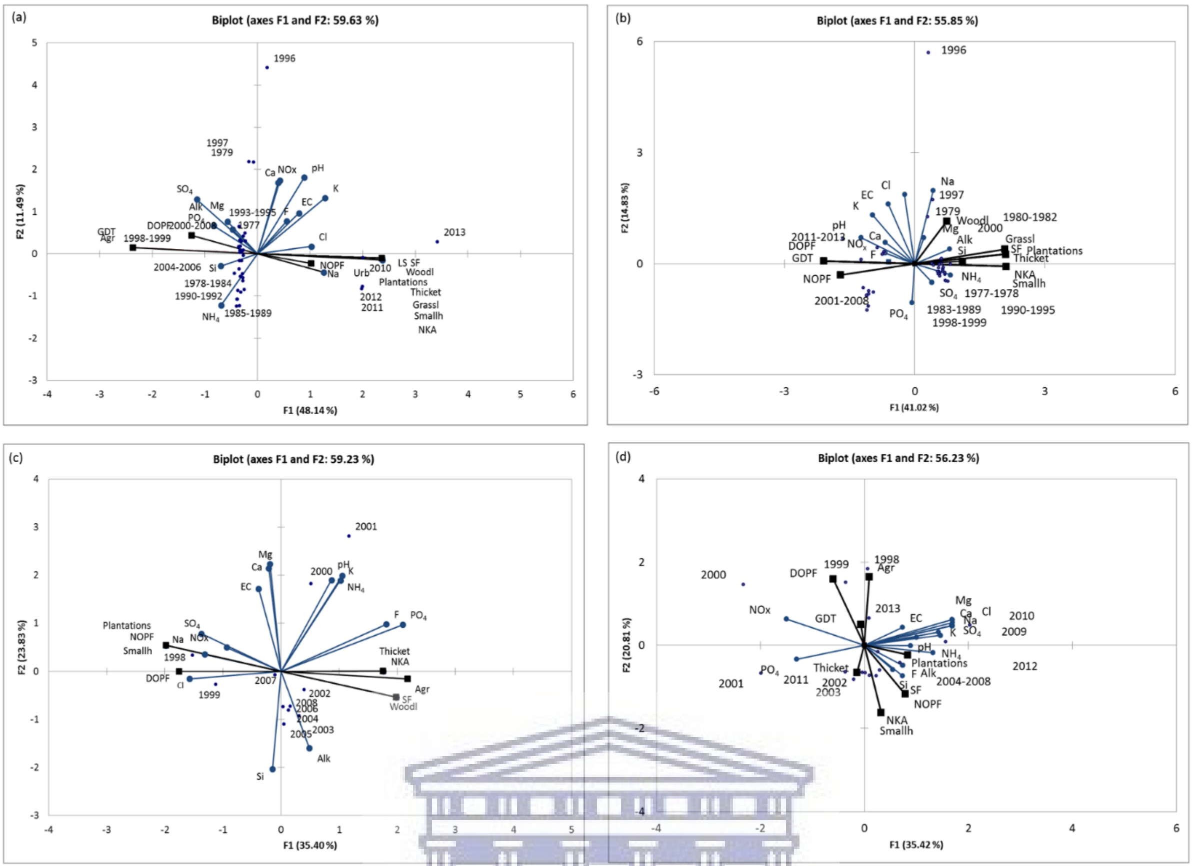
Parameter	TWQR	Units	Maximum measured concentrations	
			Touws River (1980-2013)	Duiwe River (1998-2013)
Calcium	0-32 (D)	mg L^{-1}	83.7 (1996) (fire) 35.4 (1997)	33-35 (1998-1999) 37-42 (2000) 33-43 (2003-2005) 34-50 (2007-2008) 33-64 (2009) 32-36 (2011-2012)
Chloride	0-100 (D) 0-105 (I)	mg L^{-1}	3234.1 (1996) (fire) 1292.3 (1997)	Entire period > 100
Fluoride	< 0-1 (D) 0-2 (I) 0.075 (AE)	mg L^{-1}		Entire period < 1
Magnesium	0-30 (D)	mg L^{-1}	221.4 (1996) (fire) 87.2 (1997)	31-33 (1998-1999) 34-37 (2000) 30-33 (2003) 33-40 (2005-2007) 34-68 (2009-2010) 32-33 (2011-2012)
Potassium	0-50	mg L^{-1}	66.84 (1996) (fire) 29.17 (1997)	Entire period < 50
Sodium	0-100 (D) 0-70 (I)	mg L^{-1}	1874.5 (1996) (fire) 736.4 (1997)	103-257 (1998-1999) 141-290 (2000) 105-290 (2000-2003) 104-252 (2003-2004) 151-298 (2005) 154-311 (2006-2007) 120-397 (2008-2009)

Parameter	TWQR	Units	Maximum measured concentrations	
			Touws River (1980-2013)	Duiwe River (1998-2013)
				151-258 (2011-2013)
Sulphate	0-200	mg L^{-1}	551.4 (1996) (fire) 167.3 (1997)	209 (2010)
NO_x-N	0-6 (D) 0-5 (I) <0.5 (AE) Oligotrophic 0.5-2.5 Mesotrophic	mg L^{-1}	0.2 (1996-1997) (fire) 0.13 (2007) (fire)0.24 (2008) (fire) 0.14 (2011)	0.7-2.3 (1999-2000) 0.8-1.2 (2003)
Ammonium	0-1 (D) <0.5 (AE) Oligotrophic 0.5-2.5 Mesotrophic	mg L^{-1}	0.84 (2007) (fire) 0.75 (2008) (fire)	0.37 (2007) 0.21 (2007) 0.28 (2010) 0.32 (2011after flood)
Phosphate	0.02-0.1 (AE)	mg L^{-1}	0.15 (1980,1983) (fire) 0.43 (1982) (fire) 0.14 (1987)	0.42 (2001) 0.14 (2003) 0.11 (2005)

3.3.3 Links between land cover and surface physico-chemistry

PCA analysis for the Touws River sub-catchment revealed all vegetation land cover categories including plantations, as well as urban and smallholdings were positively correlated with Cl^-, K^+, pH and Na^+. The agriculture and Goukamma dune thicket vegetation showed a negative correlation with Cl^-, K^+, and Na^+ (Figure 3.7a, Appendix 3.1). In the buffer zone (riparian and adjacent land) in the Touws catchment (Figure 3.7b, Appendix 3.2), degraded Outeniqua plateau fynbos and Goukamma dune thicket showed a positive correlation to K^+ and pH. Plantations were positively correlated to Si and alkalinity while Knysna Afromontane forests and smallholdings were positively correlated to NH_4^+, Si and alkalinity. Grasslands were positively correlated to Na^+, Si and alkalinity. Shrubland fynbos, thicket and dense bush showed a positive correlation to Si and alkalinity and woodland to Na^+.

In the Duiwe River sub-catchment, PCA analyses showed that water quality was better correlated with land cover in a 100 m buffer area than in the sub-catchment. In the sub-catchment Knysna Afromontane forests, thicket and agriculture were positively correlated with PO_4-P (Figure 3.7c, Appendix 3.3). In the Duiwe buffer shrubland fynbos was positively correlated to NH_4^+-N and F^- (Figure 3.7d, Appendix 3.4). Natural Outeniqua fynbos showed a negative correlation to NO_x-N. Degraded Outeniqua Plateau fynbos, which was invaded by black wattle, was positively correlated to NO_x-N. Plantations were positively correlated to EC, Cl^-, and SO_4^{2-} whereas thicket and dense bush was closely associated with PO_4-P.



Land cover category

Degraded Outeniqua Plateau Fynbos (DOPF)	Grassland (Grassl)
Natural Knysna Afromontane (NKA)	Low shrubland (LS)
Natural Outeniqua Plateau Fynbos (NOPF)	Woodland/Open bush (Woodl)
Shrubland Fynbos (SF)	Agriculture (Agr)
Thicket /Dense bush (Thicket)	Smallholdings (Smallh)
Goukamma Dune Thicket (GDT)	Urban (Urb)

Figure 3.7 PCA of mean water quality variables (dots) and land cover (squares) in the (a) Touws River (1980-2013) sub-catchment, (b) Touws River buffer, (c) Duiwe River sub-catchment (1998-2013) and (d) Duiwe River buffer

Land cover classes are associated with water quality variables. The orientation of the water quality variables lines (dots) reflects direction of maximum change of the variable. The longer the lines the greater the association of water quality variable with a land cover. Land cover with perpendicular projections near to or beyond the tip of a dot will be strongly positively correlated with the water quality variable represented. Those projections of water quality variables near the origin have a lesser association with a particular land cover.

3.4 Discussion

The Duiwe River catchment became anthropogenically influenced with agricultural pressures in terms of dairy and vegetable farming, irrigation, abstraction changing flow volumes and fertilizer application altering water quality. Kapp et al. (1995) estimated that the mean annual pre-development runoff of 6.6 10^6 m^3 per year in the Duiwe River has been reduced by 48.5% due to agricultural abstractions whereas updated rainfall-runoff modelling data estimate suggests a reduction closer to 80%. Farmers in the area made use of in-channel dams on the Duiwe River and tributaries on both the Touws and Duiwe Rivers. There is limited use of borehole water (DWAF, 2004) since the quartzitic sandstones of the TMG such as in the Touws and upper Duiwe catchments are highly fractured and faulted limiting the storage of groundwater (Lin et al., 2014). The changing land use to pastures in the lower Duiwe River sub-catchment (Figure 3.1b) coupled with gentler slopes also decrease runoff (Singh et al., 2014). The rainfall data (Figure 3.2) shows a downward trend in the number of rainy days but an increase in the rainfall intensity. Interviews with farmers by De Lange and Mahumani (2013) indicated that the average rainfall per year (790 - 800 mm) is insufficient for optimal production of pasture areas as an average of 1000 mm per year is required (De Lange and Mahumani, 2013). Changes in rainfall patterns and associated evaporation rates (Hoffman et al., 2011) are directly influencing farming activities so that irrigation scheduling changed and the amount of pasture presently being irrigated was reduced (De Lange and Mahumani, 2013).

The changing land use in the Duiwe River catchment illustrated by the extensive mapped agricultural areas and changing irrigation systems (Figure 3.1b) reduced river flows shown by the decreasing trend in flow data, which was at zero flows at times. This had a cumulative impact on water quality in the Duiwe River, which also negatively impacts the Wilderness Lake (Eilandvlei) it feeds (Russell, 2013). The reductions in flows in the Duiwe River significantly altered the balance of the flows into the Lake System whereby through flows in the Lakes were reduced causing the Touws estuary, naturally a temporary open/closed system, to remain closed for longer periods. The biophysical and chemical properties of the Lakes are therefore changing since water is retained without any significant level rise (Russell, 2013).

On average the nutrients NO_x-N and PO_4-P were higher in the Duiwe River with mean concentrations of 0.24 mg L^{-1} and 0.04 mg L^{-1} respectively. Both followed the trend of increasing with increasing flows with the highest concentrations for NO_x-N recorded at 2.3 mg L^{-1} (1999) and 0.42 mg L^{-1} (2001) for PO_4-P after increased stream flows. This trend was also observed by Monaghan et al. (2007) where nitrogen accumulated in the top soil during the dry season and was transported with runoff to streams during the wet season. De Lange and Mahumani (2013) reported that farmers indicated increased bank erosion during high intensity storms and floods, evidenced by siltation of their instream storage dams. Much of the Duiwe River catchment consist of sandy topsoil that overlays clayey subsoil (Table 2.2, section 2.2). Nitrogen in its soluble form (NO_3^-) is readily transported but can accumulate in soil and phosphorus (PO_4^{-3}) with its affinity to mineral surfaces, will readily attach to soil particles, especially fines such as clay (Makarova et al., 2004). This can lead to a build-up in soils on land and in farm dams so that movement will depend on remobilisation of soils by erosion (Stanley and Doyle, 2002, Chattopadhyay et al., 2005). The pastures are covered with perennial grasses throughout the year reducing sediment loss while vegetables (with fertilizers applied) are grown as row crops with limited or no vegetation stabilising sediment between them (De Lange and Mahumani, 2013). The increase in nutrients in both systems after high rainfall and flow events implies that the supply may be via runoff and erosion from the land and flushing from sediments trapped by dams in the Duiwe River catchment. The high density of farm dams in the Duiwe River was likely to have a cumulative impact in this regard as well as in reducing base flows as also shown by Mantel et al. (2010).

After 2006 levels decreased with mean concentrations for NO_x-N at 0.09 mg L^{-1}. Concentrations still spiked with increased flows but were considerably lower (Figure 3.6). The decreasing trend observed in nutrients after 2006 was due to reduced, more efficient use of the amounts of fertilisers applied by farmers to contain costs. This was noted during interviews by De Lange and Mahumani (2013). Farmers use minimum tillage in the pasture management and wastewater produced by dairy farms containing cow manure is used from settling pans to spread directly onto pastures. This practice significantly reduced the nitrogen requirements for the pastures where application was reduced from 500 kg to 179 kg per hectare per application (De Lange and Mahumani, 2013).

PCA analysis of the Duiwe River buffer showed that degraded Outeniqua Plateau fynbos and thicket and dense bush were associated with PO_4-P and shrubland fynbos was associated with NH_4^+-N, indicating that both alien and indigenous vegetation supply nutrients to the Duiwe River. Plantations were positively correlated with EC, Cl^-, and SO_4^{2-} concentrations, which have been shown to increase in rivers with clearfelling practices (Lesch, 1995). The land use change from natural to alien vegetation probably resulted in changes in the nutrient concentrations observed. Legume invasive species such as *Acacias* are efficient nitrogen fixers (Tye and Drake, 2012) and are known to contribute nitrogen via soil and groundwater due to leaching, adding to nutrient loads in the rivers, especially if these species are cleared. Jovanovic et al. (2009) found that NO_x-N was quickly released into groundwater after clearing *Acacia saligna* from plots due to high residual nitrogen reserves in the rooting zone, decreased evapotranspiration due to plant removal and increased groundwater recharge. This may also explain the increases in NO_x-N concentrations in the rivers related to clearing of vegetation, as farming areas or smallholdings were established during the expansion of agricultural practices in the area, or in later years (after 1995) with the initiation of the Working for Water Programme when invasive plants were cleared (Binns et al., 2001). Alien vegetation are known to impact water quality and quantity and has replaced a large percentage of natural vegetation in the Duiwe River catchment with the river and tributaries invaded by *Acacia mearnsii*, *A. melanoxylon* and *Eucalyptus* species (GRI, 2008).

Much of the exotic vegetation in the Touws catchment is linked to afforestation (Le Maitre, 2000) with species such as *Hakea* and *Pinus*, as well as *Acacia mearnsii*, being introduced for the production of timber, fuelwood or tan-bark in the form of plantations. Pine plantations are known to reduce river flows in South Africa by using more water than indigenous vegetation (Scott et al., 1998b, Le Maitre et al., 2004) and Kapp et al. (1995) estimated that the mean annual pre-development runoff at 18 10^6 m^3 per year in the Touws catchment was reduced by approximately 21.7% due to pine plantations in the upper catchment. The results of DWS flow data showed an increasing trend in the Touws River catchment. When a percentage of the plantations were removed by one of the largest wildfires on record in 1996, no replanting occurred and the area was partly rehabilitated as part of the exit strategy of the MTO Forestry company where land phased out are either used for conservation or agriculture (Pauw, 2009). The removal of this section of pine plantation coupled with low storage capacity of the catchment is likely responsible for the increasing trend.

The PCA buffer analysis of the Touws River a close association was observed between smallholdings, NH_4^+-N and Si. No large scale farming practices occur on smallholdings but some do have livestock present, with subsistence farming of crops occurring. Others offer accommodation in the form of tourism cottages located close to or on riverbanks and dams (WLSDF, 2015). These activities explain the increases in nutrient supply. Silica concentrations are linked to soil and bedrock from a catchment and these land uses are associated with clearing of vegetation exposing soil to rainfall and runoff thereby increasing concentrations in rivers (Lesch, 1995). Average levels of NH_4^+-N ranged at 0.03 mg L^{-1} and although analysis showed a decreasing trend, levels spiked with flow increases. The highest concentrations were at 0.84 mg L^{-1} and 0.75 mg L^{-1} in 2008, which coincided with the expansion of the Touwsranten settlement. Wastewater discharge may reach the Touws River from this low cost housing settlement adding to increased NH_4^+ concentrations. The population has grown since 1996 to the extent that informal settlements developed on the fringe areas and are not connected to the water-borne sewage system (i.e. water being the transport medium for solid waste removal) (WLSDF, 2015). Table 3.5 shows that the period 1996-1997 had the highest concentrations in most solutes and nutrients in the Touws River. The year 1996 is seen as an outlier observation point in the PCA plots for both the Touws sub-catchment and Touws River buffer (Figure 3.7a and b). The wildfire that occurred during this period resulted in all solutes, except NO_x-N, exceeding the TWQR for domestic and irrigation use, which was coupled with increased rainfall. Other studies such as Smith et al. (2011) and Bergh and Compton (2015) also showed increased concentrations of nutrients to rivers after fires.

During 1996 rainfall above the 75th percentile (896 mm to 1007 mm) was recorded at all rain gauges. The concentrations of NO_x-N and PO_4-P for the Touws River increased during wet years and more so during wet and fire years (Tables 3.2 and 3.5). Fire intensity in alien vegetation is greater than in fynbos due to a greater fuel load causing water repellency of soils, which promotes increased runoff (Le Maitre et al., 2004). PCA analysis in the Touws River buffer shows, plantations, grasslands, shrubland fynbos and thicket/dense bush were positively correlated to Si, Na^+ and alkalinity. Natural Afromontane forests were positively correlated to NH_4^+-N (Figure 3.7b). Large percentages of these vegetation types were removed with the fire exposing soil to runoff. Lesch (1995) found that total phosphorus, total nitrogen and the same water quality variables assessed in the current study increased in concentrations after fires, which included plantations and indigenous fynbos vegetation.

Increased runoff and reduced infiltration and soil water storage was also found by Scott (1997) and Scott et al. (1998b) in other areas of the southwestern Cape with similar land use and catchment characteristics as the Touws River catchmnet. Le Maitre et al. (2014) showed that simulation of clear-felling of pines and degrees of water repellency of soils after fires in invaded *Acacia* areas in a neighbouring catchment to the Touws River, produced the same hydrological response. The increases in concentrations in the Touws River were short-lived and levels returned to a more constant state with natural variation within a few months. This was also observed in the study by Bergh and Compton (2015). The described catchment analysis methods proved to be effective in analysing long-term secondary databases for trends and impact on water quality at both the sub-catchment and 100 m buffer scale. It can also provide a desk top method to identify possible areas or "hotspots" for investigation in the field minimising costs and logistics. This was also concluded by other studies such as Sliva and Williams (2001).

3.4 Conclusion

The link between land cover/use and water quality and the role of temporal (historical data analysis) and spatial scales (sub-catchment and buffer areas) in the understanding of the influence of land cover and management of land use activities on water quality was demonstrated. Both the catchment land cover and land use characteristics as well as the 100 m buffer area was useful in determining impact on water quality. The databases used showed that when the spatial heterogeneity of the catchment was altered either by human influences such as agriculture or by natural events such as fires, it was reflected by changes in the water quality and quantity available as stream flow. Changes in land management could have a substantial impact in water quality improvement, especially relevant where cost-efficient solutions to water quality impairment are a requirement. This could include more efficient use of fertilizers in agricultural land management by reducing quantities of fertilizer applied or maintaining and/or establishing riparian buffer areas to mitigate the impacts to streams from the land based activities. Vegetation (indigenous or alien) as riparian buffers can also influence water quality and a proper understanding of that role is necessary for effective land and water management.

Chapter 4: Linkages between the physical river template and biological water quality indicators

The chapter was presented as a paper at the Southern African Association of Geomorphology (SAAG), University of Swaziland, Kwaluseni (Manzini), Swaziland during 2017.

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4.1 Introduction

The natural landscapes in which rivers occur determine the physical river habitat template (Jacobson, 2013). Catchment constraints such as geology, topography or land use will not only determine the river structure but also how it will function. This form-process relationship influences the riverine biota and flora associated with river systems, which is why links between the physical river structure and biota have become as important as the chemical-biotic links for an improved overall understanding of river functioning (King and Schael, 2001). River geomorphology influences habitat with varying hydrology (hydrodynamic) or changes to river substrate or channel morphology (morphodynamic), thereby influencing the distribution, variation and structure in aquatic communities (Elosegi et al., 2010, Jacobson, 2013). For example, changing hydrodynamics (e.g. timing of flow pulses) and morphodynamics (e.g. sediment removal from gravel beds or floodplain-connecting floods) may be a cue for life-history events in certain species, such as spawning and migration for fish (Jacobson, 2013).

River systems globally have been classified for numerous purposes, one of which is to relate the physical instream processes to biological processes in order to assess ecological integrity (Buffington and Montgomery, 2013). According to Rowntree et al. (2000) rivers are classified according to geomorphological zones based on river discharge, sediment load and regional slope. The longitudinal zonation identifies geomorphologically similar river reaches occurring in uniform geology, associated with particular morphological units each having their own flow characteristics but still retaining the concept of downstream changes longitudinally (Rowntree et al., 2000). However, river ecology (species assemblages), hydrology and the physical habitat template are not always continuous along river gradients and rather consist of hydrogeomorphic patches (formed by catchment geomorphology and flow characteristics), each with their own chemical and physical conditions, controlling ecological structure and function (Thorp et al., 2006). The longitudinal zonation, however, still provides a useful geomorphological classification for river systems with which to associate river type and behaviour.

Aquatic organisms themselves have diverse and complex community structure and functioning. Using multiple groups of organisms from different trophic levels in an aquatic ecosystem to assess impacts on water quality, therefore has value, in that each will have their own biological functioning providing a comprehensive view of ecological integrity (Li et al., 2010, Chon et al., 2013). When including fauna and flora such as algae and macroinvertebrates, when assessing ecological conditions of a river, a better indication of the overall river health and water quality is provided.

According to Oberholster (2011) periphyton algae play a pivotal role in ecosystem functioning by providing energy sources to higher trophic levels in rivers and can act as good indicators of nutrient enrichment. Furthermore, the use of aquatic macroinvertebrates as an indication of water quality and ecological integrity in rivers is common practice and has been efficiently used to illustrate anthropogenic impacts spatially and temporally (Márquez et al., 2015). There are limitations in their use since macroinvertebrates are affected by habitat (hydrology, substrate), food availability and seasonality as well as by obstructive impacts such as dams (Harding et al., 2005). The use of algae and diatoms as bioindicators has overcome these limitations, with a host of advantages of their use as indicated by Harding et al. (2005).
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In South Africa algae and diatoms have only recently gained momentum in their use in ecological river studies and others. Epilithic diatom community indices in rivers were applied in numerous studies in Europe, the USA and Australia (Oberholster et al., 2016). The dominance of endemic diatom species makes the use of these commonly applied assessments in Europe less applicable in Africa (Taylor et al., 2007b). Diatom trophic indices also exclude filamentous green algae. A study by De la Rey et al. (2004) combined an European diatom index with SASS 5 and concluded that both gave a good indication of general water quality conditions but no significant correlation was found with nutrients. Diatom trophic indices exclude filamentous green algae, which are an important component in biological river processes providing food and habitat for other organisms (Wehr and Sheath, 2003). Studies have shown that benthic filamentous algae were more responsive to nutrient enrichment in rivers than diatoms (Chételat et al., 1999, Figueroa-Nieves et al., 2006, Oberholster et al., 2016). The current study was designed for the assessment of macroinvertebrates and the full consortium of benthic algal communities, along the gradient of a river system, from a minimally impacted reach to a cumulative impacted reach by agricultural activities. Studies using the full consortium of benthic algae as indicators of ecological integrity in agriculturally influenced coastal rivers in Africa are scant. Both macroinvertebrates and algae were used concurrently in relation to the physical river habitat to assess water quality and ecological integrity.

This chapter is designed to address objective 2 of the study and partially objective 3 and 4 (Chapter 1, Section 1.4). Specifically, the objectives of this chapter were: 1) to characterize the distribution and community assemblages of macroinvertebrates and algae down a longitudinal river gradient in relation to chemical stream composition, land use and the physical river template (geomorphic zone, morphology, stream order, stream width); 2) to determine the environmental variables that affect macroinvertebrates and algae assemblage distribution and to; 3) determine the suitability of the selected bioindicators in relation to environmental conditions. Outcomes from the current study provide meaningful insights into the environmental drivers of algal and macroinvertebrate communities and their use as bioindicators for predicting and monitoring ecological integrity and water quality of short, steep, coastal rivers which can be applicable to other river systems.

4.2 Methods: Data collection and analysis

4.2.1 Sampling sites and land cover/use

The sites used and their descriptions are as listed and shown in chapter 2 section 2.3 (Figure 2.9). The Klein Keurbooms and Duiwe River catchments have largely been transformed by agriculture, mainly pastures with numerous storage dams, altering the river flow. The land use activities were employed as indicators of natural and anthropogenic effects on the distribution of algae and macroinvertebrate communities and on the physical river habitat template. Land use and especially agricultural activity, is closely linked to water quality in the catchment (Petersen et al., 2017). Land use activities were determined according to Petersen et al. (2017) as demonstrated in Chapter 3. Sampling sites were located in different geomorphological zones, which were related to differences in factors such as river substrate and velocity, vegetation and land cover and use (Chapter 2, Table 2.4). The data presented are from seasonal sampling over two years. Sampling commenced from March 2014 to March 2016 (dates of sampling available in Appendix 4.1).

4.2.2 River water chemistry

Samples collected from the river for water quality parameters were described in Chapter 2, section 2.3.2. The water quality summary is presented in the Results section 4.3.2 and detailed results of the short and long term water quality data are presented in the Results section 4.3.3 and river water quality samples are available in Appendix 4.1.

4.2.3 Macroinvertebrate sampling

Benthic macroinvertebrates were sampled seasonally (wet season = September to March; dry season = April to August) (Appendix 4.2). Benthic macroinvertebrates were sampled using the rapid bioassessment method SASS (South African Scoring System) version 5 (Dickens and Graham, 2002). The SASS metrics were included in the analysis to relate the macroinvertebrate scores to water quality. A kick net was used and held downstream so as to collect any macroinvertebrates dislodged. The method separates habitat or biotopes so that 3 different instream habitats are sampled, which included stones-in-and-out-of-current (SIC + SOOC), marginal and aquatic vegetation (AQ/MV) and gavel-sand-mud (GSM). The ASPT was calculated by dividing the number of taxa by the total SASS score. The scores are obtained from the list of macroinvertebrates as each has a sensitivity weighting assigned based on their tolerance to water quality impairment (Dickens and Graham, 2002). The net contents were analysed in the field where the macroinvertebrates were identified to family taxonomic level while recording the abundance groupings, number of individuals and the average score per taxon. The SASS 5 method identifies macroinvertebrates to the family taxonomic level (except Nematodes), but macroinvertebrates were also considered according to their potential functional feeding groups (FFGs) (Table 4.2) based on dominant food sources and mode of feeding, which provides a broader typological approach than species data, as the patterns that result are less region-specific (Santoul et al., 2005). Studies both locally and internationally found very similar results when comparing family to species level data in distribution patterns of macroinvertebrate assemblages down a longitudinal river profile (Bournaud et al., 1996, Smith et al., 1999, Dallas, 2002). Macroinvertebrate habitat was assessed using the Invertebrate Habitat Assessment System (IHAS) developed by McMillan (1998). The macroinvertebrates with algae as their dominant food source were collectively termed ‘grazers’. The macroinvertebrates were separated into FFGs as defined by Merritt and Cummins (1996) and Schael (2005) . Predators were classified as those macroinvertebrates that prey on other macroinvertebrates (Merritt and Cummins, 1996).

Table 4.1 Macroinvertebrate taxa sampled during the study period for all sites with functional feeding groups used to determine grazers and predators. The primary food source for grazers is included

Data code	FFG	Primary food source
Ancylidae	scraper	Algae
Baetidae	collector gatherer/deposit feeder	Most large algae and detritus
Barbarochthonidae	gatherer/shredder	Grazers on algae or shred detritus on leaf packs

Data code	FFG	Primary food source
Caenidae	collector gatherer/deposit feeder	Fine particulate detritus
Ceratopogonidae	grazer	Plant and soil detritus, algae
Chironomidae	scraper/predator	Algae and some detritus
Culicidae	filterer	Small algae and detritus
Dixidae	filterer /collector	Fine particulates and planktonic algae
Elmidae	scraper	Algae and some detritus
Glossosomatidae	scraper	Algae and some detritus
Helodidae	collector gatherer/brusher	Mostly large algae
Hydraenidae	scraper	Algae
Hydrophilidae	shredder	Filamentous green and red algae, epilithic, micro-algae, periphyton and filamentous diatoms
Hydropsychidae	filterer /collector	Algae, organic particles, small invertebrates
Hydroptilidae	scraper	Algae and some detritus
Leptoceridae	collector gatherer	Grazers on algae or shred detritus on leaf packs
Leptophlebiidae	scraper	Most large algae and detritus
Lymnaeidae	scraper	Algae
Notonemouridae	shredder	Most large algae and detritus
Oligochaeta	detrivore	Diatoms, algal, plant, silt/mud
Perlidae	shredder	Algae, leaves and detritus
Petrothrincidae	scraper	Grazers on algae or shred detritus on leaf packs
Philopotamidae	filterer /collector	Algae, organic particles, small invertebrates
Physidae	scraper	algae
Pisuliidae	shredder	Grazers on algae or shred detritus on leaf packs
Planorbinae	scraper	algae
Sericostomatidae	shredder	Grazers on algae or shred detritus on leaf packs
Simuliidae	filterer /collector	fine particulate detritus

Data code	FFG	Primary food source
Teloganodidae	grazer	Most large algae and detritus
Tipulidae	shredder/gatherer/predator	Decaying plant matter, plant fragments, micro-organisms
Turbellaria	scraper/predator	grazers on microflora and algae
Predators		
Aeshnidae, Athericidae, Belostomatidae, Chlorolestidae, Coenagrionidae, Corduliidae, Corixidae, Corydalidae, Dytiscidae, Gomphidae, Gyrinidae, Hydracarina, Libellulidae, Naucoridae, Nepidae, Notonectidae, Platycnemidae, Pleidae, Potamonautidae, Tabanidae, Veliidae		

4.2.4 Periphyton sampling

Benthic algae were sampled seasonally (wet season = September to March; dry season = April to August) from instream habitats, which included sand substrates where available. Duplicate ceramic 30 x 30 cm tiles were randomly placed instream and submerged below water level (10 – 50 cm depth), at each site, which were used instead of cobbles/boulders to obtain a standard surface area. Tiles (900 cm^2), were brushed to remove attached algae and the material was re-suspended in a plastic tube with 50 ml river water. At each site algae from both tiles were pooled together. The composite sample was divided in three subsamples: (i) unpreserved for benthic chlorophyll (mg m^{-2}) analyses; (ii) unpreserved for the culturing of questionable filamentous algae in the laboratory for identification; and (iii) preserved for microscope epilithic filamentous algae and diatom identification (Oberholster et al., 2017). The latter sub-sample was fixed in 2.5% glutaraldehyde in the field and kept at 4^{o}C in the dark until laboratory analysis (Oberholster et al., 2016).

Aliquots (10-100 ml) were sedimented depending on the abundance of the benthic algae in the samples. Diatoms were identified after clearing in acid persulfate. Strip counts were made until at least 100 - 300 individuals of each of the dominant algal species were counted (APHA, 2006). All algae were identified using a compound microscope at 1250 times magnification according to Wehr and Sheath (2003), Van Vuuren et al. (2006) and Komarek and Anagnostidis (1999; 2005). Epilithic algae abundance in the samples was determined by counting the presence of each species (as cells in a filament or equal number of individual cells). The surface water column was also sampled for suspended chlorophyll-*a* (chl-*a*) using 1 L polyethylene bottles. Samples were processed for analyses of suspended chl-*a* (benthic and suspended chl-*a* in water column) according to Porra et al. (1989). The trophic status of the river systems were evaluated using the benthic chl-*a* mg m^{-2} as proposed by Biggs (2000) where systems are considered: oligotrophic (0 – 1.7 chl-*a* mg m^{-2}), mesotrophic (1.7 – 21 chl-*a* mg m^{-2}), eutrophic

(> 21 – 84 chl-*a* mg m^{-2}) and hypertrophic (> 84 chl-*a* mg m^{-2}). Nutrient levels favouring algal growth as proposed by Dodds et al. (2002) were used; a TP threshold of 0.03 mg L^{-1} and a TN threshold of 0.5 mg L^{-1}.

4.2.5 Statistical analysis

Multivariate statistics were used to determine the spatial and temporal patterns of similarity or dissimilarity in the ecological structure (distribution, composition, abundance) in assemblages of macroinvertebrates and algae in relation to chemical stream composition and the physical river habitat template. Statistical analysis was applied to water quality parameters using Spearman's rank correlation (r_s) test in XLSTAT (2016 Addinsoft). PRIMER (v6) and its add-on package PERMANOVA+ were used to perform multivariate analyses as algal and macroinvertebrate density was not normally distributed (Clarke and Warwick, 2001, Anderson et al., 2008). When the general patterns of temporal change in the variables were established, relations among variables and associations between algal and macroinvertebrate assemblages were determined using Principle Component Analysis (PCA). Where necessary the environmental data were log (x+1) transformed and normalized to limit variances in the data prior to any analysis. The algal abundance data were square-root transformed prior to ordination to reduce variability among samples (Ewart-Smith and King, 2012). Macroinvertebrate data were presence/absence transformed.

Analysis of similarities (ANOSIM) (Clarke and Gorley, 2006), which is a non-parametric permutation procedure similar to ANOVA, was used to determine if there were any significant differences between the sites, seasons of sampling and habitat in terms of the macroinvertebrate and algal assemblages. A similarity profile (SIMPROF) permutation test was run *a priori* on all site samples to assess statistically significant evidence of true clusters in samples together with ordination by non-metric multidimensional scaling (MDS) based on Bray-Curtis similarities. All analyses were considered statistically significant at $p \leq 0.05$. The SIMPER (similarity percentages) (Clarke and Gorley, 2006) routine was used to discern typical or distinguishing species of the groups identified by the cluster analysis for the macroinvertebrate and algal assemblages. Distance-based Linear Modelling (DISTLM) was used to determine which environmental variables were mostly responsible for the difference and variation in species assemblages of macroinvertebrates and algae between sites using PERMANOVA+. The distance-based redundancy analysis (dbRDA), which is a constrained ordination, was used to fit the values from the linear model with an overlay of those variables.

4.3 Results

4.3.1 Macroinvertebrate composition

A total of 59 taxa occurred at all sites sampled over two years (Appendix 4.2). At site K1 the most abundant families were Pisuliidae, Leptoceridae, Teloganodidae, Leptophlebiidae, Sericostomatidae, Glossostomatidae, Philopotamidae, Elmidae, Notonemouridae, Barbarochthonidae, Baetidae, Caenidae, Corixidae, Naucoridae, Veliidae, Simuliidae and Chironomidae. Site K2 had the same families occurring except that increased abundances of Helodidae, Aeshnidae, Chlorolestidae, Coenagrionidae, Ceratopogonidae, Hydroptilidae and Dytiscidae occurred. Site K1 had very limited instream and marginal vegetation sampling habitat compared to site K2 where the marginal vegetation habitat was abundant. Both sites K1 and K2 had similar abundances of grazers during all sampling seasons but more predators occurred at site K2 (Figure 4.1). At site K3 much of the same taxa occurred except that Gyrinidae were also abundant at this site and the composition of grazers and predators were very similar to site K2. Taxa occurring at sites K1 and K2 were mostly sensitive to water quality impacts and therefore were high scoring, indicating sensitivity to water quality. High scoring taxa also occurred at site K3 but in lower abundances and more of the less sensitive macroinvertebrate families were abundant. The SASS and ASPT scores also reflected this observation (Table 4.1).

Site K4 had a completely different set of taxa compared to the upstream sites. Throughout all sampling seasons the most abundant taxa that occurred were Oligochaete, Turbellaria, Hirudinea, Notonectidae, Physidae, Lymnaeidae, Baetidae, Corixidae, Ancylidae and Coenagrionidae. Site K4 also had the highest abundances of predators with less abundance of grazers except during autumn when grazer abundance was decreased at all sites. All sampling biotopes were available at site K4 but macroinvertebrate taxa were still low in abundance and scores during all sampling events due to water quality and flow impacts.

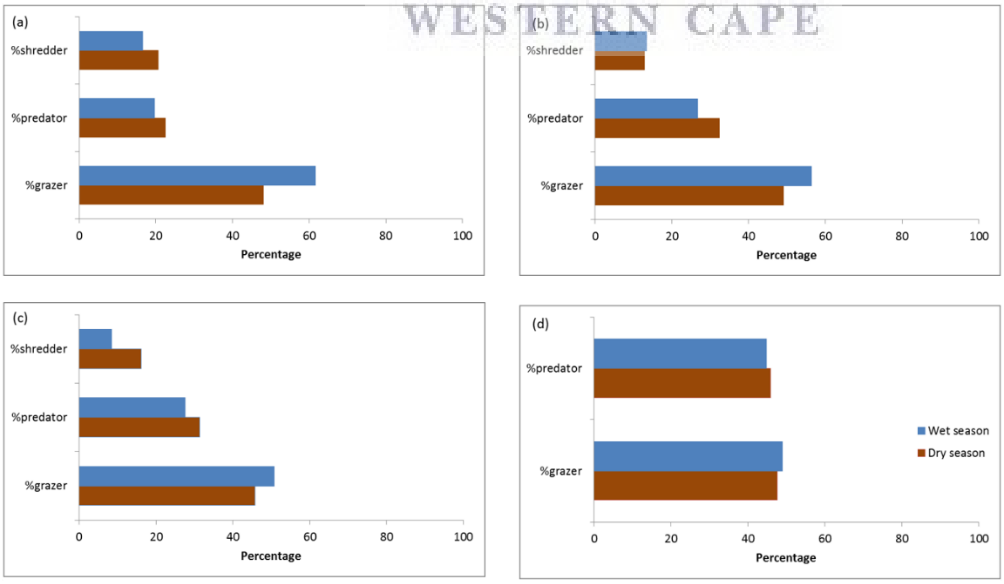

Figure 4.1 Macroinvertebrate grazer, predator and shredder abundances at (a) site K1, (b) site K2, (c) site K3 and (d) site K4

Table 4.2 Average SASS 5 metrics (Average Score Per Taxon-ASPT and Invertebrate Habitat Assessment System-IHAS) for the dry and wet seasons for all sampling sites over the sampling period in the Klein Keurbooms and Duiwe Rivers

Sampling sites	SASS	Number of taxa	ASPT	Habitat (IHAS %)
		Dry season		
K1	137	16	8.6	72
K2	149	21	6.9	74
K3	134	20	6.8	63
K4	65	14	4.6	54
		Wet season		
K1	148	20	7.4	74
K2	149	20	7.5	76
K3	124	19	6.6	60
K4	64	16	4	60

Statistical analysis of macroinvertebrate composition

The SIMPROF ($p < 0.001$) cluster analysis and MDS ordination showed a clear separation between two groups, with sites K1, K2 and K3 (Group 1) and another with site K4 (Group 2) (Figure 4.2a). The main groups, in the dendogram, are identified by the black lines and the branches in red indicate that these sites are statistically similar. Group 1, consisting of samples from sites K1, K2 and K3 and Group 2, consisting mainly of site K4 samples. ANOSIM performed for all sites determined that there were significant differences between the sites with a Global $R^2 = 0.419$ ($p = 0.001$). Pairwise tests showed site K1 and K2 were not significantly different but site K1 differed from the cumulative impact site (K4) and from site K3 (invaded with alien vegetation). Site K4 was most dissimilar to sites K1, K2 and K3 (Table 4.3). The MDS plot also showed a clear separation between the two groups (Figure 4.2b).

Table 4.3 Fraction of variation explained (R^2) for ANOSIM pair-wise tests of differences between sampling sites, geomorphic zones and habitat for macroinvertebrates. A significance level of ≤ 0.05 (*) is indicated

Sampling sites	**K2**	**K3**	**K4**
K1	0.05	0.26*	0.85*
K2		0.07	0.72*
K3			0.64*
Geomorphological zones	**Upper foothill**	**Transitional**	
Mountain stream	0.001	0.93*	
Upper foothill		0.84*	
Transitional			
Habitat	**Vegetation (marginal/aquatic)**	**Gravel/sand/mud**	
Stones (in and out of current)	0.28*	0.44*	
Vegetation (marginal/aquatic)		0.21	

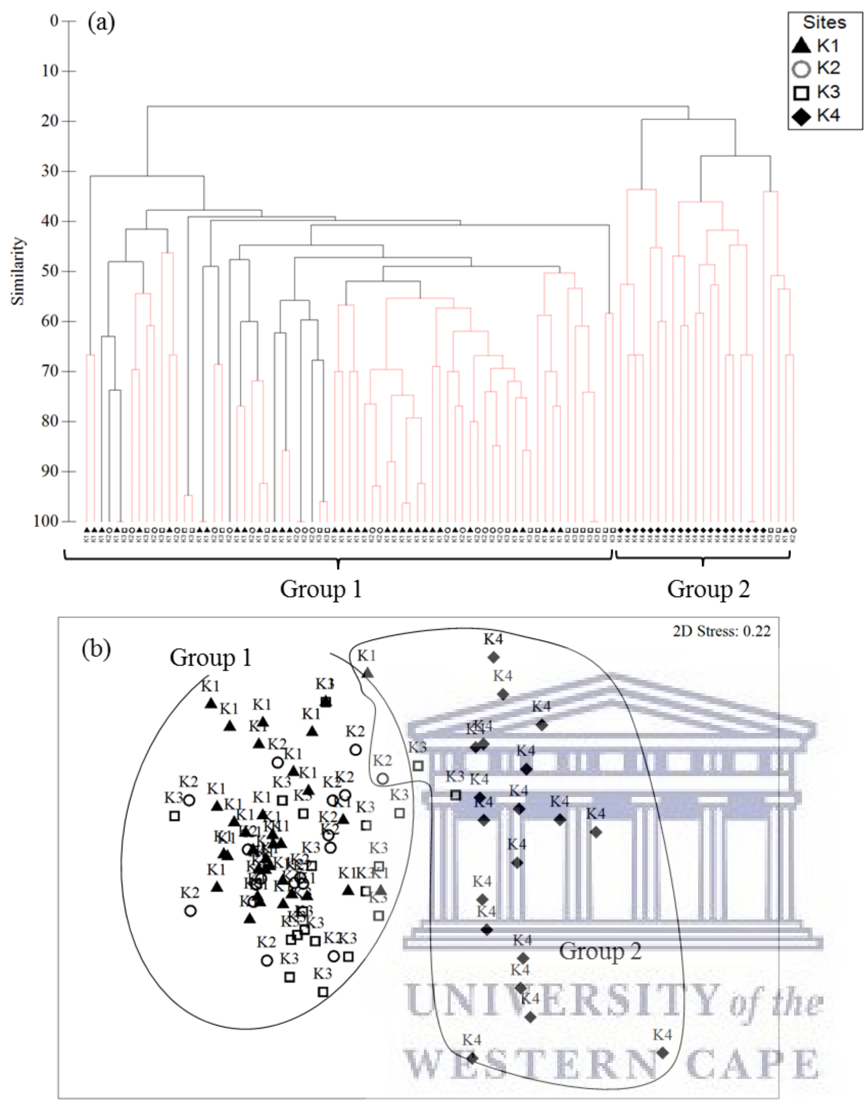

Figure 4.2 (a) Cluster dendogram and (b) MDS ordination of Bray-Curtis similarity between macroinvertebrate species composition at all the sites. Sites codes: K1 = site 1; K2 = site 2; K3 = site 3; K4 = site 4

The same results were obtained when testing geomorphological zones. A one-way nested PERMANOVA between geomorphological zones indicated that macroinvertebrate assemblages between both geomorphological zones and sampling habitat (habitat nested within zones) were statistically different.

Table 4.4 Results of one –way nested PERMANOVA of macroinvertebrate assemblages between geomorphological zone and biotope (habitat). Significant differences are at p ≤ 0.05

Groups	df	SS	MS	Pseudo-F	p
Zone	2	35644	17822	5.4847	0.0012
Biotope (zones)	12	55691	4640.9	3.2992	0.0001
Residuals	80	112530	1406.7		
Total	94	226650			

ANOSIM showed no significant difference of macroinvertebrate assemblages sampled between seasons at each site (Global $R^2 = 0.03$; $p = 0.197$). The macroinvertebrate taxa typifying the upstream sites were pollution-sensitive taxa (reflected in high SASS and ASPT scores in Table 4.2) from the orders Ephemeroptera, Trichoptera and Plecoptera (EPT), which occurred in acidic waters (pH: 4 to 5) that were low in nutrients, temperature and EC.
The impact on water quality is reflected in the macroinvertebrate taxa where the SASS and ASPT scores at site K3 were lower than those recorded at site K2 with a lower abundance of sensitive taxa, especially during the wet season. River hydromorphology influenced the physical habitat diversity for macroinvertebrates. The number of macroinvertebrate taxa sampled at site K1 during the dry season was lower than that sampled in the wet season and in the foothills (Table 4.2) due to the presence of fewer sampling habitats. The wetted channel was narrower and so sampling of the marginal vegetation was restricted (Figure 2.12, section 2.3.1). No instream vegetation occurred and only the stones habitat was available to sample. The topographical nature of the area meant that site K4 was still in a transitional zone with good habitat diversity. An increase, although not significant, was observed in the habitat scores and macroinvertebrate taxa, from the dry to the wet season.

The SIMPER routine in Primer was used to discern the species typifying the groupings formed as well as identifying the discriminate species resulting in group differences. The results showed that the average similarity for Group 1 was 45% and for Group 2 it was 30% while the average dissimilarity between the two groups was 83%. The typical families across all sites that occurred at both sites K1 and K2 were Leptoceridae, Chironomidae, Pisuliidae, Notonemouridae, Elmidae, Naucoridae and Leptophlebiidae (Figure 4.3). Similar macroinvertebrate assemblages occurred at site K3, with the addition of Gyrinidae and Corydalidae. These sites were dominated by grazers, which included scrapers and shredders with increasing predators occurring at site K2 and K3 (Figure 4.1). Site K4 was the most dissimilar in the species contributing to the average similarity with Physidae (grazer) (16%), Coenagrionidae (predator) (13%) and Oligochaeta (10%) contributing the most. The taxa present were low-scoring and pollution tolerant and no shredders were recorded as this site.

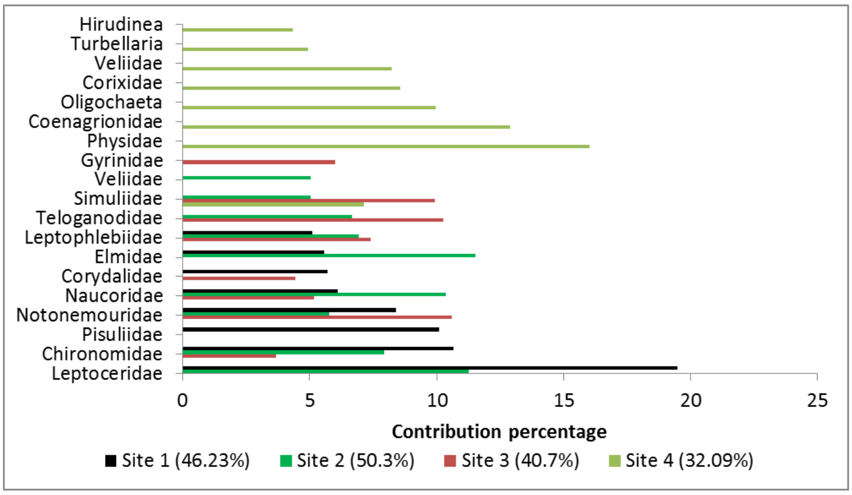

Figure 4.3 SIMPER results of macroinvertebrate taxa all sites that together contributed at least 70% to the overall similarity. Average similarity percentages are shown in brackets

The taxa typifying each of the sites within the two groups during the various sampling events were analysed separately. SIMPER analysis showed that autumn samples at site K1 were typified by Leptoceridae, Naucoridae and Pisuliidae (Figure 4.4a). Typical species in winter still included Leptoceridae as dominant but Notonemouridae contributed equally with Naucoridae, and more or less similar contributions with Pisuliidae and other grazers at a similarity of 41%. During spring similar contributions were made by the same species. Site K1 samples in summer were dominated by the same grazers, which also included Ceratopogonidae with a small contribution (6%) and the predator Corydalidae (7%).

During autumn site K2 had a very similar species composition to site K1 except that the predator Naucoridae (27% contribution), Baetidae (14%) and Elmidae (13%) was more dominant (Figure 4.4a). During the winter the highest number of families was present including Barbarochthonidae and Simuliidae making contributions. During spring and summer there was not much change in the species composition except that fewer families were contributing to the similarity at site K2 (Figure 4.4c, d). At site K3, during autumn, Leptoceridae again typified the site but there were also families present that did not typify site K1 or K2, which included Gyrinidae (7% contribution), Coenagrionidae (8%) and Caenidae (11%). The presence of these families were due to the habitats available at site K3, which included gravel/sand/mud and aquatic and marginal vegetation. The number of families at site K3 remained more or less similar during winter, spring and summer. Site K3 does have an increase in the number of predators contributing to the similarity than at site K1 or K2. Site K4 was most different to all the sites. During autumn the dominant family typifying the site was Coenagrionidae and this sample was dominated by predators (Figure 4.4a). The least number of families contributed to the spring sample, which was dominated by the grazer Physidae at a contribution of 28%. This trend continued into the summer sample with Physidae contributing 24% and the two predators Coenagrionidae and Corixidae contributing 15% and 14% respectively. The only sample dominated by grazers at site K4 was during winter (Figure 4.4b) with Turbellaria (16% contribution), Physidae (13% contribution) and Baetidae (14% contribution) contributing to the sample similarity.

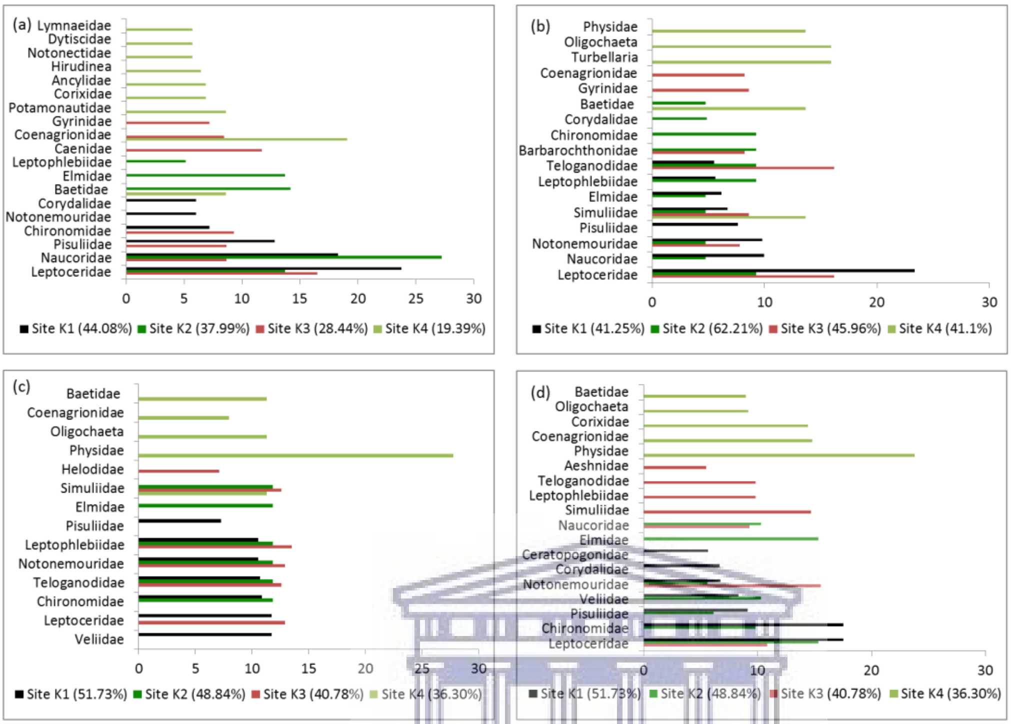

Figure 4.4 SIMPER results for macroinvertebrate composition in comparison at all sites for all sampling events that together contributed at least 70% to the overall similarity (a) autumn, (b) winter, (c) spring and (d) summer. The contribution percentages made to the average similarity by each taxon are shown. Average similarity for each site per sampling event is shown in brackets.

The habitat availability as well as water quality determined the macroinvertebrate assemblages at the sites. SIMPER results across sampling habitat/biotopes between group 1and 2 showed that in both groups the stones biotopes were most dissimilar to the vegetation and gravel/sand/mud biotopes (Figure 4.5). The results showed that typical species in group 1 were not similar to those typifying group 2, which were dominated by predators and scrapers. The dissimilarity percentage and those discriminating species between the two groups and the sampling biotopes are summarized in Table 4.5

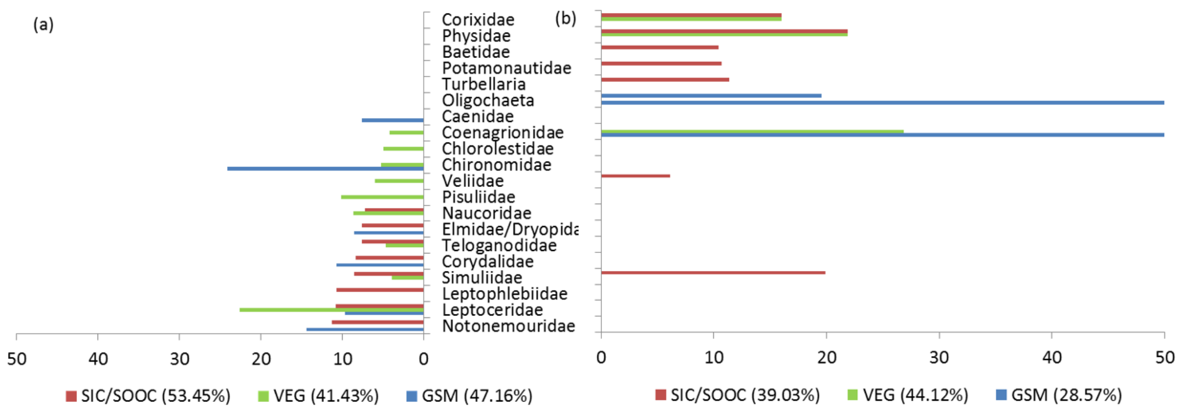

Figure 4.5 SIMPER results of macroinvertebrate taxa in different sampling biotopes that together contributed at least 70% to the overall similarities (a) Group 1 and (b) Group 2. Average similarity percentages within biotopes are shown in brackets

Table 4.5 SIMPER results of the dissimilarity of macroinvertebrate taxa that contributed to differences between Group 1 and 2 based on sampling biotopes (habitat). Differentiating species are limited to the three highest dissimilarity coefficient/standard deviation

Group 1			
Biotopes	**Av. diss (%)**	**Species**	**Diss/SD**
SIC/SOOC & AQ/MV	58.79	Corydalidae Leptophlebiidae	1.3 1.13
SIC/SOOC & GSM	56.89	Simuliidae Naucoridae	1.34 1.25
AQ/MV & GSM	65.89	Pisuliidae Corydalidae Naucoridae	1.23 1.11 1.06
Group 2			
SIC/SOOC & AQ/MV	75.14	Coenagrionidae Oligochaeta Simuliidae	2.27 1.43 1.27
SIC/SOOC & GSM	78.42	Coenagrionidae Simuliidae Turbellaria	2.52 1.74 1.06
AQ/MV & GSM	98.30	Oligochaeta Corixidae Gomphidae	2.27 1.64 0.93

4.3.2 Periphyton communities and biomass

The benthic algal taxa sampled at all four sites over the study period totalled 67 taxa of which 40 were Bacillariophyta taxa (diatoms), 16 were Chlorophyta (green algae), 8 were Cyanophyta (cyanobacteria), 1 Xanthophyta (yellow-green alga), 1 Euglenophyta (yellow-green alga) and 1 Rhodophyta (red alga) (Appendix 4.3). The taxon composition and abundance varied considerably between sites and over time and taxa richness were very similar across sites except at site K4 where the highest numbers were recorded (Table 4.6). The biomass of the four most sampled genera is shown in Figure. 4.9.

Table 4.6 Summary of benthic algal taxa recorded over the study period

Division	Common name	Site 1	Site 2	Site 3	Site 4
Bacillariophyta	Diatoms	20	20	17	20
Chlorophyta	Green algae	4	4	5	12
Euglenophyta	Yellow green algae	1	0	0	1
Cyanophyta	Cyanobacteria	2	4	5	4
Xanthophyta	Yellow green algae	1	1	1	0
Rhodophyta	Red algae	0	1	1	0
Total		**28**	**30**	**29**	**37**

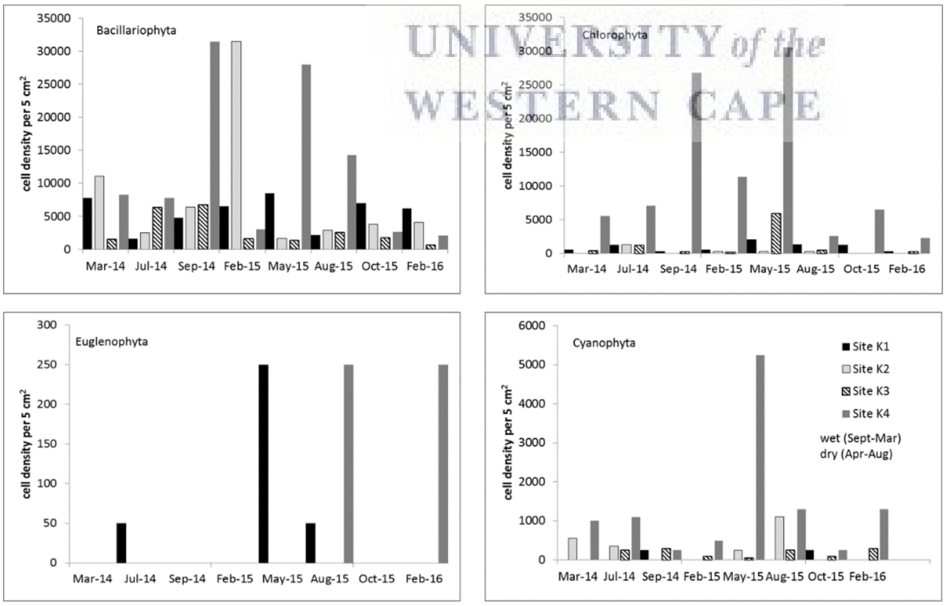

Figure 4.6 Cell densities (abundance) of benthic algal divisions per site over the wet and dry seasonal sampling period (2014-2016)

Benthic chl-*a* samples were consistently low at site K1 in comparison to other sites but increased during the wet season (Table 4.7, Appendix 4.1) with the highest recorded biomass in the wet season being 34.48 mg m^{-2} (eutrophic). The suspended chl-*a* in the water column followed the same trend with elevated values observed during the wet season in comparison to the dry season. At site K2 the benthic chl-*a* peaked during the wet season with not much difference observed in the suspended chl-*a* between wet and dry seasons. The same trend occurred at site K3, with benthic chl-*a* increasing on average during the wet season (3.5 mg m^{-2}) compared to the dry season (2.7 mg m^{-2}). Benthic chl-*a* levels remained similar, on average, at site K4 between wet and dry seasons (2 to 2.4 mg m^{-2}) with a slight increase in suspended chl-*a* during the dry season compared to the wet season. Mesotrophic conditions were recorded at site K1, K2 and K3 during the dry and wet seasons in 2014 and 2016. The benthic chl-*a* samples ranged from 3 mg m^{-2} to 5.5 mg m^{-2} at site K1, 6 mg m^{-2} to 17.5 mg m^{-2} at site K2 and 6.8 mg m^{-2} to 9.85 mg m^{-2} at site K3. However, oligotrophic conditions were more frequently recorded, especially at site K1. Mesotrophic conditions occurred throughout the sampling events at site K4 except when oligotrophic conditions occurred during increased water levels during August, September and October 2015 together with increased mean monthly river flows (Figure 4.7).

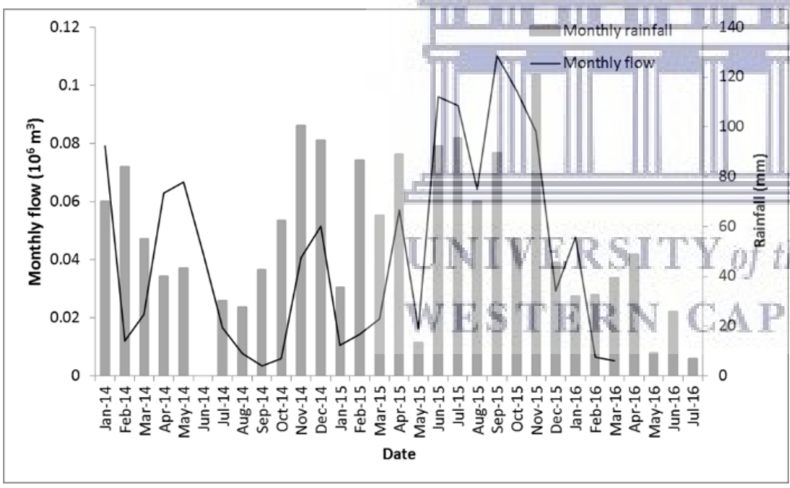

Figure 4.7 Mean monthly flows and rainfall at site K4

Table 4.7 Mean and standard deviation values for physical and chemical parameters measured at the sampling locations during the dry and wet seasons for the Klein Keurbooms and Duiwe Rivers (2014-2016) (n = 40)

Variables	Unit	Site K1		Site K2		Site K3		Site K4	
		Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
Na^{+}	mg L^{-1}	17.3 ± 1.8	17.6±1.1	21.2±2.3	22.0±1.63	36.0±2.3	40.2±8.6	172.2±65.0	151.7±48.3
Ca^{2+}	mg L^{-1}	0.8 ± 0.3	1.3±0.8	1.1±0.4	1.40±0.9	2.5±0.2	3.4±1.3	23.6±9.1	21.2±6.7
Mg^{2+}	mg L^{-1}	2.1±0.1	2.1±0.2	2.7±0.3	2.8±0.3	4.9±0.2	5.2±0.9	21.7±8.7	18.2±5.4
Alkalinity	mg L^{-1}	2.5±0.7	2.1±0.5	3.8±1.0	3.5±0.9	5.5±0.9	32.3±30.9	51.4±16.5	32.6±19.0
NO_x-N	mg L^{-1}	0.1±0.0	0.1±0.1	0.1±0.04	0.1±0.0	0.1±0.0	0.2±0.1	0.4±0.4	0.3±0.4
EC	mS m^{-1}	12.3±0.8	13±0.8	14.8±0.9	16.3±1.7	25.0±2.0	29.2±7.0	123.6±47.8	108.5±33.1
pH		4.9±0.1	6.3±2.8	5.2±0.1	5.1±0.2	5.8±0.2	6.0±0.3	7.5±0.3	7.1±0.8
COD	mg L^{-1}	18.4±6.1	16.5±14.3	25.2±9.0	18.5±15.4	22.0±3.8	17.8±8.7	52.0±3.8	46.2±14.9
TN	mg L^{-1}	0.5±0.1	0.5±0.1	0.5±0.3	0.6±0.3	0.6±0.2	0.7±0.3	1.4±0.6	0.9±0.2
TP	mg L^{-1}	0.1±0.0	0.1±0.01	0.1±0.00	0.1±0.0	0.1±0.0	0.1±0.0	0.1±0.1	0.2±0.2
Si	mg L^{-1}	2.4±0.2	2.6±0.3	2.6±0.1	3.0±0.4	3.2±0.2	3.7±0.5	1.7±1.1	2.2±0.9
Temperature	ºC	15.1±2.01	12.3±3.6	15.6±1.3	13.0±3.3	15.5±1.8	11.2±2.6	15.8±2.1	10.8±2.7
Benthic chl-*a*	mg m^{-2}	1.1±2.6	1.4±1.3	1.1±7.4	0.9±3.1	3.5±3.7	2.7±3.7	2.0±1.9	2.4±2.3
Suspended chl-*a*	µg L^{-1}	0.5±0.6	0.3±0	0.2±0.3	0.3±0.0	0.2±0.3	0.4±0.2	0.2±0.4	0.3±0.2
Water level	m	0.1±0.04	0.1±0.03	0.02±0.03	0.09±0.03	0.2±0.0	0.4±0.3	0.2±0.1	0.2±0.1
Turbidity	NTU	1.9±0.8	1.8±0.8	1.9±0.8	1.8±0.8	3.8±3.7	1.8.±0.7	50.7±69.2	41.2±32.8
Water hardness $CaCO_3$	mg L^{-1}	10.4±0.9	11.7±1.6	13.7±1.6	14.8±2.4	25.0±1.3	28.4±6.2	148.8±58.4	113.3±29.5

The SIMPROF ($p < 0.001$) cluster analysis and MDS ordination showed a separation between two groups; with sites K1, K2 and K3 (Group 1) separate from site K4 (Group 2). Group 1 separated into two sub-groups between sites K1 and K2 and K3 (Figure 4.8). ANOSIM performed for all sites showed that there were significant differences in benthic algae composition between the sites sampled ($R^2 = 0.76$; $p = 0.001$). Pairwise tests revealed that site K1 and K2 differed significantly from site K3 and site K4. No significant difference occurred between site K1 and K2. Sites K2 and K3 were only moderately different. Significant differences occurred between sites K2 and K4 and between sites K3 and K4 (Table 4.8).

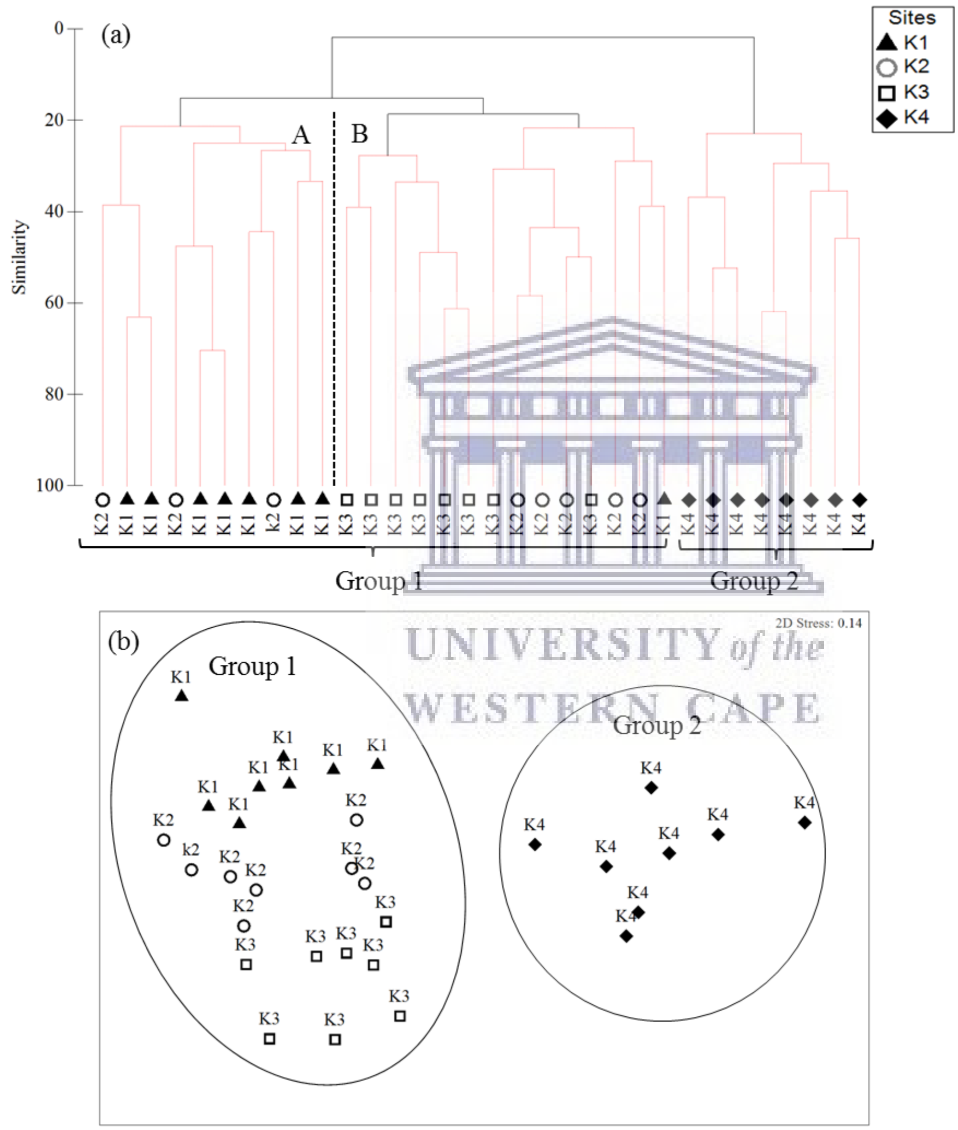

Figure 4.8 (a) Cluster dendogram with sub-group 1A and 1B and (b) MDS ordination of Bray-Curtis similarity between algal species composition at all the sites. Sites codes: K1 = site 1; K2 = site 2; K3 = site 3; K4 = site 4

There were significant differences between the mountain stream, upper foothills and transitional zones, as well as between the upper foothills and transitional zones. The site location and geomorphological zones rather than the sampling season, were responsible for algal assemblage groupings.

Table 4.8 R^2 values for pair-wise tests of differences between sites, geomorphic zones and habitat for algal communities. A significant level of ≤ 0.05 (*) is indicated

Sites	**K2**	**K3**	**K4**
K1	0.294	0.862*	0.998*
K2		0.073	0.985*
K3			0.984*
Geomorphological zones	**Upper foothill**	**Transitional**	
Mountain stream	0.385*	0.998*	
Upper foothill		0.964*	

Results from the SIMPER analysis showed that the average similarity for Group 1 was 20% and for Group 2 was 28% while the average dissimilarity between the two groups was 98%. Species that typified Group 1 for at least 70% of the sample period were dominated by diatoms indicative of low EC and low pH values. Typical species included *Eunotia formica* Ehrenberg, *Eunotia exigua* Rabenhorst, *Navicula radiosa* Kützing, *Tabellaria flocculosa* (Roth) Kützing, *Eunotia incisa* W.Smith ex W.Gregory, *Achnanthes standeri* Cholnoky, *Frustulia saxonica* Rabenhorst and *Navicula heimansioides* Lange-Bertalot (Figure 4.9). A change in community structure occurred during the wet season in 2016 when the average similarity at site K1 was driven by the green filamentous algal species; *Ulothrix zonata* Kützing. Species typical of Group 2 (site K4) had an algal composition of diatoms, green algae and cyanobacteria. The diatoms with the highest abundance and contributions to similarity were *Melosira varians* Agardh, and other commonly occurring species included *Gyrosigma attenuatum* (Kützing) Rabenhorst, *Cocconeis placentula* Ehrenberg, *Nitzschia filiformis* (W.J.E.Smith) Van Heurck , which are all indicative of brackish and saline water, occurring in meso-eutrophic conditions (Taylor et al., 2007a). The green filamentous algae *Spirogyra scripta* Link, *Spirogyra* no 1 Link and *Scenedesmus quadricauda Meyen* had the next highest average abundances and are associated with increased levels of pollution.

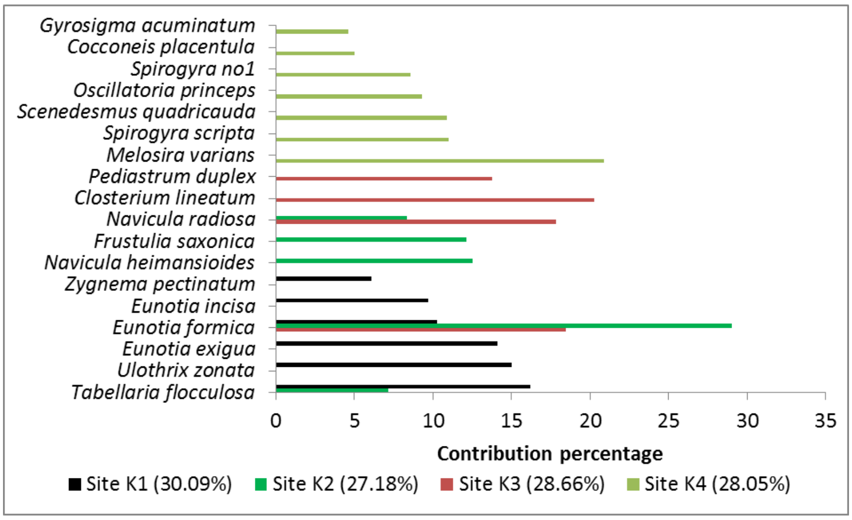

Figure 4.9 SIMPER results of algal taxa at all sites that together contributed at least 70% to the overall similarity. Average similarity percentages are shown in brackets

The average dissimilarity between the two groups was 98%. Although the Diss/SD values were below 1.5 the values above 1.0 distinguished the discriminating taxa between them as shown in Table 4.9. Different algal communities occurring between sampling events were driving the dissimilarities between the sites.

Table 4.9 SIMPER results of the dissimilarity between Groups 1 and 2 based on algal composition taxa. Only the species contributing to at least 70% of the dissimilarity are indicated (90% in total). Differentiating species are limited to the highest dissimilarity coefficient/standard deviation (Diss/SD) values. Species are arranged highest to lowest values.

Average dissimilarity (%)	98.24
Species	**Diss/SD**
Scenedesmus quadricauda	1.48
Melosira varians	1.31
Cocconeis placentula	1.17
Oscillatoria princeps	1.04
Gyrosigma acuminatum	1.06
Oscillatoria sancta	1.01
Eunotia formica	1.01

The taxa typifying each of the sites within the two groups during the various sampling events were analysed separately. SIMPER analysis showed that autumn samples were typified by *Eunotia spp.* and *Tabellaria flocculosa* (Roth) Kützing with an average similarity of 39% at site K1 (Figure 4.10). Typical species in winter still included *Eunotia spp.* with contributions from *Ulothrix zonata* (F.Weber & Mohr) Kützing and *Zygnema no 1* (Vaucher) C.Agardh at a similarity of 64%. During spring similar contributions were made by *Eunotia exigua* and *Tabellaria flocculosa*. Site K1 samples in summer were typified by *Ulothrix zonata*.

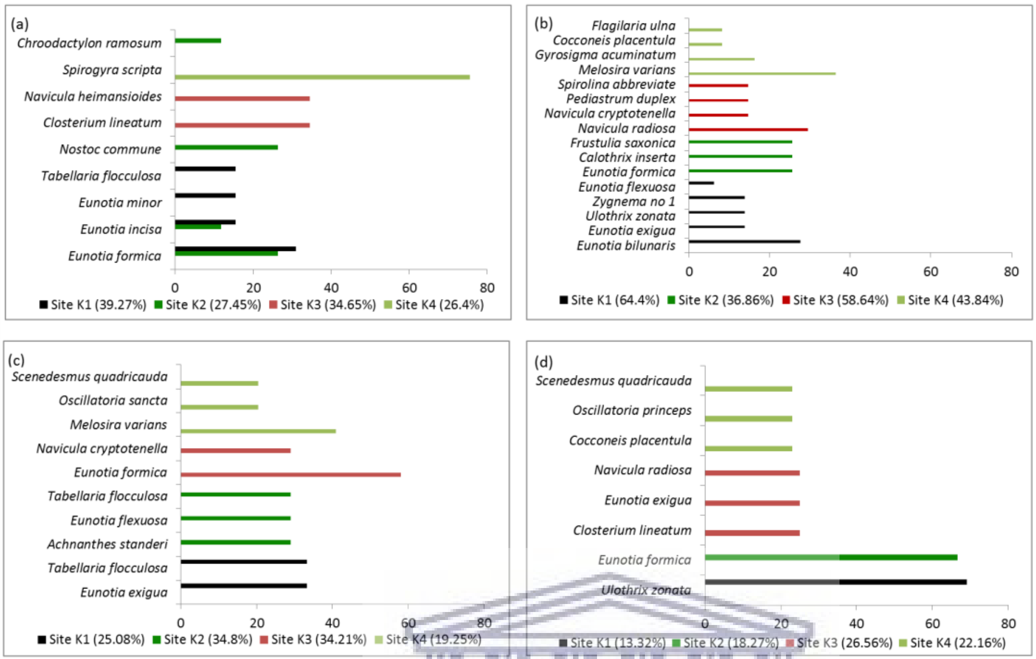

Figure 4.10 SIMPER results for algal composition in comparison at all sites for all sampling events that together contributed at least 70% to the overall similarity (a) autumn, (b) winter, (c) spring and (d) summer. The contribution percentages made to the average similarity by each taxon are shown. Average similarity for each site per sampling event is shown in brackets

At site K2 *Eunotia formica* typified the algal composition during both autumn (26% contribution) and summer (69%) (Figure 4.10a and d). During the winter sampling *Eunotia formica* contributed equally along with *Calothrix inserta* and *Frustulia saxonica* (26%). *Calothrix inserta*, *Tolypothrix*, *Scytonema crispum* and *Nostoc commune*, which are nitrogen fixing filamentous cyanobacteria occurred in the dry season (autumn and winter) at site K2, indicating that nitrogen was possibly a limiting factor at this site. During spring *Achnanthes standeri*, *Eunotia flexuosa* and *Tabellaria flocculosa* were the typical species with equal contributions at 29%. Site K3 was typified by *Navicula heimansioides* and *Closterium lineatum* during autumn and winter *Navicula spp.*, *Pediastrum duplex* and *Spirolina abbreviate*. During the spring sample *Eunotia formica* contributed 58% to the average similarity of 34% while *Navicula cryptonella* contributed 29%. The summer sample showed that C*losterium lineatum*, *Eunotia exigua* and *Navicula radiosa* were typical species contributing at site K3. In these wet seasons *Scytonema crispum* and *Nostoc commune* also occurred at site K3.

Site K4 was most different to all the sites with *Melosira varians* C.Agardh typifying the sites during spring and winter sampling (Figure 4.10b, c). During autumn 76% of abundance was contributed by *Spirogyra scripta* Nygaard. The typical species driving the average similarity during summer were *Cocconeis placentula* Ehrenberg, *Oscillatoria princeps* Vaucher & Gomont and *Scenedesmus quadricauda* (Turpin) Brébisson. The species *Aulacoseira granulate* (Ehrenberg) Simonsen was also sampled at this site but only during one winter sampling, which meant that abundance was not high and therefore did not drive any grouping. The species did however form one of the species resulting in dissimilarity between sites K2 and K4 and sites K3 and K4 during the winter season. The differentiating species between sites contributing to the average dissimilarities are summarized in Table 4.10.

Table 4.10 SIMPER results of the dissimilarity between sites based on algal compositions between sampling events. Differentiating species are limited to the three highest dissimilarity coefficient/standard deviation (Diss/SD) values. Species are arranged highest to lowest values

Sites	Av. diss (%)	Species	Diss/SD
Autumn			
1 & 2	76.31	*Nostoc commune* *Ulothrix zonata* *Eunotia formica*	6.35 3.69 2.25
1 & 3	91.68	*Zygnema no 1* *Eunotia formica* *Ulothrix zonata*	4.82 4.75 3.9
1 & 4	98.98	*Spirogyra scripta* *Eunotia formica* *Zygnema no 1*	7.79 2.28 2.28
2 & 3	77.03	*Nostoc commune* *Closterium lineatum* *Pediastrum duplex*	5.86 1.52 1.52
2 & 4	98.95	*Spirogyra scripta* *Nostoc commune*	12.56 2.25
3 & 4	100	*Spirogyra scripta* *Navicula heimansioides*	6.29 1.76
Winter			
1 & 2	83.77	*Scytonema crispum* *Eunotia bilunaris* *Calothrix inserta*	8.41 4.06 3.19
1 & 3	85.52	*Eunotia bilunaris* *Spirolina abbreviate* *Eunotia exigua*	11.92 11.92 11.92

Sites	Av. diss (%)	Species	Diss/SD
1 & 4	98.30	*Trachelomonas intermedia* *Melosira varians*	19.81 9.51 9.51
2 & 3	86.94	*Spirolina abbreviate* *Eunotia exigua* *Calothrix inserta*	10.31 4.72 3.25
2 & 4	99.34	*Melosira varians* *Gyrosigma acuminatum* *Scenedesmus quadricauda*	9.08 9.08 9.08
3 & 4	93.85	*Melosira varians* *Gyrosigma acuminatum* *Scenedesmus quadricauda*	10.26 10.26 10.26
Spring			
1 & 2	64.43	*Eunotia flexuosa* *Ulothrix punctate*	11.95 11.95
1 & 3	86.83	*Navicula cryptotenella* *Ulothrix punctate* *Scytonema crispum*	5.97 5.97 5.97
1 & 4	100	*Melosira varians* *Gyrosigma acuminatum* *Oscillatoria sancta*	5.46 5.46 3.49
2 & 3	87.18	*Navicula cryptotenella* *Scytonema crispum* *Eunotia flexuosa*	6.22 6.22 5.45
2 & 4	100	*Gyrosigma acuminatum* *Eunotia flexuosa* *Oscillatoria sancta*	6.19 3.37 3.37
3 & 4	98.29	*Melosira varians* *Gyrosigma acuminatum* *Navicula cryptotenella*	5.45 5.45 3.14
Summer			
1 & 2	86.67	*Ulothrix zonata* *Frustulia saxonica* *Eunotia formica*	7.67 3.76 2.66
1 & 3	95.08	*Ulothrix zonata* *Eunotia exigua* *Closterium lineatum*	27.29 10.03 2.16

Sites	Av. diss (%)	Species	Diss/SD
1 & 4	98.49	*Cocconeis placentula* *Ulothrix zonata* *Scenedesmus quadricauda*	11.54 11.54 3.25
2 & 3	91.64	*Eunotia exigua* *Nostoc commune* *Frustulia saxonica*	27.06 6.5 4.1
2 & 4	97.58	*Eunotia formica* *Spirolina abbreviate* *Oscillatoria princeps*	2.77 2.56 2.17
3 & 4	94.93	*Cocconeis placentula* *Scenedesmus quadricauda*	9.97 3.39 2.82

4.3.3 Environmental variables responsible for variance in communities

All water quality variables at sites K1, K2 and K3 were below the Target Water Quality Ranges (TWQR) as determined by DWAF (1996c), although concentrations at site K3 were slightly more elevated than at sites K1 and K2. The pH at sites K1 and K2 was acidic (4.8 and 5.3), becoming more alkaline further downstream (site K3: 5.7 - 6; site K4: 7 - 8) (Table 4.7, Appendix 4.1). The EC at site K1 and K2 ranged between 12 - 16 mS m^{-1}, increasing at site K3 (20 - 30 mS m^{-1}), with the highest recorded at site K4 (> 100 mS m^{-1}), which was above the guideline limits for aquatic ecosystems (DWAF, 1996c). The only exceptions occurred during August 2015 (64 mS m^{-1}) and October 2015 (54 mS m^{-1}), when there was an increase in rainfall, river water levels and river flows. Calcium, magnesium and sodium exceeded the guidelines for domestic water use at site K4 according to DWAF (1996a). Silica was positively correlated with river flows, reaching the highest concentrations during August and October 2015, and was mostly higher at site K3 than at sites K1 and K2. Nutrient concentrations remained low at site K1 compared to site K2 and site K3, where increased NO_x-N concentrations were observed during August 2015. At site K3, NO_x-N concentrations increased from 0.1 mg L^{-1} to 0.3 mg L^{-1} which coincided with increased flows and water levels. Total nitrogen concentrations were always < 1 mg L^{-1}. Total phosphorus concentrations were < 0.05 mg L^{-1} except during March 2014 at site K3 (0.08 mg L^{-1}).

Since a longer data record was available for the Duiwe River where site K4 was located, more detailed analysis was possible. The water quality and flow data recorded at site K4 showed significant negative correlations with river flows. This was with the exception of NO_x-N, PO_4^{3-}-P and Si, which were significantly positively correlated with river flows (Table 4.11). At site K4 the NO_x-N concentrations peaked during the dry season (0.3 to 0.8 mg L^{-1}) and wet season

(0.4 to 1 mg L^{-1}) (Table 4.7), often exceeding the DWAF (1996c) guidelines for the aquatic environment, which classifies mesotrophic conditions as between 0.5 - 2.5 mg L^{-1}. The TN followed the same trend with increased levels during September 2014 and February 2015, reaching 2 mg L^{-1}. Total phosphorus concentrations peaked in the same months as NO_x-N with the highest concentrations recorded in July 2014 (0.33 mg L^{-1}), August 2015 (0.44 mg L^{-1}) and October 2015 (0.26 mg L^{-1}). Longer term phosphate (PO_4^{3-}-P) data showed the same trends, with the highest concentrations recorded during spring (September 2014) at 3.5 mg L^{-1}. Phosphate levels exceeded the recommended DWAF (1996c) guidelines for aquatic environments of 0.02 to 0.1 mg L^{-1} in the majority of cases (Petersen et al., 2017). Alkalinity levels were always higher at site K4 than at site K1, K2 and K3. The lowest levels were recorded during August 2015 (34 mg L^{-1}) and October 2015 (24 mg L^{-1}) with associated hardness ($CaCO_3$) of 79 mg L^{-1} during August and 67 mg L^{-1} during October, which coincided with increased water levels and flows (Appendix 4.1). All other sampling periods had increased alkalinities between 48 mg L^{-1} to 63 mg L^{-1} coupled with increased TP, NO_x-N and TN. Sodium (Na^+) exceeded TWQR levels of 0 - 100 mg L^{-1} (domestic use) and even on lower concentration occurrences the TWQR for irrigation of 0 - 70 mg L^{-1} was exceeded (winter and spring months in 2015). Calcium and magnesium exceeded TWQR levels for domestic concentrations at 0 - 32 mg L^{-1} and 0 - 30 mg L^{-1} respectively, during spring sampling (September 2014). Silica also reached the highest concentrations during winter (August 2015) and spring (October 2015) at 3.5 and 3.2 mg L^{-1} respectively (increased with flows). The silica concentrations were lowest at site K4 during low river flow months.

Table 4.11 Seasonal Mann-Kendall trend analysis for Site K4 long term data set with the Spearman rank correlation (r_s) between river flow and physico-chemical parameters for the period 1998-2016

Parameter	Spearmans r_s	Trend	Kendall's Tau
Site 4 (1998 - 2016)			
Total Alkalinity	-0.595***	+	0.248***
Calcium Ca^{2+}	-0.705***	+	0.173*
Chloride Cl^-	-0.704***	+	0.125*
Electrical conductivity (EC)	-0.709***	+	0.127*
Potassium K^+	-0.588***	+	0.266***
Sodium Na^+	-0.686***	0	0.150*
Magnesium Mg^{2+}	-0.708***	0	0.171**
pH	-0.295***	0	0.149*

Parameter	Spearmans r_s	Trend	Kendall's Tau
Site 4 (1998 - 2016)			
Phosphate PO_4^{3-}-P	0.447***	-	-0.132*
Nitrite and nitrate NO_x-N	0.255*	-	-0.191***
Ammonium NH_4^+-N	-0.111	+	0.233***
Sulphate SO_4^{2-}	-0.275***	0	-0.041
Silica Si	0.577***	0	-0.090

(+), upward trend; (-), downward trend; (0), no significant trend * Significance: *** < 0.0001; ** < 0.001; * < 0.05

The PCA analysis indicated that the eigenvector, PC1, explained 45% of the variation, PC2 explained 11% while PC3 and PC4 explained 7% and 6% of the variation in environmental variables respectively, i.e. 69% of the variability between them (Figure 4.11a, Appendix 4.4). The ordination showed a shift along the primary axis from more pristine sites to more polluted sites with a clear separation between upstream sites (K1, K2 and K3) and site K4 (shown by smaller ellipsoid in Figure 4.11a). PC1 was driven by Na^+, Ca^{2+}, Mg^{2+}, alkalinity, EC and pH, SASS, ASPT and habitat scores. A gradual increase of these variables occurred along the gradient from the reference site K1 to the cumulative impact site K4, except in the SASS metric scores. The PCA showed that the less impacted sites K1, K2 and K3 were associating with the higher SASS, ASPT and habitat scores as well as the number of taxa found. PC2 and PC3 were driven by the nutrient variables (TP, TN and NO_x-N) together with benthic and suspended chl-*a*. PC2 was also driven by water temperature and macroinvertebrate grazers and predators.

The DISTLM marginal tests showed that the greatest amount of variation in species data was explained by pH (43.3%). This was closely followed by EC (41.2%), Ca^{2+} (41.2%), Na^+ (40.8%) and Mg^{2+} (40.8%) (Appendix 4.5). Alkalinity and COD explained 36.5%, and 28.6% of variation respectively. Less of the variation was explained by flow (7.4%) and NO_x-N (9.7%) but TN (19.7%) contributed more. Silica explained 12.4% of the variation while macroinvertebrate grazers accounted for 25.8% of the variation. The co-variates that contributed to the overall model variation (99.4%) in species explained by DISTLM were shown by the sequential tests. The pH was the most significant contributor to the variation (43%), followed by macroinvertebrate taxa (7.6%) benthic chl-*a* (5.7%), water temperature (5.5%), chl-*a* (water column) (4.98%), NO_x-N (4.89%), water level (4.7%), TP (4.46%), Si (4.32%) and flow (4.1%). Together these co-variates contributed to 89.7% of the total variation. The dbRDA ordination plot (Figure 4.11b) showed site K3 associating with NO_x-N, silica and benthic chl-*a*. Grazers, SASS scores, habitat and ASPT scores were associated with sites K1, K2 and K3 along with suspended water column chl-*a* and water temperature. Site K4 again separated from the upstream sites, showing its dissimilarity, and associated with alkalinity, pH, EC and water levels.

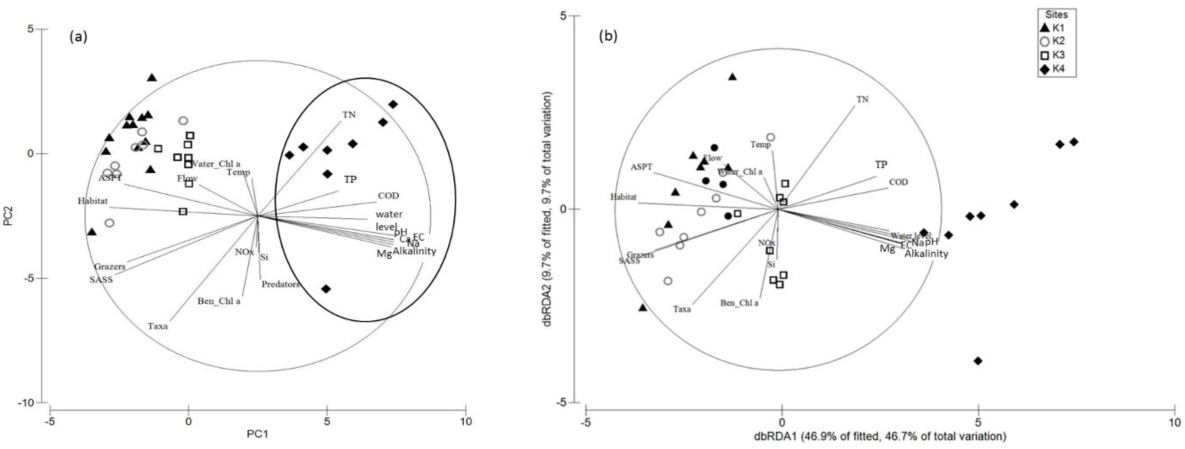

Figure 4.11 Ordination plots showing the relationship of macroinvertebrate and algal assemblages with the physico-chemical variables and vectors showing the Spearman correlation between these variables at all sites (a) PCA (Euclidean distance matrix) with the smaller ellipsoid showing the site K4 separation and (b) dbRDA plots

4.4 Discussion

The physical river template, water quality and hydromorphology were important drivers of the variation in algal biomass and composition and benthic macroinvertebrate abundance and composition. The first and second order streams had good general water quality that was low in nutrients, well oxygenated and acidic with good habitat quality. Both the algae and macroinvertebrates were indicative thereof. Algal biomass remained low, likely due to the occurrence of a high abundance of grazers (Figure 4.1) in the upstream sites, which dominated the samples during all seasons together with a closed riparian canopy. This was consistent with findings from Feminella and Hawkins (1995) and Hawkins et al. (1997) who showed that with increased abundances of grazers, algal biomass decreased.

As was expected the sites further downstream showed deterioration due to land derived impacts on water quality and the physical river condition. This was evidenced by the increase in the diatom, *Navicula cryptotenella* Lange-Bertalot, which is tolerant to moderate levels of pollution (Taylor et al., 2007a), as well as well as *Pediastrum duplex* Meyen (green algae) and *Spirulina abbreviata* Turpin (cyanobacteria). The presence of *Nostoc commune* Vaucher ex Bornet, *Calothrix inserta* Agardh ex Bornet, *Tolypothrix* sp. Kützing ex Bornet and *Scytonema crispum* Agardh ex Bornet, which are nitrogen fixing filamentous cyanobacteria, occurred at the foothill sites, indicating that nitrogen was likely to be a limiting factor, and corresponding to the low nutrient levels recorded throughout the sampling period (Table 4.7) (Oberholster et al., 2016). When an increase in the NO_x-N and TN concentrations was observed at site K3 during the dry season in 2015, these species were replaced by the diatom *N. radiosa*, which occurred in large numbers. Marks and Lowe (1989) also found *N. radiosa* associated with increased nutrient supply.

Elevated benthic algal biomass was recorded at site K4 compared to the sites in the mountain stream and foothills, except during increased flow periods (i.e. August and October 2015), when decreased alkalinity and hardness was observed. Filamentous green algal biomass is associated with alkalinity or hardness due to bicarbonate ions providing additional carbon dioxide for photosynthesis (Oberholster et al., 2016). Previous studies showed that alkalinity below 25 mg L^{-1} limits phosphorus availability to algae or prevents growth due to low mineralized carbon levels (Wurts and Durborow, 1992, Summers, 2008). The lower benthic algal biomass (Figure 4.9) during these times was a possible indication that less TP was available for algae growth and therefore more available in the water column. This was associated with elevated levels of suspended chl-*a* levels in the water column. According to Dodds et al. (2002), when suspended chl-*a* levels are increased, soluble phosphorus levels should also increase as algae contain chlorophyll and phosphorus. The low benthic algal biomass may also be attributed to the increase in water column depth when light penetration to the river bed was limited due to observed increased turbidity (Table 4.7). Munn et al. (1989) showed that turbidity had a high correlation with chl-*a* in agricultural streams, so that increased turbidity was concurrent with a decrease in light, thereby reducing chl-*a* accrual.

Changes to the river hydromorphology also contributed to changes in the algal community structure. The current study showed that when flows are reduced or natural flows are altered, benthic filamentous algal mats can form, which can reduce or eliminate sensitive macroinvertebrate taxa, which will ultimately affect ecological integrity (Ewart-Smith, 2012). In the mountain stream at site K1, after a low rainfall wet season (December 2015) moving into the dry season (2016), the typifying diatoms were replaced by benthic filamentous green algae (*Ulothrix zonata*), as conditions became favourable for their establishment (Figure 4.9). Biggs and Price (1987) also found this species associated with good quality, low EC river water.

When elevated nutrient concentrations occurred at site K4, species of the genera *Oscillatoria Vaucher ex Gomont*, *Spirogyra*, *Oedogonium* Link and *Stigeoclonium* Kützing, formed dense benthic mats, especially during the dry season. Douterelo et al. (2004) and Schneider and Lindstrøm (2011) also found these species associated with higher nutrient concentrations. The reduced river flow due to run-of-river abstraction, water storage upstream in instream dams and other instream obstructions (Petersen et al., 2017) further aided the establishment of these species. This was similar to previous findings of Ewart-Smith and King (2012) who reported that filamentous green and blue-green algae were prolific during stable base flows in nutrient enriched rivers with high temperatures. Biggs and Smith (2002) reported that increased algal richness occurred when floods occurred at lower frequencies. Increased flow periods (August 2015 and October 2015) were associated with decreased alkalinities and hardness and no algal mat formations occurred. Similar observations were made by Summers (2008) who found that hardness of water at 100 mg L^{-1} was ideal for filamentous mat growth and that hardness of 120 - 150 mg L^{-1} suppressed the mat formation. This was also observed by Oberholster et al. (2016) who found that filamentous algae associated with alkalinity rather than nutrients, while the current study shows filamentous algae, with the exception of *N. commune*, occurring in both increased nutrient and alkalinity conditions. The benthic filamentous algae gave a good indication of nutrient enrichment, which could be related to agricultural activity in the catchment. Since the autecology for algae is known from literature (Oberholster and de Klerk, 2014), it was possible to establish that nutrient enrichment occurred due to the presence of certain diatoms and benthic filamentous green algae, which was validated with observations of increasing nutrient concentrations (Table 4.7).

The macroinvertebrate taxa identified to family level were sufficient in detecting the broad scale impairment to water quality but we could not always ascertain the response to specific water quality variables, especially in more impacted sites. The macroinvertebrate taxa were indicative of deteriorating water quality moving down the longitudinal impact gradient. The same pollution tolerant macroinvertebrate taxa occurred at the nutrient enriched site K4 (Figure 4.3) during each sampling event. It may be that these families are indicative of increased nutrient concentrations. However, this would require further testing by increasing the number of sampling sites in similar river systems.

Macroinvertebrates were better indicators of habitat integrity and were sensitive to changing hydromorphology, as evidenced by changing abundance and type of taxa that occurred. The majority of predators in the upstream sites were sampled at site K2 and most taxa from the order Odonata were sampled at site K3. The latter is consistent with a previous study by Smith et al. (2007), who found fewer taxa from this order occurring in closed canopy streams compared to open canopies and that they were sensitive to riparian canopy structure rather than composition (alien versus indigenous). This may explain their increased abundances at site K3 which had a 50% open canopy.

Changes to river flow conditions at site K4 resulted in virtually dry riffle habitats during the dry season due to low water levels and river flows, creating fewer sampling habitat. This in turn changed the macroinvertebrate community to represent taxa that were less dependent on flow. During the wet season the channel filled as shown by the increase in the wetted channel width and more habitats were available to macroinvertebrate taxa (Table 2.4, Figure 2.12). Similar observations were made by Eady et al. (2014). The macroinvertebrate taxonomic richness therefore increased during the wet season rather than decreased as a result. However, no change occurred in SASS and ASPT scores at site K4 as the macroinvertebrate taxa recorded were pollution tolerant and low-scoring due to land derived impacts to the water quality.

However, this showed that there is the possibility for river self-purification with improved flow management. Further study on the sensitivity of macroinvertebrates to flow also provides the opportunity to identify indicator species for flow reductions or flow restoration (Dewson et al., 2007). Rivers naturally have patches of flow heterogeneity which different species of either macroinvertebrates or algae will occupy. These bioindicators can therefore be used in strategies such as environmental water allocations where flows can be set or restored for different hydrological zones within a river system, thereby improving the natural flow regime as well as ecosystem responses to managed flows (Thoms and Parsons, 2003, Dollar et al., 2007). The diatoms present gave some indication of the habitat integrity but were not influenced by river flows. Where the physical river condition deteriorated at sites K2 and K3, the diatoms *Nitzschia* sp., *Frustulia* sp. and *N. radiosa* were recorded. Kutka and Richards (1996) found these taxa were positively correlated with increased bank erosion and were turbidity-tolerant.

The advantages of using bioindicators compared to traditional assessment of river condition are numerous and have been explored by Li et al. (2010), Holt and Miller (2011) and Gökçe (2016), among others. The assessment of the river systems using only water chemistry will provide a snapshot view of river condition and long-term data is necessary to detect changes in river systems (Oberholster et al., 2017). Using continuous logging instruments are costly to purchase and require regular field maintenance to ensure data reliability (Gökçe, 2016). The current study showed that the use of bioindicators in addition to water chemistry provided a holistic view of river integrity, which included water quality and the structural functioning (habitat diversity, flow characteristics, biological interactions) of the aquatic ecosystem. Chessman et al. (1999) states that ecosystem health and biological integrity not only includes water quality but how these structural aspects interact with water quality and so defines river condition. The changing environmental conditions resulted in variation of biomass and community assemblages of bioindicators used as was shown in the current study. The algae and macroinvertebrates responded to stressors such as water quality impacts, reduced flows and disturbances such as increased flows and instream habitat alterations. This makes them useful in predicting ecosystem response to such events (Ewart-Smith and King, 2012), which is important when considering holistic restoration of river integrity.

4.5 Conclusion

Results over wet and dry seasons showed that less impacted sites were associated with increased pollution-sensitive macroinvertebrate and algal taxa along with increased habitat integrity. The macroinvertebrate taxa indicated water quality deterioration but not always the specific impact. Macroinvertebrates were good indicators of general river condition and habitat integrity and were sensitive to changing hydromorphology. The diatoms recorded were indicative of the pH, alkalinity and electrical conductivity in the river systems and were not influenced by river flows. The benthic filamentous algae were better indicators of nutrient enrichment which was associated with agricultural activity in the catchment. The overall conclusion of the study was that using both macroinvertebrates and the full consortium of algae as bioindicators proved beneficial in assessing aquatic ecosystem response to changing environmental conditions and land use in short coastal rivers. The bioindicators responded to reduced flows due to agricultural abstraction and instream channel obstructions, alteration to instream habitat and water quality alteration. Although water chemistry assessment of a river is able to provide information on quality, combining this with bioindicators would provide additional information on structural river functioning.

Chapter 5: Linkages between riparian morphodynamics, land cover and water quality

This chapter was submitted for publication to the African Journal of Aquatic Science and is currently in review:

Petersen, C.R., and Jovanovic, N.Z. and Grenfell, M.C. 2018. The effectiveness of riparian zones as a mitigation to water quality along a longitudinal gradient in an agriculturally impacted river system in South Africa.

5.1 Introduction

The riparian ecotone, which is the interface or transition area between the terrestrial and aquatic ecosystems, has become an important aspect in protecting water resources quality from non-point source pollution in agricultural landscapes (Vought et al., 1994, Fennessy and Cronk, 1997, Connolly et al., 2015, Dindaroğlu et al., 2015). There are numerous studies on the goods and services that riparian ecotones, or as referred to in this study, riparian buffer zones, provide in such landscapes. Among these are flood attenuation, aquifer recharge, stabilizing streambanks and reducing channel erosion, trapping or removing phosphorus, nitrogen, and other nutrients and contaminants, trapping sediment, maintaining habitats (terrestrial and aquatic), moderating water temperatures and providing recreation and aesthetics (Wenger, 1999, Steiger and Gurnell, 2002b, Steiger et al., 2005, Allen et al., 2016, Chase et al., 2016, Tanaka et al., 2016a). Riparian zones vary significantly according to their location in a landscape and develop through plant succession controlled by hydrogeomorphic parameters. These can include river discharge regimes, water table fluctuations, sediment transport regimes, erosion, deposition, sediment texture and topography, which depend on the lateral, vertical and longitudinal linkages between the channel and the floodplain (Steiger and Gurnell, 2002b, Richardson et al., 2007, Noe, 2013).

The location of the riparian zone can determine the width or extent, which is dependent on valley geometry. For example, rivers occurring in confined valleys may have narrower riparian zones with more lateral connectedness to hillslope processes, whereas rivers in unconfined valleys are less connected to hillslope processes, have wider developed riparian zones with more influence from fluvial processes (Tabacchi et al., 1998, Polvi et al., 2011). River and riparian geomorphology can therefore have an impact on the ability of riparian vegetation to perform certain functions such as the sediment and nutrient attenuation potential. For example, if the riparian zone consists of a range of morphological units, which may or may not be hydrologically connected to the subsurface flow, nutrients carried via groundwater may not come into contact with the zone where plant uptake or microbial denitrification can occur (Ranalli and Donald, 2010). The elevation at which river banks or associated morphological units form could affect where deposition of sediment and nutrients occur, that may be transported by the river channel (Steiger and Gurnell, 2002b), which can affect the type of vegetation that become established. Since the vegetation is dependent on water for survival and growth, the distance from the river channel or depth to the riparian water table, especially in arid or semi-arid areas will become important (Richardson et al., 2007). The physiography and hydrogeomorphic processes will therefore determine the physical templates that will develop to provide the riparian habitat (Steiger et al., 2005).

Changes to the riparian zone such as alien plant invasion, will also impact on the potential of riparian zones to provide ecosystem services. Often these areas become a source or a sink for pollutants depending on the flow paths of water draining to rivers (Dosskey et al., 2010). The riparian vegetation can therefore have a direct effect on water quality and on instream biological communities. Rowntree (1991) found banks invaded with Australian *Acacias* to be transformer species of habitat as they formed monospecific stands with dense canopies, which limited the growth of understorey plants and promoted bank erosion. The dense stands of *Acacia mearnsii* trees are known to increase water use from riparian zones, which results in changes to catchment hydrology and ultimately reductions in river flows (Le Maitre et al., 2016). Samways et al. (2011) reported changes in water quality variables such as lower temperature, increasing light and fine sediments, conductivity, pH, dissolved oxygen, habitat quality and availability, allochthonous input, river shading and bank stability.

Numerous studies exist on appropriate riparian buffer widths and vegetation types that are effective in the mitigation of agricultural impacts to surface and groundwater quality. Very few of these studies occurred in a South African context with a Mediterranean-type climate and rainfall throughout the year. Although guidelines for riparian zones as buffers to rivers, wetlands and estuaries have been developed by Blanchè (2002) and Macfarlane et al. (2014), they are primarily based on international studies with limited field-based information from South Africa.

The aim of this chapter was to assess linkages between riparian morphodynamics and water quality and to determine the effectiveness of the riparian zone in mitigating land derived impacts on water quality (nutrients from pastures) along an anthropogenic impact gradient (from the reference site, to the agricultural pastures and to the alien vegetation riparian zones). The objectives were: 1) to examine, characterise and identify the plant community distribution patterns at the selected sites; 2) to determine the water and sediment quality emanating from variations in land cover (the buffer zone land cover/use types described above) along the Klein Keurbooms River using runoff plots and 3) to examine the influence and effectiveness of the riparian vegetation in mediating nutrient fluxes and improving river water quality in comparison to adjacent pastures associated with minimum tillage. The chapter was designed to meet objective 3 of the study (Chapter 1, section 1.4).

5.2 Methods: Data collection and analysis

5.2.1 Sampling sites

Three riparian zones were selected which were located along a 450 m reach of the Klein Keurbooms River, which were the same sites as used for data collection in Chapter 4 shown in Figure 2.8a and b (Chapter 2, section 2.3). The study sites were representative of the various land covers (pastures, contrasting riparian vegetation types) and land uses (agriculture and natural) on water quality.The most upstream site was representative of reference water quality and vegetation, located in an indigenous forest upstream of all impact (site K1), further downstream a site was located in a riparian zone partially invaded by alien vegetation (semi-indigenous) (site K2) and the third site furthest downstream was a riparian zone almost completely invaded by alien vegetation (degraded) (site K3). Two separate areas were located in the pasture field, one adjacent to the semi-indigenous riparian zone and one adjacent to the degraded riparian zone site (Figure 2.8b). Although the riparian vegetation analysis included the cumulative impact site K4 on the Duiwe River, the runoff plots were only located along the Klein Keurbooms River. The riparian vegetation of the three riparian zone areas were assessed and characterised along each of the cross-sections surveyed and used in Chapter 4. The perennial *Pennisetum clandestinum* (kikuyu grass) and *Lolium perenne* (perennial rye grass) are utilised in pastures, which are fertilized with a nitrogen/potassium mixture after the growing season. Pasture management involves minimum tillage with annual replanting during April-August. Site and runoff plot characteristics are summarised in Table 2.4 and 2.5 (Chapter 2, section 2.3).

5.2.2 Runoff plots and runoff determination

To understand the runoff potential and water quality associated with the runoff from the pastures and in relation to riparian zone areas, 1m^2 runoff plots were installed (Figure 5.1). Three replicate plots were installed at each land cover: indigenous forest (K1), two pasture areas adjacent to each riparian zone site, semi-indigenous riparian zone (K2) and degraded riparian zone (K3), (15 in total, Table 2.5, section 2.3). Each runoff plot consisted of a galvanized metal frame that was inserted 0.1 m into the ground and 0.1 m above ground. The pastures were undulating areas of land and runoff plots were specifically located in a downslope direction (toward the Klein Keurbooms River and riparian zone) at the edge of the pastures. Overland flow from the pasture toward the riparian zones were also noted during periods of higher rainfall events while sampling. The downstream frame consisted of a canal with a line of drain holes to funnel runoff water, which was installed at surface level (Figure 5.1). The canal was covered with a protective cover to prevent rain or other undesirable runoff to add to that collected in the receiving tank. A pipe was connected from the canal collector to the receiving container buried below the ground surface. A rain cover was placed over the tank to prevent rain water from collecting in the hole. The volume of surface runoff was determined by measuring the water in the collecting tank. A composite sample of runoff was sampled quarterly, at the end of rainfall events, after thorough mixing of the sample to suspend the contents. The runoff plots located in the pastures were maintained to represent the surrounding conditions in the field.

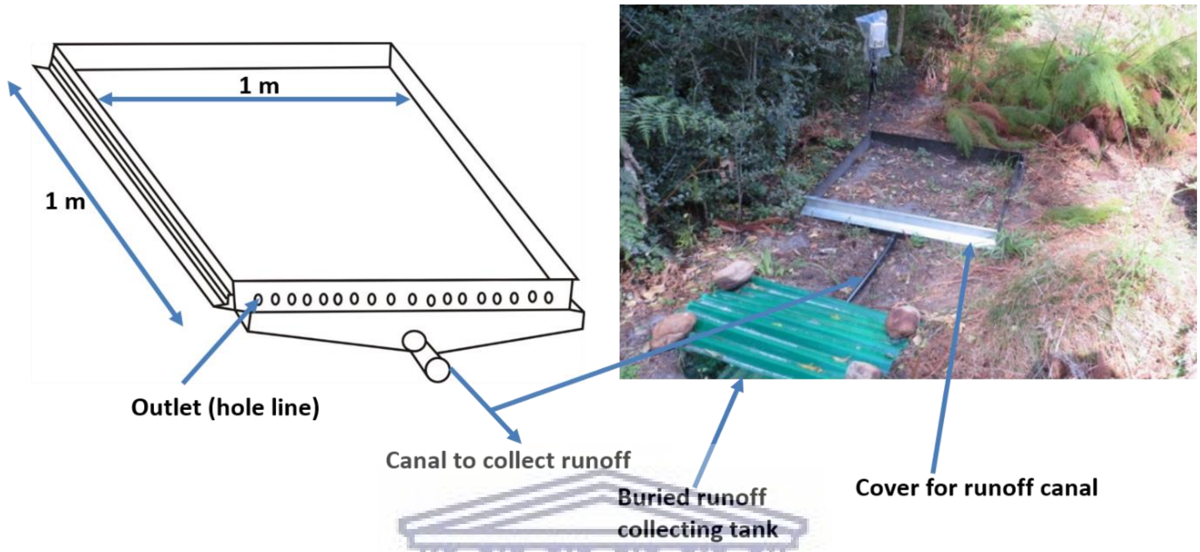

Figure 5.1 Overview of the 1 m² metallic frame. Adapted from Patin et al. (2012)

The analysis of the water samples represented the average content of nutrients of surface runoff over the runoff plot during preceding rainfall events at the time of sampling. The length of time between runoff and sample collection may have resulted in alteration of the nutrient concentrations. Moore and Locke (2013) tested various storage and preservation methods for nitrogen and phosphorus and reported that no single method worked best for all species of nutrients. TN and TP were used as proxies for possible alteration, and provided suitable indications of the inter-site variation in total nutrient loads. Similar ambient conditions for storage of runoff water were kept using dark storage containers, which were covered, below ground to minimise alterations. The surface water runoff coefficient was determined by dividing the collected runoff volume by the recorded daily rainfall from the rain gauges located at runoff plots. The temperature (ºC) and average accumulated rainfall was measured hourly with the use of three rain gauges and the data were recorded with portable ONSET HOBO Pendant Event data loggers (Onset Computer Corp.) that were installed at the runoff plots: one in the indigenous forest, one in the semi-indigenous riparian strip (canopy throughfall) and one in the pasture (open sky rainfall) to represent differences in rainfall occurring with and without tree canopy interception.

5.2.4 Runoff water quality

Runoff water samples were collected quarterly (2014-2017) to represent different seasons from all containers to assess water quality from the land covers. Sample collection coincided with river water sampling. A detailed description of the water quality parameters sampled was described in Chapter 2, section 2.3.2.

5.2.5 River water quality

Samples for river water quality analysis were collected quarterly (2014-2017), coinciding with runoff water sampling. All water samples were collected in pre-rinsed, 1 L polyethylene bottles and placed on ice in the dark, with no addition of preservatives. The river water samples data set was common to the analysis in Chapter 4 as well as Chapter 5. The methods and parameters measured was described in Chapter 2, section 2.3.2.

5.2.6 Sediment sampling

River bank sediments and sediment in the runoff plots were sampled during 2014, 2015 and 2016. A detailed description and characterization of the river bank sediment are presented in Chapter 2, section 2.3.1. The data set was common to the sediment data set used in Chapter 4. The data set with grain sizes and sediment chemistry is shown in Appendix 2.1 and Appendix 2.2. The volumetric soil water content in each runoff plot site was measured using Decagon EC-5 soil moistures sensors installed at 0.1 m and at 0.3 m depths to relate to the rainfall and runoff events that occurred. Decagon EM50 loggers were used to record soil water content measurements hourly over the entire study period.

5.2.8 Statistical analysis

Vegetation

Multivariate statistics were used to determine the spatial and temporal patterns of similarity or dissimilarity in the distribution, composition and abundance in riparian vegetation in relation to the downstream land cover/use. PRIMER (version 6) and its add-on package PERMANOVA+ were used to perform multivariate analyses (Clarke and Warwick, 2001, Anderson et al., 2008). Analysis of similarities (ANOSIM) was used to determine if there were any significant differences in riparian plant species composition between the sites, seasons of sampling and bank position (Clarke and Gorley, 2006). A similarity profile (SIMPROF) permutation test was run a priori on all site samples to assess statistically significant evidence of true clusters in samples together with ordination by non-metric multidimensional scaling

(MDS) based on Bray-Curtis similarities. Tests were considered significant at $p \leq 0.05$. The SIMPER (similarity percentages) (Clarke and Gorley, 2006) routine was used to discern typical or distinguishing species of the groups identified by the cluster analysis for the plant communities. The riparian vegetation data were 4th root transformed to account for rarer species present (Reinecke et al., 2013). Distance-based Linear Modelling (DISTLM) in PERMANOVA+ was used to determine which environmental variables best explained the variation in vegetation assemblages at the sites. The environmental variables included the sediment analyses already mentioned, elevation above and distance away from the active channel thalweg. Where necessary the environmental data were log (x+1) transformed and normalized to limit variances in the data prior to analysis (Clarke and Warwick, 2001, Clarke and Gorley, 2006).

Runoff plots

Analysis of Variance (ANOVA) and correlation analyses in XLSTAT (Addinsoft 2017) among runoff volume, slope, and land cover; rainfall and sampling season (wet and dry) was used to detect changes in specific environmental variables between sites. Where significant differences were found ($p \leq 0.05$) in ANOVA, Tukey's multiple post-hoc comparisons of differences between sites and seasons were applied. Correlation analysis was performed between rainfall and volumetric soil moisture for sites where runoff plots were located to improve the understanding of hydrological and runoff generation processes. Data, where necessary, were log (x+1) transformed to fulfil the assumptions of normality and to reduce variance before analysis. To determine if there were any significant correlations (spatially) between water quality from generated runoff and land cover, principal component analysis (PCA) was performed using XLSTAT. The PCA analyses were also applied to the surface water quality variables.

5.3 Results

5.3.1 Plants community distribution patterns: General vegetation composition

A total of 72 quadrats were sampled at 12 transects across the four sampling sites on the Klein Keurbooms and Duiwe Rivers. The number of species sampled totaled to 115 species from 61 families (Appendices 5.1 and 5.2). Trees were the most commonly recorded species, thereafter herbaceous species followed by shrubs (Figure 5.2). Climbers, graminoids, geophytes, hydrophytes and helophytes were recorded in lower but equal numbers. The majority of species were perennial, which were mostly trees, shrubs and herbs and were indigenous (82%) while a small percentage was endemic (6%). Most of the endemic species occurred at sites K1 and K2 with a smaller percentage occurring at site K4. Overall few exotic species (8%) were recorded mostly trees which included *Acacia melanoxylon*, *Acacia mearnsii*, *Acacia dealbata* and some herbaceous species, which included *Cirsium vulgare*, *Urtica urens* and *Urtica dioica* (Appendix 5., 5.2). The forested sites K1 and the riparian zone at site K2 were the most species rich consisting of the highest percentage recorded indigenous trees and herbaceous species on both left and right banks (Table 5.1).

Table 5.1 Species richness per sampled transect

Vegetation growth form categories	Site K1			Site K2			Site K3			Site K4		
	1.1	**1.2**	**1.3**	**2.1**	**2.2**	**2.3**	**3.1**	**3.2**	**3.3**	**4.1**	**4.2**	**4.3**
Climbers	3	1	3	3	1	1	2	1	0	1	2	2
Geophytes	1	2	1	2	1	2	2	0	0	0	2	1
Graminoids	2	1	2	1	0	0	1	0	0	0	4	0
Helophytes	0	0	1	2	1	2	0	1	0	1	1	0
Herbs	4	6	6	10	4	5	5	3	1	2	7	3
Hydrophytes	0	0	0	0	0	0	3	1	0	0	2	1
Scrambler	1	0	0	0	0	0	0	0	0	0	0	0
Shrubs	5	4	5	5	5	3	4	2	0	0	3	4
Succulents	0	0	0	0	0	0	0	0	0	1	0	0
Trees	12	11	11	6	10	4	6	7	4	3	5	5
Total number of species	28	25	29	29	22	17	23	15	5	8	26	16

The riparian zone at site K2 and K3 was adjacent to extensive pasture areas on the right bank and accommodation cottages occurred on the left bank at site K3. Pastures were planted with the grasses *Pennisetum clandestinum* (kikuyu grass) and *Lolium perenne* (perennial rye grass). Vegetation at site K1 and partly at site K2 (transect 2.1) was heterogeneous with mature species that provided good structure to banks. Undergrowth was common on the forest floor at site K1 and at site K2 beneath the indigenous trees. Very little to no undergrowth occurred beneath the alien trees at transects 2.2, 2.3 and transects 3.2 on the right bank. The left bank at site K2 consisted of indigenous vegetation. Shrubs were more abundant at sites K1 and K2 while most graminoids were recorded at site K4. Herbaceous species occurred at all sites but the most were recorded at site K2 and K3. Climbers were most abundant at site K1.

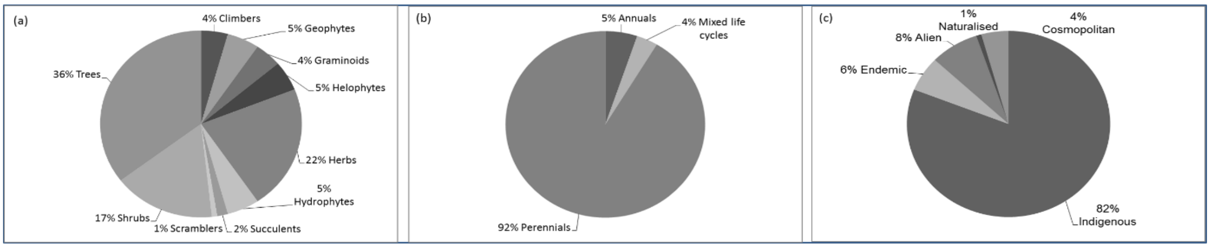

Figure 5.2 Vegetation composition from sampling sites from the Klein Keurbooms and Duiwe Rivers showing (a) growth forms, (b) life cycle and (c) alien or indigenous

Species richness declined at sites K3 and K4 due to the increase in alien vegetation. The right bank at site K3 was dominated by *Acacia mearnsii* trees and severe erosion caused bank collapse due to undercutting and scouring from previous flood events and trampling (Figure 5.3). This was also the cause of the narrow riparian zone at transect 3.1 on the right bank. The majority of the *Acacia mearnsii* (black wattle) trees were mature and young (1-2 years). The left bank was disturbed by human influence (trampling, vegetation removal), which resulted in homogenous vegetation with the top and middle plots dominated by *Pennisetum clandestinum* at transect 3.1 and *Pteridium aquilinum* and *Rubus pinnatus* at transects 3.2 and 3.3.

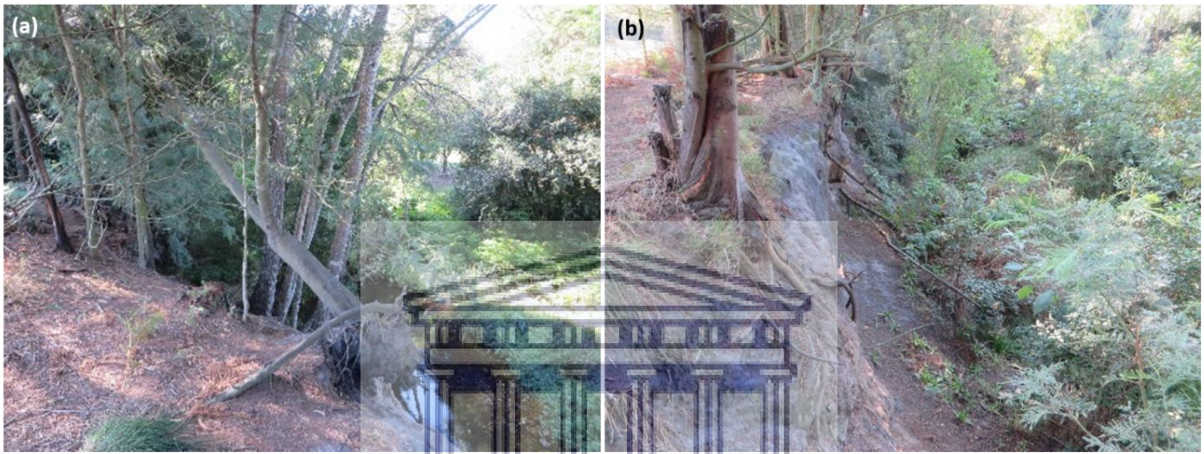

Figure 5.3 Erosion of the right bank at site K3 (a) top bank view looking downstream showing alien tree toppling and (b) exposed alien tree roots and bank scour looking downstream

Site K4 occurred in a protected area with approximately 40% indigenous vegetation occurring at transects assessed. Transects at 4.1 and 4.2 were mostly natural sites with no disturbance but clearing of vegetation occurred at 4.3 and as a result more invasion by alien vegetation. A higher percentage of alien vegetation was recorded at site K4 than K3 but at site K4 the aliens were shrub and herb dominated while at site K3 they were tree dominated. Overall, fewer species were recorded at site K4 due to more homogenous vegetation naturally occurring but also because fewer plots were available to sample. The right bank at this site was a steep hillslope and accessibility was made difficult. Site K2 and site K4 were the only sites where hydrophytes were present.

5.3.2 Vegetation comparison between riparian sites

The SIMPROF ($p < 0.001$) cluster analysis and MDS ordination showed a clear separation in plant species composition between sites K1 (reference) and K2 (semi-indigenous) (Group 1) and site K3 (degraded) (Group 2). The ANOSIM analysis revealed that there was a significant difference between sites with a global R^2 value of 1 (p = 0.01). The analysis showed three groups forming as shown in the cluster dendogram and MDS plot with a stress value of 0.01 (Figure 5.4). The black lines show the groups that were significantly different while the red lines indicate no significant difference between the samples. Group 1 formed at 43% similarity with two sub-groups forming with the reference indigenous forest site K1 (62%) (1A) and with the semi-indigenous site K2 (73%) (1B), Group 2 formed at 22% similarity with the degraded site K3 samples. Group 1 and 2 were further analysed for patterns within site transects looking at differences between the left and right banks, middle, upper and lower bank plots and differences between the vegetation occurring at these site transects.

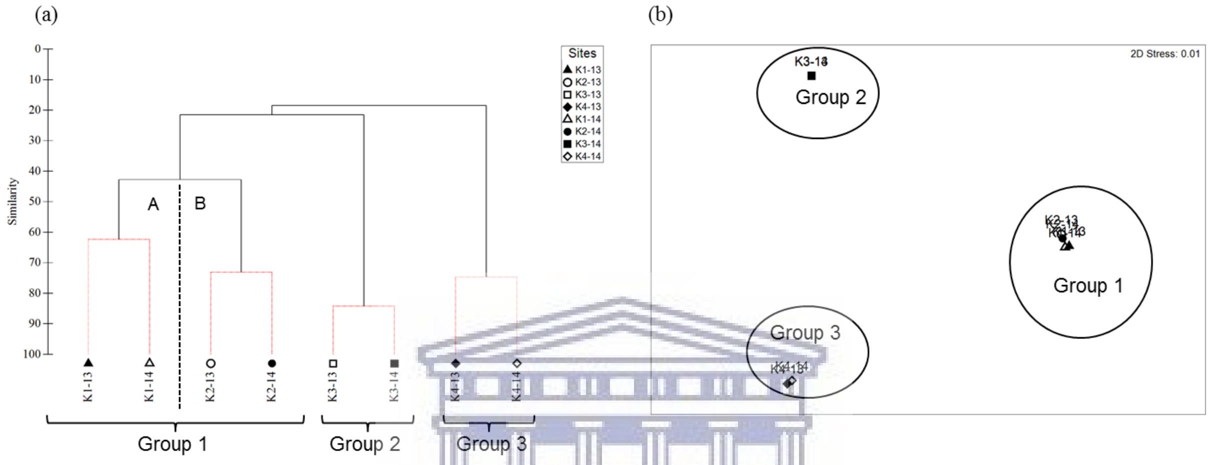

Figure 5.4 (a) Cluster dendrogram and (b) MDS ordination of Bray-Curtis similarity between vegetation species composition at all the sites. Sites codes: K1 = Indigenous forest; K2 = Semi-indigenous; K3 = Degraded; K4 = Cumulative site; Numbers indicate the year of sampling: 14 = 2014; 16 = 2016

Group 1

Two way ANOSIM analyses did not reveal significant plant species differences between the left and right banks (R^2 = 0.093). There was a moderate difference between plant species at transects between site K1 and site K2 (Overall, the global R^2 = 0.39). The pairwise tests however revealed R^2 values > 0.6 (values in bold in Table 5.2) between transects at site K1 and those occurring further downstream at site K2 (increased invasion with aliens).

No significant differences occurred between the left and right banks between the sampling years at both site K1 and site K2. Although both the reference site K1 banks were natural as well as the left bank at site K2, significant differences in the vegetation species driving the groupings occurred. At site K2 species were dominated by herbaceous and shrub species while at site K1 they were tree dominated. At site K2 the indigenous riparian zone was conserved on the left bank while the right bank was adjacent to dairy pastures and prone to invasion by alien vegetation. *Acacia mearnsii* was responsible for 15% contribution to the group and 49% of the cumulative percentages with a smaller percentage (1%) of *Acacia dealbata* present (Figure 5.5).

Table 5.2 Table: R^2 values for pair-wise tests of differences between transects at site K1 and K2 (Group 1A and 1B). Significant differences are at p ≤ 0.05 (bold)

Transect	R statistic	Significance level (p value)
1.1, 2.1	0.343	0.0001
1.1, 2.2	**0.565**	**0.0001**
1.1, 2.3	**0.612**	**0.0001**
1.2, 2.1	0.364	0.0001
1.2, 2.2	**0.653**	**0.0001**
1.2, 2.3	**0.597**	**0.0001**
1.3, 2.1	0.312	0.0002
1.3, 2.2	**0.561**	**0.0001**
1.3, 2.3	**0.698**	**0.0001**
2.1, 2.2	0.303	0.0003
2.1, 2.3	0.288	0.001
2.2, 2.3	0.136	0.015

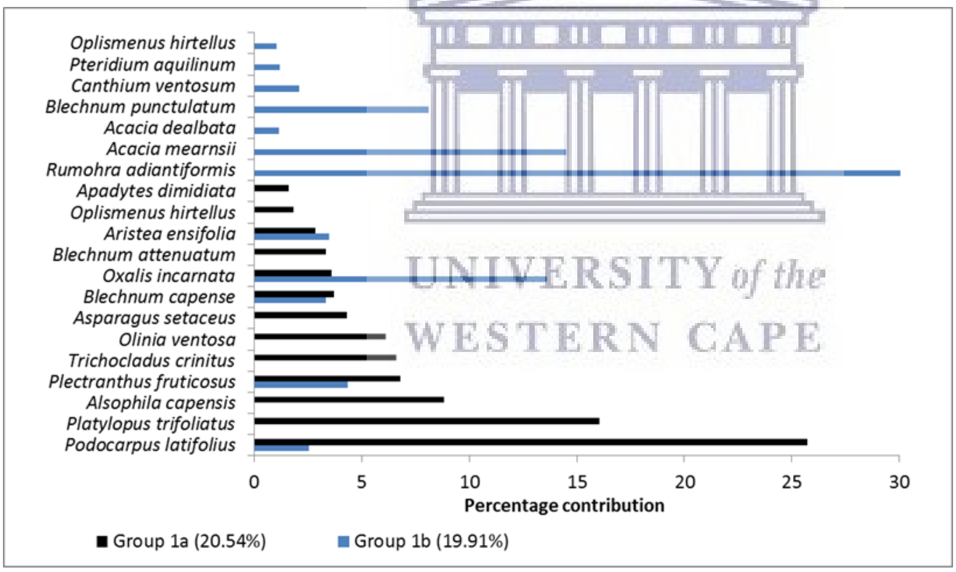

Figure 5.5 SIMPER results for vegetation assemblage composition that together contributed at least 70% to the overall similarities between sub-group 1A and B. The contribution percentages made to the average similarity by each species are shown and the average similarity for 1A and 1B are shown in brackets

When the plots at different positions on the banks were tested, ANOSIM results for Group 1 showed that overall no significant differences occurred ($R^2 = 0.11$; $p = 0.0002$). Pairwise tests, however, revealed that differences, although not very distinguishable, did occur between the upper and lower plots with some differences occurring between the years ($R^2 = 0.225$; $p = 0.0001$). The middle and lower plots were even less distinguishable ($R^2 = 0.14$). A one way nested PERMANOVA between left and right banks and bank position (nested within banks) also indicated a significant difference in plot positions (Table 5.3). The main plant species responsible for dissimilarity between the two banks and various plot positions within Group 1 were *Podocarpus latifolius*, *Alsophila capensis*, *Aristea ensifolia*, *Canthium ventosum*, *Rumohra adiantiformis*, *Plectranthus fruticosus*, *Acacia mearnsii*, *Blechnum punctulatum*, and *Juncus capanse* (Table 5.4). These species remained the drivers for dissimilarity between both sampling years. Mature tree species such as *Podocarpus latifolius* occurred on the upper, middle and lower plots at site K1 and K2 with the hydrophytic herb *Juncus capanse* occurring only on the lower wet banks. The tree species occurring at sites K1 and K2 were well established and most recorded trees were mature with heights between 7 to 25 m. Other herbaceous species and graminoids present occurred as undergrowth below trees.

Table 5.3 Results of one –way nested PERMANOVA of vegetation assemblages between river banks and position within the bank. Significant differences are at p ≤ 0.05

Groups	df	SS	MS	Pseudo-F	p
Position	2	20352	10176	1.3941	0.1971
Bank (position)	3	21898	7299.3	2.1437	0.0005
Residuals	66	224730	3405		
Total	71	266980			

Table 5.4 Differentiating species contributing to within-group dissimilarity for Group 1 (Site K1 and K2). L = Left bank, R = Right bank. Differentiating species are limited to the three highest dissimilarity coefficient/standard deviation (Diss/SD) values. Species are arranged highest to lowest values

Species	Sites occurred	Average Dissimilarity (%)	Bank	Bank plot position	Diss/SD
Podocarpus latifolius	1	92.95	L	Mid & Up	4.94
		82.60	L	Up & Mid	2.10
		87.01	LR	Mid	1.96
	1,2	94.11	LR	Up	5.49
		95.48	LR	Up & Low	2.34
		96.76	LR	Up	2.21
Alsophila capensis	1	90.02	RL	Low & Up	10.34
		85.58	LR	Mid & Low	5.69
		81.30	R	Low & Up	9.35
	1,2	89.36	L	Low & Up	14.15
		91.06	L	Low	13.19
		95.92	LR	Low & Mid	6.85
Aristea ensifolia	1	96.17	RL	Low & Mid	4.33
		77.77	R	Low & Mid	4.14
	1,2	88.41	RL	Low	4.16
		98.43	R	Low & Up	4.01

Species	Sites occurred	Average Dissimilarity (%)	Bank	Bank plot position	Diss/SD
		87.78	LR	Low & Mid	3.97
Canthium ventosum	1,2	82.09	RL	Mid & Up	1.53
Rumohra adiantiformis	1,2	88.41	RL	Low	9.49
		89.07	L	Low	8.94
		89.25	RL	Up & Low	8.51
Oplismenus hirtellus	1	73.62	LR	Up & Up	1.61
	2	88.33	RL	Up & Mid	2.04
	1,2	79.72	L	Up	1.55
Oxalis incarnata	1	85.89	R	Low & Up	8.17
		85.50	LR	Low & Up	7.98
		79.01	R	Up & Up	7.67
	2	87.09	R	Low & Up	4.67
		66.95	R	Mid & Up	4.37
		83.84	LR	Mid & Low	2.59
	1,2	96.85	R	Up & Low	7.31
		94.55	R	Up & Mid	6.86
		100	R	Up	4.84
		98.67	LR	Low & Up	4.11
Plectranthus fruticosus	1,2	96.97	L	Low & Mid	23.97
		93.71	R	Mid	5.08
		88.72	RL	Up & Mid	1.98
	2	82.44	RL	Up & Mid	12.16
		88.29	RL	Low & Mid	11.30
		79.00	LR	Mid & Up	10.93
Acacia mearnsii	1,2	95.92	LR	Low & Mid	11.17
		95.69	R	Low & Mid	11.04
		91.91	R	Mid & Low	10.94
	2	91.24	LR	Up & Mid	11.88
		88.33	RL	Mid & Up	10.41
		79.88	LR	Mid	5.01
Juncus capanse	2	74.51	LR	Low & Low	1.74
Blechnum punctulatum	2	80.75	L	Mid & Up	1.79
	1,2	85.78	RL	Mid & Low	1.62

Group 2

Group 2 only consisted of transects located at site K3. ANOSIM revealed a moderate difference between transects at the degraded site (R^2 = 0.364; p = 0.001). The pairwise test indicated a significant difference between transects 3.1 and 3.3 (R^2 = 0.682; p = 0.001). Dissimilarity between these two transects were driven by the species *Pteridium aquilinum* (Diss/SD = 3.09) with *Acacia mearnsii* and *Rubus pinnatus* contributing the next highest percentages to the average dissimilarity of 96 %. There were no significant differences between the vegetation occurring in the upper, middle and lower bank positions between both years (R^2 = 0.055; p = 0.09) and only a small distinguishable difference between the left and right banks between the two years (R^2 = 0.216). A one way nested PERMANOVA between left and right banks and bank position (nested within banks) also indicated no significant difference between bank position (p = 0.861) (Table 5.5). The alien tree species *Acacia mearnsii*, was driving the grouping at transect 3.1 (contributing 36%) and transect 3.2 (15%) followed by *Pennisetum clandestinum*, contributing 23.6% at transect 3.1. This was mostly due to the development on the right bank where the grass *Pennisetum clandestinum* (kikuyu), was planted for recreation purposes. *Pteridium aquilinum* was driving the grouping at transect 3.2 with a contribution of 43% (Figure 5.6). The least species diversity occurred at transect 3.3 as this site was dominated by *Pteridium aquilinum*. The distinguishing species contributing to the within group dissimilarity were driven by *Pteridium aquilinum, Rubus pinnatus, Carex aethopica, Commelina benghalensis, Oplismenus hirtellus, Oxalis incarnata, Acacia mearnsii, Juncus capanse* and *Carpha glomerata* (Table 5.6).

Table 5.5 Results of one –way nested PERMANOVA of vegetation assemblages between river banks and position within the bank. Significant differences are at p ≤ 0.05

Groups	df	SS	MS	Pseudo-F	p
Position	2	10520	5259.9	0.65421	0.861
Bank (position)	3	24155	8051.8	2.577	0.004
Residuals	30	93733	3124.4		
Total	35	128530			

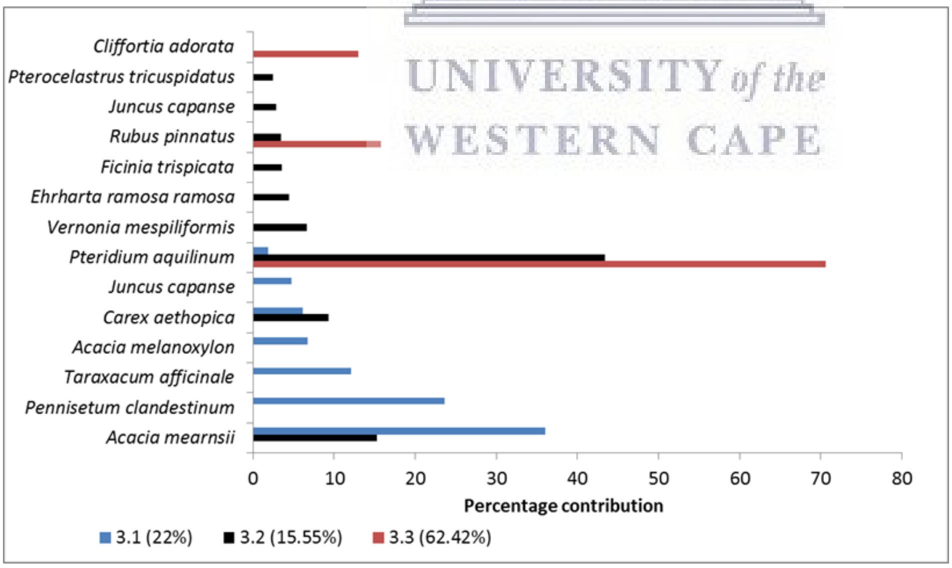

Figure 5.6 SIMPER results for vegetation assemblage composition that together contributed at least 70% to the overall similarities to Group 2 (Site K3, all transects). The contribution percentages made to the average similarity by each species are shown and the average similarity for each transect is shown in brackets

Table 5.6 Differentiating species contributing to within-group dissimilarity for Group 2 (Site K3). L = Left bank, R = Right bank. Differentiating species are limited to the three highest dissimilarity coefficient/standard deviation (Diss/SD) values. Species are arranged highest to lowest values.

Species	Average Dissimilarity (%)	Bank	Position	Diss/SD
Pteridium aquilinum	79.88	L	Low & Mid	1.45
	80.51	L	Low & Up	1.44
	83.42	L	Low & Mid	1.42
Rubus pinnatus	83.67	RL	Up	1.33
	88.29	RL	Mid & Up	1.30
	88.07	RL	Mid & Up	1.26
Carex aethopica	84.87	L	Low & Mid	1.33
	83.96	RL	Up & Low	1.32
	86.43	LR	Low	1.20
Commelina benghalensis	88.33	RL	Up & Mid	2.04
Oplismenus hirtellus	73.62	LR	Up & Up	1.61
	66.40	L	Low	1.30
	79.72	L	Up	1.55
Oxalis incarnata	88.63	RL	Mid	1.31
	77.36	R	Up & Mid	1.30
	71.51	R	Mid	1.29
Acacia mearnsii	83.40	RL	Up & Low	1.59
	84.24	RL	Up & Low	1.58
	83.96	RL	Up & Low	1.36
Juncus capanse	85.81	R	Mid & Low	1.10
	88.51	RL	Low & Mid	1.08
	85.80	RL	Low	1.04
Carpha glomerata	84.08	R	Mid & low	1.11
	82.18	R	Mid & Low	1.10
	87.72	LR	Mid & Low	1.09

Group 3

ANOSIM revealed a significant difference between the three transects at this site between both sampling years ($R^2 = 0.405$). The pairwise tests showed a difference between bank vegetation and the instream vegetation as well as between transects 4.1 and 4.3 (Table 5.7). Site K4 was the only site with extensive instream vegetation especially downstream of the causeway as a result of reduced flows.

Table 5.7 R^2 values for pair-wise tests of differences between transects at site 4 (Group 3). Significant differences are at p ≤ 0.05 (bold)

Transect	R statistic	Significance level (p value)
4.1, 4.1 instream	**0.652**	**0.015**
4.1, 4.2	0.34	0.0009
4.1, 4.2 instream	**0.652**	**0.015**
4.1, 4.3	**0.505**	**0.0002**
4.1, 4.3 instream	**0.528**	**0.015**
4.1 instream, 4.2	0.478	0.015
4.1 instream, 4.3	**0.793**	**0.022**
4.2, 4.2 instream	0.478	0.015
4.2, 4.3	0.133	0.065
4.2, 4.3 instream	0.478	0.015
4.2 instream, 4.3	**0.793**	**0.022**
4.3, 4.3 instream	**0.793**	**0.022**

Although this site occurred in a protected the area the vegetation was disturbed and sometimes cleared by SANParks management, especially closer to transects 4.2 and 4.3. The species driving the average similarity at each of these transects was different as indicated in Figure 5.7 and Table 5.11. At transect 4.1 the tree species *Rhamnus prinoides* contributed to the grouping with 25%, while at transect 4.2 it was *Senecio quiquelobus* that had the highest contribution at 32%. At transect 4.3 *Urtica urens* dominated the contribution at 34%. The upper and middle bank positions showed no significant differences but some differences did occur between the upper and lower bank positions (R^2 = 0.266; p = 0.011) of both the right and left banks, and instream vegetation differed from the bank vegetation (R^2 values between 0.5 and 0.767) driven by *Potamogetan*. A one way nested PERMANOVA between left and right banks and bank position (nested within banks) also indicated a significant difference (p = 0.045) (Table 5.8). The vegetation species differentiating the bank vegetation included *Urtica urens*, *Wachendorfia thyrsiiflora*, *Potamogetan*, *Rhamnus prinoides*, *Rhoicissus tomentosa*, *Juncus capanse*, *Persicaria decipiens*, *Carex aethopica*, and *Oplismenus hirtellus* (Table 5.9). The right bank at this site was a steep hillslope, which made sampling of the upper and middle bank plots inaccessible.

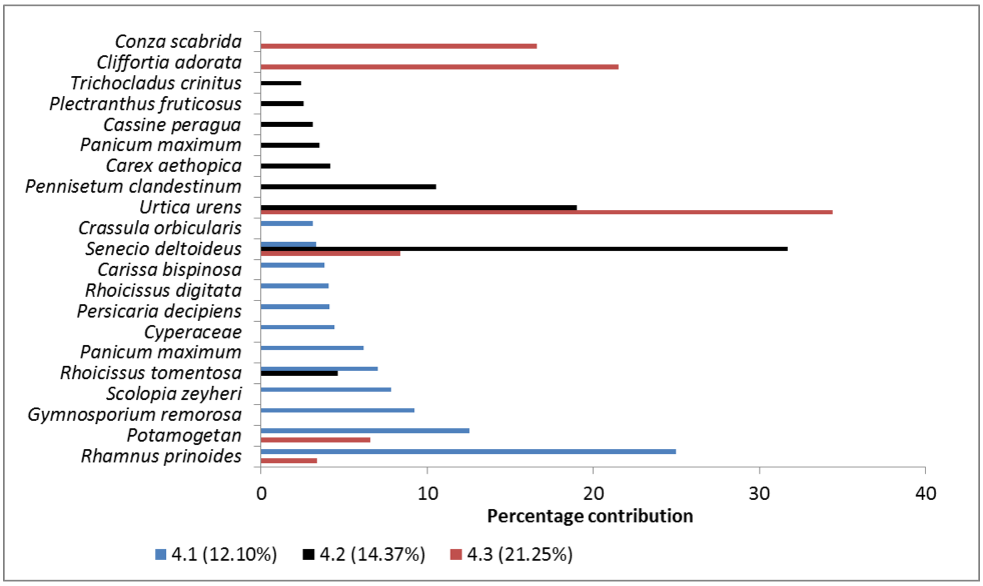

Figure 5.7 SIMPER results of typical species contributing to Group 3 (Site 4, all transects)

Table 5.8 Results of one –way nested PERMANOVA of vegetation assemblages between river banks and position within the bank. Significant differences are at p ≤ 0.05

Groups	df	SS	MS	Pseudo-F	p
Position	3	43999	14666	2.0762	0.1098
Bank (position)	2	13506	6753.1	2.3067	0.0045
Residuals	28	81975	2927.7		
Total	33	13920			

Table 5.9 Typical or distinguishing species contributing to within-group dissimilarity for Group 3 (Site K4). Differentiating species are limited to the three highest dissimilarity coefficient/standard deviation (Diss/SD) values

Species	Average Dissimilarity (%)	Bank	Position	Diss/SD
Urtica urens	100	RL	Low & Mid	1.32
	93.86	LR	Mid & Low	1.30
	77.48	L	Mid & Up	1.29
Wachendorfia thyrsiiflora	86.16	LR	Low	1.33
	85.11	L	Mid & Low	1.31
	94.57	LR	Low & Mid	1.29
Potamogetan	100	L	Low & Instream	12.35
	100	R	Mid & Instream	9.55
	100		Instream	9.27
Rhamnus prinoides	94.26	RL	Low	1.32
	97.31	LR	Up & Low	1.27
	96.75	RL	Low & Up	1.17
Rhoicissus tomentosa	97.47	RL	Mid & Up	1.28
	88.09	LR	Up & Mid	1.32

Species	Average Dissimilarity (%)	Bank	Position	Diss/SD
Juncus capanse	80.77	LR	Low	1.41
	68.30	L	Low	1.28
	86.61	L	Up &Low	1.27
Persicaria decipiens	12.43		Instream	0.58
Carex aethopica	82.49	R	Low	1.33
	98.94	LR	Up & Low	1.31
	92.88	R	Mid & Low	1.29
Wachendorfia thyrsiiflora	83.57	LR	Low	1.31
Oplismenus hirtellus	95.57	LR	Low & Mid	11.23
	73.15	R	Mid	9.17
	95.46	LR	Low & Mid	8.44

5.3.3 Environmental variables responsible for variance in vegetation communities

The DISTLM results showed that the greatest amount of variation in species data was explained by TP and phosphate at 25.9% and 23% respectively (Marginal tests) (Appendix 5.3). The elevation and pH both explained 21.2 % of the variation. This was followed by sand (20%) and gravels (19.4%) grain sizes, organic carbon (18.2%) and TN (17.4%). Less of the variation was explained by silt (12.5%) and the distance from the channel thalweg (10.5%).

The co-variates that contributed to the overall model variation in species explained by DISTLM were shown by the sequential tests (Appendix 5.3). Total phosphorus (TP) (variation explained: 26%) was the most significant contributor to the variation followed by pH (18%), gravels (15.7%), distance from channel thalweg (10.2%), silt (8.8%), TN (6.3%) and organic carbon (4.9%). Together these co-variates contributed to 90% of the total variation. The dbRDA ordination plot (Figure 5.8) showed site K4 associating with pH and PO_4^{3-} and almost completely separating from the upstream sites. Mostly the lower bank plots at site K4 were associated with TP and PO_4^{3-}. Plots from the upper, middle and lower banks at sites K1, K2 and K3 were sand dominated and therefore were associated with the sand fraction while a few from site K1 (lower bank) and K3 (upper bank) associated with the silt fraction, TN, carbon and distance to the channel thalweg (Figure 5.8). The transects closely associated to elevation were 2.2, 2.3, 3.1 and 3.2, which were also the transects where the heights of the right banks were increased due to erosion (bank scour, undercutting and bank collapse). Transects at 3.1 and 3.2 were most associated with TN (Figure 5.8).

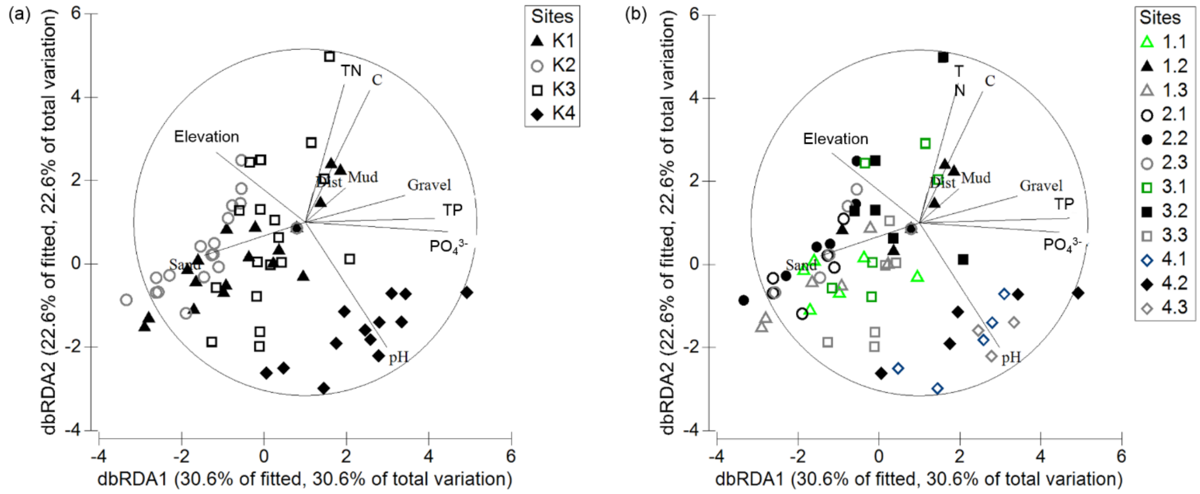

Figure 5.8 dbRDA ordination plot showing the relationship between vegetation composition and environmental variables with vectors showing the Spearman correlation between environmental variables at; (a) all sites and (b) all transects

5.3.4 Rainfall

The rainfall was monitored from mid - 2014 to mid - 2017 with the cumulative rainfall recorded (total rainfall that occurred throughout the monitoring period) in the pastures being 1893.6 mm, 928 mm in the semi-indigenous riparian zone (K2) and degraded riparian zone (K3) and 799.6 mm in the indigenous forest (K1) (Figure 5.9). High variability occurred between rainfall figures recorded from the three gauging stations (riparian zone CV: 97% and pasture CV: 57%). As expected lower rainfall on average with greater variability was recorded from the rain gauges located beneath the tree canopies in the forest, semi-indigenous and degraded riparian areas when compared to the rainfall occurring in the open pasture area. This was also true for the rainfall intensity, which in turn resulted in high variability in the rainfall-runoff response from the runoff plots. The wettest year occurred during 2015 with 2 of the 5 events of rainfall > 100 mm (wet season) and 8 events > 50 mm (wet and dry season) occurring in the pastures. The rain lost to interception below tree canopies in the indigenous forest riparian zone was calculated at 65% and at 58% in the semi-indigenous and degraded riparian zones. The two-way ANOVA (between sampling seasons and land cover) showed significant differences in rainfall (MS= 1.829, df = 5, F=3.337, $p < 0.010$). The Tukey post-hoc analysis of differences showed no significant difference between the wet and dry season but there was a significant difference between the rainfall occurring in the pasture land cover and the riparian zones.

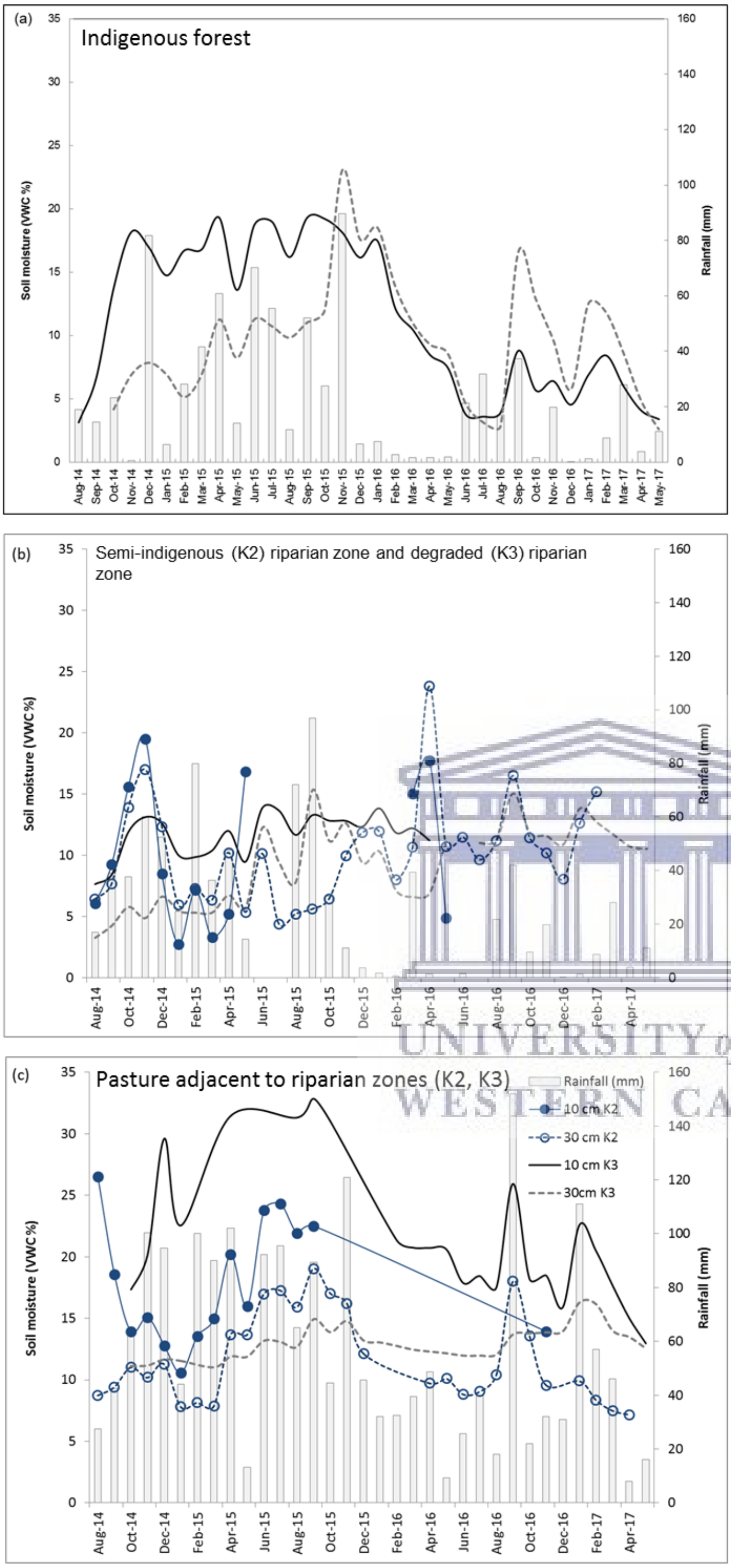

Figure 5.9 Monthly rainfall and average volumetric soil moisture content at the (a) indigenous forest (solid line = 0.1 m and broken line = 0.3 m depth), (b) semi-indigenous (K2) riparian zone and degraded riparian zone (K3) and (c) pasture adjacent to riparian zones (K2, K3). For (b) and (c) closed grey circle = K2 at 0.1 m and open grey circles = K2 at 0.3 m; solid line = K3 at 0.1 m and broken line = K3 at 0.3 m

5.3.5 Sediment characteristics

The volumetric soil water content was responsive to rainfall events (Figure 5.9) where rainfall showed a significantly positive correlation to soil water content at 0.1 m and 0.3 m depths at all runoff plots. According to the Tukey post-hoc multiple comparisons, the soil moisture content at the surface in the pastures differed significantly from the riparian areas, with the highest percentages occurring in the pasture when compared to the riparian zone sites. Site K3 (degraded riparian zone) displayed the lowest soil moisture content at both depths compared to all sites, which may be related to the increase water use by the alien trees (Le Maitre et al., 2016), or the difference in soil texture where pastures displayed lower sand content compared to riparian zone sites (Chapter 2, section 2.3, Table 2.4).

When considering the soil chemistry, the percentage of TN concentrations observed in the near surface and at the 0.3 m depth at site K1 was the lowest compared to the other sites (Figure 5.10b). A gradual increase was observed at site K2 riparian zones at both depths compared to site K1. The highest concentrations occurred at the alien invaded site K3 with percentage concentrations increasing more than two-fold at both depths compared to site K1 and K2. Similar results were observed with the percentage of organic carbon and organic matter content in the degraded riparian zone (K3) (Figure 5.10c and d). The pastures showed higher concentrations of TP in the surface sediment when compared to the riparian zone areas (Figure 5.10a). This may be due to past fertilisation but it could also be due to the process of mineralisation. According to Makarova et al. (2004) and Dodd and Sharpley (2015), in most soils, the TP can comprise up to 90% in organic compounds, which are associated with increased soil organic matter and carbon, made available to plants during mineralisation. During this process, phosphorus is made available to the soil solution explaining the higher concentrations recorded. The increases in TP at the sites, especially the pastures, are associated with increases in organic matter and carbon. TP concentrations in the riparian zones followed the same trend as TN concentrations with the surface sediment having higher concentrations than at the 0.3 m depth, with the exception of site K3.

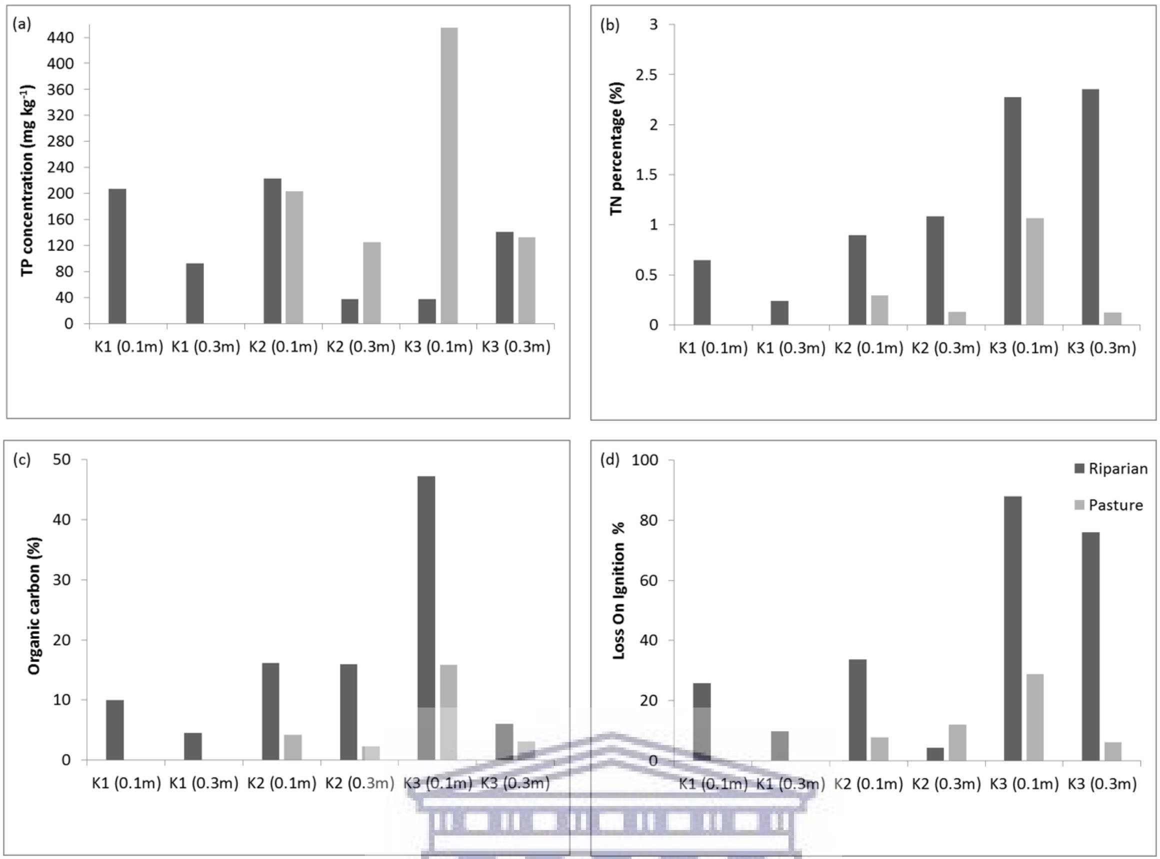

Figure 5.10 Average chemical sediment quality for the different land covers at the runoff plots at the 0 - 0.1 m and at the 0.3 m depth profiles for (a) TP, (b) TN, (c) organic carbon and (d) organic matter by loss on ignition %

5.3.6 Runoff characteristics and water quality

The runoff from the plots from the different land covers correlate to the rainfall distribution. The lowest runoff volumes were recorded from plots at site K1. Statistically no significant differences in runoff volume occurred during the sampling seasons with low surface runoff coefficients (< 1% – 3%), which increased during October 2016 and May 2017 (7% - 55%). (Figure 5.11a). During both these sampling events higher preceding rainfall occurred and soil moisture in the deeper profiles exceeded that near the profile surfaces (Figure 5.9a). In the semi-indigenous riparian zone (K2) runoff consistently occurred from the plots located in the predominantly indigenous vegetation (indicated by open circles in Figure 5.12b) (Appendix 5.3). Runoff was less consistent, occurring only during wet seasons, from the plot located beneath predominantly alien trees at site K2 (indicated by the solid black circles in Figure 5.12b) (Appendix 5.4), with the lowest runoff volumes on average occurring from this plot, indicated by the low runoff coefficients. Runoff events occurred at site K2 riparian zone following increased rainfall events with associated increases in soil moisture conditions (Figure 5.9b). Runoff was more frequent at site K3 with higher (21% - 34%) runoff coefficients than at site K2 (Figure 5.11b). The largest volume of runoff throughout the sampling period occurred from the combined total volume from the pasture plots (runoff from the six replicate plots in the pasture) associated with higher soil water content (Figure 5.9c). Fewer runoff events occurred from the pastures adjacent to site K3 but surface runoff coefficients were generally higher than those from pastures adjacent to site K2.

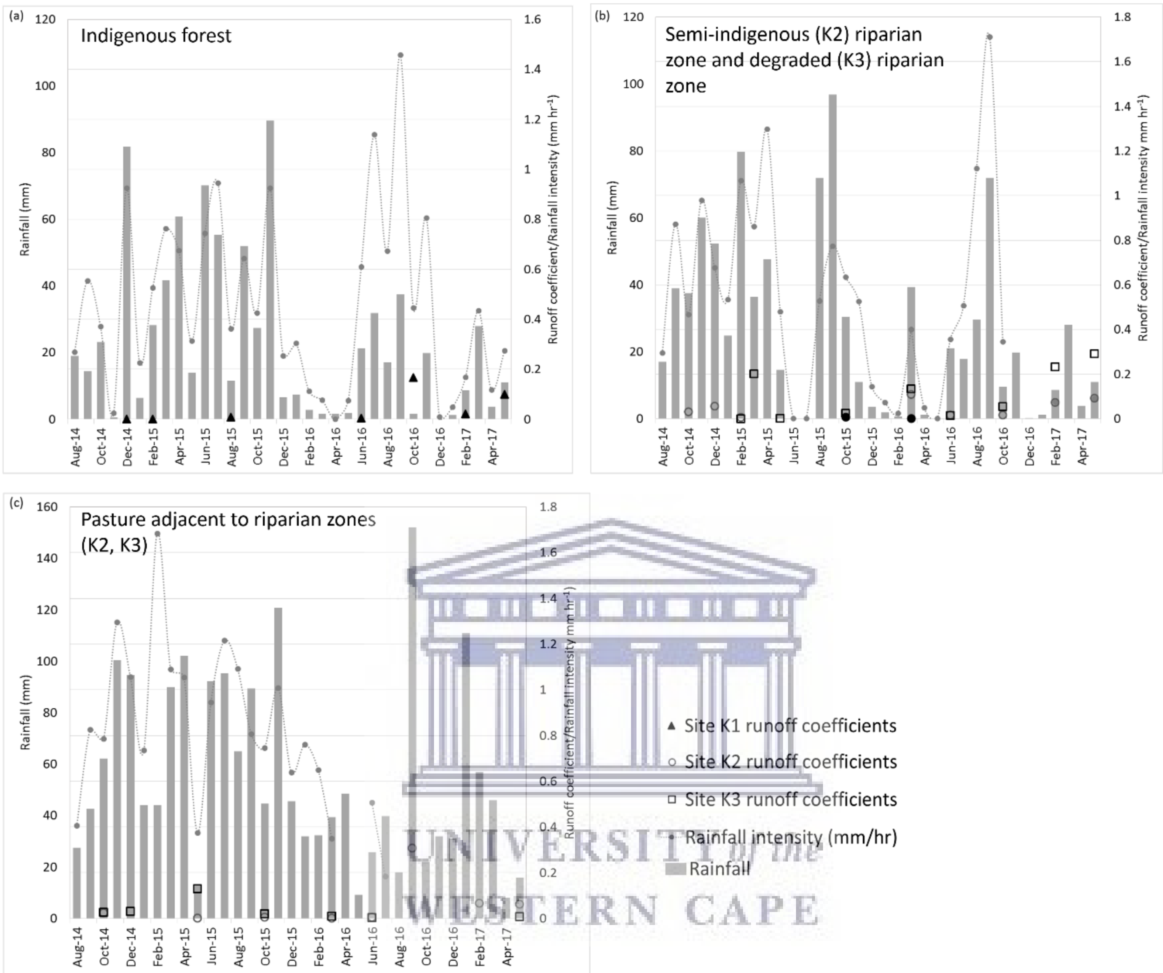

Figure 5.11 Monthly average rainfall (mm) and intensity with surface runoff coefficients per runoff event for runoff plots at (a) indigenous riparian zone (K1) = solid triangle, (b) semi-indigenous (K2) riparian zone (open circle) and degraded riparian zone (K3) and (c) pasture adjacent to riparian zones (K2, K3)

The average Na^+, EC, pH and DOC were at similar concentrations in the indigenous forest runoff and the semi-indigenous riparian zone runoff. There was a close correlation between the suspended solids (SS) and turbidity recorded from runoff water at all plots. The highest concentrations for suspended solids were consistently recorded from the semi-indigenous and degraded riparian zone plots when compared to the reference site and pasture plots (Table 5.10). The ANOVA showed no significant differences ($p > 0.05$) when applying Tukey's test between sampling season between runoff water quality from different land covers and river water samples between sites, regarding the alkalinity, EC, Na^+ and NO_x-N. The Tukey's comparison test showed a significant difference in pH between runoff water from the pasture and the semi-indigenous riparian zone plots and between runoff water from plots located in the pastures and river water samples.

At the indigenous forest plots, the NH_4^+-N was relatively low but the highest TN concentrations were recorded compared to the other riparian zone plots (Figure 5.12a). Despite the increased TN concentrations in the runoff water, much lower concentrations were recorded from the river water at site K1 (Table 5.11). In the semi-indigenous riparian runoff plots, on average, higher concentrations of NH_4^+-N (12.3 mg L^{-1}) and NO_x-N (14.4 mg L^{-1}) occurred during the wet season compared to the dry season (NH_4^+-N: 9.5 mg L^{-1}, NO_x-N: 9.3 mg L^{-1}). The highest NO_x-N concentrations (53 mg $^{-1}$) and NH_4^+-N concentrations (30 mg L^{-1}) were recorded from the runoff plot located beneath predominantly black wattles in the wet season at site K2 riparian zone (Figure 5.12c). The TN and TP concentrations were lower during the wet season than the dry season at the semi-indigenous plots, which was attributed to the dilution effect of higher rainfall.

The overall TN concentrations were much higher in the K3 riparian zone than in the K2 riparian zones. The pastures adjacent to site K2 showed much higher concentrations of NO_x-N and TP in runoff compared to the riparian zone. The TP concentrations were higher in the K3 riparian zone (Figure 5.12e) than in the K1 and K2 riparian zones (Figure 5.12a and c).The two-way ANOVA showed a significant difference in TP concentrations between land covers (MS = 40.069, df = 8, F = 3.982 $p < 0.001$). A significant difference was also observed using Tukey's post-hoc analysis of differences test between runoff water in the pasture adjacent to site K3 riparian zone and the river water samples at site K3, K2 and K1. The ANOVA showed a significant difference in TN concentrations between runoff plot water and river water samples (MS = 619.7, df = 8, F = 4.64 $p < 0.0002$). The Tukey's post-hoc analysis revealed differences between the indigenous forest runoff and the river water at site K1, K2 and K3 as well as between runoff water from the degraded riparian zone. No significant difference occurred between sampling season or with any of the other water quality variables between the runoff water and the river water quality.

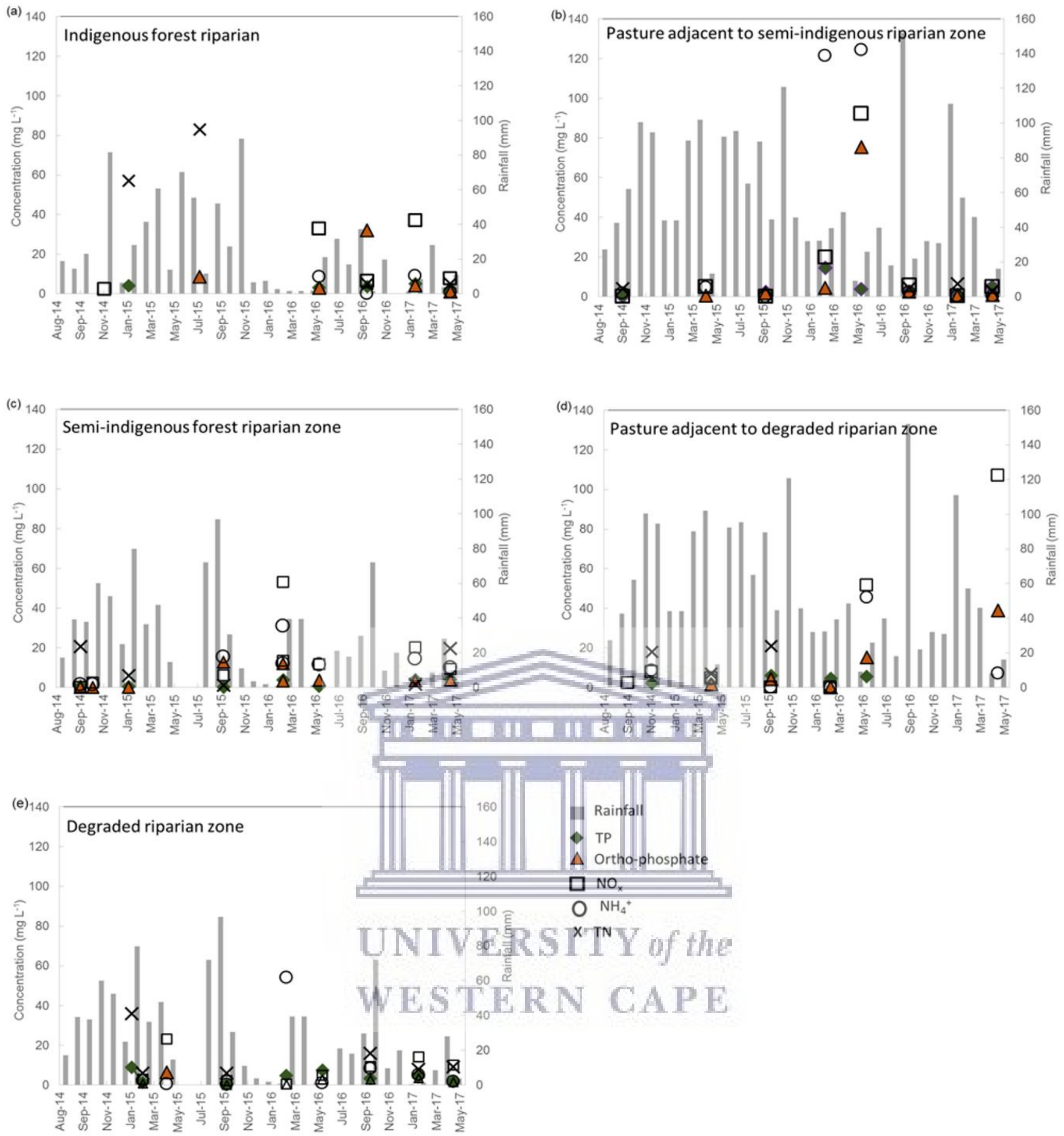

Figure 5.12 Rainfall and associated sampling events from plots for riparian zone sites and adjacent pastures showing nutrient concentrations (mg L^{-1})

Table 5.10 Summary for chemical parameters measured at runoff plots (2014-2017) (n = 68)

Variables	Unit	Indigenous forest (Site K1)			Pastures (site K2)			Semi-indigenous riparian zone (Site K2)			Pastures (site K3)			Degraded riparian zone (Site K3)		
		Mean/stddev	Min	Max	Mean/stddev	Min	Max	Mean/stddev	Min	Max	Mean/stddev	Min	Max	Mean/stddev	Min	Max
Na^+	mg L^{-1}	15.5 ± 5.4	8.3	21	43.3±72.7	3.9	230	16.1±16.5	2.3	59	43.8±57.9	11	160	35.7±19.4	9.7	76
Alkalinity	mg L^{-1}	9.2±12.3	0.3	27	50±72.6	0.5	200	3.7±3.6	0.5	14	22.2±33.2	0.4	80	52±128.6	0.3	41
NO_x-N	mg L^{-1}	18.7±16.5	2.4	44	31.8±78.6	0.1	321	12.6±16.1	0.1	53	25±38.3	0.1	107	8.3±7.9	0.1	23
NH_4^+	mg L^{-1}	4.7±5.3	0.1	13	46.2±87	0.1	280	11.3±12.7	0.6	38	13.9±23.7	0.1	70	7.4±15.1	0.1	54
PO_4^{3-}	mg L^{-1}	7.7±11	0.9	32	30.3±98	0.3	383	6.2±7.5	0.2	25	10.8±15.9	0.1	39	2.4±2.1	0.1	6.4
TN	mg L^{-1}	31±36.8	4	83	4.4±2	2	7	14.8±12.8	2	39	15.3±7.4	7	21	13.9±10.7	6	36
TP	mg L^{-1}	3.4±2	1	7	5±6.2	0.3	23	2.7±3.7	0.1	10	5±3.4	1.8	11	3.9±2.7	0.5	8.8
EC	mS m^{-1}	30.3±19.4	8	54	96.3±181.9	7	520	30.8±30.7	3	98	30±24.1	8	76	53.1±29.5	11	110
pH		5.5±1.6	4	7.4	7±1.7	4.4	9.4	5.1±0.9	3.7	6.5	5.8±1.5	4.3	7.4	5.7±1.6	4.1	8.4
DOC	mg L^{-1}	22.3±7.7	15	35	39.1±39.4	5.1	118	28.7±27.2	5.2	93	80.3±63.7	14	200	43.6±16.6	18	77
Turbidity	NTU	19±14.7	7.5	44	37.9±45.4	1.6	136	61.8±83	5.2	225	98.3±61.9	27	138	39.9±35.6	6.4	113
SS	mg L^{-1}	57±45.3	25	89	110.7±125.1	5	377	250.6±352.1	34	1044	99±116	31	233	201.7±255.5	26	790
Volume	L	0.5±0.7	0.02	1.8	1.8±2.8	0.01	8	1.3±1.9	0.05	8.2	0.9±1	0.05	3	1.3±2.1	0.02	7.3

Table 5.11 Summary for chemical parameters measured at the river sampling locations (2014-2016) (n = 40)

Variables	Unit	Site K1			Site K2			Site K3		
		Mean/Stddev	**Min**	**Max**	**Mean/Stddev**	**Min**	**Max**	**Mean/Stddev**	**Min**	**Max**
Na^+	mg L^{-1}	17.5 ± 1.5	16	20	21.6±1.9	19	25	37.9±5.9	30	51
Alkalinity	mg L^{-1}	2.3±0.6	1.6	3.8	3.5±0.9	2.4	4.8	17.4±23.6	4.5	66
NO_x-N	mg L^{-1}	0.1±0.03	0.1	0.2	0.1±0.07	0.1	0.3	0.1±0.07	0.1	0.3
TN	mg L^{-1}	0.5±0.1	0.4	0.6	0.6±0.3	0.3	1	0.7±0.2	0.5	1
TP	mg L^{-1}	0.1±0.01	0.04	0.1	0.05±0.01	0.03	0.1	0.1±0.1	0.05	0.1
EC	mS m^{-1}	12.6±0.8	11	14	15.3±1.5	14	18	26.9±5.4	20	37
pH		5±0.1	4.7	5.1	5.2±0.2	4.8	5.4	6±0.6	5.3	7
Si	mg L^{-1}	2.5±0.3	2.1	3	2.7±0.4	2.3	3.4	3.4±0.42	3	4.2
Turbidity	NTU	1.9±0.8	0.8	1.9	1.9±0.8	0.8	1.9	2.8±1.4	1.4	2.8

The nutrient loads from the runoff plots were calculated by multiplying the concentrations by the volume of water yielded by the 1 m^2 runoff plots. Although the TN concentrations in the indigenous forest were the highest of all runoff sites, the overall TN loads were the lowest (Figure 5.12a and Table 5.12). This could be attributed to the dilution capacity associated with the higher rainfall during the wet season compared to the dry season. The NH_4^+-N loads were much higher in the pastures and degraded riparian zone sites than those recorded in the reference site (Table 5.12), which could be attributed to the agricultural activities and the alien vegetation impact. The TN loads decreased from the pastures where higher runoff volumes occurred to the semi-indigenous riparian zone where the runoff volumes decreased in comparison (Table 5.12). Higher TN loads and NO_x-N loads were observed in the degraded riparian zone with the encroachment by *Acacia mearnsii* trees when compared to the adjacent pastures. The same trend was followed with TP loads. The annual nutrient loading was higher in 2016 than in the wetter 2015 in the indigenous (K1), semi-indigenous (K2) riparian zones and in the adjacent pastures (Table 5.12) as a result of higher surface runoff volumes (coefficients) during 2016 (Table 5.12). This could be attributed to the dilution capacity associated with the increased rainfall during 2015. At the adjacent pastures to the degraded riparian zone, annual nutrient loading increased with the increased rainfall during 2015 associated with increased surface runoff volumes and coefficients (Table 5.12, Figure 5.11c). The nutrient loading is therefore dependent on land cover, land use, rainfall amount and intensity and the associated surface runoff.

Table 5.12 Annual rainfall (mm) and associated nutrient loads (g m^{-2}) from runoff plots

Years sampled	No. months	Annual Runoff (L)	Annual Rainfall (mm)	Total loads (g m^{-2}) PO_4^{3-}	TP	TN	NO_x-N	NH_4^+-N
		Site K1						
2014	5	0.04	139	0	0	0	0	0
2015	12	0.12	463.8	0.9	0.08	9.4	0	0
2016	12	1.9	144.2	57.8	6.9	9	15	1
2017	5	2.6	52.6	1.9	2.2	5	15.6	2.6
Total	**34**	**4.7**	**799.6**	**60.6**	**9.2**	**23.4**	**30.6**	**3.6**
		Pastures (Site K2)						
2014	5	1.5	327.2	0	0.4	6	0.5	2.7
2015	12	0.5	846.2	0.4	0.7	0	0.2	0.2
2016	12	21.1	482.2	48.4	19.7	27.2	69.8	29.9
2017	5	8.2	238	3.6	7.8	39.1	12	2.03
Total	**34**	**31.3**	**1893.6**	**52.4**	**28.6**	**72.3**	**82.5**	**34.8**
		Semi-indigenous riparian zone (Site K2)						
2014	5	8.1	206	0	0.7	22.1	5.8	5.7
2015	12	1.5	417	1.2	0.2	3.4	0.8	1.6
2016	12	9.3	252.4	9.3	1.7	0	10.4	9.1
2017	5	4.9	52.6	3.7	4.2	20.1	14.2	11.6
Total	**34**	**23.8**	**928**	**14.2**	**6.8**	**45.6**	**31.2**	**28**
		Pastures (Site K3)						
2014	5	4.7	327.2	0	6.3	54	27.4	26.1
2015	12	2.6	846.2	6.3	8.5	30.8	8.1	7.4
2016	12	1	482.2	1.5	2.5	0	4.3	4.1
2017	5	1.3	238	3.9	0	0	10.7	0.7
Total	**34**	**9.6**	**1893.6**	**11.7**	**12.3**	**84.8**	**50.5**	**38.3**

Years sampled	No. months	Annual Runoff (L)	Annual Rainfall (mm)	Total loads (g m^{-2})				
				PO_4^{3-}	TP	TN	NO_x-N	NH_4^+-N
Degraded riparian zone (Site K3)								
2014	5	0	206	0	0	0	0	0
2015	12	0.8	417	5.2	10.1	49.4	23.2	16.1
2016	12	2.8	252.4	2.5	5.2	7.7	8.4	25.2
2017	5	14.5	52.6	9.1	10.6	43.6	48.1	14.3
Total	**34**	**18.1**	**928**	**16.6**	**22.9**	**100.7**	**79.7**	**55.6**

The efficiency of the different land covers in reducing nutrient concentrations were obtained by relating all measured data (collected for the period of study) for nutrient concentrations in runoff at the start point of observation (pastures), to each of the riparian zones, to the end point of observation (river). Regression analysis showed a decreasing trend for all nutrient concentrations at both sites K2 and K3 (Figure 5.13a and b) from the start point of observation (pasture) to the end point of observation (river). The NO_x-N concentrations recorded in runoff from the semi-indigenous riparian zone were 60% lower (on average throughout the study period) than those recorded in the runoff from the adjacent pasture plots. The TP concentrations were 50% lower (on average throughout the study period) in runoff from the semi-indigenous riparian zone compared to the adjacent pasture.

Even though river water quality could not directly be compared to the quality from runoff plots as sampling occurred on a quarterly basis, inferences could be made regarding the buffering capacity of the vegetation. The bioindicators sampled from the river gave a good indication of the water quality as shown in Chapter 4 with elevated levels of nutrients indicated by increases in the diatom species, *N. radiosa*. Results from Chapter 4 also indicated that the bioindicators used in combination with Concentrations of TN and TP recorded in river water were more than 90% lower than those recorded in the semi-indigenous riparian zone at site K2 (supplementary Table S1 and S2). The NH_4^+-N was excluded as this was not measured consistently in the river water samples. Concentrations of NO_x-N and TP were 72% and 27% lower, respectively, in the riparian zone than in the adjacent pasture at site K3. The TN concentration showed only a 10% reduction from the pasture runoff to the runoff in the degraded riparian zone. Furthermore, overall nutrient concentrations at site K3 were 95-98% lower in river water than in the adjacent degraded riparian zone (supplementary Table S1 and S2). The calculated loads between the pastures and the semi-indigenous riparian zone for NO_x-N was reduced by 78% and the TN loads were reduced by 28%. The TP loads between the pastures and the semi-indigenous riparian zone was reduced by 77%. At site K3 all total nutrient loads were higher in the degraded riparian zone than those recorded in the adjacent pasture (Table 2). The TN loads increased by 13%, NO_x-N loads increased by 26% while the TP loads increased by 50%.

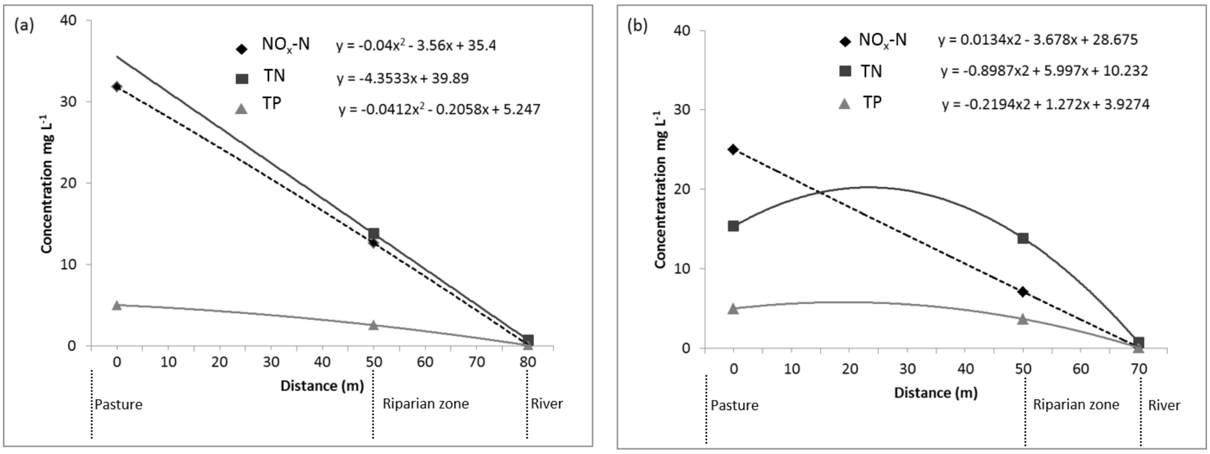

Figure 5.13 The relation between the nutrient concentrations and the distance between sampling plots as measured in (a) pastures adjacent to the semi-indigenous riparian zone (site K2) and (b) pastures adjacent to the degraded riparian zone (site K3)

5.3.8 Links between runoff water quality and land covers

Principle Components Analysis (PCA) was used to explain the relations among water quality variables and the association with the land covers between seasons. During the wet season runoff occurred in the plot located beneath black wattle in the semi-indigenous riparian zone, which showed NO_x-N and NH_4^+-N associating with this plot (Figure 5.14a). The PO_4^{3-} -P and EC also associated more with the semi-indigenous riparian plots during the wet season. The TN associated with the degraded riparian zone during the wet season and included Na^+ and DOC. Runoff from the pastures showed a strong association with suspended solids, TP, turbidity, alkalinity and pH. During the dry season there was a clear association of the nutrients, NO_x-N, NH_4^+-N, TN, and PO_4^{3-}-P and TP, with the pastures (Figure 5.14b). The TN was associated with the pastures and the degraded riparian zone plots as well as Na^+, pH, alkalinity, suspended solids, turbidity and DOC.

When considering the PCA with runoff quality for the pastures adjacent to the riparian zones at the semi-indigenous site plots (K2) and the degraded riparian zone (K3), similar results were obtained (Figure 5.14c-d). The pastures were associated with turbidity, suspended solids, pH and alkalinity as well as TP, PO_4^{3-}-P and NH_4^+-N. The semi-indigenous riparian zone at site K2 was associated with NO_x-N, TN and EC. The plot in the semi-indigenous riparian zone dominated by black wattle showed a strong association to Na^+ and DOC and runoff volume was also associated with the semi-indigenous plots. At site K3 the riparian zone plot beneath black wattle was associated with NO_x-N, NH_4^+-N, EC, DOC and Na^+. The adjacent pastures were associated with PO_4^{3-}-P, TP and TN, alkalinity, pH, suspended solids and turbidity. The PCA analysis of the river water samples shows site K3 associated with silica, alkalinity, hardness, pH, EC, calcium, magnesium and sodium. The chemical oxygen demand, TN, NO_x-N and TP were associated with site K2.

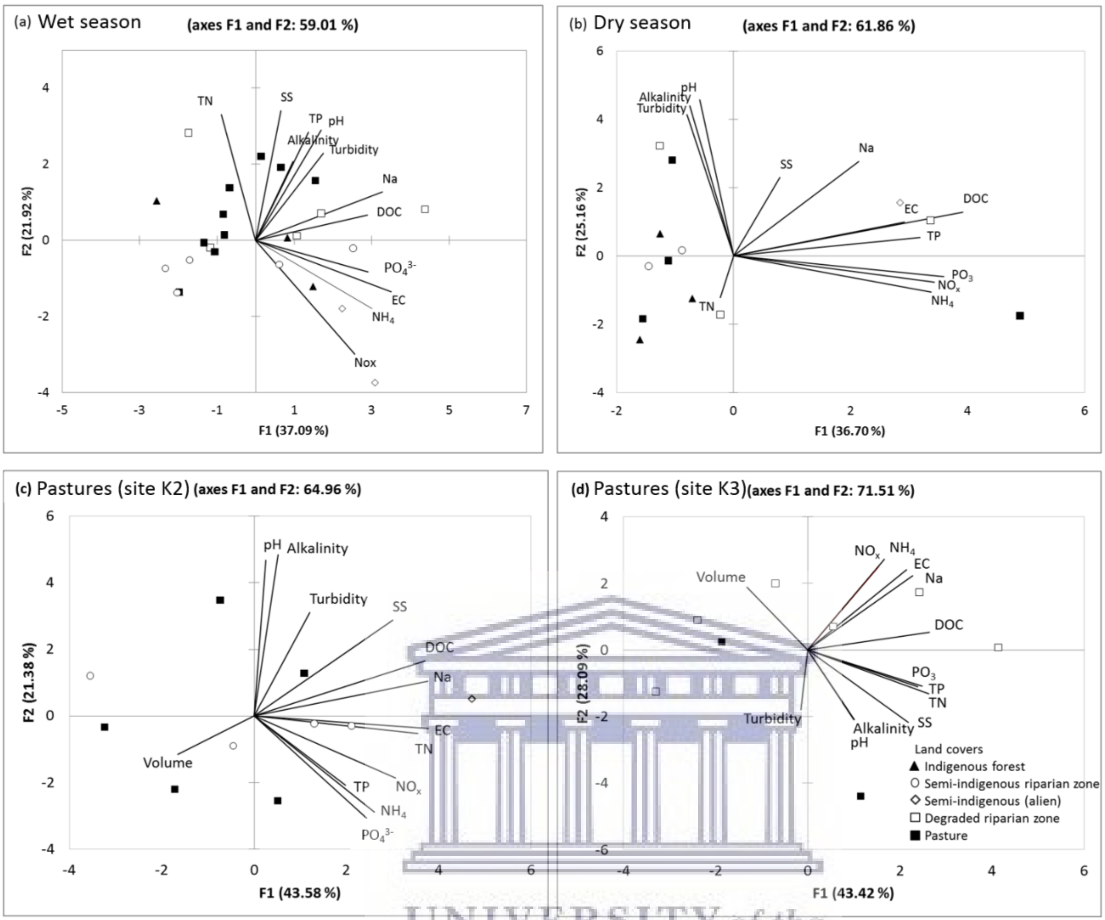

Figure 5.14 PCA analysis of seasonal runoff water quality variables between all sites and land covers

5.4 Discussion

A clear anthropogenic impact gradient was observed in vegetation and water quality as a result of land cover and use at the sites assessed. All sites consisted of woody, evergreen vegetation but the type and composition differed, which influenced the sediment quality, water and runoff quality and runoff volumes generated. The least runoff events and volumes occurred in the indigenous forest. The closed canopy in the forest results in lower rain drop impact and rainfall intensity due to increased interception of water by vegetation. Some rain water may also be intercepted by the canopy and flow as stemflow down the trees which further facilitates water infiltration into the surface and sub-soil reducing surface runoff (Cayuela et al., 2018). The forest vegetation increases transpiration which allows more infiltration before soils become saturated (Dosskey et al., 2010). The soils in the indigenous forest had the lowest concentration of TN compared to the other riparian sites sampled. According to Geldenhuys (1993) and Van der Colff et al. (2017) the soils in the southern Cape indigenous forests are generally nutrient poor as shown in Figure 5.10 at K1. Daalen (1984) showed that the evergreen forest trees have abscission and production of leaves year-round so that the nutrients are retained in the leaves rather than in the soil. The process of recycling nutrients (nitrogen and phosphorus) before leaf abscission is not unique to the indigenous forest species and is shared with invasive species such as Australian *Acacias* (Morris et al., 2011, Van der Colff et al., 2017). A dense root mat was observed during sampling in the top 0.2 m of the soil surface, which is the area where plant litter accumulates. Studies have shown that these roots are responsible for nutrient cycling and conservation (Lamont, 1982, Daalen, 1984, Kotze and Geldenhuys, 1992). Any nutrients released from decomposing leaf litter are utilized by the forest vegetation allowing for efficient nutrient re-absorption (Daalen, 1984). This explains why the highest TN concentrations occurred in runoff water from the indigenous forest but the loads in this runoff were the lowest, as well as the low concentrations sampled in river water at site K1, draining an area of relatively pristine indigenous forest (Table 5.10 and 5.11).

The change in vegetation down the longitudinal river gradient resulted in a decreasing canopy cover at the riparian sites from the indigenous forest to the degraded riparian zone (Table 2.5, Chapter 2, section 2.3). Although the leaf litter and understory vegetation limited sediment removal from runoff plots, high suspended solids were recorded from the semi-indigenous riparian plots. However, the highest values were recorded during October 2014, which was shortly after installation of the plots, causing disturbance of the soil surface. The degraded riparian zone, however, had no understory vegetation and more suspended solids were consistently recorded throughout the study period compared to the other sites, with the exception of the runoff plots at riparian zone site K2 (Table 5.10). This was also shown by the increased runoff and runoff coefficients from this riparian zone site K3 (Table 5.12, Figure 5.11). Sediment removal from the pastures were limited due to the perennial grass cover and replanting occurred during low to no rainfall periods. Runoff from these plots occurred when soils were completely saturated after increased rainfall periods or when antecedent soil water content was high.

A clear distinction between the nutrient dynamics in runoff from the unimpacted reference site compared to the sites further downstream where agricultural impact was observed. The effects of agricultural land use and alien vegetation encroachment by *Acacia mearnsii* in the riparian zone of the semi-indigenous and degraded riparian sites were detected by the increased nutrient concentrations in runoff during the wet and dry seasons of especially NO_x-N and NH_4^+-N illustrated by the PCA analysis. Runoff plots influenced by *Acacia mearnsii* were also associated with increased TN concentrations in the sediment and TN loads in runoff water.

The pastures adjacent to the semi-indigenous and degraded riparian zones were fertilized with a mixture of nitrogen and potassium between April-November (J Crowther 2017, personal communication, 7 August). The amount of fertilizer applied to pastures was reduced in the recent past (since 2006) as farming practices changed to limit costs (Petersen et al., 2017). Residual nitrogen can, however, remain in the saturated zone (Chen and Hong, 2011, Aguiar et al., 2015), as was observed in the current study with the relatively increased concentrations and loads of NH_4^+-N, NO_x-N and TN in the runoff from the pastures (Figure 5.12) compared to the riparian zones. Higher TP and TN concentrations also occurred in the pasture soils at site K3. This is also evident, as according to the land owner, the pastures where the runoff plots were located were not fertilized during 2016 due to lower rainfalls than the previous year's wet season (J Crowther 2017, personal communication, 7 August). The higher intensity rainfall over the pasture areas combined with any residual or accumulated nitrogen in the surface zones is likely responsible for the increased nutrient concentrations and loads from this land cover.

The percentage nitrogen concentration present in the deeper soil profiles, were much lower than in the surface soils at all runoff plots, with the exception of the reference site. The pastures were always grass covered so that nutrients were utilized by the vegetation. The pasture management entailed minimum tillage, which meant less soil disturbance influencing the nutrient dynamics where the organic matter can provide increased contact with colloids and phosphorus and nitrogen ions, increasing adsorption, mineralization and nitrification processes (Aguiar et al., 2015). The nitrogen concentrations recorded from the lower river bank sediments and river water were consistently low throughout the study period. The NO_x-N and TN concentrations in the river at site K3 only increased during the wet season and historical water quality data showed that nitrogen and phosphates in the catchment are positively correlated with increased rainfall and associated increased river flows (Petersen et al., 2017). The slope of the study area is low enough to generate overland sheet flow as reported by Peterjohn and Correll (1984) and Blanchè (2002). Therefore, although there is the possibility that nutrients are leached to groundwater with high rainfall and intensity, it is more likely that nutrients are supplied to the river channel via surface runoff and shallow through-flow and therefore the supply to the channel is event based.

The effectiveness of the riparian zones in nutrient mitigation was influenced by the inter-site contrasts in riparian vegetation. Although the runoff from the pasture to the semi-indigenous riparian zone showed a 60% reduction in NO_x-N concentration compared to the 72% reduction between the pasture and the degraded riparian zone, the TN concentrations in runoff from the degraded riparian zone were similar to those of the pastures, with only a 10% difference. The increase in alien *Acacia mearnsii* tree invasions resulted in an increase in TN percentage concentrations in both the surface and deeper soil profiles from the semi-indigenous and degraded riparian zones (Figure 5.10). The *A. mearnsii* trees are a nitrogen-fixing legume species that increase nitrogen pools in soils both through biogeochemical fixing and surface-loading of N-enriched leaf litter, and residual nitrogen has been found to remain in the sediment where such species occur (Stock et al., 1995, Corbin and D'Antonio, 2004, Marchante et al., 2008, Jovanovic et al., 2009). Marchante et al. (2008) and Morris et al. (2011) also reported that *Acacia* tree species have increased nitrogen in their leaves compared to indigenous species studied and that nitrogen percentages increased in leaf litter, which was subsequently leached to the soil as the leaves decompose. Dense leaf litter from the *Acacia mearnsii* trees occurred in the runoff plots at site K3. The higher TN concentrations in the degraded riparian zone sediment compared to the semi-indigenous and reference sites are consistent with the observations and interpretations of these previous studies. Other alien species are also capable of creating their own nutrient-rich environments to further their invasive potential as shown by Ehrenfeld (2003). The *Acacia mearnsii* trees are therefore ecosystem engineers in that they transformed the riparian zone in physical habitat and soil chemistry creating conditions suitable for further invasion of the riparian zone and potentially the river by other species (Gonzalez et al., 2008, Rilov et al., 2012). Although the riparian buffer zones had a similar vegetation structure, differences in species composition within the buffer strongly influence the sediment and runoff water quality.

The decrease in riparian width and vegetation type along the downstream river gradient impacted on the effectiveness of the riparian zones in nutrient mitigation from land derived impacts. At K2 the nutrient concentrations recorded in runoff from the semi-indigenous riparian zone were reduced compared to concentrations recorded in runoff from the adjacent pasture plots. The calculated loads between the pastures and the semi-indigenous riparian zone for all nutrients recorded were also reduced (Table 5.12). In buffer zone guidelines developed by Macfarlane et al. (2014), it is assumed, based on certain risks scores, that a buffer width of 5 m has a 50% nutrient efficiency removal rate and a buffer of 25 m has a 80% nutrient efficiency removal rate. In this study the width of the semi-indigenous riparian zone was on average narrower than 25 m and the nutrient loads were reduced at relatively high rates as those shown by Macfarlane (2014) with regards to TP (77% less) and NO_x-N (78% less). The TN loads occurred at a lesser removal rate at 28%. At site K3 all nutrient loads were increased in the degraded riparian zone plots in comparison to those recorded in the pastures as shown in Table 2.

However, the river water at both sites K2 and K3 had very low TN and TP concentrations compared to runoff sampled in the riparian zones. The changing nutrient conditions in the river water were illustrated by the biological indicators sampled at site K2 and K3 (Chapter 4). When higher nutrient concentrations and turbidity occurred during increased rainfall and river flow conditions, increased concentrations of diatoms associated with such conditions occurred. So even though river water was sampled every quarter of the year, the use of bioindicators in addition to water chemistry provided information on water quality and the structural functioning (habitat diversity, flow characteristics, biological interactions). Nutrient concentrations sampled in the river were always below recommended Target Water Quality Ranges (DWAF, 1996a, b, c) and increases were only evident during higher rainfall events. The nutrient concentrations were very similar to reference conditions with only slight increases in TN concentrations at site K3 (Table 5.11). This implies that a large percentage of the nutrients are retained by the riparian zones, even with the narrow width (< 15 m) and the presence of the alien trees as in the case of the degraded riparian zone. The infiltration and nutrient retention in the riparian zones is further enhanced by the uniform surface runoff and shallow through-flow that occurred through the riparian zone as also found by Blanchè (2002).

Site specific conditions as outlined in Macfarlane et al. (2014) play an important role when delineating riparian buffer widths for different climatic regions. It also illustrates that some degree of ecosystem service function in relation to water quality was performed by the degraded riparian buffer zone, despite being invaded by alien vegetation. Others such as Stromberg et al. (2009) and Corenblit et al. (2014) showed that alien species, under certain circumstances, were able to perform ecological functions in river systems such as sediment, nutrient and organic matter retention as well as conservation/restoration of riparian plant diversity. Since many riparian zones are invaded by alien vegetation and where few opportunities are available to restore indigenous vegetation, systematic process based restoration may be applicable (Stromberg et al., 2009).

5.5 Conclusion

The results illustrate a clear distinction between the nutrient dynamics in runoff from the reference site compared to the sites with anthropogenic impact. The contrasting land covers generated different runoff volumes, water quality concentrations and associated loads. Where agricultural impacts and alien vegetation occur, nutrient loads were increased in the riparian zones and pastures. Nutrient concentrations in runoff were lower in the semi-indigenous and the degraded riparian zones than in their respective adjacent pastures. However, the TN loads were higher in the degraded riparian zone invaded by *Acacia mearnsii* than in the adjacent pastures and in comparison to the riparian zones with predominantly indigenous vegetation. Although riparian vegetation retained a percentage of nutrients, low concentrations in the river was also a result of dilution with increasing river flows. The width of riparian zones as a mitigation to water quality should be assessed on a site-by-site basis. It is recommended that indigenous vegetation is used in the establishment of riparian buffer zones while the use of nitrogen fixing legumes be avoided.

Chapter 6: Discussion: Inter-relationships between hydrogeomorphology, ecology, anthropogenic impacts, and ecosystem service provision

6.1 Introduction

River landscapes and their associated riparian zones are essential to humans and have been providing life sustaining services for millennia. However, the growing demands of increasing populations have brought about serious degradation of these ecosystems and so also a shift in the functions and related ecosystem services (Schmutz and Sendzimir, 2018). It is accepted that multidisciplinary research is necessary in river science in order to understand structure and function and the role this plays in the provision and delivery of goods and services to society. Ecosystem services can be defined as direct and indirect contributions by ecosystems as benefits to human well-being that provide either provisioning (e.g. freshwater), regulating (e.g. water purification) or cultural (e.g. ecotourism) services (Vidal-Abarca et al., 2016).

The concept of ecosystem services has been entrenched in policies internationally, such as the United Nations Millennium Ecosystem Assessment (MEA, 2005) (Large and Gilvear, 2015). Ecological infrastructure is gaining momentum in water and land use management especially in the form of riparian vegetation, for its potential to deliver ecosystem services. However, there is often a lag from policy into practice as ecological infrastructure is not always easily quantifiable compared to others such as built infrastructure (Schmutz and Sendzimir, 2018). The ecosystem service concept has filtered into national policy instruments such as the National Development Plan and the National Infrastructure Development Plan as well as the National Water Resources Strategy in South Africa (UNEP, 2015). This came about with the realisation that water is a limited resource in South Africa and that natural capital can be used in its conservation, so various projects developed with this focus. One such project was the Project for Ecosystem Services (ProEcoServ-SA) focussing on ecosystem services and the integration of ecosystem service approaches into resource management and decision making (UNEP, 2015). The project was a Global Environment Fund (GEF) project in partnership with various worldwide organisations, which included the Council for Scientific and Industrial Research (CSIR) in South Africa. This project, in partnership with the South African National Biodiversity Institute (SANBI), developed a framework for investment in ecological infrastructure, with a focus on the regulating services that support and improve water quality and quantity using natural capital as a priority to socio-economic development in South Africa (SANBI, 2014). Another project investing in ecological infrastructure is the removal of invasive alien plants, especially stands of invasive alien trees that use more water than indigenous vegetation by the Working for Water Programme (Binns et al., 2001, Mander et al., 2017). Others have focussed on targeted rehabilitation of sub-tropical thicket on hillslopes denuded by grazing in the Baviaanskloof catchments (Mander et al., 2017). The consideration of ecological infrastructure in water and land use planning is beneficial and can reduce risk to societies. For example, riparian and other green infrastructure (e.g. networks of waterways, parks or conservation areas) can provide services that reduce flooding in a catchment by attenuating flood waters thereby reducing damage to built- infrastructure such as houses or roads, provide water purification or recreation (Benedict and McMahon, 2002, MEA, 2005).

Ecosystem services generally stem from naturally functioning ecosystems where ecological processes are responsible for driving the linkages between materials, nutrients, energy and organic matter (Vidal-Abarca et al., 2016). In riverine and riparian environments these linkages occur between hydrology, geomorphology and ecology where water, sediment and biodiversity control how river processes interact with the physical river template to provide ecosystem services (Thorp et al., 2006, Vidal-Abarca et al., 2016). Riparian zones as ecological infrastructure provide an array of ecosystem services. Vidal-Abarca et al. (2016) reported that with analysis of hydrogeomorphological and biological elements, riparian zones were responsible for the provisioning, regulating and cultural services but that most services they provided were in the regulating category. The provisioning services included clean surface water for drinking and material to algae and other animals for direct processing and use. Regulating services included mediation of toxic chemicals, mediation of flows, maintenance of physical, chemical and biological conditions, among others (Vidal-Abarca et al., 2016). In order to determine which ecosystem services will be provided by river and riparian ecosystems it is necessary to understand the structure and functioning of these systems and the pressures which impact them. Since many ecosystems are altered due to human and other impacts, there is a lack of baseline data on natural functioning and the related ecosystem services rendered. However, with shifting perceptions of what a "natural" functioning river is due to changing conditions over time, anthropogenically altered or ecologically degraded river systems or riparian zones may still be able to provide forms of ecosystem services (Stromberg et al., 2009, Corenblit et al., 2014, Large and Gilvear, 2015) and adaptive management and systematic process-based principles for restoration of river ecosystems may become more applicable (Beechie et al., 2010).

The preceding chapters of the current study outlined the field data collected during 2014 - 2017. It was essential to understand the catchment influences on the geomorphology, hydrology and ecology as well as the linkages between them in order to determine the ecosystem services that riparian buffer zones provided. Chapter 4 evaluated the linkages between riparian morphodynamics, the physical river template and utilised aquatic organisms as indicators of water quality and river ecological integrity. Chapter 5 investigated the controls on the morphodynamics of riparian zones and assessed the effectiveness of the riparian vegetation in mediating nutrient fluxes from agricultural land use and improving river water quality. The aim of this chapter is to interpret these results collectively by examining the hydrogeomorphic process-form, riparian vegetation relationships and aquatic biota responses in light of study objective four (Chapter 1, section 1.4). This chapter will also draw from the biophysical and anthropogenic information from Chapter 2 and the historical catchment land cover and land use data from Chapter 3. A systems understanding of the interactions between the different ecosystems and land cover/use as well as past and current land use management practices in the Duiwe River catchment is provided to understand ecosystem service provision by riparian buffer zones as ecological infrastructure.

Dollar et al. (2007) developed a conceptual framework for linking the three primary subsystems or hierarchies of river ecosystems, namely, fluvial geomorphology, hydrology and ecology in an interdisciplinary approach using scale as the common link. The approach used a multi-level (hierarchical) flow-chain model, which consisted of the driver (abiotic/biotic agent of change), the template or substrate which the driver acts on, controllers of the driver or agent of change and a process or entity that responds to the driver. The conceptual framework for river ecosystems provided a method to combine separate disciplines in an integrated manner by ordering phenomena and materials to reveal patterns (Dollar et al., 2007). According to Pickett et al. (1994) and Dollar et al. (2007), successful interdisciplinary science can be achieved by joining many areas of understanding into a single conceptual-empirical structure. The current interdisciplinary study demonstrates strong links between climate, catchment topography, river template, hydrogeomorphology, land cover and use, human activities and their influence on overall river ecological integrity. A hierarchical conceptual framework flow-chain model for river structure and function (Figure 6.1), was developed in the context of the current study, using the Dollar et al. (2007) flow-chain model as a basis. It demonstrates the linkages and processes between hydrogeomorphology and ecology and goes a step further than the Dollar et al. (2007) model by demonstrating the link between river ecosystems, goods and services they provide and how they are impacted on by human disturbance, illustrated with riparian buffer zones. The flow-chain model developed in this study also illustrates the role of water quality as a driver of river ecological integrity, which is lacking in other models and frameworks for river ecosystems and river integrity such as Thoms and Parsons (2002) and Dollar et al. (2007).

Similarly to the multi-level flow-chain model proposed by Dollar et al. (2007), the higher levels of the current framework also places constraint on the lower levels while allowing for bottom-up process integration. The system drivers and controllers of drivers, places constraint on the habitat drivers (templates), which determines response of biological and abiotic entities and ultimately which ecosystem services are provided. In the model, ecosystem services are considered a product of the interactions between the drivers (agent of change) acting on material (riparian zones and aquatic ecosystems) within the boundary templates (light grey box in Figure 6.1) or as a response to the products of the interactions between drivers, material and templates (dashed green box in Figure 6.1). The ecosystem service provision is therefore dependant on the context provided by the top-down constraints and the integration of bottom-up interactions, but also allowing for feedback loops. Understanding of top-down constraints and bottom-up process interactions, are beneficial as they capture the continuum of hierarchical influences (Dollar et al. 2007) necessary for the determination of ecosystem service provision.

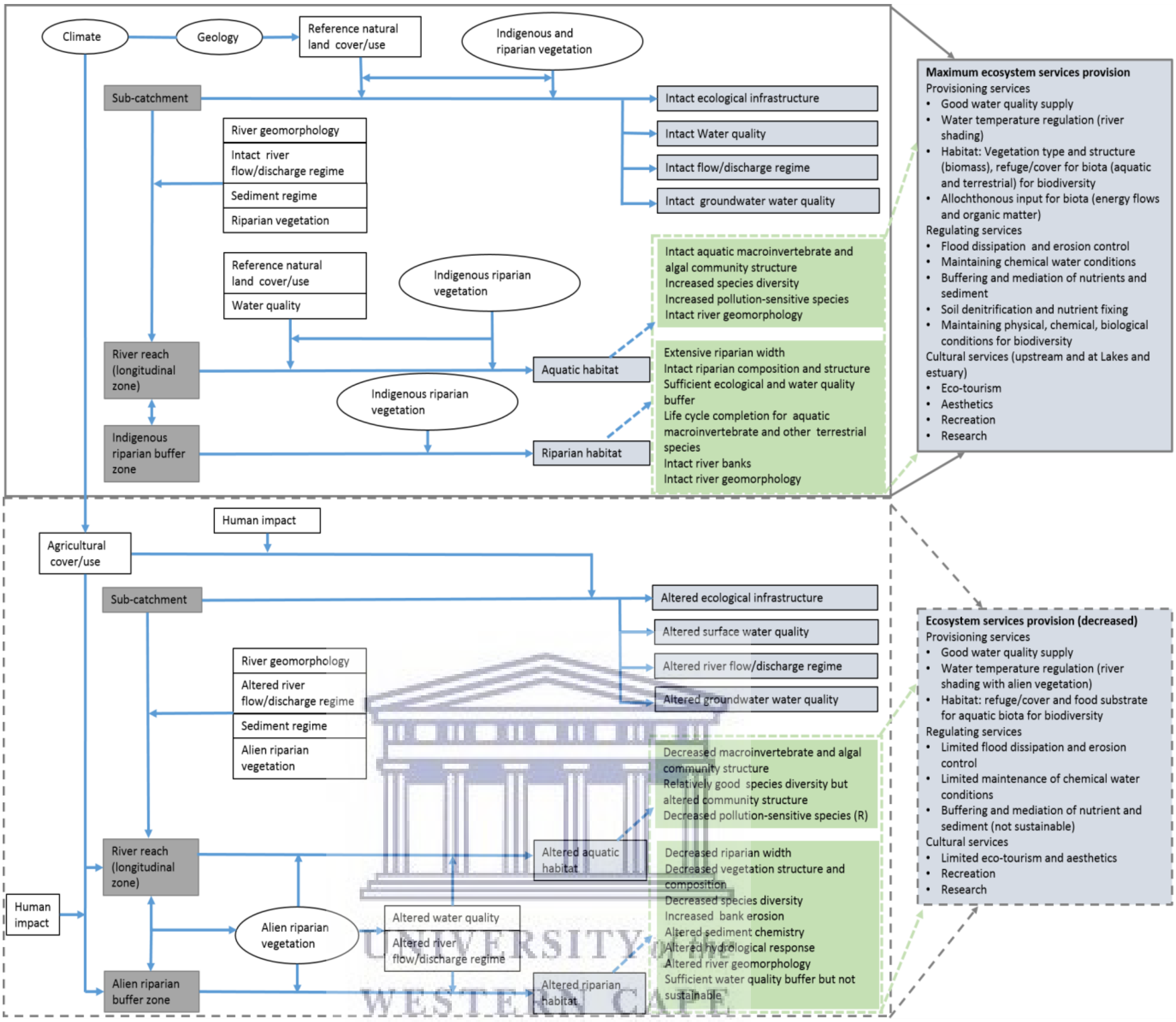

Figure 6.1 A conceptual framework flow-chain model of the links between system drivers, controllers of drivers, habitat drivers and ecosystem response (riparian and aquatic) in the provision of ecosystem services (in the context of riparian buffer zones).

Templates = dark grey box, Drivers or agents of change = white box, Controllers = ellipsoids, Products = light grey box (solid black line), Ecosystem response (abiotic/biotic) = green box and green dashed box. Solid grey border represents ecosystem services provided by natural/reference ecosystems while the dashed grey border represents ecosystem services provided by degraded or human impacted ecosystems.

6.2 Ecosystem services derived from riparian buffer zones

In order to determine ecosystem provision of riparian buffer zones, different scales of investigation was necessary. A sub-catchment scale determined that climate and geology in the study area (described in Chapter 2) are the overarching system controllers within the conceptual framework, which determine the river type and associated aquatic and riparian habitat, land cover, land use and vegetation distribution. (Figure 6.1). This is illustrated where the steeper slopes of the Table Mountain Sandstone in the Touws catchment and upper reaches of the Duiwe River catchment produced shallow, less fertile soils, which were conducive to alien tree plantations (in the Touws catchment) and natural forests while the coastal platform in which most of the Duiwe River catchment occur produced gentle slopes and fertile soils suitable for dairy pastures and vegetable crops, as shown in the catchment analysis in Chapter 3. The aseasonal rainfall meant an abundant water supply, both to the agricultural industry and to the vegetation biomes that occur in the catchments. The climate and geology determines the vegetation biomes and distribution so that the indigenous forests are confined in gullies and bergwind shadow areas where they are protected against fires that occur in the fire-adapted fynbos vegetation that surrounds the forests (Geldenhuys, 1994).

The land cover and land use with associated anthropogenic impacts became a driver or agent of change to the vegetation occurring in the riparian buffer zones at the different study sites and therefore the way in which riparian zones provided the ecosystem service of water quality enhancement (an ecosystem service product, Figure 6.1). At a sub-catchment scale where natural or reference land cover acts as an agent of change with no or limited anthropogenic influence, intact ecological infrastructure, good water quality, natural river flow/discharge and groundwater quality is produced. Indigenous vegetation and riparian vegetation acts as a controller within the natural landscape thereby contributing to the products produced (smaller light grey boxes in Figure 6.1). This becomes evident at the lower scales of the hierarchical framework at the physical river template and the indigenous riparian buffer zone (templates in Figure 6.1).

Minimal anthropogenic impact at the reference site K1 resulted in natural indigenous vegetation cover. The site occurred in a forest with intact vegetation species composition and structure consisting of an evergreen and deciduous canopy and sub-canopy of trees, shrubs, herbs and graminoids. The geology and the topography of the area resulted in an unconfined valley, a step-pool and pool-riffle river morphology, sandy river bank soils and an extensive riparian zone. The vegetation structure and composition of the riparian zone ensured that erosion of the sandy soils was minimal where the dense root mat occurred in top 0.2 m depth in the indigenous forest that protected the soil surface (Abernethy and Rutherfurd, 2000, Pollen-Bankhead et al., 2013). The combination of vegetation species and growth forms on both the left and right banks and in all plot positions along the banks at site K1 also provided increased bank stability due to different root systems (depths and tensile strength) as well as essential riparian habitat to aquatic and other species (Abernethy and Rutherfurd, 2001, Simon and Collison, 2002, Samways and Sharratt, 2010, Camporeale et al., 2013, Merritt, 2013). The forest vegetation therefore play a role in determining channel form and landforms present at the site but the landforms can also determine the type of vegetation that occur (therefore feedback arrows in Figure 6.1 between riparian vegetation and the river reach). This was illustrated at the degraded site K3 where the collapsed bank material at the base of the bank formed a flood bench that was colonized by mostly herbs and sedges due to the frequent inundation during increased river flows.

Table 4.7 shows the low turbidity levels recorded at site K1 throughout the study period, which is an indication of the low sediment yields supplied to the channel. The indigenous forest was adapted to nutrient poor soils where the surface root mats were responsible for nutrient recycling and conservation so that the forest vegetation was efficient at nutrient re-absorption from decomposing leaf litter (Daalen, 1984) (Chapter 5). The result of this was an effective buffering capability of the indigenous forest of nutrients to river water as illustrated by the very low nutrient concentrations recorded during all sampling events in the Klein Keurbooms River reaches flowing through the indigenous forest (Table 5.11). The product of these interactions were provisioning ecosystem services, which included the enhancement of water quality and habitat to aquatic and terrestrial species and regulating services, which included flood dissipation, erosion control, buffering and mediation of nutrients and sediment, soil denitrification and nutrient fixing (Figure 6.1).

The land cover/use are drivers of the habitat response created by the river template and riparian buffer zones (Figure 6.1). In the natural landscape, the unaltered river geomorphology, flow regime and indigenous riparian vegetation resulted in natural aquatic and riparian habitat. The water quality (controlled by indigenous vegetation) became a driver of river ecological integrity. The bioindicators (macroinvertebrates and algae) as well as the riparian vegetation were identified as the biotic responders to the habitat drivers within the boundary templates in the framework. Riparian structure and composition were important determining factors in the diversity and abundance of macroinvertebrates, which includes the functional feeding groups, as well as water quality (Vought et al., 1998, Fierro et al., 2017, Braun et al., 2018). However, even with taxonomic alteration in the riparian zones, macroinvertebrate communities may not always illustrate change. For example, alien trees may form a dense closed canopy having the same effect as a dense closed canopy formed by indigenous trees (Smith et al., 2007). The widest riparian zone of indigenous forest at the reference site K1 displayed macroinvertebrate assemblages of high-scoring, pollution-sensitive families from the orders Ephemeroptera, Trichoptera and Plecoptera (EPT), which occurred in acidic waters (pH: 4 to 5) that were low in nutrients, temperature and EC (Chapter 4). Water temperatures were regulated by the dense riparian canopy cover and nutrient loads in the river were regulated by the efficient nutrient use by the indigenous riparian vegetation. The ecological buffer to water quality provided by the intact riparian zones supported a diverse and species rich biotic community assemblage as the physical, chemical and biological conditions for biodiversity were maintained (ecosystem service product in Figure 6.1). Similar conditions occurred at the semi-indigenous site (K2) and algae and macroinvertebrate species reflected this.

Further down the longitudinal impact gradient at site K2 agricultural impacts and human disturbance resulted in a vegetation response where the riparian zone changed from indigenous vegetation to a mixture of indigenous and alien *Acacia mearnsii* trees with an average decreased riparian width of 17 m, while at site K3 the narrowest riparian width (<15 m) occurred, almost completely invaded by alien acacia trees. Dairy pastures adjacent to site K2 and K3 resulted in high nutrient concentrations recorded in runoff sampled. However, a decreasing trend in nutrients from the adjacent pastures to each riparian zone at both sites K2 and K3 occurred (Chapter 5, Figure 5.13). A much narrower riparian buffer at the semi-indigenous site occurred compared to site K1, with some invasion of alien vegetation, which still resulted in efficient nutrient retentions of NO_x-N and TP and therefore provided a provisioning and regulating service to water quality enhancement.

The more densely invaded riparian site K3 illustrated how vegetation can act as a controller of the driver or agent of change as described by Dollar et al. (2007). At site K3 a lower reduction in TN occurred in the runoff between the pasture and the degraded riparian zone compared to site K1 and K2 (Chapter 5). This was attributed to *Acacia mearnsii* trees being a nitrogen-fixing legume species that increases nitrogen pools in soils as shown in studies by Stock et al. (1995), Corbin and D'Antonio (2004), Marchante et al. (2008) and Jovanovic et al. (2009). Witkowski (1991), Forrester et al. (2005). Marchante et al. (2008) and Morris et al. (2011) reported that acacia tree species have increased nitrogen in their leaves, which is leached to the soil as the leaves decompose. The TN concentrations and loads at site K3 was due to extensive leaf litter in combination with already increased nutrient concentrations in soils and high nutrient content runoff from the adjacent pasture.

The changing riparian buffer zone with increased alien invasion and reduced width, affected the macroinvertebrate and algal communities as well as the riparian habitat (biotic response in Figure 6.1). Although the macroinvertebrate community assemblages at site K3 remained similar to site K2, the abundances of sensitive taxa at site K3 decreased, illustrating an impact on water quality (Table 4.7). The abundance of shredders also decreased compared to the upstream sites K1 and K2. Moraes et al. (2014) reported similar results with a greater abundance of shredders where wider riparian buffers occurred compared to narrower buffers. Site K2 and site K3 both showed higher abundances of macroinvertebrate predators, especially from the family Odonata at site K3, which had increased luminosity despite the alien tree invasion. This result was in agreement with those reported by Smith et al. (2007). The more open canopy at this site due to the shrub and grass dominated left bank and the eroded, alien tree dominated right bank, was likely to provide more allochthonous input from the alien acacias in comparison to the upstream sites. Mesa (2014) showed a lower species richness and diversity in macroinvertebrate taxa due to allochthonous input from alien acacias and citrus riparian vegetation compared to input from indigenous riparian species. Studies have shown that leaf litter from alien vegetation alters water quality, nutrient input and pH (Graça, 2006). At site K3 most of the water quality variables were elevated in comparison to the sites upstream, which included TN, turbidity and silica, especially during increased rainfall and flow events (Figure 3.6). During these events, changes occurred in the macroinvertebrate and algal assemblages where increased abundances of turbidity-tolerant species were observed. The algal species associated with increased nutrient supply were also more abundant. However, with the dilution capacity of the river, increased nutrient concentrations were short-lived.

The riparian habitat at site K3 was altered as acacia trees changed the vegetation structure and composition by forming tall stands and closed canopies on the right bank that resulted in no undergrowth of indigenous plants to provide increased bank stability. The acacias were directly responsible for the severe bank erosion observed. Souza et al. (2013) showed that changes in the canopy structure can determine the presence of undergrowth or change the hydrological regime, thereby impacting on runoff responses. The acacias can have a dimorphic root system where deeper root systems access deeper stored water and nutrient sources during dry periods in addition to a shallow, dense root biomass that have access to surface water during wetter periods (Le Maitre et al., 1999, Morris et al., 2011). The invaded area at site K2 and K3 displayed a dense root mat in the 0 - 0.3 m range of sediment depth and the deeper roots were exposed on the right bank after scouring (Figure 5.3b). The shallow root system of Australian acacias are associated with increased bank erosion where they are unable to withstand flash floods which uproot the trees and cause bank collapse and tree toppling (Rowntree, 1991, Holmes et al., 2005). This was observed at the site K3 where a steep right bank developed after previous flood events due to channel bank collapse and scour (Figure 5.3) as the rooting depth of the trees did not develop below the plane of failure (Rowntree, 1991, Holmes et al., 2005).

The soil moisture was generally decreased at site K3 throughout the study period when compared to the more natural banks upstream (Figure 5.9). According to Dye and Jarmain (2004) and Rebelo et al. (2013) *Acacia mearnsii* transpiration rates are increased due to increased biomass linked to the dense stands resulting in decreased soil water content. This is likely to alter the groundwater tables of the riparian zone. Similar to the reference site, the sandy banks at the degraded site were protected from erosion of sediment at the surface due to stability provided by the surface root biomass (Tickner et al., 2001). This was consistent with results reported by Rowntree (1991) where roots of *Acacia mearnsii* increased the shear strength of bank material resulting in the maintenance of a steeper cross-section as at site K3.

The alien trees can also be considered as a controller when they modified river cross-sections and flow patterns (Figure 6.1). Toppled *Acacia mearnsii* trees caused the development of an instream wood debris dam resulting in a narrower channel downstream of the debris dam (Figure 2.12 c, transect 3.3) and an increase in channel cross-section area upstream of it, which contributed to the channel erosion (Figure 2.21 c, transect 3.2) (Rowntree, 1991, Nakamura and Swanson, 1993, Tabacchi et al., 2000, Gurnell, 2013). Studies by Rowntree (1991), Richardson et al. (1997), Tickner et al. (2001), Holmes et al. (2005), Bruton (2010) and Rebelo et al. (2013) showed that alien vegetation can have a direct impact on river channel morphology whereas others such as Le Maitre et al. (2015b), Scott et al. (1998a) and Scott et al. (1998b) showed that indirect impacts can result from runoff reduction and sedimentation. The toppled *Acacia mearnsii* trees at site K3, provided dead wood to the channel and created habitat, shelter and flow diversity for aquatic organisms as well a food source to macroinvertebrates by providing a substrate for algae, fungi and bacteria (Naiman et al., 2005). This was responsible for some of the diversity of biota still sampled at the degraded site. The toppled trees provided a provisioning ecosystem service to aquatic biota by creating a substrate for algae and a food source to macroinvertebrates as well as a refuge area for other aquatic biota such as fish, increasing biodiversity and thereby river ecological integrity.

The riparian vegetation at all sites acted as controllers of the physical riparian habitat, water and soil chemistry. However, the *Acacia mearnsii* trees were able to transform their environment to create suitable conditions for further invasion (Tickner et al., 2001, Gonzalez et al., 2008, Rilov et al., 2012). Without any intervention, the possibility exists that the acacias will spread further into the semi-indigenous riparian zone site K2. The location of this site is currently in an area that serves as an area of recreation and tourism. Further spread of the invasive acacias into the semi-indigenous riparian zone will continue to transform the riparian ecosystem. This will not only impact on the ability of the riparian zone to attenuate nutrients from the pasture but also places the cultural ecosystem service at risk and therefore the economic benefit to the landowner.

Despite deterioration in riparian width and structure along the downstream river gradient, an ecological buffer was still provided evidenced by the water quality, intact instream habitat and the presence of pollution-sensitive macroinvertebrate and algal taxa, (although decreased) at the alien invaded site K3. This regulatory and provisional services to water quality provided by the alien invaded riparian buffer zone (shown by the dashed grey box in Figure 6.1), can be impacted on in the long-term. Currently, the nutrient and turbidity increases are event-based, but based on current field observations during the study, the possibility exists that the riparian width will decrease further with high or flood flows due to further bank collapse, especially at site K3. An even narrower riparian width may not have the ability to mitigate the nutrient enriched runoff from the adjacent pastures in combination with the additional TN accumulated in bank sediment related to the nitrogen-fixing acacias and their increased nitrogen content in leaf litter. Macfarlane et al. (2014) and Macfarlane and Bredin (2017) showed that nutrient efficiency removal rates dropped with narrower buffers and that wider buffers were more efficient. Any accumulated nitrogen and phosphorus in the bank material will increase concentrations in the river water with further bank erosion impacting not only on water quality but also on the aquatic biodiversity (Tabacchi et al., 2000). The results from Chapter 4 showed that during increased rainfall and river flows the macroinvertebrate and algae species were altered when water quality deteriorated. A decreased riparian width may therefore alter the macroinvertebrate and algal communities as habitat and water quality deteriorates (Braun et al., 2018). Studies have shown that for biodiversity of aquatic and semi-aquatic macroinvertebrate species, wider riparian buffer zones are required to successfully complete their life-cycles (Samways and Sharratt, 2010, Macfarlane and Bredin, 2017). Braun et al. (2018) reported that EC concentrations in rivers with narrower riparian buffer widths (< 5 m) increased and that dominant genera or life stages of macroinvertebrates were affected due to the changing water quality. The boundary templates (habitat drivers) and agent of change impacting these templates, shown in Figure 6.1, will therefore largely influence the response of aquatic ecosystems and thereby the river ecological integrity.

The interactions and processes that occurred upstream between model components (drivers, material and templates) determined the responses of components downstream. This was clearly illustrated at the cumulative impact site K4. The combination of changing river hydrogeomorphology and land cover and use in the upstream catchment were large drivers of influence on aquatic ecosystems at this site. The riparian zone was natural in vegetation and structure (average width 25 m) but impact from upstream resulted in the most deterioration in water quality, especially in terms of EC, nitrogen and phosphorus (Figure 3.6). The altered flow regime (driver) (Figure 6.1) due to agricultural abstraction and water quality due to nutrient enrichment, was reflected in the algal and macroinvertebrate species that occurred. The macroinvertebrate communities decreased in species richness with pollution-tolerant taxa dominating as well as species which were less dependent on flow. The river bed substrate (material) was altered as more constant low flows resulted in coarse substrate being embedded by finer material reducing the aquatic habitat available and thereby impacting on this provisioning ecosystem service. It may be that the material transported from upstream and deposited at site K4 was nutrient-rich, and although the instream deposited material was not analysed, the lower bank plots at this site had the highest concentrations of TP (Figure 2.14). This altered riverine state likely influenced the instream ecosystem services such as nitrogen or organic matter processing (Sweeney et al., 2004) resulting in a change in the provision of this ecosystem service (Figure 6.1). The open riparian canopy in combination with a reduced flow regime, limited the interaction between the river and the riparian zone to the occurrence of increased river and flood flows during the wet seasons (increased rainfall periods). This reduced interaction with the riparian zone and floodplain will impact the hydrodynamics and nutrient, organic matter and decomposition processes (Lamers et al., 2006). During high and flood flows an improvement in water quality and macroinvertebrate species richness was observed at site K4 and the algal community changed from being dominated by benthic filamentous algae associated with increased nutrients, to diatoms.

Site K4 illustrated that although riparian zones are intact and of adequate width for buffer functions, their roles may be reduced by inappropriate or inadequate land management in upstream catchments. Despite land management improvements made upstream such as reduced amount fertilizer applied to pastures and minimum tillage (Petersen et al., 2017), the cumulative impact of land use to water quality was reflected in the low-scoring pollution-tolerant macroinvertebrate biota and algal assemblages at site K4. The water quality and river hydrogeomorphology remained the driving factor of species richness and river self-purification at this site as demonstrated in Chapter 4.

Poor land management in the upstream catchments and the altered river structure and function has implications for the downstream Ramsar Wilderness Lake Eilandvlei, which the Duiwe River feeds. Increased nutrient runoff and bank degradation of nutrient rich sediment will increase the supply of nutrients to downstream river reaches and the Wilderness Lakes and estuary, thereby increasing the likelihood of eutrophic conditions. Russell (2013) reported that water inflow from rivers to Lakes were the primary drivers in changing water quality which affected the biophysical and chemical properties. A further study by Taljaard et al. (2018) suggests that river inflows may not be the primary driver of inter-annual and seasonal trends in dissolved inorganic nutrients but that catchment derived fluxes are still significant especially during flood events, as shown by Allanson and Whifield (1983) or through secondary mechanisms (remineralization).

The Wilderness Lake System have recently been experiencing nuisance algal blooms accompanied by fish and bird kills (Russell, 2013, Taljaard et al., 2018). Currently the Lake System displays resilience as it is able to assimilate the impact on water quality. However, the Lakes have a low flushing potential and with incremental nutrient enrichment due to poor land use management, the potential exists for a threshold exceedance with an irreversible change in trophic status (Taljaard et al., 2018). This will ultimately impact on the cultural benefits such as ecotourism and recreation currently derived from this Lake System. Furthermore, changes to the upstream ecological infrastructure such as the increasing alien tree invasion and the associated hydrological impacts, will alter the flood attenuating service provided by natural vegetation structure and processes. In the recent past seasonal estuary mouth closure in combination with floods of the Lakes have caused significant damage to infrastructure such as holiday houses and businesses (Le Maitre et al., 2014). Climate change predictions estimated that the frequency of high intensity rainfall will increase in the Southern Cape area (New et al., 2006, Nel et al., 2011a) increasing risk to communities in the area. Le Maitre et al. (2014) showed that with increasing invasion of acacias and pines in the Southern Cape catchments flood damage risk will increase with even less intense rainfalls than those which were previously recorded.

Interpretation of the land cover/use, vegetation and aquatic biota characterization at different scales allowed for the understanding of the underlying processes involved in ecosystem service provision, change or reduction thereof. The flow-chain model explained the processes and interactions that occurred over nested geomorphological levels with a description of abiotic, vegetation and aquatic responses and feedback in the provision of ecosystem services. Furthermore, the model illustrated that the vegetation structure and composition are important variables in the riparian buffer zones and that together with vegetation attributes and land use management, the hydrological and nutrient cycles were influenced, which ultimately affected water quality and water quantity, which were also drivers of river ecological integrity. Similar findings were reported in Tabacchi et al. (2000), Dosskey et al. (2010), Roberts et al. (2012) and Souza et al. (2013).

6.3 Concluding remarks

Similarly to the river ecosystems hierarchical flow-chain model of Dollar et al. (2007), the hierarchical conceptual framework flow-chain model developed in Figure 6.1 identified and integrated physical drivers or agents of change at different scales. The biological entities were identified and integrated into the model as responders to the agents of change (Dollar et al. 2007). The flow-chain model illustrates that the system drivers and controllers of drivers places constraint on the habitat drivers (templates), which is coupled with anthropogenic impacts to determine the response of biological entities (riparian vegetation and aquatic biota). The riparian vegetation were responders but also became a controller (indigenous and alien vegetation) depending on the scale of investigation.

The model illustrates the interdisciplinary understanding of interactions and processes between hydrogeomorphology and ecology within boundary templates to produce ecosystem services (products). Ecosystem services can be considered a product of the interactions between the drivers (agent of change) acting on material (riparian zones and aquatic ecosystems) within the boundary templates or in the context of a response to a product of the interactions between top-down constraints and the bottom-up processes within the boundary templates. This was illustrated where ecosystem structure and function was altered at site K2 and site K3 thereby affecting the potential of riparian buffer zones to attenuate nutrients from the agricultural environment. The model structure proposed can assist in the appropriate management of rivers to maintain ecological integrity and their riparian ecosystems to provide maximum ecosystem services in light of shifting perceptions of naturally functioning rivers.

Chapter 7: General conclusions and recommendations

7.1 Key research findings

The primary aim of this thesis was to develop a systems understanding of the interactions between different ecosystems, land cover and the influence of past and current land use management practices in the Touws and Duiwe River catchments. The focus was on the biophysical interactions, linkages and drivers of change to water quality and quantity and ecological river integrity. This formed the process-based understanding of the hydrogeomorphic controls on the provision of goods and services that vegetation, and especially riparian vegetation, perform as ecological infrastructure, as illustrated in the conceptual framework flow-chain model (Figure 6.1). This framework was informed by the field research data, which provided an understanding of the processes and interactions between hydrogeomorphology and ecology to provide ecosystem services as products (Figure 6.1), and changes thereto, in the context of changing land cover/use and riparian zone composition and structure. Given the significant influence riparian zones have on fluvial systems, its ecological importance and its ability to provide numerous goods and services, it was necessary to understand the linkages between these ecotones, rivers and their terrestrial environments. In doing so the research provided knowledge to inform appropriate river management for ecological integrity and characteristics for desirable riparian buffer zones to provide maximum ecosystem services in a South African environment.

This section summarizes the key findings or conclusions of the core chapters (Chapters 3 - 6) in light of the research objectives and knowledge gaps identified in the introduction to this thesis (Chapter 1, section 1.3).

7.1.1 Objective 1: To examine historical influences of land cover and land use changes on water quality (Chapter 3)

Chapter 3 presented a historical analysis based on available secondary data sources for the Touws and Duiwe River catchments. Each of the sub-catchments studied have unique characteristics which influence water quality and each one was separately analysed. These two rivers are an important water supply source for the human population, economically and for the provision of ecosystem goods and services along the Southern Cape coast. The primary aim was to deduce how land cover/use changed over time and how the direct and indirect effects of these changes influenced water quality and flows in the past and present. The approach used mixed methods and focused on land cover and use and how it changed on a catchment-wide scale and within 100 m buffer zone (including the riparian and adjacent area). The following broad findings can be drawn from this research.

The combination of the historical aerial photograph interpretation, land cover change analysis and historical water quality, chemistry, surface flows and historical rainfall records, provided a more holistic approach to catchment analysis and improved the spatial and temporal understanding gained. Studying the past allowed a gain on current perspective of recent and ongoing changes to water quality. The historical mapping analysis provided an initial context for water quality comparison in each catchment.

The study demonstrated a clear link between land cover/use and water quality and the role of temporal (historical data analysis) and spatial scales (buffer areas) in the understanding of the influence of land cover and management of land use activities on water quality. A large proportion of the Touws River catchment remained natural and much of the changes in the land cover and use occurred in the Duiwe River catchment. Changes in the rainfall patterns and intensity with associated evaporation rates influenced land use in the catchment. Farmers reduced the percentage of pasture being irrigated and changed to centre pivots for irrigation efficiency. Water quantity and quality in the Duiwe River catchment deteriorated with time as land cover and use were altered. The secondary databases used showed that when the spatial heterogeneity of the catchment was altered either by human influences such as agriculture or by natural events such as fires, it was reflected by changes in the water quality and quantity available as stream flow.

The land cover/use in the 100 m buffer area provided a slightly better indication of the impact on water quality. The sub-catchments heterogeneity masked any signal from land cover/use impact on water quality and the impacts were therefore more pronounced from the area closest to the rivers. Even with the differences between the catchments, both were very responsive to rainfall. Nutrients increased in both river systems after high rainfall and flow events supplied via runoff and erosion from the land and flushing from sediments trapped by dams, with a cumulative impact of nutrients in the Duiwe River catchment. The buffer analysis indicated that both alien and indigenous vegetation were a source of nutrient supply in the Duiwe River catchment. The historical water quality records in the Touws River illustrated the assimilative capacity of the river as nutrient influxes were short-lived.

Changes in land management in the recent past had a substantial impact in water quality improvement, especially in the Duiwe River. After 2006, a decreasing trend in nutrients were observed, which was attributed to reduced and more efficient use of fertilizers applied by farmers to minimise costs. In interviews conducted with farmers it was noted that minimum tillage was used in pasture management and wastewater produced by dairy farms containing cow manure was used from settling pans to spread directly onto pastures, which reduced nitrogen fertilizer requirements significantly. These changes were especially important since they demonstrated cost-efficient solutions to water quality impairment. Long-term flow, water quality and rainfall databases paired with historical land cover and land use data proved to be essential in the understanding of the impacts on water quality and water quantity in both catchments analysed. Both the catchment land cover and land use characteristics as well as the 100 m buffer area was useful in determining impact on water quality. The above mentioned method proved to be cost-effective as it provided a desk top method to identify areas in the field where nutrient pollution may be occurring and so assisted in site selection in the study.

7.1.2 Objective 2: To evaluate linkages between riparian morphodynamics, the physical river template and using aquatic organisms as indicators of water quality and river ecological integrity (Chapter 4)

Chapter 4 investigated the response of aquatic communities (macroinvertebrates and algae) to physical and chemical water quality parameters in the agriculturally dominated Duiwe River catchment. The goal of this chapter was to determine the inter-relationships between the physical river template (including vegetation effects) and environmental drivers of algal and macroinvertebrate communities and suitability in their use as bioindicators for predicting and monitoring river ecological integrity and water quality. The sites were selected to reflect the changing land cover/use and riparian vegetation along the river gradient. The chapter revealed that the physical river template, water quality and hydrogeomorphology were important drivers of the variation in algal and benthic macroinvertebrate abundance and composition. The land derived impacts affected the physical river condition and water quality down the longitudinal river gradient.

Natural and anthropogenic changes to river hydrogeomorphology contributed to changes in the algal community structure. Anthropogenic changes and agricultural activities reduced flows by abstraction and so created favourable conditions for the growth of benthic filamentous algal mats, which in combination with water quality impact, reduced sensitive macroinvertebrate taxa especially at the cumulative site K4. However, at the reference site K1, during the natural dry season period, the typifying diatoms were replaced by the benthic filamentous green algae, *Ulothrix zonata*. The reduced river flow at site K4 changed the macroinvertebrate community to represent taxa that were less dependent on flow. A significant finding however, was that with increased flows during the wet season at site K4 the macroinvertebrate taxonomic richness increased as well as the removal of benthic filamentous algal mats, which were replaced by an increase in diatom species. This showed the possibility for river self-purification with improved flow management.

The decrease in abundance and diversity of both macroinvertebrates and algae and increase in pollution-tolerant taxa indicated that the bioindicators responded to stressors such as water quality impacts. Macroinvertebrates indicated water quality deterioration but not the specific impact and were better indicators of habitat integrity than algae. In the study certain macroinvertebrate species occurred in certain habitats associated with certain flow conditions. At times habitat availability was flow–dependent and with further study of flow sensitivity of macroinvertebrates, the opportunity to identify indicator species for flow reductions or flow restoration could be provided. The patches of flow heterogeneity that naturally occur in rivers will be occupied by different species of either macroinvertebrates or algae. The presence or absence of these bioindicators can therefore be used to determine instream flow requirements where flows can be determined or restored for the maintenance of different flow types within a river system.

Since the autecology for algae was known from literature, algae gave a better indication of nutrient enrichment, which was validated with observations of increasing nutrient concentrations. Diatoms were not flow dependant, could be sampled in any condition and gave a good indication of water quality. Sampling of bioindicators therefore provided a cost-effective and rapid method of assessing water quality as well as river ecological integrity compared to costly logging instrumentation, which would only give information on the water quality parameters, would require regular field maintenance to ensure data reliability and would require a form of security against possible damage or theft. The current study showed that the use of bioindicators in addition to water chemistry provided a holistic view of river ecological integrity, which included water quality and the structural functioning related to habitat diversity, flow characteristics and biological interactions of the aquatic ecosystem.

7.1.3 Objective 3: To investigate controls on the morphodynamics of riparian zones (this incorporates vegetation effects) and to assess the linkages between agricultural land use and water quality (Chapter 5)

Chapter 5 investigated the linkages between riparian morphodynamics and water quality to evaluate the effectiveness of riparian zones in mitigating land derived impacts on water quality (nutrients from pastures) along a longitudinal river gradient. The same study sites as in Chapter 4 were assessed in the Duiwe River catchment, where the study sites consisted of riparian buffer zones differing in vegetation types and land use impact. Each riparian site was characterised by identifying the plant community distribution patterns. Runoff plots were used to determine the water and sediment quality emanating from variations in land cover (the riparian zone and agricultural fields) along the Klein Keurbooms River. The runoff water and sediment quality were evaluated for differences in nutrient flux in the agricultural pasture and again within each riparian zone. This provided an opportunity to examine the influence and effectiveness of the different riparian vegetation in mediating nutrient fluxes and improving river water quality in comparison to adjacent pastures under minimum tillage.

The research showed a clear longitudinal gradient in vegetation and water quality as a result of land cover and use. The water quality and vegetation changed as land cover and use changed.

Although all sites consisted of woody, evergreen vegetation, the type and composition differed, which influenced the sediment quality, water and runoff quality and runoff volumes generated. The results showed that the rainfall amount and intensity decreased below riparian canopies due to interception compared to rainfall in the open-sky agricultural sites. The results illustrate a clear distinction between the nutrient dynamics in runoff from the reference site compared to the sites further downstream with agricultural impact. The reference site K1 was located in the indigenous forest riparian zone with no anthropogenic impact. The forest structure and composition was intact and high transpiration rates resulted in the least runoff. Runoff quality from this site showed the highest TN concentrations compared to all other sites but the loads were the lowest and nutrient conservation and cycling was efficient. Low concentrations in nutrients were observed in the river draining the near pristine forest.

The effects of agricultural land use and alien vegetation encroachment by *Acacia mearnsii* in the riparian zone of the semi-indigenous (site K2) and degraded (site K3) riparian sites were detected by the increased nutrient concentrations in runoff during the wet and dry seasons of especially NO_x-N and NH_4^+-N. The nitrogen-fixing *Acacia mearnsii* trees transformed the riparian zone in physical habitat and soil chemistry creating conditions suitable for further invasion of the riparian zone. The composition and structure of the riparian zone was altered with no vegetation undergrowth below dense canopies, extensive bank erosion and large amount of leaf litter. The suspended solids in runoff and the runoff coefficients were the highest at these sites. The soil below the invasive trees displayed high TN concentrations in the surface and deeper sediment profiles. Sediment sampled from lower bank plots in the absence of acacias displayed consistently low concentrations. Long-term historical data (Chapter 3) illustrated that NO_x-N and TN concentrations in the river at site K3 mostly increased during the wet season and that nitrogen and phosphates in the catchment are positively correlated with increased rainfall and associated increased river flows. Therefore, although there is the possibility that nutrients are leached to groundwater with high rainfall and intensity it was concluded that the nutrient supply to the river channel was via event based surface runoff and shallow through-flow.

The effectiveness of the riparian zones in nutrient mitigation was shown by the longitudinal contrasts in riparian vegetation. A decreasing trend in nutrient concentrations in runoff were observed from the pastures to the semi-indigenous and degraded riparian zones. However, high TN loads were associated with the degraded riparian zone and were less dissimilar to loads recorded from the agricultural pastures. This was explained by the already increased nutrient concentrations in soils, high nutrient content runoff from the adjacent pasture and nutrient supply from acacia leaf litter. This demonstrated that the different species composition and structure within the buffer is an important consideration in its establishment.

Even with event-based increases, the nutrient concentrations in the river at site K2 and site K3 were always below recommended Target Water Quality Ranges. The riparian widths at the semi-indigenous and degraded riparian sites were much narrower than in the indigenous forest but nutrient retention capacities were similar. This implies that a large percentage of the nutrients are retained by the riparian zones, even with the narrow width (< 15 m) and the presence of the alien trees as in the case of the degraded riparian zone. The degraded riparian buffer zone showed similar results to the nutrient retention of the semi-indigenous riparian buffer, implying some degree of regulatory ecosystem service to the enhancement of water quality. Although riparian vegetation retained a percentage of nutrients, low concentrations in the river were also a result of dilution with increasing river flows.

7.1.4 Objective 4: To synthesise the information from these objectives to develop an ecological infrastructure framework assessing ecosystem service provision of riparian zones (Chapter 6, with inputs from Chapters 3, 4 and 5)

The aim of this chapter was to interpret results collectively from previous chapters by examining the hydrogeomorphic process-form, riparian vegetation relationships and aquatic biota responses in light of study objective 4. The inter-relationships between geomorphology, hydrology and ecology was examined in order to understand the structure and function of river and riparian ecosystems as ecological infrastructure in providing ecosystem services. Riparian zones as ecological infrastructure, provide an array of ecosystem services including provisioning services (e.g. clean surface water for drinking), regulating services (e.g. mediation of nutrients and sediment from agricultural surfaces) and cultural services (e.g. recreation).

The current study used a conceptual framework flow-chain model as a basis for evaluating ecosystem services provided by river ecosystems and in particular riparian buffer zones. The model provided a method to combine and integrate the disciplines of hydrogeomorphology and ecology to reveal process and pattern. The flow-chain model explained the processes and interactions that occurred over nested geomorphological levels with a description of abiotic and biotic (vegetation and aquatic) responses and feedback in the provision of ecosystem services. The maximum ecosystem services benefits are derived from the wider indigenous riparian buffer zones and the lack of additional land use pressure. The riparian vegetation were responders but also became controllers (indigenous and alien vegetation) depending on the scale of investigation. The wider riparian buffer zone of indigenous vegetation with intact composition and structure, ensured that water quality in the river was enhanced for the benefit of humans and the aquatic environment, associated with increased habitat and aquatic biodiversity.

Agricultural land cover and use, over time, placed pressure on riparian areas by altering the width, structure and composition of the vegetation. The narrower riparian buffer (17 m) with indigenous vegetation still efficiently mediated nutrient fluxes from agricultural areas and the macroinvertebrate and algal species were an indication of this. The human disturbance brought about invasive alien species, which transformed not only the riparian habitat but ultimately the water quality, riparian habitat, aquatic biota and the physical river channel. This illustrated how alien vegetation became a controller. A decrease in provisioning ecosystem services occurred observed in altered bank conditions (due to erosion) and ultimately water quality and the aquatic ecosystem shown in Figure 6.1 by the dashed grey box. However, the presence of some high-scoring macroinvertebrates indicated that the narrow riparian buffer (< 15 m) invaded by alien trees still provided some form of regulatory service to water quality. The riparian buffer zones at sites K2 and K3 were therefore able to reduce the impacts from the adjacent agricultural land use. The study illustrated the river's assimilative capacity coupled with dilution effects, improved the water quality, both at the degraded site K3 and the cumulative impact site K4.

With agricultural land cover/use the model illustrated that land use management was an essential component in the ability of riparian ecosystems to provide ecosystem services. Flow alteration in the upstream catchment affected the instream habitats and water quality at the cumulative site K4 and the algae and macroinvertebrates were indicative thereof. Water quality in combination with river flow was an important driver of river ecological integrity in the model. When the flows and water levels increased the macroinvertebrate and algal diversity improved, which presents the possibility of identifying indicator species for flow reductions or restoration. Further alteration to the ecological infrastructure will not only lead to further negative environmental impacts but ultimately the loss of essential ecosystem services.

The flow-chain model developed in Figure 6.1 provided a basis for an understanding of the linkages, processes and interactions that allows or prevents ecosystem service provision by river ecosystems and in particular, riparian buffer zones. The model illustrated how the integration of multiple disciplines can be combined to produce abiotic and biotic responses and how ecosystem services can be evaluated, how the services are altered or reduced. This model structure can be utilised to assist in the appropriate management of rivers and their riparian ecosystems to provide maximum ecosystem services even in an altered riverine environment.

In summary, the following key findings are presented:

- The use of secondary databases proved valuable in the interpretation of the catchment analysis and the land cover/use were primary drivers in the past and present on water quality and quantity in both the Touws and Duiwe River catchments. The secondary databases were also an important tool in linking natural events that altered water quality in the catchments such as floods and fires (Chapter 3).
- The geology and climate were large controllers of the land cover/use and anthropogenic impact altered not only water quality but also the riparian vegetation.
- The bioindicators, macroinvertebrates and algae, used in combination with water chemistry, improved the understanding of the structural functioning (habitat diversity, flow characteristics, biological interactions) of the aquatic ecosystem (Chapter 4). Both the macroinvertebrates and algae species were influenced by the riparian vegetation present. The bioindicators responded to stressors such as water quality impacts, reduced flows and disturbances such as increased flows and instream habitat alterations.

Macroinvertebrates were good indicators of general river condition and habitat integrity and were sensitive to changing hydromorphology whereas benthic filamentous algae were better indicators of nutrient enrichment. Diatoms were a good indication of water quality and were available to sample in most river conditions.

- The bioindicators illustrated that their sensitivity to flow provides the opportunity to identify indicator species for flow alteration or flow restoration.
- Riparian vegetation was an important controller of water quality and instream/terrestrial habitat. Indigenous riparian vegetation were efficient ecological buffers to water quality enhancement by efficiently reusing and recycling nutrients (Chapter 5). The indigenous vegetation maintained the natural physical, chemical and biological components of the river ecosystem and provided the most ecosystem services.
- Riparian buffer zones were altered by nitrogen-fixing alien acacias that transformed riparian habitat, soil and water chemistry (riparian runoff and instream) (Chapter 5). The alien riparian vegetation were responsible for the bank erosion at the downstream sites and for the morphological channel changes caused by an instream debris dam created by toppled acacia trees. The debris dam, however, also provided instream habitat and flow diversity for macroinvertebrates and algae. The alien acacias are therefore currently still providing ecosystem services in an altered river environment. A sufficient buffer (although not sustainable) to water quality enhancement between agricultural land and the river is still provided by the alien invaded riparian zone (Chapter 6).
- Riparian vegetation structure and composition are important variables in the riparian buffer zones and that together with vegetation attributes and land use management, the hydrological and nutrient cycles were influenced, which ultimately affected water quality and water quantity, which were also drivers of river ecological integrity.
- The outcomes from the main study objectives were combined in a conceptual framework flow-chain model for an interdisciplinary understanding of how river ecosystems and in particular riparian buffer zones, are able to provide ecosystem services and how anthropogenic influences are able to alter such services (Chapter 6). In the context of the current study, a number of ecosystem services were identified related to riparian buffer zones:
    - *Provisioning services*
        - Indigenous riparian vegetation ensured good river water quality.
        - Water temperature was regulated by a closed riparian canopy shading of indigenous and alien riparian vegetation.
        - Habitat was provided for aquatic and terrestrial biota. Habitat was also provided by the alien riparian vegetation especially instream where toppled trees provided substrate for algae and substrate and food sources for macroinvertebrates as well as refuge/cover to biota such as fish.
        - Allochthonous input was provided from riparian vegetation for biota to ensure energy flows and organic matter (food source). This service was altered by alien riparian vegetation.

- *Regulating services*
    - Intact ecological infrastructure provided erosion control and dissipation of flood waters.
    - Indigenous riparian zones provided efficient buffering and mediation of nutrients and sediment. The indigenous vegetation was capable of efficiently utilizing, conserving and recycling nutrients. The alien riparian buffer zone was less efficient at attenuating nutrients and was directly responsible for the erosion of river banks. The altered composition and structure of the alien riparian buffer zone still provided a regulatory function to water quality enhancement, although not sustainable in the long-term.
- *Cultural services*
    - The indigenous riparian vegetation provided aesthetics and recreation.
    - Both indigenous and alien vegetation provided areas for research.
    - The regulating and provisioning services in the upstream catchment allows for the provision of cultural services downstream in the catchment such as the recreational, aesthetics, research and ecotourism activities provided by the Ramsar Wilderness Lakes System and estuary.

7.2 Recommendations and implications for management

- Catchment scale assessment is crucial in the understanding of where impacts to water resources are originating from, for maximum riparian buffer effectiveness and by association provisioning and regulating ecosystem services such enhanced water quality. In the current study, natural land cover occurred at the cumulative impact site K4 and all the impact on water quality and aquatic ecology was originating from the catchment upstream. Both catchment scale and local scale (site-specific) land cover/use will influence the efficiency of riparian buffers to mitigate nutrient loads. As the conceptual framework flow-chain model in Chapter 6 illustrates, a catchment scale assessment will provide insight into the catchment heterogeneity such as geology, soils, vegetation or land cover and use and the river scale as well as the scale at which these various aspects become relevant. If riparian buffers are to be established the type of land cover/use (e.g. pasture vs row crops) will determine the type of riparian buffer (e.g. grass vs woody) or even if riparian buffer zones are an appropriate management tool.
- Site-specific conditions are therefore an important consideration when delineating riparian buffer widths for mitigation measures in different climatic regions. The use of secondary databases such as long-term rainfall records as presented in Chapter 3 provided an effective way of linking rainfall and land cover/use to water quality. The study sites were located on private land and the land owners were an important source of site information regarding rainfall records and past and current land management.

- The potential exists for establishing and maintaining riparian buffers by clearing invasive alien species and allowing for the establishment or planting of indigenous vegetation. Where nitrogen-fixing legumes were present the increased nutrient content in sediment should be considered, which could potentially provide a source of nutrients to rivers if river banks are left bare, where there is a potential for erosion to occur. If alien vegetation is replaced with indigenous vegetation on river banks, continued monitoring of this process will provide knowledge on the progress and the processes involved that determines successful vegetation re-establishment and ecosystem service maintenance.
- The vegetation structure (mixture of trees, shrubs and herbs) and composition (vegetation taxonomic type) of the riparian buffer zone play an important role in the buffering capabilities (vegetation control shown in Figure 6.1). The indigenous riparian buffer provided the maximum ecosystem services to water quality enhancement by reducing nutrient and sediment content provided to the river and thereby also enhanced the aquatic environment as illustrated in Figure 6.1. Indigenous vegetation to the particular region where riparian buffer zones are being established/restored should be utilised.
- Narrower riparian buffers (17 m average) with indigenous vegetation provided similar nutrient retention capabilities to that of the wider riparian buffers (25 m average) with indigenous vegetation. The same ecosystem service benefits were provided at both these sites, which included water quality enhancement, sufficient riparian habitat with good instream habitat to aquatic biota. A wider riparian buffer with predominantly alien acacias provided increased nutrient loads in runoff as did the smallest riparian zone (< 15 m) with predominantly alien acacias. Smaller buffer widths with indigenous species or a mixture of species are able to provide greater ecosystem services (nutrient and sediment attenuation) than wider alien invaded riparian buffers. However, the alien invaded riparian buffer zones still provided some regulatory form of ecosystem service to water quality evidenced by good river ecological integrity, although this will not be sustainable in the long-term.
- If overland flow and shallow through-flow is the main transport mechanism of runoff from agricultural fields, river water quality can be improved with adequate riparian buffers. If the main transport mechanisms of nutrient delivery to rivers are via groundwater flows that bypasses riparian vegetation roots thereby preventing nutrient uptake or denitrification, alternative management options for water quality improvement must be considered, such as controlling pollutant input at the source as the riparian buffer will be ineffective. This again illustrates the importance of site information. An adequate groundwater assessment is essential if no such information exists to determine groundwater flow paths, depth/levels and quality. The ERT surveys can provide a rapid and relatively cost-effective method to obtain information on groundwater depth. Boreholes or the more cost-effective piezometers can be used in groundwater quality monitoring depending on groundwater depth and site conditions. This information can be used in combination with riparian vegetation type and

characteristic assessments as different vegetation types will have different root systems and depths.

7.3 Future research

Based on the research findings and limitations of the study the following opportunities for further research were identified:

- Although some groundwater monitoring occurs in the two study catchments in the upper mountain catchment, no groundwater data were available in the vicinity of the study sites on the coastal plateau. Further research on surface-groundwater interactions (data on groundwater use, levels, flow paths and quality) are required related to riparian buffer zones, which are essential to understanding the geohydrology of these systems.
- It is essential to evaluate what the impact of different agricultural practices (e.g. pastures vs row crops) in different geographical areas are on groundwater quality, to what extent leaching of contaminants (e.g. nutrients or pesticides) occur within agricultural land and riparian zones and to what extent riparian vegetation root systems assist in the process of regulation. This will improve on the knowledge gain regarding the buffering potential of different riparian buffer zone types.
- A limitation in the current study was that macroinvertebrates were not identified to species level so conclusions could not be drawn regarding the response of macroinvertebrate biota to specific water quality variables, especially at the more impacted sites. Future studies should include a higher taxonomic resolution of macroinvertebrate biota so that knowledge could be gained regarding the effect of different riparian zone vegetation types on different macroinvertebrate species. This can inform river restoration efforts and knowledge would be gained on the autecology of macroinvertebrate species in South Africa.

REFERENCES

ABERNETHY, B. & RUTHERFURD, I. D. 2000. The effect of riparian tree roots on the mass-stability of riverbanks. *Earth Surface Processes and Landforms,* 25**,** 921-937.

ABERNETHY, B. & RUTHERFURD, I. D. 2001. The distribution and strength of riparian tree roots in relation to riverbank reinforcement. *Hydrological processes,* 15**,** 63-79.

AGUIAR, T. R., RASERA, K., PARRON, L. M., BRITO, A. G. & FERREIRA, M. T. 2015. Nutrient removal effectiveness by riparian buffer zones in rural temperate watersheds: The impact of no-till crops practices. *Agricultural Water Management,* 149**,** 74-80.

AL-HUMAID, A. I. & WARRAG, M. O. A. 1998. Allelopathic effects of mesquite (Prosopis juliflora) foliage on seed germination and seedling growth of bermudagrass (Cynodon dactylon). *Journal of Arid Environments,* 38**,** 237-243.

ALLANSON, B. R. & WHIFIELD, A. K. 1983. The limnology of the Touw River floodplain: Part I: Ecological structure in relation to planning and management. *South African National Scientific Programmes Report.* Pretoria: Council for Scientific and Industrial Research.

ALLEN, D. C., CARDINALE, B. J. & WYNN-THOMPSON, T. 2016. Plant biodiversity effects in reducing fluvial erosion are limited to low species richness. *Ecology,* 97**,** 17-24.

ANDERSON, M. J., GORLEY, R. N. & CLARKE, K. R. 2008. PERMANOVA+ for PRIMER: Guide to software and statistical methods. Plymouth, UK: PRIMER-E.

ANDERSON, R. J., BLEDSOE, B. P. & HESSION, W. C. 2004. Width of streams and rivers in response to vegetation, bank material and other factors. *Journal of the American Water Resources Association,* 40**,** 1159-1172.

APHA 2006. *Standard Methods for Examination of Water and Wastewater 20th edn,* Washington, DC., American Public Health Association.

ARC 2005. Land Type Survey Staff. 1972 – 2005. Land Types of South Africa: Digital map (1:250 000 scale) and soil inventory databases. Pretoria, South Africa: ARC-Institute for Soil, Climate and Water.

ATALAY, A. 2001. Variation in Phosphorus Sorption with soil particle size. *Soil and Sediment Contamination,* 10**,** 317-335.

BARBIER, E. B., HACKER, S. D., KENNEDY, C., KOCH, E. W., STIER, A. C. & SILLIMAN, B. R. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs,* 81**,** 169-193.

BEECHIE, T. J., SEAR, D. A., OLDEN, J. D., PESS, G. R., BUFFINGTON, J. M., MOIR, H., RONI, P. & POLLOCK, M. M. 2010. Process-Based Principles for Restoring River Ecosystems. *BioScience,* 60**,** 209-222.

BENDIX, J. & HUPP, C. R. 2000. Hydrological and geomorphological impacts on riparian plant communities. *Hydrological Processes,* 14**,** 2977-2990.

BENEDICT, M. A. & MCMAHON, E. T. 2002. Green infrastructure: smart conservation for the 21st century. *Renewable Resources Journal,* 20**,** 12-17.

BERGH, E. W. & COMPTON, J. S. 2015. A one-year post-fire record of macronutrient cycling in a mountain sandstone fynbos ecosystem, South Africa. *South African Journal of Botany,* 97**,** 48-58.

BIGGS, B. J. F. 2000. Eutrophication of streams and rivers: dissolved nutrients—chlorophyll relationships for benthic algae. *Journal of North American Benthological Society,* 19**,** 17-31.

BIGGS, B. J. F. & PRICE, G. M. 1987. A survey of filamentous algal proliferations in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research,* 21**,** 175-191.

BIGGS, B. J. F. & SMITH, R. A. 2002. Taxonomic Richness of Stream Benthic Algae: Effects of Flood Disturbance and Nutrients. *Limnology and Oceanography,* 47**,** 1175-1186.

BINNS, J. A., ILLGNER, P. M. & NEL, E. L. 2001. Water shortage, deforestation and development: South Africa's Working for Water Programme. *Land Degradation & Development,* 12**,** 341-355.

BLANCHÈ, C. 2002. *The use of riparian buffer zones for the attenuation of nitrate in agricultural landscapes.* Unpublished MSc thesis, University of Natal.

BLETTLER, M. C. M., AMSLER, M. L., EBERLE, E. G., SZUPIANY, R., LATOSINSKI, F. G., ABRIAL, E., OBERHOLSTER, P. J., ESPINOLA, L. A., PAIRA, A., POZA, A. & CAPITULO, R. A. 2016. Linking hydro-morphology with invertebrate ecology in diverse morphological units of a large river-floodplain system. *Water Resources Research,* 52**,** 9495-9510.

BOURNAUD, M., CELLOT, B., RICHOUX, P. & BERRAHOU, A. 1996. Macroinvertebrate Community Structure and Environmental Characteristics along a Large River: Congruity of Patterns for Identification to Species or Family. *Journal of the North American Benthological Society,* 15**,** 232-253.

BRAUN, B. M., PIRES, M. M., STENERT, C., MALTCHIK, L. & KOTZIAN, C. B. 2018. Effects of riparian vegetation width and substrate type on riffle beetle community structure. *Entomological Science,* 21**,** 66-75.

BRIERLEY, G., FRYIRS, K., OUTHET, D. & MASSEY, C. 2002. Application of the River Styles framework as a basis for river management in New South Wales, Australia. *Applied Geography,* 22**,** 91-122.

BRIERLEY, G. J. & FRYIRS, K. 2000. River Styles, a Geomorphic Approach to Catchment Characterization: Implications for River Rehabilitation in Bega Catchment, New South Wales, Australia. *Environmental Management,* 25**,** 661-679.

BROMILOW, C. 2010. *Problem plants and alien weeds of South Africa*, Briza.

BRUMMER, T. J., BYROM, A. E., SULLIVAN, J. J. & HULME, P. E. 2016. A Quantitative Framework to Derive Robust Characterization of Hydrological Gradients. *River Research and Applications,* 32**,** 1517-1529.

BRUTON, S. 2010. *The impacts of woody invasive alien plants on stream hydrogeomorphology in small headwater streams of Kwazulu-Natal.* Unpublished MSc, University of KwaZulu-Natal.

BUFFINGTON, J. M. & MONTGOMERY, D. R. 2013. 9.36 Geomorphic Classification of Rivers. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

BUNN, S. E., THOMS, M. C., HAMILTON, S. K. & CAPON, S. J. 2006. Flow variability in dryland rivers: Boom, bust and the bits in between. *Rivers Research and Applications,* 22**,** 179–186.

BUNTE, K. & ABT, S. R. 2001. Sampling surface and subsurface particle-size distributions in wadable gravel- and cobble-bed streams for analysis in sediment transport, hydraulics and streambed monitoring Colorado, USA: Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

BUNTE, K. A. A., S.R. 2001. Sampling frame for improving pebble count accuracy in coarse gravel-bed streams. *Journal of the American water resources association,* 37**,** 1005-1014.

CALLISTO, M., GOULART, M., MEDEIROS, A., MORENO, P. & ROSA, C. 2004. Diversity assessment of benthic macroinvertebrates, yeasts, and microbiological indicators along a longitudinal gradient in Serra do Cipó, Brazil. *Brazilian Journal of Biology,* 64**,** 743-755.

CAMPOREALE, C., PERUCCA, E., RIDOLFI, L. & GURNELL, A. M. 2013. Modeling the interactions between river morphodynamics and riparian vegetation. *Reviews of Geophysics,* 51**,** 379-414.

CAPE. 2011. *CapeNature fire management* [Online]. http://bgis.sanbi.org/cape/fire.asp. [Accessed July 2015].

CAPON, S. J., CHAMBERS, L. E., MAC NALLY, R., NAIMAN, R. J., DAVIES, P., MARSHALL, N., PITTOCK, J., REID, M., CAPON, T., DOUGLAS, M., CATFORD, J., BALDWIN, D. S., STEWARDSON, M., ROBERTS, J., PARSONS, M. & WILLIAMS, S. E. 2013. Riparian Ecosystems in the 21st Century: Hotspots for Climate Change Adaptation? *Ecosystems,* 16**,** 359-381.

CARPENTER, K. D., CZUBA, C., MAGIRL, C., MARINEAU, M., SOBIESZCZYK, S., CZUBA, J. & KEITH, M. 2012. Geomorphic Setting, Aquatic Habitat, and Water-Quality Conditions of the Molalla River, Oregon, 2009–10. Virginia: U.S. Department of the Interior and U.S. Geological Survey (USGS).

CAYUELA, C., LLORENS, P., SÁNCHEZ-COSTA, E., LEVIA, D. F. & LATRON, J. 2018. Effect of biotic and abiotic factors on inter- and intra-event variability in stemflow rates in oak and pine stands in a Mediterranean mountain area. *Journal of Hydrology,* 560**,** 396-406.

CHAMIER, J., SCHACHTSCHNEIDER, K., LE MAITRE, D. C., ASHTON, P. J. & VAN WILGEN, B. W. 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. *Water SA,* 38**,** 345-356.

CHASE, J. W., BENOY, G. A., HANN, S. W. R. & CULP, J. M. 2016. Small differences in riparian vegetation significantly reduce land use impacts on stream flow and water quality in small agricultural watersheds. *Journal of Soil and Water Conservation,* 71**,** 194-205.

CHATTOPADHYAY, S., RANI, L. A. & SANGEETHA, P. V. 2005. Water quality variations as linked to landuse pattern: a case study in Chalakudy river basin, Kerala. *Current Science,* 89**,** 2163-2189.

CHEN, N. & HONG, H. 2011. Nitrogen export by surface runoff from a small agricultural watershed in southeast China: seasonal pattern and primary mechanism. *Biogeochemistry,* 106**,** 311-321.

CHESSMAN, B., GROWNS, I., CURREY, J. & PLUNKETT-COLE, N. 1999. Predicting diatom communities at the genus level for the rapid biological assessment of rivers. *Freshwater Biology,* 41**,** 317-331.

CHÉTELAT, J., PICK, F. R., MORIN, A. & HAMILTON, P. B. 1999. Periphyton biomass and community composition in rivers of different nutrient status. *Canadian Journal of Fisheries and Aquatic Sciences,* 56**,** 560-569.

CHON, T. S., QU, X., CHO, W. S., HWANG, H. J., TANG, H., LIU, Y., CHOI, J. H., JUNG, M., CHUNG, B. S., LEE, H. Y. & CHUNG, Y. R. 2013. Evaluation of stream ecosystem health and species association based on multi-taxa (benthic macroinvertebrates, algae, and microorganisms) patterning with different levels of pollution. *Ecological Informatics,* 17**,** 58-72.

CHU, H.-J., LIU, C.-Y. & WANG, C.-K. 2013. Identifying the relationships between water quality and land cover changes in the tseng-wen reservoir watershed of Taiwan. *International journal of environmental research and public health,* 10**,** 478-489.

CLARKE, K. R. & GORLEY, R. N. 2006. Primer V6: user manual/tutorial. Plymouth, UK.

CLARKE, K. R. & WARWICK, R. M. 2001. Change in marine communities: an approach to statistical analysis and intepretation. 2nd Edition. Plymouth, UK.

CLASSIFICATION, S., WORKING & GROUP 1991. *Soil classification, A taxonomic system for South Africa.,* South Africa, Department of Agricultural Development.

COMPIN, A. & CÉRÉGHINO, R. 2007. Spatial patterns of macroinvertebrate functional feeding groups in streams in relation to physical variables and land-cover in Southwestern France. *Landscape Ecology,* 22**,** 1215-1225.

CONNOLLY, N. M., PEARSON, R. G., LOONG, D., MAUGHAN, M. & BRODIE, J. 2015. Water quality variation along streams with similar agricultural development but contrasting riparian vegetation. *Agriculture, Ecosystems & Environment,* 213**,** 11-20.

CORBIN, J. D. & D'ANTONIO, C. M. 2004. Effects of exotic species on soil nitrogen cycling: Implications for restoration. *Weed Technology,* 18**,** 1464-1467.

CORENBLIT, D., STEIGER, J., TABACCHI, E., GONZÁLEZ, E. & PLANTY-TABACCHI, A. M. 2014. Ecosystem engineers modulate exotic invasions in riparian plant communities by modifying hydrogeomorphic connectivity. *River Research and Applications,* 30**,** 45-59.

DAALEN, J. C. 1984. Distinguishing features of forest species on nutrient-poor soils in the Southern Cape. *Bothalia,* 15**,** 229-239.

DABROWSKI, J., BRUTON, S., DENT, M., GRAHAM, G., HILL, T., MURRAY, K., RIVERS-MOORE, N. & VAN DEVENTER, H. 2013. Linking land use to water quality for effective water resource and ecosystem management. Water Research Commission, WRC Report No. 984/1/13, Pretoria, South Africa.

DALLAS, H. F. 1997. A preliminary evaluation of aspects of SASS (South African Scoring System) for the rapid bioassessment of water quality in rivers, with particular reference to the incorporation of SASS in the National Biomonitoring Programme. *Southern African Journal of Aquatic Sciences,* 23**,** 79-94.

DALLAS, H. F. 2002. Spatial and temporal heterogeneity in lotic systems: Implications for defining reference conditions for macroinvertebrates. Water Research Commission, WRC Report No. KV 138/02, Pretoria, South Africa.

DANIELS, R. B. & GILLIAM, J. W. 1996. Sediment and Chemical Load Reduction by Grass and Riparian Filters. *Soil Science Society of America Journal,* 60**,** 246-251.

DE LA REY, P. A., TAYLOR, J. C., LAAS, A., VAN RENSBURG, L. & VOSLOO, A. 2004. Determining the possible application value of diatoms as indicators of general water quality: a comparison with SASS 5. *Water SA,* 30**,** 325-332.

DE LANGE, W. J. & MAHUMANI, B. K. 2013. Report on the agricultural economic significance of deteriorating water quality in the Hoekwil study area. Stellenbosch, South Africa: CSIR, CSIR/NRE/GES/IR/2013/0088/A.

DELA-CRUZ, J., PRITCHARD, T. I. M., GORDON, G. & AJANI, P. 2006. The use of periphytic diatoms as a means of assessing impacts of point source inorganic nutrient pollution in south-eastern Australia. *Freshwater Biology,* 51**,** 951-972.

DESCY, J. P. & COSTE, M. 1991. A test of methods for assessing water quality based on diatoms. *Verhandlungen des Internationalen Verein Limnologie,* 24**,** 2112-2116.

DEWSON, Z. S., JAMES, A. B. & DEATH, R. G. 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. *Journal of the North American Benthological Society,* 26**,** 401-415.

DICKENS, C. W. S. & 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.

DINDAROĞLU, T., REIS, M., AKAY, A. E. & TONGUÇ, F. 2015. Hydroecological approach for determining the width of riparian buffer zones for providing soil conservation and water quality. *International Journal of Environmental Science and Technology,* 12**,** 275-284.

DING, J., YUAN, J., LAN, F., QI, L., QIUZHI, P. & MUYI, K. 2015. Impacts of Land Use on Surface Water Quality in a Subtropical River Basin: A Case Study of the Dongjiang River Basin, Southeastern China. *Water,* 7**,** 4427-4445.

DODD, R. J. & SHARPLEY, A. N. 2015. Recognizing the role of soil organic phosphorus in soil fertility and water quality. *Resources, Conservation and Recycling,* 105**,** 282-293.

DODDS, W. K., SMITH, V. H. & LOHMAN, K. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. *Canadian Journal of Fisheries and Aquatic Sciences,* 59**,** 865-874.

DOLÉDEC, S., PHILLIPS, N., SCARSBROOK, M., RILEY, R. H. & TOWNSEND, C. R. 2006. Comparison of structural and functional approaches to determining landuse effects on grassland stream invertebrate communities. *Journal of the North American Benthological Society,* 25**,** 44-60.

DOLLAR, E. S. J., C.S. JAMES , C. S., ROGERS, K. H. & THOMS, M. C. 2007. A framework for interdisciplinary understanding of rivers as ecosystems. *Geomorphology,* 89**,** 147-162.

DOODY, D. G., WITHERS, P. J. A., DILS, R. M., MCDOWELL, R. W., SMITH, V., MCELARNEY, Y. R., DUNBAR, M. & DALY, D. 2016. Optimizing land use for the delivery of catchment ecosystem services. *Frontiers in Ecology and the Environment,* 14**,** 325-332.

DOSSKEY, M. G., VIDON, P., GURWICK, N. P., ALLAN, C. J., DUVAL, T. P. & LOWRANCE, R. 2010. The Role of Riparian Vegetation in Protecting and Improving Chemical Water Quality in Streams. *JAWRA Journal of the American Water Resources Association,* 46**,** 261-277.

DOUTERELO, I., PERONA, E. & MATEO, P. 2004. Use of cyanobacteria to assess water quality in running waters. *Environmental Pollution* 127**,** 377-384.

DU TOIT, J. C. O. & O'CONNOR, T. G. 2014. Changes in rainfall pattern in the eastern Karoo, South Africa, over the past 123 years. *Water SA,* 40**,** 453-460.

DUDENSKI, H. V. 2007. Outeniqua coast water situation study: Main Report Volume 2: Appendices. Pretoria, South Africa: DWAF, Report No: P WMA 16 / 000 / 00 / 0407.

DUNAWAY, D., SWANSON, S. R., WENDEL, J. & CLARY, W. 1994. The effect of herbaceous plant communities and soil textures on particle erosion of alluvial streambanks. *Geomorphology,* 9**,** 47-56.

DWA 2009. Resource Quality Services methods manual: Inorganic chemistry laboratory. Method 5001.5, 5001.7, and 5001.11. Department of Water Affairs, Pretoria, South Africa.

DWAF 1996a. South African Water Quality Guidelines (2nd Edition). Volume 1. Domestic Water Use. Department of Water Affairs and Forestry. Pretoria, South Africa.

DWAF 1996b. South African Water Quality Guidelines (2nd Edition). Volume 4. Agricultural Use: Irrigation. Pretoria, South Africa.

DWAF 1996c. South African Water Quality Guidelines (2nd Edition). Volume 7: Aquatic Ecosystems. Pretoria, South Africa.

DWAF 1999. *1:500 000 Hydrogeological map series of the Republic of South Africa. Oudtshoorn 3321*.

DWAF 2004. Gouritz Water Management Area: Internal Strategic Perspective *In:* PLANNING, N. W. R. (ed.). Ninham Shand (Pty) Ltd in association with Jakoet & Associates and Umvoto Africa, P WMA16/000/00/0304.

DWS. 2015a. *Hydrological services – surface water (data, dams, floods and flows)* [Online]. Pretoria, South Africa. Available: http://www.dwaf.gov.za/iwqs/wms/data/K_reg_WMS_nobor_map.htm.

DWS. 2015b. *Resource Quality Services water quality monitoring sites grouped by water management area* [Online]. Pretoria, South Africa. Available: www.dwaf.gov.za/iwqs/wms/data/WMS_pri_txt.asp.

DYE, P. & JARMAIN, C. 2004. Water use by black wattle (Acacia mearnsii): implications for the link between removal of invading trees and catchment streamflow response. *South African Journal of Science* 100**,** 40-44.

EADY, B. R., HILL, T. R. & RIVERS-MOORE, N. A. 2014. Shifts in aquatic macroinvertebrate community structure in response to perenniality, southern Cape, South Africa. *Journal of Freshwater Ecology,* 29**,** 475-490.

EHRENFELD, J. G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems,* 6**,** 503-523.

ELOSEGI, A., DÍEZ, J. & MUTZ, M. 2010. Effects of hydromorphological integrity on biodiversity and functioning of river ecosystems. *Hydrobiologia,* 657**,** 199-215.

EWART-SMITH, J. & KING, J. 2012. The Relationship Between Periphyton, Flow and Nutrient Status in South-Western Cape Foothill Rivers and the Implications for Management. Water Research Commission report no. 1676/1/12, Pretoria, South Africa.

EWART-SMITH, J. L. 2012. *The relationship between periphyton, flow and nutrients in foothill rivers of the south-western Cape, South Africa.* Unpublished PhD thesis, University of Cape Town.

FEMINELLA, J. W. & HAWKINS, C. P. 1995. Interactions between Stream Herbivores and Periphyton: A Quantitative Analysis of past Experiments. *Journal of the North American Benthological Society,* 14**,** 465-509.

FENNESSY, M. S. & CRONK, J. K. 1997. The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. *Critical Reviews in Environmental Science and Technology,* 27**,** 285-317.

FIERRO, P., BERTRÁN, C., TAPIA, J., HAUENSTEIN, E., PEÑA-CORTÉS, F., VERGARA, C., CERNA, C. & VARGAS-CHACOFF, L. 2017. Effects of local land-use on riparian vegetation, water quality, and the functional organization of macroinvertebrate assemblages. *Science of The Total Environment,* 609**,** 724-734.

FIGUEROA-NIEVES, D., ROYER, T. V. & DAVID, M. B. 2006. Controls on chlorophyll-a in nutrient-rich agricultural streams in Illinois, USA. *Hydrobiologia,* 568**,** 287-298.

FORRESTER, D. I., BAUHUS, J. & COWIE, A. L. 2005. Nutrient cycling in a mixed-species plantation of *Eucalyptus globulus* and *Acacia mearnsii*. *Canadian Journal of Forest Research,* 35**,** 2942-2950.

FÖRSTNER, U. & SOLOMONS, W. 1980. Trace metal analysis on polluted sediments. Part 1. Assessment of sources and intensities. *Environmental Technology Letters,* 1**,** 494-505.

FRISSELL, C. A., LISS, W. J., WARREN, C. E. & HURLEY, M. D. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management,* 10**,** 199-214.

GELDENHUYS, C. J. 1993. Floristic composition of the southern Cape forests with an annotated check-list. *South African Journal of Botany,* 59**,** 26-44.

GELDENHUYS, C. J. 1994. Bergwind Fires and the Location Pattern of Forest Patches in the Southern Cape Landscape, South Africa. *Journal of Biogeography,* 21**,** 49-62.

GERTENBACH, W. 2007. Dairy farming in South Africa-Where to now South Africa: Institute for Animal Production Western Cape Department of Agriculture.

GÖKÇE, D. 2016. Algae as an indicator of water quality. *In:* DHANASEKARAN, D. (ed.) *Algae - Organisms for Imminent Biotechnology.* InTech.

GOLDBLATT, P. & MANNING, J. 2000. *Cape plants: a conspectus of the Cape flora of South Africa. Strelitzia 9*, National Botanical Institute.

GONZALEZ, A., LAMBERT, A. & RICCIARDI, A. 2008. When does ecosystem engineering cause invasion and species replacement? *Oikos,* 117**,** 1247-1257.

GORDON, N. D., MCMAHON, T. A., FINLAYSON, B. L., GIPPEL, C. J. & NATHAN, R. J. 2004. *Stream hydrology: An introduction for ecologists* Chichester, John Wiley & Sons, Ltd.

GÖRGENS, A. H. M. & HUGHES, D. A. 1981. Hydrological investigations on the Wilderness catchments Prepared for the Cooperative research Programmes. *Touw River Floodplain Report. Part II. Hydrology and Hydraulics.* Rhodes University: Institute for Freshwater Studies.

GRABOWSKI, R. C. & GURNELL, A. M. 2016. Hydrogeomorphology—Ecology Interactions in River Systems. *River Research and Applications,* 32**,** 139-141.

GRAÇA, M. A., PINTO, P., CORTES, R., COIMBRA, N., OLIVEIRA, S., MORAIS, M. & MALO, J. 2004. Factors Affecting Macroinvertebrate Richness and Diversity in Portuguese Streams: a Two-Scale Analysis *International Review of Hydrobiology,* 89**,** 151-164.

GRAÇA, M. A. S. 2006. Allochthonous organic matter as a food resource for aquatic invertebrates in forested streams. *In:* RIVERA, A. C. (ed.) *Forests and dragonflies.* Sophia, Bulgaria: Pensoft.

GRI 2008. Land cover mapping for the Garden Route Initiative, Cape Action for People and Environment. Data supplied by Mr A. Brown then GRI coordinator, South African National Parks Board, Knysna.

GURNELL, A. 2014. Plants as river system engineers. *Earth Surface Processes and Landforms,* 39**,** 4-25.

GURNELL, A. M. 2013. 9.11 Wood in Fluvial Systems. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

HARDING, J. S., YOUNG, R. G., HAYES, J. W., SHEARER, K. A. & STARK, J. D. 1999. Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology,* 42**,** 345-357.

HARDING, W. R., ARCHIBALD, C. G. M. & TAYLOR, J. C. 2005. The relevance of diatoms for water quality assessment in South Africa: A position paper. *Water SA*, 31**,** 41-46.

HARDISON, E., C, O'DRISCOLL, M. A., DELOATCH, J. P., HOWARD, R. J. & BRINSON, M. M. 2009. Urban land use, channel incision, and water table decline along coastal plain streams, North Carolina. *Journal of American Water Resources Association,* 45**,** 1032-1045.

HARVEY, A. M. 2002. Effective timescales of coupling within fluvial systems. *Geomorphology,* 44**,** 175-201.

HASHOLT, B. Influence of erosion on the transport of suspended sediment and phosphorous. Sediment and Stream Water Quality in a Changing Environment:Trends and Explanation, 1991 Vienna IAHS Publ.

HAWES, E. & SMITH, M. 2005. Riparian Buffer Zones: Functions and Recommended Widths. Yale: Yale School of Forestry and Environmental Studies.

HAWKINS, C. P., HOGUE, J. N., DECKER, L. M. & FEMINELLA, J. W. 1997. Channel Morphology, Water Temperature, and Assemblage Structure of Stream Insects. *Journal of the North American Benthological Society,* 16**,** 728-749.

HELSEL, D. R. & HIRSCH, R. M. 1992. *Statistical Methods in Water Resources,* Amsterdam, Elsevier.

HENDERSON, L. 1998. Invasive alien woody plants of the southern and southwestern Cape region, South Africa. *Bothalia,* 28**,** 91-112.

HOFFMAN, M. T., CRAMER, M. D., GILLSON, L. & WALLACE, M. 2011. Pan evaporation and wind run decline in the Cape Floristic Region of South Africa (1974–2005): implications for vegetation responses to climate change. *Climatic Change* 109**,** 437-452.

HOFFMAN, M. T. & ROHDE, R. F. 2007. From pastoralism to tourism: The historical impact of changing land use practices in Namaqualand. *Journal of Arid Environments,* 70**,** 641-658.

HOLMES, P. M., RICHARDSON, D. M., ESLER, K. J., WITKOWSKI, E. T. F. & FOURIE, S. 2005. A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. *South African Journal of Science,* 101**,** 553-564.

HOLT, E. A. & MILLER, S. W. 2011. Bioindicators: using organisms to measure environmental impacts. *Nature Education Knowledge,* 3**,** 8.

HUPP, C. R. A. O., W.R. 1996. Riparian vegetation and fluvial geomorphic processes. *Geomorphology,* 14**,** 277-295.

IERODIACONOU, D., LAURENSON, L., LEBLANC, M., STAGNITTI, F., DUFF, G., SALZMAN, S. & VERSACE, V. 2005. The consequences of land use change on nutrient exports: a regional scale assessment in south-west Victoria, Australia. *Journal of Environmental Management,* 74**,** 305-316.

JACOBSON, R. B. 2013. 12.2 Riverine Habitat Dynamics. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

JOVANOVIC, N. Z., ISRAEL, S., TREDOUX, G., SOLTAU, L., LE MAITRE, D., RUSINGA, F., ROZANOV, A. & VAN DER MERWE, N. 2009. Nitrogen dynamics in land cleared of alien vegetation (*Acacia saligna*) and impacts on groundwater at Riverlands Nature Reserve (Western Cape, South Africa). *Water SA,* 35**,** 37-44.

JUNK, W. J., BAYLEY, P. B. & SPARKS, R. E. 1989. The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences,* 106**,** 110-127.

KAPP, J., FIJEN, A. P. M. & VAN ZYL, F. 1995. Towards a Water Management Strategy for an environmentally sensitive and popular tourist region. *Water Science and Technology,* 32**,** 245-254.

KENT, M. & COKER, P. 1992. *Vegetation description and analysis.,* Chichester, John Wiley.

KING, J. M. & SCHAEL, D. M. 2001. Assessing the ecological relevance of a spatially-nested geomorphological hierarchy for river management. Water Research Commission Report No. 754/1/01, Pretoria, South Africa.

KLATT, J. G., MALLARINO, A. P., DOWNING, J. A., KOPASKA, J. A. & WITTRY, D. J. 2003. Soil Phosphorus, Management Practices, and Their Relationship to Phosphorus Delivery in the Iowa Clear Lake Agricultural Watershed *Journal of Environmental Quality,* 32**,** 2140-2149.

KOMAREK, J. & ANAGNOSTIDIS, K. 1999. Cyanoprokaryota: Chroococcales. *In:* ETTL, H., H.H.GARTNER, AND D.MOLLENHAUER (ed.) *Susswasserflora von Mitteleuropa 19/1.* Gustav Fischer: Stutgart.

KOMAREK, J. & ANAGNOSTIDIS, K. 2005. Cyanoprokaryota: Oscillatoriales. *In:* BUDEL, B., G. GARTNER , L. KRIENITZ, AND M. SCHAGER (ed.) *Susswasserflora von Mitteleuropa 19/2.* Munchen: Elsevier.

KOTZE, H. & GELDENHUYS, C. J. 1992. Root Systems of Some Southern Cape Indigenous Forest Trees. *South African Forestry Journal,* 163**,** 21-25.

KRAAIJ, T., COWLING, R. M. & VAN WILGEN, B. W. 2011. Past approaches and future challenges to the management of fire and invasive alien plants in the new Garden Route National Park. *South African Journal of Science,* 107**,** 16-26.

KUCHAY, N. A. & BHAT, M. S. 2013. Automated drainage characterization of Dudganga watershed in western Himalayas. *European Scientific Journal,* 9**,** 126-138.

KUTKA, F. J. & RICHARDS, C. 1996. Relating Diatom Assemblage Structure to Stream Habitat Quality. *Journal of the North American Benthological Society,* 15**,** 469-480.

LAMERS, L. P. M., LOEB, B. R., ANTHEUNISSE, A. M., MILETTO, M., LUCASSEN, E. C. H. E. T., BOXMAN, A. W., SMOLDERS, A. J. P. & ROELOFS, J. G. M. 2006. Biogeochemical constraints on the ecological rehabilitation of wetland vegetation in river floodplains. *Hydrobiologia,* 565**,** 165-186.

LAMMERT, M. & ALLAN, J. D. 1999. Assessing Biotic Integrity of Streams: Effects of Scale in Measuring the Influence of Land Use/Cover and Habitat Structure on Fish and Macroinvertebrates. *Environmental Management,* 23**,** 257-270.

LAMONT, B. 1982. Mechanisms for Enhancing Nutrient Uptake in Plants, with Particular Reference to Mediterranean South Africa and Western Australia. *Botanical Review,* 48**,** 597-689.

LARGE, A. R. G. & GILVEAR, D. J. 2015. Using Google Earth, A Virtual-Globe Imaging Platform, for Ecosystem Services-Based River Assessment. *River Research and Applications,* 31**,** 406-421.

LE MAITRE, D., FORSYTH, G., DZIKITI, S. & GUSH, G. 2013. Estimates of the impacts of invasive alien plants on water flows in South Africa. Stellenbosch, South Africa, CSIR/NRE/ECO/ER/2013/0067/B: CSIR.

LE MAITRE, D., NEL, J., TALJAARD, S., VAN NIEKERK, L., GENTHE, B., OBERHOLSTER, P., SOMERSET, V., PETERSEN, C., SMITH-ADAO, L., MAHERRY, A., DE LANGE, W. & MAHUMANI, B. 2015a. Understanding the system dynamics of the Touws River Catchment and how they influence water quality of the rivers and lakes. South Africa: Unpublished report, CSIR.

LE MAITRE, D. C. 2000. Pines in cultivation: A global review. *In:* RICHARDSON, D. M. (ed.) *Ecology and biogeography of Pinus.* Cambridge, UK: Cambridge University Press.

LE MAITRE, D. C., FORSYTH, G. G., DZIKITI, S. & GUSH, M. B. 2016. Estimates of the impacts of invasive alien plants on water flows in South Africa. *Water SA,* 42**,** 659-672.

LE MAITRE, D. C., GUSH, M. B. & DZIKITI, S. 2015b. Impacts of invading alien plant species on water flows at stand and catchment scales. *AoB Plants,* 7**,** plv043; doi:10.1093/aobpla/plv043.

LE MAITRE, D. C., KOTZEE, I. M. & O'FARRELL, P. J. 2014. Impacts of land-cover change on the water flow regulation ecosystem service: Invasive alien plants, fire and their policy implications. *Land Use Policy* 36**,** 171- 181.

LE MAITRE, D. C., SCOTT, D. F. & COLVIN, C. 1999. Review of information on interactions between vegetation and groundwater. *Water SA,* 25**,** 137-152.

LE MAITRE, D. C., VAN WILGEN, B. W., GELDERBLOM, C. M., BAILEY, C., CHAPMAN, R. A. & NEL, J. A. 2002. Invasive alien trees and water resources in South Africa: case studies of the costs and benefits of management. *Forest Ecology and Management* 160 143-159.

LE MAITRE, D. C., VERSFELD, D. B. & CHAPMAN, R. A. 2004. The impact of invading alien plants on surface water resources in South Africa: A preliminary assessment. *Water SA,* 26**,** 397-408.

LEE, K.-H., ISENHART, T. M., SCHULTZ, R. C. & MICKELSON, S. K. 2000. Multispecies Riparian Buffers Trap Sediment and Nutrients during Rainfall Simulations. *Journal of Environmental Quality,* 29**,** 1200-1205.

LEE, K., ISENHART, T. & SCHULTZ, R. 2003. Sediment and nutrient removal in an established multi-species riparian buffer *Journal of Soil and Water Conservation*, 58**,** 1-8.

LEMLEY, D. A., TALJAARD, S., ADAMS, J. B. & STRYDOM, N. 2014. Nutrient characterisation of river inflow into the estuaries of the Gouritz Water Management Area, South Africa. *Water SA*, 40**,** 687-698.

LESCH, W. 1995. The development of guidelines for the design of streamwater quality monitoring strategies in the forestry industry. Water Research Commission, WRC Report No. 524/1/95, Pretoria, South Africa.

LI, L., ZHENG, B. & LIU, L. 2010. Biomonitoring and Bioindicators Used for River Ecosystems: Definitions, Approaches and Trends. *Procedia Environmental Sciences,* 2**,** 1510-1524.

LIN, L., LIN, H. & XU, Y. 2014. Characterisation of fracture network and groundwater preferential flow path in the Table Mountain Group (TMG) sandstones, South Africa. *Water SA*, 40**,** 263-272.

LITE, S. J., BAGSTAD, K. J. & STROMBERG, J. C. 2005. Riparian plant species richness along lateral and longitudinal gradients of water stress and flood disturbance, San Pedro River, Arizona, USA. *Journal of Arid Environments*, 63**,** 785-813.

LIU, X., ZHANG, X. & ZHANG, M. 2008. Major Factors Influencing the Efficacy of Vegetated Buffers on Sediment Trapping: A Review and Analysis. *Journal of Environmental Quality*, 37**,** 1667-1674.

LOKE, M. H. & BARKER, R. D. 1996. Rapid least-squares inversion of apparent resistivity pseudosections by a quasi-Newton method. *Geophysical Prospecting*, 44**,** 131-152.

LUBKE, R. & DE MOOR, I. 1998. *Field Guide to the Eastern and Southern Cape Coasts* Cape Town, University of Cape Town Press.

LYONS, J., THIMBLE, S. W. & PAINE, L. K. 2000. Grass vs trees: managing riparian areas to benefit streams of the central North America. *Journal of the American Water Resources Association*, 36**,** 920-930.

MACFARLANE, D. M. & BREDIN, I. P. 2017. Buffer zone guidelines for rivers, wetlands and estuaries. Part 1: Technical manual. Water Research Commission, WRC Report No TT 715-1-17, Pretoria, South Africa.

MACFARLANE, D. M., BREDIN, I. P., ADAMS, J. B., ZUNGU, M. M., BATE, G. C. & DICKENS, C. W. S. 2014. Preliminary guideline for the determination of buffer zones for rivers, wetlands and estuaries. Final Consolidated Report. Water Research Commission, WRC Report No TT 610/14, Pretoria, South Africa.

MAHERRY, A. M., HORAN, M. J. C., SMITH-ADAO, L. B., VAN DEVENTER, H., NEL, J. L., SCHULZE, R. E. & KUNZ, R. P. 2013. Delineating river network quinary catchments for South Africa and allocating associated daily hydrological information. Water Research Commission, WRC Report No. 2020/1/12, Pretoria, South Africa.

MAKAROVA, M. I., HAUMAIERB, L., ZECHB, W. & MALYSHEVAA, T. I. 2004. Organic phosphorus compounds in particle-size fractions of mountain soils in the northwestern Caucasus. *Geoderma*, 118**,** 101-114.

MANDER, M., JEWITT, G., DINI, J., GLENDAY, J., BLIGNAUT, J., HUGHES, C., MARAIS, C., MAZE, K., VAN DER WAAL, B. & MILLS, A. 2017. Modelling potential hydrological returns from investing in ecological infrastructure: Case studies from the Baviaanskloof-Tsitsikamma and uMngeni catchments, South Africa. *Ecosystem Services*, 27**,** 261-271.

MANDER, Ü., HAYAKAWA, Y. & KUUSEMETS, V. 2005. Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds. *Ecological Engineering*, 24**,** 421-432.

MANNING, J. & GOLDBLATT, P. 2012. *Plants of the greater Cape Floristic Region, Vol. 1: The core Cape flora. Strelitzia 29,* South Africa, South African National Biodiversity Institute.

MANTEL, S. K., HUGHES, D. A. & MULLER, N. W. 2010. Ecological impacts of small dams on South African rivers Part 1: drivers of change - water quantity and quality. *Water SA,* 36**,** 351-360.

MARCHANTE, E., KJØLLER, A., STRUWE, S. & FREITAS, H. 2008. Short- and long-term impacts of Acacia longifolia invasion on the belowground processes of a Mediterranean coastal dune ecosystem. *Applied soil ecology,* 40**,** 210-217.

MARKER, M. E. & HOLMES, P. J. 2010. The geomorphology of the Coastal Platform in the southern Cape. *South African Geographical Journal,* 92**,** 105-116.

MARKS, J. C. & LOWE, R. L. 1989. The independent and interactive effects of snail grazing and nutrient enrichment on structuring periphyton communities. *Hydrobiologia,* 185**,** 9-17.

MÁRQUEZ, J. A., CIBILS, L., PRINCIPE, R. E. & ALBARIÑO, R. J. 2015. Stream macroinvertebrate communities change with grassland afforestation in central Argentina. *Limnologica - Ecology and Management of Inland Waters,* 53**,** 17-25.

MCMILLAN, P. H. 1998. An Integrated Habitat Assessment System (IHAS v2), for the rapid biological assessment of rivers and streams. CSIR, Pretoria, South Africa.

MEA 2005. Millennium ecosystem assessment: Ecosystems and human well-being: Synthesis. Washington, DC: Island Press.

MEHDI, B., LEHNERA, B., GOMBAULTA, C., MICHAUDB, A., BEAUDINB, I., SOTTILEC, M.-F. & BLONDLOTC, A. 2015. Simulated impacts of climate change and agricultural land use change on surface water quality with and without adaptation management strategies. *Agriculture, Ecosystems and Environment,* 213**,** 47-60.

MERRITT, D. M. 2013. 9.14 Reciprocal Relations between Riparian Vegetation, Fluvial Landforms, and Channel Processes. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

MERRITT, R. W. & CUMMINS, K. W. 1996. *An Introduction to the Aquatic Insects of North America, third ed.,* Dubuque, IA, Kendal/Hunt.

MESA, L. M. 2014. Influence of riparian quality on macroinvertebrate assemblages in subtropical mountain streams. *Journal of Natural History,* 48**,** 1153-1167.

MILLER, R. B., FOX, G. A., PENN, C. J., WILSON, S., PARNELL, A., PURVIS, R. A. & CRISWELL, K. 2014. Estimating sediment and phosphorus loads from streambanks with and without riparian protection. *Agriculture, Ecosystems and Environment,* 189**,** 70-81.

MONAGHAN, R. M., WILCOCK, R. J., SMITH, L. C., TIKKISETTY, B., THORROLD, B. S. & COSTALL, D. 2007. Linkages between land management activities and water quality in an intensively farmed catchment in southern New Zealand. *Agriculture, Ecosystems & Environment,* 118**,** 211-222.

MOORE, M. T. & LOCKE, M. A. 2013. Effect of Storage Method and Associated Holding Time on Nitrogen and Phosphorus Concentrations in Surface Water Samples. *Bulletin of Environmental Contamination and Toxicology,* 91**,** 493-498.

MORAES, A. B., WILHELM, A. E., BOELTER, T., STENERT, C., SCHULZ, U. H. & MALTCHIK, L. 2014. Reduced riparian zone width compromises aquatic macroinvertebrate communities in streams of southern Brazil. *Environmental Monitoring and Assessment,* 186**,** 7063-7074.

MORRIS, T. L., ESLER, K. J., BARGER, N. N., JACOBS, S. M. & CRAMER, M. D. 2011. Ecophysiological traits associated with the competitive ability of invasive Australian acacias. *Diversity and Distributions,* 17**,** 898-910.

MUCINA, L. & RUTHERFORD, M. C. 2006. *The vegetation of South Africa, Lesotho and Swaziland. Strelitzia, 19.*, Pretoria, South Africa, South African National Biodiversity Institute.

MUNN, M. D., OSBORNE, L. L. & WILEY, M. J. 1989. Factors influencing periphyton growth in agricultural streams of central Illinois. *Hydrobiologia,* 174**,** 89-97.

MUNYIKA, S., KONGO, V. & KIMWAGA, R. 2014. River health assessment using macroinvertebrates and water quality parameters: A case of the Orange River in Namibia. *Physics and Chemistry of the Earth,* 76-78**,** 140-148.

NAIMAN, R. J., DECAMPS, H. & MCCLAIN, M. E. 2005. *Riparia-Ecology, Conservation and Management of Streamside Communities,* Academic Press, London.

NAKAMURA, F. & SWANSON, F. J. 1993. Effects of coarse woody debris on morphology and sediment storage of a mountain stream system in western Oregon. *Earth Surface Processes and Landforms,* 18**,** 43-61.

NEKTARIOS, P. A., ECONOMOU, G. & AVGOULAS, C. 2005. Allelopathic effects of Pinus halepensis needles on turfgrasses and biosensor plants. *Hortscience,* 40**,** 246-250.

NEL, J., ARCHIBALD, S., LE MAITRE, D., FORSYTH, G. & THERON, A. 2011a. Understanding the implications of global change for the insurance industry: The Eden Case Study. Final report for component 1: understanding the risk landscape. Stellenbosch: Natural Resources and the Environment, CSIR Report No. CSIR/NRE/ECOS/IR/2011/0063/B.

NEL, J., ARCHIBALD, S., LE MAITRE, D., FORSYTH, G. & THERON, A. 2011b. Understanding the implications of global change for the insurance industry: The Eden Case Study. Final report for component 1: understanding the risk landscape. Stellenbosch: Natural Resources and the Environment, CSIR, CSIR/NRE/ECOS/IR/2011/0063/B.

NEL, J., COLVIN, C., LE MAITRE, D. & SMITH, J. 2013. Strategic water source areas. Stellenbosch, South Africa: Natural Resources and the Environment, CSIR. CSIR/NRE/ECOS/ER/2013/0031/A.

NEL, J. L., DRIVER, A. L., STRYDOM, W. F., MAHERRY, A. M., PETERSEN, C., HILL, L., ROUX, D. J., NIENABER, S., VAN DEVENTER, H., SWARTZ, E. R. & SMITH-ADAO, L. B. 2011c. ATLAS of Freshwater Ecosystem Priority Areas in South Africa: Maps to support sustainable development of water resources Water Research Commission, WRC Report No.TT 500/11, Pretoria, South Africa.

NEL, J. L., LE MAITRE, D. C., NEL, D. C., REYERS, B., ARCHIBALD, S., VAN WILGEN, B. W., FORSYTH, G. G., THERON, A. K., O'FARRELL, P. J., KAHINDA, J.-M. M., ENGELBRECHT, F. A., KAPANGAZIWIRI, E., VAN NIEKERK, L. & BARWELL, L. 2014. Natural hazards in a changing world: a case for ecosystem-based management. *PLoS One 9, e95942*.

NEW, M., HEWITSON, B., STEPHENSON, D. B., TSIGA, A., KRUGER, A., MANHIQUE, A., GOMEZ, B., COELHO, C. A. S., MASISI, D. N., KULULANGA, E., MBAMBALALA, E., ADESINA, F., SALEH, H., KANYANGA, J., ADOSI, J., BULANE, L., FORTUNATA, L., MDOKA, M. L. & LAJOIE, R. 2006. Evidence of trends in daily climate extremes over southern and west Africa. *Journal of Geophysical Research,* 111**,** D14102, doi:101029/2005jd006289.

NOE, G. B. 2013. 12.21 Interactions among Hydrogeomorphology, Vegetation, and Nutrient Biogeochemistry in Floodplain Ecosystems. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

NOSETTO, M. D., JOBBÁGY, E. G., BRIZUELA, A. B. & JACKSON, R. B. 2012. The hydrologic consequences of land cover change in central Argentina. *Agriculture, Ecosystems & Environment,* 154**,** 2-11.

OBERHOLSTER, P. J. 2011. Using epilithic filamentous green algae communites as indicators of water quality in the headwaters of three South African river systems during high and medium flow periods. *In:* KATTEL, G. (ed.) *Zooplankton and Phytoplankton.* Nova Science Publishers, Inc.

OBERHOLSTER, P. J., BOTHA, A. M., HILL, L. & STRYDOM, W. F. 2017. River catchment responses to anthropogenic acidification in relationship with sewage effluent: An ecotoxicology screening application. *Chemosphere* 189**,** 407-417.

OBERHOLSTER, P. J. & DE KLERK, A. R. Aquatic ecosystems in the coal mining landscape of the Upper Olifants River, and the way forward. 21st Century challenges to the southern African coal Sector, 2014 Johannesburg. The Southern African Institute of Mining and Metallurgy, 237-251.

OBERHOLSTER, P. J., GENTHE, B., HOBBS, P., CHENG, P. H., DE KLERK, A. R. & BOTHA, A. M. 2013. An ecotoxicological screening tool to prioritise acid mine drainage impacted streams for future restoration. *Environmental Pollution,* 176**,** 244-253.

OBERHOLSTER, P. J., SOMERSET, V. S., TRUTER, J. C. & BOTHA, A.-M. 2016. The Interplay between Environmental Conditions and Filamentous Algae Mat Formation in Two Agricultural Influenced South African Rivers. *River Research and Applications,* 33**,** 388-402.

OSTERKAMP, W. R. & HUPP, C. R. 2010. Fluvial processes and vegetation: Glimpses of the past, the present and perhaps the future. *Geomorphology,* 116**,** 274-285.

PALMER, C. G. & O'KEEFFE, J. H. 1992. Feeding patterns of four macroinvertebrate taxa in the headwaters of the Buffalo River, eastern Cape. *Hydrobiologia,* 228**,** 157-173.

PARSONS, H. & GILVEAR, D. 2002. Valley floor landscape change following almost 100 years of flood embankment abandonment on a wandering gravel-bed river. *River Research and Applications,* 18**,** 461-479.

PARTRIDGE, T. C., DOLLAR, E. S. J., MOOLMAN, J. & DOLLAR, L. H. 2010. The geomorphic provinces of South Africa, Lesotho and Swaziland: A physiographic subdivision for earth and environmental scientists. *Transactions of the Royal Society of South Africa,* 65**,** 1 - 47.

PATIN, J., MOUCHE, E., RIBOLZI, O., CHAPLOT, V., SENGTAHEVANGHOUNG, O., LATSACHAK, K. O., SOULILEUTH, B. & VALENTIN, C. 2012. Analysis of runoff production at the plot scale during a long-term survey of a small agricultural catchment in Lao PDR. *Journal of Hydrology,* 426**,** 79-92.

PAUW, J. 2009. Challenges to sustainability in the Garden Route:Water, land and economy. Nelson Mandela Metropolitan University, George, South Africa.

PETERJOHN, W. T. & CORRELL, D. L. 1984. Nutrient Dynamics in an Agricultural Watershed: Observations on the Role of A Riparian Forest. *Ecology,* 65**,** 1466-1475.

PETERSEN, C. R., JOVANOVIC, N. Z., LE MAITRE, D. C. & GRENFELL, M. C. 2017. Effects of land use change on streamflow and stream water quality of a coastal catchment. *Water SA,* 43**,** 551-564.

PHILLIPS, J. F. V. 1963. The Forests of George, Knysna and the Zitzikama: A Brief History of Their Management: 1778-1939 South Africa: Department of Forestry.

PICKETT, S. T. A., KOLASA, J. & JONES, C. G. 1994. *Ecological Understanding: The Nature of Theory and the Theory of Nature,* Academic Press, San Diego.

POFF, N. L., ALLAN, J. D., BAIN, M. B., KARR, J. R., PRESTEGAARD, K. L., RICHTER, B. D., SPARKS, R. E. & STROMBERG, J. C. 1997. The natural flow regime: a paradigm for river conservation and restoration. *BioScience,* 47**,** 769-784.

POLLEN-BANKHEAD, N. & SIMON, A. 2010. Hydrologic and hydraulic effects of riparian root networks on streambank stability: Is mechanical root-reinforcement the whole story? *Geomorphology,* 116**,** 353-362.

POLLEN-BANKHEAD, N., SIMON, A. & THOMAS, R. E. 2013. 12.8 The Reinforcement of Soil by Roots: Recent Advances and Directions for Future Research. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

POLVI, L. E., WOHL, E. E. & MERRITT, D. M. 2011. Geomorphic and process domain controls on riparian zones in the Colorado Front Range. *Geomorphology,* 125**,** 504-516.

PORRA, R. J., THOMPSON, W. A. & KRIEDEMANN, P. E. 1989. Determination of accurate extinction coefficient and simultaneous equations for assaying chlorophylls a and b extracted with four different solvents: verification of the concentration of chlorophyll standards by atomic absorption spectrometry. *Biochimica et Biophysica Acta,* 975**,** 384-394.

PORTER, S. D., MUELLER, D. K., SPAHR, N. E., MUNN, M. D. & DUBROVSKY, N. M. 2008. Efficacy of algal metrics for assessing nutrient and organic enrichment in flowing waters. *Freshwater Biology,* 53**,** 1036-1054.

QU, X., ZHANG, H., ZHANG, M., LIU, M., YU, Y., XIE, Y. & PENG, W. 2016. Application of multiple biological indices for river health assessment in northeastern China. *Annales de Limnologie-International Journal of Limnology,* 52**,** 75-89.

RANALLI, A. J. & DONALD, L. M. 2010. The importance of the riparian zone and in-stream processes in nitrate attenuation in undisturbed and agricultural watersheds – A review of the scientific literature. *Journal of Hydrology* 389**,** 406-415.

REBELO, A. J. 2012. *An ecological and hydrological evaluation of the effects of restoration on ecosystem services in the Kromme River System, South Africa.* Unpublished MSc thesis, Stellenbosch University.

REBELO, A. J., LE MAITRE, D., ESLER, K. J. & COWLING, R. M. 2013. Are we destroying our insurance policy? The effects of alien invasion and subsequent restoration. *Landscape Ecology for Sustainable Environment and Culture.* Dordrecht: Springer.

REINECKE, K., BROWN, C., KLEYNHANS, M. & KIDD, M. 2013. Links Between Riparian Vegetation and Flow. Water Research Commission Report No. 1981/1/13, Pretoria, South Africa.

REINWARTH, B., FRANZ, S., BAADE, J., HABERZETTL, T., KASPER, T., DAUT, G., HELMSCHROT, J., KIRSTEN, K. L., QUICK, L. J., MEADOWS, M. E. & MÄUSBACHER, R. 2013. A 700-year record on the effects of climate and human impact on the southern Cape coast inferred from lake sediments of Eilandvlei, Wilderness Embayment, South Africa. *Geografiska Annaler: Series A, Physical Geography,* 95**,** 345-360.

RICHARDSON, D. M., HOLMES, P. M., ESLER, K. J., GALATOWITSCH, S. M., STROMBERG, J. C., KIRKMAN, S. P., PYŠEK, P. & HOBBS, R. J. 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and distributions,* 13**,** 126-139.

RICHARDSON, D. M., MACDONALD, I. A. W., HOFFMANN, J. H. & HENDERSON, L. 1997. Alien plant invasions. *In:* COWLING, R. M., RICHARDSON, D. M. & PIERCE, S. M. (eds.) *Vegetation of Southern Africa.* Cambridge: Cambridge University Press.

RILOV, G., MANT, R., LYONS, D., BULLERI, F., BENEDETTI-CECCHI, L., KOTTA, J., QUEIRÓS, A. M., CHATZINIKOLAOU, E., CROWE, T. & GUY-HAIM, T. 2012. How strong is the effect of invasive ecosystem engineers on the distribution patterns of local species, the local and regional biodiversity and ecosystem functions?. *Environmental Evidence,* 1**,** 1-8.

ROBERTS, W. M., STUTTER, M. I. & HAYGARTH, P. M. 2012. Phosphorus Retention and Remobilization in Vegetated Buff er Strips: A Review. *Journal of Environmental Quality,* 41**,** 389-399.

RODRÍGUEZ-BLANCO, M. L., TABOADA-CASTRO, M. M. & TABOADA-CASTRO, M. T. 2010. Sediment and phosphorus loss in runoff from an agroforestry catchment, NW Spain *Land Degradation and Development*, 21**,** 161-170.

ROSGEN, D. L. 1994. A classification of natural rivers. *CATENA,* 22**,** 169-199.

ROWNTREE, K. 1991. An assessment of the potential impact of alien vegetation on the geomorphology of river channels in South Africa. *Southern African Journal of Aquatic Sciences,* 17**,** 28-43.

ROWNTREE, K. M. & DOLLAR, E. S. J. 1999. Vegetation controls on channel stability in the Bell River, Eastern Cape, South Africa. *Earth Surface Processes and Landforms,* 24**,** 127-134.

ROWNTREE, K. M. & WADESON, R. A. 1999. A hierarchical geomorphological model for the classification of selected South African rivers. Water Research Commission Report No. 497/1/99, Pretoria, South Africa.

ROWNTREE, K. M., WADESON, R. A. & O'KEEFE, J. 2000. The development of a Geomorphological classification system for the longitudinal zonation of South African rivers. *South African Geographical Journal,* 82**,** 163-172.

ROYALL, D. 2013. 13.3 Land-Use Impacts on the Hydrogeomorphology of Small Watersheds. *In:* SHRODER, J. F. (ed.) *Treatise on Geomorphology.* San Diego: Academic Press.

RUSSELL, I. A. 1996. Fish abundance in the Wilderness and Swartvlei lake. *South African Journal of Zoology,* 31 1-9.

RUSSELL, I. A. 2013. Spatio-temporal variability of surface water quality parameters in a South African estuarine lake system. *African Journal of Aquatic Science* 38**,** 53-66.

RUSSELL, I. A. & KRAAIJ, T. 2008. Effects of cutting Phragmites australis along an inundation gradient, with implications for managing reed encroachment in a South African estuarine lake system. *Wetlands Ecology and Management,* 16**,** 383-393.

RUSSELL, I. A., RANDALL, R. M., COLE, N., KRAAIJ, T. & KRUGER, N. 2012. Garden Route National Park, Wilderness Coastal Section, State of Knowledge. South Africa: SANParks Scientific Services.

SALM 2017. Total Nitrogen: Determination of Nitrogen in water samples by thermo-catalytic digestion and chemo-detector:SALM 36. Stellenbosch: CSIR, SALM 36.

SALM 2018a. Chemical Oxygen Demand in water and wastewaters: SALM 28. Stellenbosch: CSIR, SALM 28.

SALM 2018b. Determination of Cations and Sulphate in water samples with Thermo ICAP6500: MALS 6.5-A. Stellenbosch: CSIR, MALS 6.5-A.

SALM 2018c. Determination of Nitrate and Nitrite in water samples by Flow Injection Analysis Colorimetry: SALM 7. Stellenbosch: CSIR, SALM 7.

SALM 2018d. Determination of Total Phosphorus: Water and Soil and Sediment Samples by Thermo iCAP ICP: SALM 49. Stellenbosch: CSIR, SALM 49.

SALM 2019. Determination of alkalinity in water and wastewater: SALM 5. Stellenbosch: CSIR, SALM 5.

SAMWAYS, M. J. & SHARRATT, N. J. 2010. Recovery of endemic dragonflies after removal of invasive alien trees. *Conservation Biology,* 24**,** 267-277.

SAMWAYS, M. J., SHARRATT, N. J. & SIMAIKA, J. P. 2011. Effect of alien riparian vegetation and its removal on a highly endemic river macroinvertebrate community. *Biological Invasions,* 13**,** 1305-1324.

SANBI 2014. A Framework for investing in ecological infrastructure in South Africa. Pretoria, South Africa: South African National Biodiversity Institute.

SANDERCOCK, P. J. & HOOKE, J. M. 2010. Assessment of vegetation effects on hydraulics and of feedbacks on plant survival and zonation in ephemeral channels. *Hydrological Processes,* 24**,** 695-713.

SANPARKS 2010. Draft Management Plan for the Garden Route National Park. South Africa: SANParks.

SANPARKS 2014. Garden Route National Park: State of Knowledge. South Africa: South African National Parks.

SANTOUL, F., CAYROU, J., MASTRORILLO, S. & CÉRÉGHINO, R. 2005. Spatial patterns of the biological traits of freshwater fish communities in south-west France. *Journal of Fish Biology*, 66**,** 301-314.

SASIKUMAR, K., VIJAYALAKSHMI, C. & PARTHIBAN, K. 2001. Allelopathic effects of four eucalyptus species on redgram (Cajanus cajan L.). *Journal of Tropical Agriculture,* 39.

SCHAEL, D. M. 2005. *Distributions of physical habitats and benthic macroinvertebrates in Western Cape headwater streams at multiple spatial and temporal scales.* Unpublished PhD thesis, University of Cape Town.

SCHAFER, G. N. 1991. Forest Land Types of the Southern Cape, Part 1. *Division of Forest and Science Technology* Pretoria, South Africa: CSIR.

SCHLACHER, T. A. & WOOLDRIDGE, T. H. 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.

SCHMUTZ, S. & SENDZIMIR, J. 2018. Riverine Ecosystem Management: Science for Governing Towards a Sustainable Future. Springer Open.

SCHNEIDER, S. C. & LINDSTRøM, E.-A. 2011. The periphyton index of trophic status PIT: a new eutrophication metric based on non-diatomaceous benthic algae in Nordic rivers. *Hydrobiologia*, 665**,** 143-155.

SCHULZE, R. 2000. Transcending scales of space and time in impact studies of climate and climate change on agrohydrological responses. *Agriculture, Ecosystems & Environment*, 82**,** 185-212.

SCHULZE, R. E. 1995. Hydrology and Agrohydrology: a text to accompany the ARC 3.00 agrohydrological modelling system Water Research Commission, WRC Report No. TT 69/9/95, Pretoria, South Africa.

SCHULZE, R. E., MAHARAJ, M., WARBURTON, M. L., GERS, C. J., HORAN, M. J. C., KUNZ, R. P. & CLARK, D. J. 2008. South African Atlas of Climatology and Agrohydrology Water Research Commission, WRC Report No. 1489/1/08, Pretoria, South Africa.

SCOTT, D. F. 1997. The contrasting effects of wildfire and clearfelling on the hydrology of a small catchment. *Hydrological Processes,* 11**,** 543-555.

SCOTT, D. F., LE MAITRE, D. C. & FAIRBANKS, D. H. K. 1998a. Forestry and streamflow reductions in South Africa: A reference system for assessing extent and distribution. *Water SA,* 24**,** 187 199.

SCOTT, D. F., VERSFELD, D. B. & LESCH, W. 1998b. Eosion and sediment yield in realtion to afforestation and fire in the mountains of the Western Cape Province, South Africa. *South African Geographical Journal,* 80**,** 52-59.

SIEBEN, E. J. J., KHUBEKA, S. P., SITHOLE, S., JOB, N. M. & KOTZE, D. C. 2018. The classification of wetlands: integration of top-down and bottom-up approaches and their significance for ecosystem service determination. *Wetlands Ecology and Management,* 26**,** 441-458.

SIMON, A. & COLLISON, A. J. C. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. *Earth Surface Processes and Landforms,* 27**,** 527-546.

SINGH, P., GUPTA, A. & SINGH, M. 2014. Hydrological inferences from watershed analysis for water resource management using remote sensing and GIS techniques. *The Egyptian Journal of Remote Sensing and Space Sciences,* 17**,** 111-121.

SLIVA, L. & WILLIAMS, D. 2001. Buffer Zone versus Whole Catchment Approaches to Studying Land Use Impact on River Water Quality. *Water Research,* 35**,** 3462-3472.

SMITH-ADAO, L. B. 2016. *Links between valley confinement, landforms and vegetation distribution in a semi-arid valley floor environment, Baviaanskloof, South Africa.* Unpublished PhD, Rhodes University.

SMITH, H. G., SHERIDAN, G. J., LANE, P. N. J., NYMAN, P. & HAYDON, S. 2011. Wildfire effects on water quality in forest catchments: A review with implications for water supply. *Journal of Hydrology,* 396**,** 170-192.

SMITH, J., SAMWAYS, M. J. & TAYLOR, S. 2007. Assessing riparian quality using two complementary sets of bioindicators. *Biodiversity and Conservation,* 16**,** 2695-2713.

SMITH, M. J., KAY, W. R., EDWARD, D. H. D., PAPAS, P. J., RICHARDSON, K. S. J., SIMPSON, J. C., PINDER, A. M., CALE, D. J., HORWITZ, P. H. J., DAVIS, J. A., YUNG, F. H., NORRIS, R. H. & HALSE, S. A. 1999. AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. *Freshwater Biology,* 41**,** 269-282.

SNIJMAN, D. A. 2013. Plants of the Greater Cape Floristic Region volume 2: The Extra Cape Flora.

SOLARI, L., VAN OORSCHOT, M., BELLETTI, B., HENDRIKS, D., RINALDI, M. & VARGAS-LUNA, A. 2016. Advances on Modelling Riparian Vegetation—Hydromorphology Interactions. *River Research and Applications,* 32**,** 164-178.

SOLTAU, L. & PEEK, C. 2015. Resistivity interpretation: Wilderness, Western Cape. Stellenbosch, South Africa: GEOSS - Geohydrological & Spatial Solutions International, GEOSS Report Number: 2015/03-15.

SOUZA, A. L. T. D., FONSECA, D. G., LIBÓRIO, R. A. & TANAKA, M. O. 2013. Influence of riparian vegetation and forest structure on the water quality of rural low-order streams in SE Brazil. *Forest Ecology and Management*, 298**,** 12-18.

SPINK, A., HILLMAN, M., FRYIRS, K., BRIERLEY, G. & LLOYD, K. 2010. Has river rehabilitation begun? Social perspectives from the Upper Hunter catchment, New South Wales, Australia. *Geoforum,* 41**,** 399-409.

STANLEY, E. H. & DOYLE, M. W. 2002. A Geomorphic Perspective on Nutrient Retention Following Dam Removal: Geomorphic models provide a means of predicting ecosystem responses to dam removal. *BioScience*, 52**,** 693-701.

STEIGER, J. & GURNELL, A. M. 2002a. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. *Geomorphology,* 49**,** 1-23.

STEIGER, J. & GURNELL, A. M. 2002b. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. *Geomorphology,* 49**,** 1-23.

STEIGER, J., TABACCHI, E., DUFOUR, S., CORENBLIT, D. & PEIRY, J. L. 2005. Hydrogeomorphic processes affecting riparian habitat within alluvial channel–floodplain river systems: a review for the temperate zone. *River Research and Applications,* 21**,** 719-737.

STEVENSON, R. J. & BAHLS, L. L. 1999. Periphyton protocols. *In:* BARBOUR, M. T., GERRITREN, J., SNYDER, B.D., AND STRIBLING, J.B. (ed.) *Rapid Bioassessment*

Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish 2ndEPA 841-B-99-002 edn. United States Environmental Protection Agency, Washington, DC., USA.

STOCK, W. D., WIENAND, K. T. & BAKER, A. C. 1995. Impacts of invading N2-fixing Acacia species on patterns of nutrient cycling in two Cape ecosystems: evidence from soil incubation studies and 15N natural abundance values. *Oecologia,* 101**,** 375-382.

STROMBERG, J. C., CHEW, M. K., NAGLER, P. L. & GLENN, E. P. 2009. Changing Perceptions of Change: The Role of Scientists in *Tamarix* and River Management. *Restoration Ecology,* 17**,** 177-186.

SUMMERS, J. S. 2008. Assessment of filamentous algae in the Greenbrier River and other West Virginia streams West Virginia Department of Environmental Protection, West Virginia, USA.

SWEENEY, B. W., BOTT, T. L., JACKSON, J. K., KAPLAN, L. A., NEWBOLD, J. D., STANDLEY, L. J., HESSION, W. C. & HORWITZ, R. J. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America,* 101**,** 14132-14137.

TABACCHI, E., CORRELL, D. L., HAUER, R., PINAY, G., PLANTY-TABACCHI, A. M. & WISSMAR, R. C. 1998. Development, maintenance and role of riparian vegetation in the river landscape. *Freshwater Biology,* 40**,** 497-516.

TABACCHI, E., LAMBS, L., GUILLOY, H., PLANTY-TABACCHI, A.-M. M., E. & DECAMPS, H. 2000. Impacts of riparian vegetation on hydrological processes. *Hydrological Processes,* 14**,** 2959-2976.

TALJAARD, S., VAN NIEKERK, L. & LEMLEY, D. 2018. Nutrient Chemistry of the Wilderness Lake System, South Africa. *Water SA,* 44**,** 65-74.

TANAKA, M. O., SOUZA, A. L. T., MOSCHINI, L. E. & OLIVEIRA, A. D. K. 2016a. Influence of watershed land use and riparian characteristics on biological indicators of stream water quality in southeastern Brazil. *Agriculture, Ecosystems & Environment,* 216**,** 333-339.

TANAKA, M. O., SOUZA, A. L. T. D., MOSCHINI, L. E. & OLIVEIRA, A. K. D. 2016b. Influence of watershed land use and riparian characteristics on biological indicators of stream water quality in southeastern Brazil. *Agriculture, Ecosystems & Environment,* 216**,** 333-339.

TAYLOR, J. C., HARDING, W. R. & ARCHIBALD, C. G. M. 2007a. An illustrated guide to some comman diatom species from South Africa. Water Research Commission Report No.TT 282/07, Pretoria, South Africa.

TAYLOR, J. C., PRYGIEL, J., VOSLOO, A., DE LA REY, P. A. & VAN RENSBURG, L. 2007b. Can diatom-based pollution indices be used for biomonitoring in South Africa? A case study of the Crocodile West and Marico water management area. *Hydrobiologia* 592**,** 455-464.

TELFORD, W. M., GELDART, L. P. & SHERIFF, R. E. 1990. *Applied geophysics: Second edition* United Kingdom, Cambridge University Press.

THOMS, M. C. & PARSONS, M. Eco-geomorphology: an interdisciplinary approach to river science. *In:* DYER, F. J., THOMS, M. C. & OLLEY, J. M., eds. The Structure, Function and Management Implications of Fluvial Sedimentary Systems, 2002 Oxfordshire, UK. International Association of Hydrological Sciences, 113-119.

THOMS, M. C. & PARSONS, M. E. 2003. Identifying spatial and temporal patterns in the character of the Condamine Balonne River, Australia, using multivariate statistics. *River Research and Applications* 19**,** 443-458.

THORP, J. H. & DELONG, M. D. 2002. Dominance of autochthonous autotrophic carbon in food webs of heterotrophic rivers? *Oikos,* 96**,** 543-550.

THORP, J. H., FLOTEMERSCH, J. E., DELONG, M. D., CASPER, A. F., THOMS, M. C., BALLANTYNE, F., WILLIAMS, B. S., O'NEILL, B. J. & HAASE, S. C. 2010. Linking Ecosystem Services, Rehabilitation, and River Hydrogeomorphology. *BioScience,* 60**,** 67-74.

THORP, J. H., THOMS, M. C. & DELONG, M. D. 2006. The riverine ecosystem synthesis: biocomplexity in river networks across space and time. *River Research and Applications,* 22**,** 123-147.

TICKNER, D. P., ANGOLD, P. G., GURNELL, A. M. & MOUNTFORD, J. O. 2001. Riparian plant invasions: hydrogeomorphological control and ecological impacts. *Progress in Physical Geography,* 25**,** 22-52.

TONG, S. T. Y. & CHEN, W. 2002. Modeling the relationship between land use and surface water quality. *Journal of Environmental Management,* 66**,** 377-393.

TYE, D. R. C. & DRAKE, D. C. 2012. An exotic Australian Acacia fixes more N than a coexisting indigenous Acacia in a South African riparian zone. *Plant Ecology,* 213**,** 251–257.

UNEP 2015. Global Synthesis Report of the Project for Ecosystem Services. UNEP, Ecosystem Services Economics Unit, Division of Environmental Policy Implementation

VAN COLLER, A. L., ROGERS, K. H. & HERITAGE, G. L. 1997. Linking riparian vegetation types and fluvial geomorphology along the Sabie River within the Kruger National Park, South Africa. *Afrrican Journal of Ecology,* 35**,** 192-212.

VAN COLLER, A. L., ROGERS, K. H. & HERITAGE, G. L. 2000. Riparian vegetation-environment relationships: complimentarity of gradients versus patch hierarchy approaches. *Journal of Vegetation Science* 11**,** 337-350.

VAN DEN BERG, E. C., PLARRE, C., VAN DEN BERG, H. M. & THOMPSON, M. W. 2008. The South African National Land Cover 2000. Pretoria.

VAN DER COLFF, D., DREYER, L. L., VALENTINE, A. & ROETS, F. 2017. Comparison of nutrient cycling abilities between the invasive Acacia mearnsii and the native Virgilia divaricata trees growing sympatrically in forest margins in South Africa. *South African Journal of Botany,* 111**,** 358-364.

VAN NIEKERK, A. W., HERITAGE, G. L. & MOON, B. P. 1995. River classification for management: The geomorphology of the Sabie River in the Eastern Transvaal. *South African Geographical Journal,* 77**,** 68-76.

VAN VUUREN, S., TAYLOR, J. C., GERBER, A. & VAN GINKEL, C. 2006. Easy Identification of the Most Common Freshwater Algae Department of Water Affairs and Forestry, Pretoria, South Africa.

VAN WILGEN, B. W., BOND, W. J. & RICHARDSON, D. M. 1992. The ecology of fynbos: Nutrients, fire and diversity. *In:* COWLING, R. M. (ed.) *Ecosystem Management.* Cape Town: Oxford University Press.

VAN WILGEN, B. W., FORSYTH, G. G., DE KLERK, H., DAS, S., KHULUSE, S. & SCHMITZ, P. 2010. Fire management in Mediterranean-climate shrublands: a case study from the Cape fynbos, South Africa. *Journal of Applied Ecology,* 47**,** 631-638.

VANNOTE, R. L., MINSHALL, G. W., CUMMINS, K. W., SEDELL, J. R. & CUSHING, C. E. 1980. 'The river continuum concept'. *Canadian Journal of Fisheries and Aquatic Sciences,* 37**,** 130-137.

VIDAL-ABARCA, M. R., SANTOS-MARTÍN, F., MARTÍN-LÓPEZ, B., SÁNCHEZ-MONTOYA, M. M. & SUÁREZ ALONSO, M. L. 2016. Exploring the Capacity of Water Framework Directive Indices to Assess Ecosystem Services in Fluvial and Riparian Systems: Towards a Second Implementation Phase. *Environmental Management,* 57**,** 1139-1152.

VLOK, J. H. J., EUSTON-BROWN, D. I. & WOLF, T. 2008a. A vegetation map for the Garden Route Initiative. Unpublished 1:50 000 maps and report. *CAPE FSP task team.* CAPE.

VLOK, J. H. J., EUSTON-BROWN, D. I. & WOLF, T. 2008b. A vegetation map for the Garden Route Initiative. Unpublished 1:50 000 maps and report. *CAPE FSP task team.* CAPE.

VLOK, J. H. J., EUSTON-BROWN, D. I. W. & WOLF, T. 2008c. Unpublished 1:50 000 maps and report supported by CAPE FSP task team. South Africa: CAPE FSP.

VOUGHT, L. B. M., DAHL, J., PEDERSEN, C. L. & LACOURSIERE, J. O. 1994. Nutrient retention in riparian ecotones. *Ambio,* 23**,** 342-348.

VOUGHT, L. M., KULLBERG, A. & PETERSEN, R. C. 1998. Effect of riparian structure, temperature and channel morphometry on detritus processing in channelized and natural woodland streams in southern Sweden. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 8**,** 273-285.

WALSH, G. & WEPENER, V. 2009. The influence of land use on water quality and diatom community structures in urban and agriculturally stressed rivers. *Water SA,* 35**,** 579-594.

WARD, J. V. 1989. The four-dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society,* 16**,** 531-544.

WARD, J. V. & TOCKNER, K. 2001. Biodiversity: towards a unifying theme for river ecology. *Freshwater Biology,* 46**,** 807-820.

WEHR, J. D. & SHEATH, R. G. 2003. Freshwater habitats of algae. *In:* WEHR, J. D., SHEATH, R. G. & KOCIOLEK, P. (eds.) *Freshwater Algae of North America: Ecology and Classification.* San Diego, CA: Academic Press.

WENGER, S. 1999. A review of the scientific literature on riparian buffer width, extent and vegetation. Georgia: Institute of Ecology, University of Georgia.

WITKOWSKI, E. T. F. 1991. Effects of invasive alien acacias on nutrient cycling in the coastal lowlands of the Cape fynbos. *Journal of Applied Ecology*, 1-15.

WLSDF 2015. Local spatial development framework for Wilderness, the Lakes, Hoekwil and the adjacent agricultural areas. George, South Africa.: Department of Human Settlements, Land Affairs and Planning.

WOLMAN, M. G. 1954. A method of sampling coarse river-bed material *EOS, Transactions of the American Geophysical Union* 35**,** 951-956.

WURTS, W. & DURBOROW, R. 1992. Interaction of carbon dioxide, pH, alkalinity and hardness in fish ponds. United States Department of Agriculture: Southern Regional Aquaculture Centre (SRAC), Publication no. 464.

WYNN, T. & MOSTAGHIMI, S. 2006. The effects of vegetation and soil type on streambank erosion, southwestern Virginia, USA. *JAWRA Journal of the American Water Resources Association,* 42**,** 69-82.

YU, B., JOO, M. & CAROLL, C. Land use and water quality trends of the Fitzroy River, Australia. *In:* AL., B. A. E., ed. Understanding Freshwater Quality Problems in a Changing World Proceedings of H04, IAHS-IAPSO-IASPEI Assembly, 2013. International Association for Hydrological Sciences 1-8.

ZHANG, W. 2010. Pool Effects on Longitudinal Dispersion in Streams and Rivers. *Journal of Water Resource and Protection,* 02**,** 960-971.

ZHANG, X. & YANG, F. 2004. *Rclimdex (1.0): User manual,* Ontario, Canada, Climate Research Branch of Meteorological Service.

APPENDICES

Appendix 2.1 Grain sizes and sorting at all sites. RB-right bank, LB-left bank, U-upper, middle, L-lower plots

Site	Percentile (mm)				%						
	D_{16}	D_{50}	D_{84}	D_{95}	Sand	Gravel	Mud/ silt	Cobble	Boulder	Sorting	Geomorphic feature
1.1	0.27	0.14	0.076	0.006	95.4	1.5	3.1	0	0	0.28	LBU
	0.32	0.17	0.095	0.064	93.4	5.1	1.5	0	0	0.30	LBM
	0.28	0.18	0.11	0.066	96.8	2.2	1	0	0	0.39	LBL
	52	**170**	**230**	**250**	**0**	**19**	**0**	**81**	**0**	**4.42**	**Instream**
	0.28	0.18	0.59	0.066	99.7	0	0.3	0	0	2.11	RBL
	0.61	0.23	0.13	0.071	88.2	11.5	0.3	0	0	0.21	RBM
	0.54	0.25	0.092	0.055	96.7	1.6	1.8	0	0	0.17	RBU
1.2	0.3	0.15	0.075	0.0058	94.9	2.1	3	0	0	0.25	LBU
	0.85	0.24	0.099	0.036	88.4	9.7	1.9	0	0	0.12	LBM
	0.93	0.81	0.6	0.28	99.2	0	0.8	0	0	0.65	LBL
	11	**64**	**370**	**460**	**0**	**50**	**0**	**19**	**31**	**33.64**	**Instream**
	0.28	0.8	0.59	0.28	99.5	0	0.5	0	0	2.11	RBL
	1.1	0.3	0.18	0.12	88.2	11.5	0.3	0	0	0.16	RBM
	0.54	0.2	0.093	0.057	96.7	1.6	1.8	0	0	0.17	RBU
1.3	0.31	0.16	0.089	0.025	96.1	2.1	1.8	0	0	0.29	LBU
	0.59	0.19	0.093	0.024	98	0	2	0	0	0.16	LBM
	0.25	0.16	0.083	0.0058	92	7	1	0	0	0.33	LBL
	9.4	**19**	**100**	**300**	**0**	**79**	**0**	**11**	**10**	**10.64**	**Instream**
	0.28	0.18	0.083	0.0058	96	3.3	0.7	0	0	0.30	RBL
	0.65	0.27	0.15	0.081	99.3	0	0.7	0	0	0.23	RBM
	0.79	0.27	0.15	0.088	96.1	3.7	0.3	0	0	0.19	RBU
2.1	0.28	0.16	0.085	0.034	96.7	1.2	2.1	0	0	0.30	LBU
	0.28	0.16	0.085	0.034	97.2	1	1.8	0	0	0.30	LBM
	0.59	0.24	0.16	0.1	96	3.8	0.2	0	0	0.27	LBL
	10	**32**	**180**	**230**	**0**	**65**	**0**	**35**	**0**	**18.00**	**Instream**

Site	D16	D50	D84	D95	Sand	Gravel	Mud/ silt	Cobble	Boulder	Sorting	Geomorphic feature
		Percentile (mm)					**%**				
	0.54	0.25	0.16	0.1	97.6	1.9	0.6	0	0	0.30	RBL
----	0.56	0.15	0.064	0.0077	97.3	2.7	0	0	0	0.11	RBM
	0.25	0.18	0.11	0.066	96.1	2.7	1.2	0	0	0.44	RBU
2.2	0.21	0.14	0.079	0.014	96.5	1.4	2.1	0	0	0.38	LBU
	0.27	0.18	0.12	0.069	97.6	1.4	1.1	0	0	0.44	LBM
	0.96	0.35	0.21	0.14	90.5	9.3	0.1	0	0	0.22	LBL
	31	**61**	**130**	**210**	**0**	**52**	**0**	**48**	**0**	**4.19**	**Instream**
	0.63	0.27	0.19	0.13	99.8	0	0.2	0	0	0.30	RBL
	0.24	0.15	0.096	0.064	97.5	1	1.5	0	0	0.40	RBM
	0.28	0.14	0.075	0.0071	93.1	4.2	2.8	0	0	0.27	RBU
2.3	0.25	0.18	0.11	0.066	90.8	3.7	5.5	0	0	0.44	LBU
	0.23	0.15	0.093	0.045	97.1	1.4	1.6	0	0	0.40	LBM
	0.45	0.21	0.13	0.069	96.1	2.9	1	0	0	0.29	LBL
	44	**86**	**190**	**300**	**0**	**35**	**0**	**56**	**9**	**4.32**	**Instream**
	1.4	0.55	0.23	0.18	74.3	25.7	0	0	0	0.16	RBL
	0.23	0.14	0.074	0.0077	96.4	1.1	2.5	0	0	0.32	RBM
	0.18	0.12	0.065	0.0022	91.6	2.8	5.6	0	0	0.36	RBU
3.1	0.56	0.25	0.14	0.079	96	2	2	0	0	0.25	LBU
	0.27	0.17	0.083	0.019	93.8	4.5	1.7	0	0	0.31	LBM
	0.51	0.22	0.13	0.066	98.5	0	1.5	0	0	0.25	LBL
	13	**32**	**99**	**140**	**0**	**65**	**0**	**35**	**0**	**7.62**	**Instream**
	0.57	0.29	0.18	0.096	99.2	0	0.8	0	0	0.32	RBL
	0.89	0.28	0.23	0.11	90.7	8.9	0.4	0	0	0.26	RBM
	0.27	0.17	0.083	0.0097	94.7	2.7	2.7	0	0	0.31	RBU
3.2	0.24	0.14	0.084	0.023	96.6	1.5	1.9	0	0	0.35	LBU
	0.3	0.2	0.096	0.021	96.1	1.8	2.1	0	0	0.32	LBM
	0.79	0.28	0.16	0.083	99.3	0	0.7	0	0	0.20	LBL
	11	**44**	**110**	**180**	**0**	**63**	**0**	**37**	**0**	**10.00**	**Instream**
	0.31	0.14	0.072	0.0044	92.9	6.8	2	0	0	0.23	RBL
	0.78	0.21	0.075	0.0037	86.6	9.7	3.7	0	0	0.10	RBM
	0.43	0.16	0.067	0.0022	93.6	3.1	3.3	0	0	0.16	RBU

Site	Percentile (mm)				%					Sorting	Geomorphic feature
	D_{16}	D_{50}	D_{84}	D_{95}	Sand	Gravel	Mud/ silt	Cobble	Boulder		
3.3	0.28	0.18	0.1	0.057	88	4	8	0	0	0.36	LBU
	1	0.26	0.14	0.08	90.1	9.7	0.2	0	0	0.14	LBM
	1	0.24	0.15	0.097	90	10	0	0	0	0.15	LBL
	10	**42**	**110**	**160**	**0**	**68**	**31**	**1**	**1**	**11.00**	**Instream**
	1.3	0.33	0.19	0.1	81	18.7	0.4	0	0	0.15	RBL
	0.51	0.21	0.13	0.07	99.1	0	0.9	0	0	0.25	RBM
	0.56	0.2	0.028	0.07	97.2	0.9	1.9	0	0	0.05	RBU
4.1	0.65	0.21	0.12	0.066	97.2	1.4	1.4	0	0	0.18	LBU
	0.3	0.15	0.073	0.0059	92.4	4.6	3	0	0	0.24	LBM
	1.1	0.61	0.22	0.11	99.4	0	0.6	0	0	0.20	LBL
	9.7	**17**	**75**	**270**	**0**	**82**	**0**	**6**	**2**	**7.73**	**Instream**
	0.7	0.27	0.23	0.069	98.5	0.5	1	0	0	0.33	RBL
	0.56	0.22	0.11	0.055	94.5	3.3	2.2	0	0	0.20	RBM
4.2	0.65	0.15	0.072	0.0045	90.3	6.3	3.4	0	0	0.11	LBU
	0.72	0.22	0.12	0.063	91.3	6.7	2.1	0	0	0.17	LBM
	0.3	0.6	0.22	0.1	42	56.7	1.3	0	0	0.73	LBL
	19	**44**	**130**	**210**	**0**	**67**	**0**	**33**	**0**	**6.84**	**Instream**
	0.65	0.15	0.072	0.0045	83.9	0	16.1	0	0	0.11	RBL
	0.7	0.27	0.23	0.069	97.5	2.1	0.5	0	0	0.33	RBM
4.3	0.82	0.22	0.083	0.0042	96.1	0	3.9	0	0	0.10	LBU
	0.98	0.44	0.19	0.1	99.5	0	0.5	0	0	0.19	LBM
	0.23	0.099	0.12	0.005	80.2	0	19.8	0	0	0.52	LBL
	8.9	**26**	**74**	**120**	**0**	**79**	**0**	**21**	**0**	**8.31**	**Instream**

Appendix 2.2 Sediment chemistry for all river bank site plots (all sampled periods). U-upper, M-middle, L-lower plots, L=Left bank, R=Right bank

Site code	TN %	TP (mg/kg)	PO_4^{-3} (mg/kg)	PO_4-P (mg/kg)	Organic carbon	pH (KCI)
1-1-RBL	0.24	193.90	255.57	83.52	4.46	4.60
1-2-RBL	0.34	112.59	337.61	110.33	7.01	4.33
1-3-RBL	0.21	64.46	128.09	41.86	1.49	4.97
1-1-LBL	0.12	110.24	247.43	80.86	4.39	4.60
1-2-LBL	0.73	383.78	637.02	208.17	16.45	4.33
1-3-LBL	0.15	84.76	78.98	25.81	3.66	4.97
2-1-RBL	0.19	83.36	264.80	86.53	5.75	4.13
2-2-RBL	0.39	91.29	214.00	69.93	4.68	4.20
2-3-RBL	0.18	88.51	262.44	85.76	3.84	3.97
2-1-LBL	0.32	95.50	152.25	49.75	5.81	4.60
2-2-LBL	0.53	121.49	179.27	58.58	7.93	4.33
2-3-LBL	*0.23*	*116.50*	*309.84*	101.25	6.73	4.93
3-1-RBL	1.15	251.86	570.05	186.29	13.71	4.07
3-2-RBL	0.63	154.19	329.81	107.78	8.65	5.13
3-3-RBL	0.24	221.30	723.54	236.45	5.65	5.50
3-1-LBL	0.39	127.84	341.99	111.76	3.58	5.80
3-2-LBL	0.61	214.38	321.50	105.07	16.34	4.73
3-3-LBL	0.11	152.28	529.15	172.92	2.92	4.67
						4.67
4-1-RBL	0.15	401.06	850.94	278.08	2.94	4.00
4-2-RBL	0.27	313.66	709.88	231.99	7.54	4.03
4-1-LBL	0.23	306.80	1051.74	343.71	5.16	4.40
4-2-LBL	0.24	498.69	534.58	174.70	6.86	5.90
4-3-LBL	0.13	359.36	905.90	296.05	3.84	5.60
1-1-RBM	0.23	131.23	326.93	106.84	6.18	4.37
1-2-RBM	0.54	209.43	433.45	141.65	9.40	4.27
1-3-RBM	0.24	92.47	271.44	88.71	2.94	4.73
1-1-LBM	0.21	57.74	415.11	135.66	6.62	4.37
1-2-LBM	0.68	313.32	797.06	260.48	17.01	4.73
1-3-LBM	0.48	194.89	469.23	153.34	9.65	4.37
2-1-RBM	0.20	81.85	149.27	48.78	5.53	4.20
2-2-RBM	2.82	115.47	207.17	67.70	4.65	4.37
2-3-RBM	0.12	112.58	409.64	133.87	3.32	4.13
2-1-LBM	0.13	84.66	237.12	77.49	4.00	4.93
2-2-LBM	0.19	45.46	95.46	31.19	3.21	4.20
2-3-LBM	0.49	163.74	205.34	67.10	14.94	4.07
3-1-RBM	1.15	176.70	407.12	133.05	11.05	4.60

Site code	TN %	TP (mg/kg)	PO_4^{-3} (mg/kg)	PO_4-P (mg/kg)	Organic carbon	pH (KCI)
3-2-RBM	1.47	177.94	472.50	154.41	7.91	4.20
3-3-RBM	0.23	225.28	556.69	181.92	7.38	4.03
3-1-LBM	0.23	208.70	329.12	107.55	4.83	3.97
3-2-LBM	0.40	438.85	811.76	265.28	5.61	3.70
3-3-LBM	0.14	183.85	439.72	143.70	3.16	5.10
4-1-RBM	0.09	145.38	500.51	163.57	2.67	3.77
4-2-RBM	0.28	170.69	223.44	73.02	4.12	4.90
4-1-LBM	0.42	353.78	901.35	294.56	13.56	4.33
4-2-LBM	0.22	261.43	686.18	224.24	5.29	5.40
4-3-LBM	0.24	287.92	543.58	177.64	9.50	6.30
1-1-RBU	0.18	111.05	240.43	78.57	5.55	5.43
1-2-RBU	0.70	260.89	394.25	128.84	28.00	3.93
1-3-RBU	0.31	127.83	395.49	129.25	4.31	4.73
1-1-LBU	0.37	290.70	691.71	226.05	8.26	5.43
1-2-LBU	0.68	313.32	797.06	260.48	10.17	4.73
1-3-LBU	0.40	132.77	370.73	121.15	4.10	4.83
2-1-RBU	0.50	165.05	341.41	111.57	10.73	4.20
2-2-RBU	0.51	169.56	408.02	133.34	10.77	4.83
2-3-RBU	0.44	176.56	464.62	151.83	11.74	4.23
2-1-LBU	0.22	76.96	178.83	58.44	4.12	3.93
2-2-LBU	0.40	71.45	168.82	55.17	4.50	3.97
2-3-LBU	0.49	287.29	865.77	282.93	10.75	4.27
3-1-RBU	0.64	168.21	449.27	146.82	7.79	3.63
3-2-RBU	2.15	254.28	665.32	217.42	23.82	4.43
3-3-RBU	0.63	248.35	722.25	236.03	11.18	3.57
3-1-LBU	0.31	182.88	489.05	159.82	19.94	3.97
3-2-LBU	0.55	277.49	787.12	257.23	5.61	5.20
3-3-LBU	0.12	185.16	441.59	144.31	2.89	3.43
						4.20
4-1-LBU	0.17	276.27	749.43	244.91	5.58	5.50
4-2-LBU	0.21	242.39	815.77	266.59	3.94	4.27
4-3-LBU	0.20	282.23	830.33	271.35	4.78	5.73

Appendix 3.1 Eigenvectors for PCA: Touws River sub-catchment (only first 4 Factors are shown) (Abbreviations as per Figure 3.7)

	F1	F2	F3	F4
NH_4^+-N	-0.080	-0.291	0.126	0.072
NO_x-N	0.045	0.401	-0.032	0.186
PO_4-P	-0.096	0.159	-0.178	0.319
EC	0.091	0.228	0.423	0.066
Ca^{2+}	0.050	0.414	-0.082	-0.059
F^-	0.064	0.183	-0.208	-0.057
Cl^-	0.119	0.041	0.503	-0.006
K^+	0.149	0.315	0.096	0.049
Mg^{2+}	-0.055	0.137	0.364	0.263
Na^+	0.119	-0.054	0.497	-0.125
pH	0.103	0.432	-0.070	0.013
Si	-0.081	-0.069	0.197	0.178
SO_4^{2-}	-0.134	0.307	-0.059	0.085
Alk	-0.066	0.181	0.137	-0.310
DOPF	-0.146	0.105	0.024	-0.549
Plantations	0.275	-0.025	-0.036	-0.053
GDT	-0.276	0.035	0.036	-0.013
Agr	-0.276	0.035	0.036	-0.013
NKA	0.275	-0.025	-0.036	-0.053
Smallh	0.276	-0.035	-0.036	0.013
Urb	0.275	-0.025	-0.036	-0.053
NOPF	0.146	-0.105	-0.024	0.549
Grassl	0.275	-0.025	-0.036	-0.053
LS	0.275	-0.025	-0.036	-0.053
SF	0.275	-0.025	-0.036	-0.053
Thicket	0.275	-0.025	-0.036	-0.053
Woodl	0.275	-0.025	-0.036	-0.053

Appendix 3.2 Eigenvectors for PCA: Touws River buffer (Abbreviations as per Figure 3.7)

	F1	F2	F3	F4
NH_4^+-N	0.122	-0.075	-0.278	0.161
NO_x-N	-0.101	0.072	0.348	-0.057
PO_4-P	-0.009	-0.256	0.254	0.265
EC	-0.090	0.396	0.079	0.301
Ca^{2+}	-0.101	0.141	0.371	-0.191
F^-	-0.089	0.010	0.191	-0.174
Cl^-	-0.035	0.460	-0.117	0.242
K^+	-0.143	0.324	0.167	-0.170
Mg^{2+}	0.031	0.173	0.149	0.580
Na^+	0.063	0.486	-0.155	0.036
pH	-0.183	0.173	0.322	-0.090
Si	0.162	0.015	0.071	0.287
SO_4^{2-}	0.058	-0.124	0.471	0.140
Alk	0.119	0.097	0.267	-0.217
DOPF	-0.311	0.018	-0.094	-0.034
Plantations	0.310	0.064	0.030	-0.054
GDT	-0.311	0.018	-0.094	-0.034
NKA	0.311	-0.018	0.094	0.034
Smallh	0.311	-0.018	0.094	0.034
NOPF	-0.253	-0.075	-0.055	0.134
Grassl	0.307	0.097	0.016	-0.107
SF	0.310	0.064	0.030	-0.054
Thicket	0.310	0.064	0.030	-0.054
Woodl	0.110	0.284	-0.161	-0.344

Appendix 3.3 Eigenvectors for PCA: Duiwe River sub-catchment (only first 4 Factors are shown) (Abbreviations as per Figure 3.7)

	F1	F2	F3	F4
EC	-0.054	0.291	0.061	-0.034
NH_4^+-N	0.147	0.338	-0.050	0.137
NO_x-N	-0.130	0.085	-0.211	-0.407
PO_4-P	0.291	0.165	-0.121	-0.047
Ca^{2+}	-0.029	0.363	0.157	0.090
Cl^-	-0.219	-0.026	0.282	0.301
K^+	0.142	0.320	0.165	-0.144
Mg^{2+}	-0.027	0.379	0.158	0.074
Na^+	-0.183	0.060	0.299	0.317
SO_4^{2-}	-0.191	0.133	0.187	0.419
F^-	0.252	0.166	0.002	0.004
pH	0.121	0.322	0.188	-0.080
Alk	0.068	-0.271	0.233	0.075
Si	-0.020	-0.345	0.151	0.185
DOPF	-0.244	-0.001	0.254	-0.299
Agr	0.303	-0.027	-0.152	0.244
NKA	0.244	0.001	-0.254	0.299
NOPF	-0.276	0.092	-0.255	0.085
Smallh	-0.276	0.092	-0.255	0.085
Plantations	-0.276	0.092	-0.255	0.085
SF	0.276	-0.092	0.255	-0.085
Thicket	0.244	0.001	-0.254	0.299
Woodl	0.276	-0.092	0.255	-0.085

Appendix 3.4 Eigenvectors for PCA: Duiwe River buffer (Abbreviations as per Figure 3.7)

	F1	F2	F3	F4
EC	0.142	0.109	-0.283	0.101
NH_4^+	0.171	-0.003	0.129	-0.187
NO_x-N	-0.289	0.158	0.048	0.152
PO_4-P	-0.251	-0.084	0.156	0.230
Ca^{2+}	0.327	0.138	-0.028	0.074
Cl^-	0.324	0.118	-0.110	0.071
K^+	0.281	0.060	0.232	0.084
Mg^{2+}	0.326	0.155	0.003	0.044
Na^+	0.327	0.118	-0.072	0.051
SO_4^{2-}	0.275	0.079	-0.212	0.100
F^-	0.142	-0.122	0.272	-0.201
pH	0.193	0.049	0.301	0.209
Alk	0.254	-0.046	0.298	0.112
Si	0.106	-0.148	0.100	0.097
DOPF	-0.115	0.401	0.160	0.088
Agr	0.018	0.416	0.048	-0.021
NKA	0.062	-0.407	-0.041	0.245
NOPF	0.151	-0.294	-0.017	-0.379
Smallh	0.062	-0.407	-0.041	0.245
Plnatations	0.160	-0.057	-0.297	0.290
Thicket	-0.028	-0.165	0.262	0.478
SF	0.141	-0.186	0.252	-0.389
GDT	-0.014	0.126	0.495	0.067

Appendix 4.1 Chemical water quality parameters, flow and water levels measured at the sampling locations during the dry and wet seasons for the Klein Keurbooms and Duiwe Rivers (2014-2016) (Alk-Alkalinity)

Year	Season	Wet/ Dry season	Date	Site	Na^{+}	Ca^{2+}	Mg^{2+}	Alk	NO_x	EC	pH	COD	TN	TP	Si	Benthic Chl-*a*	Suspended Chl-*a*	Flow (m^3s^{-1})	Water Level (m)
2014	Autumn	Wet	Mar-14	1	17.0	0.5	2.10	1.70	0.10	13.0	4.80	19	1	0.05	2.1	0.01	0.00	0.068	0.125
2014	Autumn	Wet	Mar-14	2	21.0	0.8	2.50	3.30	0.10	16.0	5.10	35	1	0.05	2.3	0.11	0.00	0.051	0.128
2014	Autumn	Wet	Mar-14	3	39.0	2.5	4.70	4.50	0.10	27.0	5.70	20	1	0.08	3.1	0.12	0.00	0.51	0.174
2014	Autumn	Wet	Mar-14	4	138.0	19.0	17.00	48.00	0.60	100.0	7.70	53	1	0.22	2.3	0.02	0.00	0.016	0.17
2014	Winter	Dry	Jul-14	1	16.0	2.7	1.85	2.0	-	13.0	5.00	34.5	1	0.05	3	0.4	0.32	0.063	0.1
2014	Winter	Dry	Jul-14	2	22.0	2.7	2.60	3.00	0.10	17.0	5.20	7	1	0.05	3.4	0.6	0.26	0.004	0.089
2014	Winter	Dry	Jul-14	3	39.0	5.1	5.10	5.00	-	30.0	6.00	16	1	0.05	4.2	0.7	0.26	0.003	0.199
2014	Winter	Dry	Jul-14	4	146.0	23.0	17.00	48.00	-	114.0	7.90	65	1	0.33	2	0.5	0.58	0.005	0.13
2014	Spring	Wet	Sep-14	1	19.5	1.0	2.20	2.3	0.10	13.0	5.05	21	1	0.05	2.7	0.5	1.69	0.068	0.11
2014	Spring	Wet	Sep-14	2	25.0	1.3	3.10	2.50	0.10	15.0	5.10	26	1	0.05	2.7	0.84	0.73	0.05	0.11
2014	Spring	Wet	Sep-14	3	38.0	2.8	4.80	4.60	0.10	27.0	5.90	28	1	0.05	3.4	0.19	0.67	0.51	0.172
2014	Spring	Wet	Sep-14	4	220.0	34.0	32.00	63.00	1.00	175.0	7.90	48	2	0.13	1	0.18	0.997	0.018	0.09
2015	Summer	Wet	Feb-15	1	16.0	0.6	2.05	3.8	0.10	12.0	5.10	25	1	2.6	2.3	0.0	0.37	0.001	0.189
2015	Summer	Wet	Feb-15	2	19.0	0.8	2.50	4.60	0.10	14.0	5.40	12	1	0.06	2.5	0.1	0.07	0.03	0.125
2015	Summer	Wet	Feb-15	3	35.0	2.5	4.60	6.20	0.10	25.0	6.00	18	1	0.06	3	0.1	0.12	0.035	0.189
2015	Summer	Wet	Feb-15	4	190.0	24.0	23.00	63.00	0.10	134.0	7.50	58	2	0.11	1.2	0.23	0.1	0.013	0.162
2015	Autumn	Dry	May-15	1	18.5	0.9	2.30	2.8	0.10	13.0	4.90	21.5	1	0.05	2.6	0.018	0.3	0.12	0.125
2015	Autumn	Dry	May-15	2	22.0	1.0	2.80	4.50	0.10	16.0	5.20	41	1	0.05	2.9	0.091	0.21	0.2	0.127
2015	Autumn	Dry	May-15	3	41.0	3.0	5.40	7.30	0.10	30.0	6.00	29	1	0.05	3.5	0.064	0.6	0.016	0.228
2015	Autumn	Dry	May-15	4	155.0	22.00	19.00	43.00	0.1	112.00	7.3	30	1	0.05	1.8	0.174	0.1	0.003	0.192
2015	Winter	Dry	Aug-15	1	18.0	0.60	2.20	1.60	0.2	12.00	4.7	5	0.9	0.07	2.2	0.0349	0.26	0.063	0.076
2015	Winter	Dry	Aug-15	2	20.0	0.70	2.70	2.40	0.3	14.00	4.8	11	0.9	0.05	2.5	0.0317	0.28	0.004	0.076
2015	Winter	Dry	Aug-15	3	30.0	1.90	4.10	66.00	0.3	20.00	5.3	8	0.9	0.05	3.2	0.0380	0.26	0.003	0.78

Year	Season	Wet/ Dry season	Date	Site	Na^{+}	Ca^{2+}	Mg^{2+}	Alk	NO_x	EC	pH	COD	TN	TP	Si	Benthic Chl-*a*	Suspended Chl-*a*	Flow (m^3s^{-1})	Water Level (m)
2015	Winter	Dry	Aug-15	4	94.0	12.00	12.00	34.00	0.8	64.00	7.3	50	1	0.44	3.5	0.0570	0.26	0.076	0.3
2015	Spring	Wet	Oct-15	1	16	1	2.2	2	0.1	12	4.8	10	0.5	0.05	2.2	1.12	0.84	0.109	0.108
2015	Spring	Wet	Oct-15	2	21	0.8	2.7	2.8	0.2	14	5.1	16	0.5	0.05	2.5	0.45	0.08	0.188	0.17
2015	Spring	Wet	Oct-15	3	34	2.3	4.5	6.2	0.1	23	5.5	23	0.5	0.05	3.3	3.58	0.30	0.163	0.178
2015	Spring	Wet	Oct-15	4	77	11	9.7	24	0.3	54	7.3	51	1.1	**0.26**	3.2	0.87	0.01	0.001	0.35
2016	Summer	Wet	Feb-16	1	16	0.6	2	2.7	0.9	11	5	15	0.5	**0.05**	2.3	34.48	0.01	0.001	0.092
2016	Summer	Wet	Feb-16	2	20	1.7	2.5	4.8	0.1	14	5.3	21	0.5	**0.05**	2.6	17.54	0.02	0.03	0.084
2016	Summer	Wet	Feb-16	3	34	2.2	4.3	5.8	0.1	23	5.7	22	0.5	**0.05**	3	9.85	0.00	0.035	0.17
2016	Summer	Wet	Feb-16	4	236	30	27	59	0.1	155	7.1	50	0.7	**0.05**	0.6	5.15	0.09	0.053	0.444
2016	Winter	Dry	Jun-16	1	18	1.5	2.2	203	0.1	14	10.4	5	0.5	**0.04**	2.7	-	-	-	-
2016	Winter	Dry	Jun-16	2	24	1.2	3.2	3.9	0.1	18	5.2	15	0.5	**0.03**	3.2	-	-	-	-
2016	Winter	Dry	Jun-16	3	51	3.7	6.4	51	0.1	37	7.5	18	0.5	**0.05**	4	-	-	-	-
2016	Winter	Dry	Jun-16	4	212	28	25	5.4	0.1	144	6	40	0.5	**0.04**	1.4	-	-	-	-

Appendix 4.2 Macroinvertebrate data (March 2014-March 2016).

1 = 1 individual; 2 = 2-6; 3 = 7-20; 4 = 21-100 and 5 = >100. K1 – K4 = Sites; S = Stones (in current/out of current), VG = Vegetation (marginal/aquatic), G = Gravel, Sand, Mud. S = Summer, W = Winter, A = Autumn, Sp = Spring

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
TURBELLARIA														
Oligochaeta	1								1		2	1		
Hirudinea														
Amphipoda														
Potamonautidae														
Hydracarina						1		1				2		
Notonemouridae	2		2		2	1	2	1	2		2	2		2
Perlidae	1										2			
Baetidae 1sp		2							2			1		
Baetidae 2 sp			2											
Baetidae > 2 sp														
Caenidae			2		1			1	1			1		2
Leptophlebiidae	2		2		2	1	2	1	2			1		
Teloganodidae					2	2	2	2				2		2
Chlorolestidae		2			1	1	1	1					2	
Coenagrionidae				2									2	
Lestidae														
Platycnemidae														
Aeshnidae														
Corduliidae														
Gomphidae														

Data code	**k1-S-A_14**	**k1-VG-A_14**	**k1-S-W_14**	**k1-VG-W_14**	**k1-S-S_14**	**k1-VG-S_14**	**k1-S-Sp_14**	**k1-VG-Sp_14**	**k1-3-S-A_14**	**k1-3-VG-A_14**	**k1-3-G-A_14**	**k1-3-S-W_14**	**k1-3-VG-W_14**	**k1-3-G-W_14**
Libellulidae													2	
Pyralidae									1					
Belostomatidae														
Corixidae														
Gerridae														
Hydrometridae														
Naucoridae	2	2		2					1	2	2	2		2
Nepidae														
Notonectidae									2			2	2	
Pleidae										1				
Veliidae	2		1		1	2	1	2						
Corydalidae	2				1		1		2		1	1		2
Hydropsychidae 1 sp	1	1												
Hydropsychidae 2 sp														
Hydropsychidae > 2 sp														
Philopotamidae														
Barbarochthonidae SWC												3		
Glossosomatidae SWC									1					
Hydroptilidae				1										
Leptoceridae	4	2	2	2	2	2	2	2	3	3	3	2	3	2
Petrothrincidae SWC					1	2	1	2						
Sericostomatidae							2							
Pisuliidae	2	2	1	2	2	2	2	2	1		2		3	
Dytiscidae													2	
Elmidae	1			2		2		2	2		1	2		1

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Gyrinidae				1										
Helodidae									1		1			
Hydraenidae														
Hydrophilidae														
Athericidae						1		1			1	1		1
Ceratopogonidae					1		1					2		1
Dixidae														
Simuliidae	2	1	2	1	1			1	2					
Tabanidae														
Tipulidae														
Ancylidae														
Lymnaeidae														
Physidae														
Planorbinae														

Data code	**k1-S-A_14**	**k1-VG-A_14**	**k1-S-W_14**	**k1-VG-W_14**	**k1-S-S_14**	**k1-VG-S_14**	**k1-S-Sp_14**	**k1-VG-Sp_14**	**k1-3-S-A_14**	**k1-3-VG-A_14**	**k1-3-G-A_14**	**k1-3-S-W_14**	**k1-3-VG-W_14**	**k1-3-G-W_14**
TURBELLARIA														
Oligochaeta									2	1				
Hirudinea														
Amphipoda														
Potamonautidae														
HYDRACARINA	2	1		2	1									
Notonemouridae	2		2	2		2			2		2	2	2	2
Perlidae														
Baetidae 1sp										1	2	1	2	1
Baetidae 2 sp							2	2	2					
Baetidae > 2 sp						1								
Caenidae			1						2	1				
Leptophlebiidae	2		1	2		1	3		3	2	2	2	2	2
Teloganodidae	2	1		2	1				3	2	2	3	2	3
Chlorolestidae	1			1								1		1
Coenagrionidae								2		2	1		1	
Lestidae														
Platycnemidae		1			1				1	2				
Aeshnidae														
Corduliidae			1			1								
Gomphidae											1		1	
Libellulidae								2						
Pyralidae														
Belostomatidae														
Corixidae		2	2		2	2								
Gerridae														

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Hydrometridae														
Naucoridae							2	2	2		1		1	
Nepidae														
Notonectidae								2		1				
Pleidae														
Veliidae	2	1	1	2	1	1		2	1	1	2		2	
Corydalidae	2			2					2	2		2		2
Hydropsychidae 1 sp														
Hydropsychidae 2 sp														
Hydropsychidae >2 sp														
Philopotamidae									1		1		1	
Barbarochthonidae									2	2				
Glossosomatidae														
Hydroptilidae														
Leptoceridae	2	2	2	2	2	2	3	3	2	2	3	3	3	3
Petrothrincidae SWC														1
Sericostomatidae				2										
Pisuliidae		1			1			2		2		1		
Dytiscidae		2	1		2	1		1		2	2		2	
Elmidae/Dryopidae	2		2	2		2	2	1	2		1	2	1	2
Gyrinidae							3	1			2	1	2	1
Helodidae										1				
Hydraenidae														
Hydrophilidae														
Athericidae	2			2						1		2		2
Ceratopogonidae	2		1	2						1	2		2	

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Dixidae														
Simuliidae	1			1			2		1		1	2	1	2
Tabanidae														
Tipulidae														
Ancylidae														
Lymnaeidae														
Physidae														
Planorbinae														
TURBELLARIA											2			1
Oligochaeta	2			1			2		2	1			3	2
Hirudinea											2	2		2
Amphipoda												1		
Potamonautidae											2			
HYDRACARINA		2										2		
Notonemouridae			2	2	2	1		2	1					
Perlidae														
Baetidae 1sp			1				2			2				
Baetidae 2 sp		3												3
Baetidae > 2 sp	3				2			2			3			
Caenidae	1					1			1					
Leptophlebiidae			2	1	2	2	2	2	2	2				
Teloganodidae			2	1	3	2	2	3	2	2				
Chlorolestidae						2			2					
Coenagrionidae		1	1	1								2	1	
Lestidae														
Platycnemidae														

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Aeshnidae	1	2		1	1			1						
Corduliidae						2			2					
Gomphidae		2											2	
Libellulidae		1		1	1			1				2		
Pyralidae														
Belostomatidae														
Corixidae						1			1					
Gerridae														
Hydrometridae		2												
Naucoridae	1				1	2		1	2					
Nepidae														
Notonectidae		2		1								2	3	2
Pleidae		2										2		
Veliidae	1	3		1								2		2
Corydalidae			1		1			1		1				
Hydropsychidae 1 sp				1										
Hydropsychidae 2 sp														
Hydropsychidae >2 sp			1											
Philopotamidae														
Barbarochthonidae			2	2										
Glossosomatidae														
Hydroptilidae														
Leptoceridae	3	3	2	3	3	3		3	3					
Petrothrincidae SWC														
Sericostomatidae														
Pisuliidae	1			1										

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Dytiscidae		2		1		1			1			1	1	
Elmidae/Dryopidae	1				1		1	1		1				
Gyrinidae	2	2	1		2			2						
Helodidae			2		1		2	1		2				
Hydraenidae														
Hydrophilidae		2												
Athericidae														
Ceratopogonidae						1			1			2		
Dixidae														
Simuliidae	2		2		2	2	2	2	2	2		3		2
Tabanidae														
Tipulidae							1			1				
Ancylidae														
Lymnaeidae												2	2	
Physidae														2
Planorbinae														
TURBELLARIA		2		2										
Oligochaeta		1		1										
Hirudinea														1
Amphipoda														
Potamonautidae														
HYDRACARINA														
Notonemouridae						3			1	1	2		2	1
Perlidae											1			
Baetidae 1sp			2		2			2						
Baetidae 2 sp		3				2					2		3	2

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Baetidae > 2 sp				3										
Caenidae										2		1		
Leptophlebiidae													2	
Teloganodidae						2					1		1	
Chlorolestidae								1	2			2		1
Coenagrionidae			2		2								1	
Lestidae														
Platycnemidae														
Aeshnidae													1	1
Corduliidae									2					
Gomphidae														
Libellulidae														
Pyralidae														
Belostomatidae														
Corixidae			2		2				1			2		1
Gerridae														
Hydrometridae														
Naucoridae						2	2	2	2		2	2	2	1
Nepidae												1		
Notonectidae									2					2
Pleidae		1												
Veliidae				1		2			2		2	1		
Corydalidae						2		1	1	1	2		2	1
Hydropsychidae 1 sp													1	
Hydropsychidae 2 sp														
Hydropsychidae >2 sp														

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Philopotamidae						2								
Barbarochthonidae												2		
Glossosomatidae											3			
Hydroptilidae						1		1						
Leptoceridae						3	3	3	3	2	3	2	2	3
Petrothrincidae SWC														
Sericostomatidae								1						
Pisuliidae						2	2	2	2		1	1		
Dytiscidae									2					
Elmidae/Dryopidae						2		2			2	2	1	
Gyrinidae														2
Helodidae														
Hydraenidae														
Hydrophilidae														
Athericidae						1		1						
Ceratopogonidae						1	1			1				
Dixidae														
Simuliidae		3		3				2					3	3
Tabanidae														
Tipulidae														
Ancylidae														
Lymnaeidae														
Physidae		1	2	1	2									
Planorbinae														
TURBELLARIA														
Oligochaeta	2													

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Hirudinea	1	1												
Amphipoda													1	
Potamonautidae	2												1	
HYDRACARINA														
Notonemouridae			2		1		1	2		1		1		
Perlidae														
Baetidae 1sp	2					1			1		1			2
Baetidae 2 sp			2		2			2		2				
Baetidae > 2 sp													3	
Caenidae						1	2	1	1		1	2		
Leptophlebiidae			2	1	2			2		2				
Teloganodidae					2		2	2		2		2		
Chlorolestidae				1		2		2			2			
Coenagrionidae		2			1	1			2	1	1			3
Lestidae														
Platycnemidae														
Aeshnidae														
Corduliidae														
Gomphidae														
Libellulidae														2
Pyralidae														
Belostomatidae		2												
Corixidae	2	2				2	2		1		2	2	2	1
Gerridae														
Hydrometridae														
Naucoridae			2	2	1	1		2	1	1	1			

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Nepidae														
Notonectidae	1													
Pleidae														
Veliidae	1	2		2					2				2	
Corydalidae			2		2		1	2		2		1		
Hydropsychidae 1 sp			1											
Hydropsychidae 2 sp														
Hydropsychidae >2 sp														
Philopotamidae								2						
Barbarochthonidae			2	2	2			2		2				
Glossosomatidae														
Hydroptilidae														
Leptoceridae			2	1	1	1	2		3	1	1		2	
Petrothrincidae SWC														
Sericostomatidae			3											
Pisuliidae			2	2	2	2			2	2	2			
Dytiscidae										1				
Elmidae/Dryopidae			1		1				1					
Gyrinidae										1				
Helodidae														
Hydraenidae														
Hydrophilidae														
Athericidae			2		1					1				
Ceratopogonidae														1
Dixidae														
Simuliidae			2		1				2	1			2	

Data code	**k1-S-A_14**	**k1-VG-A_14**	**k1-S-W_14**	**k1-VG-W_14**	**k1-S-S_14**	**k1-VG-S_14**	**k1-S-Sp_14**	**k1-VG-Sp_14**	**k1-3-S-A_14**	**k1-3-VG-A_14**	**k1-3-G-A_14**	**k1-3-S-W_14**	**k1-3-VG-W_14**	**k1-3-G-W_14**
Tabanidae														
Tipulidae														
Ancylidae	2	1											1	2
Lymnaeidae														
Physidae		2												3
Planorbinae														
TURBELLARIA							1				1			
Oligochaeta					1		2		1					
Hirudinea								1						
Amphipoda														
Potamonautidae							1							
HYDRACARINA														
Notonemouridae	2	2	2	2	2				1	1	2	2	2	1
Perlidae														
Baetidae 1sp		1							1					1
Baetidae 2 sp	2		2	2			3	2		3	2	2		
Baetidae > 2 sp					2	2							2	
Caenidae				1	1	1		1						1
Leptophlebiidae	3	2	3	1	2				2	2	2	1	3	
Teloganodidae	3	2	3	2	3	2			3	1	3	2		2
Chlorolestidae		1		1										
Coenagrionidae			1	2		2		2				2		2
Lestidae														
Platycnemidae														
Aeshnidae				2	2									
Corduliidae														

Data code	k1-S-A_14	k1-VG-A_14	k1-S-W_14	k1-VG-W_14	k1-S-S_14	k1-VG-S_14	k1-S-Sp_14	k1-VG-Sp_14	k1-3-S-A_14	k1-3-VG-A_14	k1-3-G-A_14	k1-3-S-W_14	k1-3-VG-W_14	k1-3-G-W_14
Gomphidae						1								
Libellulidae				2				1				1		1
Pyralidae														
Belostomatidae														
Corixidae				2		1		1						2
Gerridae														
Hydrometridae														
Naucoridae	1	2	1	2		2			2		2	2		
Nepidae														
Notonectidae														
Pleidae														1
Veliidae		2		1						2			1	
Corydalidae	2	1	2		2				2			2		
Hydropsychidae 1 sp							2						2	
Hydropsychidae 2 sp					2							2		
Hydropsychidae >2 sp														
Philopotamidae			1									2		
Barbarochthonidae		2	2	1		2			2	2		1		
Glossosomatidae									1		2			
Hydroptilidae	1		1						1		2	1		
Leptoceridae	2	2	2	2	2	3				3		2	2	3
Petrothrincidae SWC														
Sericostomatidae									1	1				
Pisuliidae		2							1	2		1		
Dytiscidae			2	1										
Elmidae/Dryopidae	2		2	2	2	1			2	1	1	1		

Data code	**k1-S-A_14**	**k1-VG-A_14**	**k1-S-W_14**	**k1-VG-W_14**	**k1-S-S_14**	**k1-VG-S_14**	**k1-S-Sp_14**	**k1-VG-Sp_14**	**k1-3-S-A_14**	**k1-3-VG-A_14**	**k1-3-G-A_14**	**k1-3-S-W_14**	**k1-3-VG-W_14**	**k1-3-G-W_14**
Gyrinidae					1	2							1	2
Helodidae				1							1	2	1	
Hydraenidae														
Hydrophilidae														
Athericidae	1								2		1			
Ceratopogonidae		1		2				2						
Dixidae														
Simuliidae	2	2	1	2	2	2	1	1	1	1	2	1		1
Tabanidae									1					
Tipulidae							2							
Ancylidae														
Lymnaeidae								3						
Physidae								3						
Planorbinae														

Data code	k4-S-Sp_15	k4-Vg-Sp_15	k1-S-S_16	k1-VG-S_16	k2-S-S_16	k2-VG-S_16	k3-S-S_16	k3-VG-S_16	k4-S-S_16	k4-VG-S_16	k4-G-S_16
TURBELLARIA		1									
Oligochaeta	2						1		1		3
Hirudinea	1										
Amphipoda											
Potamonautidae	1									1	
HYDRACARINA							1		2		
Notonemouridae			2		2		2	2			
Perlidae											
Baetidae 1sp			1							2	1
Baetidae 2 sp		2						2			
Baetidae > 2 sp	3				2				2		
Caenidae				1		1			1		
Leptophlebiidae			3		2		2				
Teloganodidae			2	1	2		1				
Chlorolestidae			1								
Coenagrionidae		2				2		2	3	1	1
Lestidae											
Platycnemidae											
Aeshnidae					1		2			1	
Corduliidae		2									
Gomphidae											
Libellulidae								1	2		1
Pyralidae											
Belostomatidae										3	
Corixidae		3							2	2	
Gerridae											
Hydrometridae											

Data code	k4-S-Sp_15	k4-Vg-Sp_15	k1-S-S_16	k1-VG-S_16	k2-S-S_16	k2-VG-S_16	k3-S-S_16	k3-VG-S_16	k4-S-S_16	k4-VG-S_16	k4-G-S_16
Naucoridae				2	2	2	2				
Nepidae											
Notonectidae										1	
Pleidae										3	1
Veliidae		2	1	2	2	2	1	2		3	2
Corydalidae			3	2	2		2				
Hydropsychidae 1 sp							1				
Hydropsychidae 2 sp			2								
Hydropsychidae >2 sp											
Philopotamidae					2		2				
Barbarochthonidae			2	2	3	3					
Glossosomatidae			3		3						
Hydroptilidae										1	
Leptoceridae			3	3	3	3		2			
Petrothrincidae SWC			2		3	1					
Sericostomatidae											
Pisuliidae			3	3	1						
Dytiscidae		1		1		2			1		
Elmidae/Dryopidae			3	1	2	2					
Gyrinidae						3	1				
Helodidae			2								
Hydraenidae									1		
Hydrophilidae											
Athericidae			2	2	1					2	
Ceratopogonidae		2	1							2	2

Data code	**k4-S-Sp_15**	**k4-Vg-Sp_15**	**k1-S-S_16**	**k1-VG-S_16**	**k2-S-S_16**	**k2-VG-S_16**	**k3-S-S_16**	**k3-VG-S_16**	**k4-S-S_16**	**k4-VG-S_16**	**k4-G-S_16**
Dixidae											
Simuliidae	2		2				2		2		
Tabanidae											
Tipulidae											
Ancylidae	1	1								2	
Lymnaeidae											
Physidae		2							1	3	2
Planorbinae		1				1					

Appendix 4.3 Algal taxa sampled March 204-March 2016.

K1 – K4 = Sites. The relative abundance of each algal taxa was grouped into: 1 = ≤ 50 (rare) 2 = 51- 250 (scarce), 3 = 251-1000 (common), 4 = 1001-5000 (abundant), 5 = 5001-25 000 (predominant) cells/ 5cm^2. S = Summer, W = Winter, A = Autumn, Sp = Spring

Data code	k1-A_14	k2-A_14	k3-A_14	k4-A_14	k1-W_14	k2-W_14	k3-W_14	k4-W_14	k1-Sp_14	k2-Sp_14	k3-Sp_14	k4-Sp_14
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	250	0	0
Achnanthes standeri	0	0	0	0	0	0	0	0	1000	250	0	0
Aulacoseira granulata	0	0	0	0	0	0	0	0	0	0	0	0
Brachysira neoexillis	0	250	0	0	0	0	0	0	0	0	0	0
Calothrix inserta	0	0	0	0	0	250	0	0	0	0	250	0
Chroodactylon ramosum	0	50	0	0	0	0	0	0	0	0	0	0
Cladophora glomerata	0	0	0	0	0	0	0	1000	0	0	0	0
Closterium lineatum	0	0	250	0	0	0	250	0	0	0	250	0
Cocconeis pediculus	0	0	0	0	0	0	0	0	0	0	0	0
Cocconeis placentula	0	0	0	0	0	0	0	250	0	0	0	250
Craticula ambigua	0	0	0	0	0	0	0	250	0	0	0	0
Cymbella aspera	0	0	0	0	0	0	0	0	0	0	0	0
Cymbella cymbiformis	0	0	0	0	0	0	0	0	0	0	0	0
Epithemia adonata	250	0	0	0	0	0	0	0	0	0	1000	0
Eunotia bilunaris	0	0	0	0	1000	0	0	0	0	0	0	0
Eunotia exigua	0	0	0	0	250	250	50	0	250	0	0	0
Eunotia flexuosa	0	0	0	0	250	0	0	0	0	250	0	0
Eunotia formica	1000	5000	250	0	50	1000	50	0	1000	250	5000	0
Eunotia incisa	1000	1000	50	0	0	0	0	0	0	0	50	0
Eunotia minor	250	50	0	0	0	0	0	0	0	0	0	0
Eunotia pectinalis	0	0	0	0	0	0	0	0	1000	250	0	0
Flagilaria ulna	0	0	0	0	0	0	0	250	0	0	250	0
Frustulia saxonica	0	1000	1000	0	0	250	0	0	0	0	0	0

Data code	k1-A_14	k2-A_14	k3-A_14	k4-A_14	k1-W_14	k2-W_14	k3-W_14	k4-W_14	k1-Sp_14	k2-Sp_14	k3-Sp_14	k4-Sp_14
Gyrosigma acuminatum	0	0	0	0	0	0	0	1000	0	0	0	250
Gyrosigma attenuatum	0	0	0	0	0	0	0	0	0	0	0	1000
Kobayasiella subtillissma	0	0	0	0	0	0	0	0	0	0	0	0
Melosira varians	0	0	0	0	0	0	0	5000	0	0	0	5000
Navicula angusta	0	0	0	0	0	0	0	0	0	0	0	0
Navicula cryptotenella	0	0	0	0	0	0	250	0	0	0	250	0
Navicula heimansioides	0	1000	250	0	0	0	0	0	0	0	0	0
Navicula notha Wallace	250	0	0	0	0	0	0	0	0	0	0	0
Navicula radiosa	0	250	0	0	0	0	5000	0	0	50	0	0
Navicula ranomafenensis	0	0	0	0	0	0	0	0	0	0	0	0
Nitzchia iremissa	0	0	0	0	0	0	0	0	250	50	0	0
Nitzchia linearis	0	0	0	0	0	0	1000	50	0	0	0	25000
Nitzschia filiforms	0	0	0	0	0	0	0	1000	0	0	0	0
Nostoc commune	0	250	0	0	50	0	0	0	0	0	0	0
Oedegonium minus	0	0	0	0	0	250	0	50	0	0	0	1000
Oedegonium no 1	0	0	0	0	0	0	0	250	0	0	0	0
Oedegonium no 2	0	0	0	0	0	0	0	0	0	0	0	0
Oocystis rupestris	0	0	0	0	0	0	0	0	0	0	0	250
Oscillatoria princeps	0	0	0	1000	0	0	0	50	0	0	0	0
Oscillatoria sancta	0	0	0	0	0	0	0	1000	0	0	0	250
Oscillatoria tenuie	0	0	0	0	50	0	0	50	0	0	0	0
Pediastrum duplex	0	0	50	0	0	0	1000	0	0	0	50	0
Pinnularia divergens	0	0	0	0	0	0	0	0	0	0	0	0
Pinnularia viridiformis	0	0	0	0	0	0	0	0	0	0	250	0
Pleurosigma salinarum	0	0	0	0	0	0	0	0	0	0	0	0
Scenedesmus quadricauda	0	0	0	50	0	0	0	250	0	0	0	250
Scytonema crispum	0	250	0	0	0	50	0	0	0	0	50	0

Data code	k1-A_14	k2-A_14	k3-A_14	k4-A_14	k1-W_14	k2-W_14	k3-W_14	k4-W_14	k1-Sp_14	k2-Sp_14	k3-Sp_14	k4-Sp_14
Spirogyra no1	0	0	0	250	0	0	0	5000	0	0	0	25000
Spirogyra reinhardi	0	0	0	0	0	0	0	0	0	0	0	0
Spirogyra scripta	0	0	0	5000	0	0	0	250	0	0	0	0
Spirolina abbreviate	0	0	0	0	0	0	250	0	0	0	0	0
Stenopterobia delicatissima	0	0	0	0	0	0	0	0	0	0	0	0
Stigeoclonium tenue	0	0	0	250	0	0	0	0	0	0	0	0
Tabellaria flocculosa	5000	0	0	0	0	0	0	0	250	5000	0	0
Tolypothrix sp.	0	50	0	0	0	50	0	0	0	0	0	0
Trachelomonas intermedia	0	0	0	0	50	0	0	0	0	0	0	0
Tribonema vulgare	0	0	0	0	250	0	0	0	0	0	0	0
Ulothrix punctate	0	0	0	0	0	1000	0	0	250	0	0	0
Ulothrix zonata	250	0	0	0	1000	0	0	0	0	0	0	250
Zygnema no 1	50	0	0	0	250	0	0	0	50	0	0	0
Zygnema pectinatum	0	0	0	0	0	0	0	0	0	0	0	0

Data code	k1-S_15	k2-S_15	k3-S_15	k4-S_15	k1-A_15	k2-A_15	k3-A_15	k4-A_15	k1-Sp_15	k1-W_15
Achnanthes oblongella	0	25000	0	0	0	0	0	0	0	0
Achnanthes standeri	5000	0	0	0	1000	0	0	0	0	0
Aulacoseira granulata	0	0	0	0	0	0	0	0	0	0
Brachysira neoexillis	0	0	0	0	0	0	0	0	0	0
Calothrix inserta	0	0	0	0	0	250	0	0	0	0
Chroodactylon ramosum	0	0	0	0	0	1000	0	0	0	0
Cladophora glomerata	0	0	0	250	0	0	0	250	0	0
Closterium lineatum	0	0	250	0	0	0	5000	0	0	0
Cocconeis pediculus	0	0	0	0	0	0	0	1000	0	0
Cocconeis placentula	0	0	0	250	0	0	0	0	0	0

Data code	k1-S_15	k2-S_15	k3-S_15	k4-S_15	k1-A_15	k2-A_15	k3-A_15	k4-A_15	k1-Sp_15	k1-W_15
Craticula ambigua	0	0	0	0	0	0	0	250	0	0
Cymbella aspera	0	0	0	0	250	0	0	0	0	0
Cymbella cymbiformis	0	0	0	0	250	0	0	0	0	0
Epithemia adonata	0	0	0	0	0	0	0	250	0	250
Eunotia bilunaris	0	0	0	0	0	1000	250	0	0	1000
Eunotia exigua	0	250	50	1000	250	50	0	0	5000	250
Eunotia flexuosa	0	0	0	0	0	0	0	0	0	50
Eunotia formica	0	5000	0	0	1000	250	50	0	0	50
Eunotia incisa	1000	0	0	0	250	50	0	0	250	250
Eunotia minor	0	0	0	0	5000	250	0	0	250	0
Eunotia pectinalis	0	0	0	0	0	0	0	0	0	0
Flagilaria ulna	0	0	0	0	0	0	0	0	0	0
Frustulia saxonica	0	1000	0	0	0	0	0	0	250	0
Gyrosigma acuminatum	0	0	0	0	0	0	0	1000	0	0
Gyrosigma attenuatum	0	0	0	0	0	0	0	0	0	0
Kobayasiella subtillissma	0	0	0	0	250	0	0	0	0	0
Melosira varians	0	0	0	0	0	0	0	25000	0	0
Navicula angusta	250	0	0	0	0	0	0	0	0	0
Navicula cryptotenella	0	0	0	0	0	0	0	0	0	0
Navicula heimansioides	0	0	0	0	0	0	1000	0	0	0
Navicula notha Wallace	0	0	0	0	0	0	0	0	0	250
Navicula radiosa	0	0	1000	0	0	0	50	0	250	0
Navicula ranomafenensis	0	0	1000	250	0	0	0	0	0	0
Nitzchia iremissa	0	0	0	0	0	0	0	0	0	0
Nitzchia linearis	0	0	0	0	0	0	0	250	0	0
Nitzschia filiforms	0	0	0	0	0	0	0	0	0	0
Nostoc commune	0	0	50	0	0	250	0	0	0	250

Data code	k1-S_15	k2-S_15	k3-S_15	k4-S_15	k1-A_15	k2-A_15	k3-A_15	k4-A_15	k1-Sp_15	k1-W_15
Oedegonium minus	0	250	0	5000	0	0	0	0	0	0
Oedegonium no 1	0	0	0	0	0	250	0	250	0	0
Oedegonium no 2	0	0	0	50	0	0	0	0	0	0
Oocystis rupestris	0	0	0	50	0	0	0	0	0	0
Oscillatoria princeps	0	0	0	250	0	0	0	250	0	0
Oscillatoria sancta	0	0	0	0	0	0	0	0	0	0
Oscillatoria tenuie	0	0	0	0	0	0	0	5000	0	0
Pediastrum duplex	0	0	0	0	0	0	1000	0	0	0
Pinnularia divergens	0	0	0	250	0	0	0	0	0	0
Pinnularia viridiformis	0	250	0	0	0	0	0	250	0	0
Pleurosigma salinarum	0	0	0	1000	0	0	0	0	0	0
Scenedesmus quadricauda	0	0	0	1000	0	0	0	0	0	0
Scytonema crispum	0	0	0	0	0	0	0	0	0	0
Spirogyra no1	0	0	0	5000	0	0	0	0	0	0
Spirogyra reinhardi	0	0	0	0	0	0	0	0	0	0
Spirogyra scripta	0	0	0	0	0	0	0	25000	0	0
Spirolina abbreviate	0	0	0	250	0	0	50	0	0	0
Stenopterobia delicatissima	250	0	0	0	0	0	0	0	0	0
Stigeoclonium tenue	0	0	0	0	0	0	0	50	0	0
Tabellaria flocculosa	0	0	0	0	250	50	0	0	1000	50
Tolypothrix sp.	0	0	50	0	0	250	0	0	0	
Trachelomonas intermedia	0	0	0	0	250	0	0	0	0	50
Tribonema vulgare	0	0	0	0	0	50	0	0	0	50
Ulothrix punctate	0	0	0	0	0	0	0	0	250	
Ulothrix zonata	250	0	0	0	1000	0	0	0	0	250
Zygnema no 1	0	0	0	0	50	0	0	0	0	1000
Zygnema pectinatum	250	0	0	0	1000	0	0	0	1000	50

Data code	k2-W_15	k3-W_15	k4-W_15	k2-Sp_15	k3-Sp_15	k4-Sp_15	k1-S_16	k2-S_16	k3-S_16	k4-S_16
Achnanthes oblongella	0	0	0	0	0	0	0	0	0	0
Achnanthes standeri	50	0	0	1000	0	0	0	1000	0	0
Aulacoseira granulata	0	0	5000	0	0	0	0	0	0	0
Brachysira neoexillis	0	0	0	0	0	0	0	0	0	0
Calothrix inserta	1000	0	0	0	0	0	0	0	250	0
Chroodactylon ramosum	0	0	0	0	250	0	0	0	0	0
Cladophora glomerata	0	0	0	0	0	0	0	0	0	0
Closterium lineatum	0	50	0	0	0	0	0	0	50	0
Cocconeis pediculus	0	0	0	0	0	250	0	0	0	50
Cocconeis placentula	0	0	1000	0	0	0	0	0	0	250
Craticula ambigua	0	0	50	0	0	1000	0	0	0	50
Cymbella aspera	0	0	0	0	0	0	0	0	0	0
Cymbella cymbiformis	0	0	0	0	0	0	0	0	0	0
Epithemia adonata	0	0	0	0	0	0	250	50	50	0
Eunotia bilunaris	50	0	0	0	0	0	50	50	0	0
Eunotia exigua	0	50	0	1000	0	0	250	0	50	0
Eunotia flexuosa	0	0	0	250	50	0	50	0	0	0
Eunotia formica	250	250	0	0	1000	0	50	1000	50	0
Eunotia incisa	0	0	0	50	0	0	0	0	0	0
Eunotia minor	250	0	0	0	0	0	250	250	0	0
Eunotia pectinalis	0	0	0	0	0	0	0	0	0	0
Flagilaria ulna	0	0	5000	0	0	250	0	0	0	250
Frustulia saxonica	1000	0	0	0	0	0	0	250	0	0
Gyrosigma acuminatum	0	0	1000	0	0	50	0	0	0	0

Data code	k2-W_15	k3-W_15	k4-W_15	k2-Sp_15	k3-Sp_15	k4-Sp_15	k1-S_16	k2-S_16	k3-S_16	k4-S_16
Gyrosigma attenuatum	0	0	0	0	0	50	0	0	0	250
Kobayasiella subtillissma	0	0	0	0	0	0	0	0	0	0
Melosira varians	0	0	5000	0	0	1000	0	0	0	1000
Navicula angusta	0	0	0	0	0	0	0	0	0	0
Navicula cryptotenella	0	1000	0	0	250	0	0	0	250	0
Navicula heimansioides	1000		0	1000	0	0	0	5000	0	0
Navicula notha Wallace	0	0	0	0	0	0	250	0	0	0
Navicula radiosa	250	1000	0	250	250	0	0	250	50	0
Navicula ranomafenensis	0	0	0	0	250	0	0	0	0	0
Nitzchia iremissa	0	0	0	0	0	0	0	0	0	0
Nitzchia linearis	0	50	1000	0	0	0	0	0	0	250
Nitzschia filiforms	0	0	250	0	0	0	0	0	0	0
Nostoc commune	0	0	0	0	50	0	0	0	50	0
Oedegonium minus	0	0	0	0	0	0	0	0	0	0
Oedegonium no 1	0	0	0	0	0	250	0	0	0	0
Oedegonium no 2	0	0	250	0	0	250	0	0	0	250
Oocystis rupestris	0	0	1000	0	0	0	0	0	0	1000
Oscillatoria princeps	0	0	1000	0	0	0	0	0	0	1000
Oscillatoria sancta	0	0	50	0	0	250	0	0	0	250
Oscillatoria tenuie	0	0		0	0	0	0	0	0	0
Pediastrum duplex	0	250	50	0	0	0	0	0	250	0
Pinnularia divergens	0	250	0	0	0	0	0	0	250	0
Pinnularia viridiformis	0	0	0	0	0	0	0	0	0	0
Pleurosigma salinarum	0	0	0	0	0	0	0	0	0	0
Scenedesmus quadricauda	0	0	250	0	0	1000	0	0	0	250
Scytonema crispum	50	0	0	0	50	0	0	0	0	0
Spirogyra no1	0	0	0	0	0	0	0	0	0	0

Data code	k2-W_15	k3-W_15	k4-W_15	k2-Sp_15	k3-Sp_15	k4-Sp_15	k1-S_16	k2-S_16	k3-S_16	k4-S_16
Spirogyra reinhardi	0	0	1000	0	0	0	0	0	0	1000
Spirogyra scripta	0	0	0	0	0	5000	0	0	0	0
Spirolina abbreviate	0	250	50	0	0	0	0	0	0	50
Stenopterobia delicatissima	0	0	0	0	0	0	0	0	0	0
Stigeoclonium tenue	0	0	0	0	0	0	0	0	0	0
Tabellaria flocculosa	50	0	0	250	0	0	5000	250	0	0
Tolypothrix sp.	50	0	0	0	0	0	0	0	0	0
Trachelomonas intermedia	0	0	250	0	0	0	0	0	0	250
Tribonema vulgare	0	50	0	0	0	0	0	0	50	0
Ulothrix punctate	0	0	0	0	0	0	0	0	0	0
Ulothrix zonata	250	0	0	0	0	0	250	0	0	0
Zygnema no 1	0	250	0	0	0	0	0	0	0	0
Zygnema pectinatum	0	0	0	0	0	0	50	0	0	0

Appendix 4.4 Principle component loadings of the variables describing environmental factors measured over the sampling period at all sites. The values highlighted in bold explain the majority of the variation associated with each of the three primary axes (PC 1, 2 and 3) and PC4.

	Variable	PC1	PC2	PC3	PC4
	Variation explained %	**44.5**	**11**	**7.3**	**5.9**
Water chemistry	Sodium Na^{+}	**0.308**	-0.109	-0.016	0.031
	Calcium Ca^{2+}	**0.304**	-0.102	-0.010	-0.046
	Magnesium Mg^{2+}	**0.308**	-0.101	0.012	0.018
	Total Alkalinity	**0.293**	-0.117	0.089	-0.183
	Electrical conductivity (EC)	**0.310**	-0.091	-0.019	0.019
	pH	**0.305**	-0.060	-0.011	-0.010
	Chemical Oxygen Demand (COD)	**0.225**	-0.003	-0.166	**0.264**
	Silica Si	-0.160	0.094	-0.210	**-0.411**
	Benthic Chlorophyll *a* (Ben_chl-*a*)	-0.045	**-0.472**	0.207	0.242
	Water column Chlorophyll *a* (water_chl-*a*)	-0.017	**0.220**	0.101	0.061
	Temperature	-0.044	**0.322**	-0.059	**0.333**
	Water level	0.102	-0.155	0.045	**-0.626**
	Flow	-0.078	0.060	**-0.534**	0.090
Nutrients	Nitrite nitrate (NO_x-N)	0.099	-0.079	**0.461**	0.122
	Total Nitrogen (TN)	0.172	**0.375**	-0.018	0.108
	Total Phosphorus (TP)	0.027	**0.220**	**0.254**	-0.067
Macroinvertebrates	SASS	**-0.284**	-0.190	-0.063	-0.017
	Taxa	-0.199	**-0.357**	-0.186	-0.061
	ASPT	**-0.284**	0.028	0.047	-0.031
	Habitat	**-0.229**	0.037	**0.326**	0.104
	Grazers	**-0.226**	**-0.222**	**0.235**	0.112
	Predators	0.033	**-0.346**	**-0.311**	**0.307**

Appendix 4.5 Relationship between macroinvertebrate and algal assemblages and environmental variables at all sites based on a Euclidean Distance matrix, using the multivariate F-statistic (i.e. Pseudo-F).

'% var' indicates the percentage of samples variation explained by a variable. 'Cum. % var.' is the cumulative percentage variation explained for each additional co-variate in the sequential tests. Only significantly different ($p \leq 0.05$) relationships are shown

Variable		Adjusted R^2	SS(trace)	Pseudo-F	P	% var	Cum % var
Marginal Tests							
Physico-chemical	Na^{+}	-	299.58	23.393	0.0001	40.8	-
	Ca^{2+}	-	301.49	23.645	0.0001	40.1	-
	Mg^{2+}	-	299.6	23.396	0.0001	40.8	-
	Alkalinity	-	268.01	19.513	0.0001	36.5	-
	EC	-	302.86	23.829	0.0001	41.2	-
	pH	-	318.52	26.003	0.0001	43.3	-
	COD	-	210.52	13.647	0.0001	28.6	-
	Si	-	91.002	4.8045	0.0017	12.4	-
	Benthic (Chl-*a*)	-	57.562	2.889	0.027	7.8	-
	Water (Chl-*a*)	-	39.08	1.9093	0.0928	5.3	-
	Temperature	-	51.341	2.5533	0.0392	6.9	-
	Water level	-	72.745	3.7347	0.0134	9.8	-
	Flow	-	54.907	2.745	0.0324	7.4	-
Nutrients	NO_x-N	-	71.049	3.6383	0.0144	9.7	-
	TN	-	144.77	8.3394	0.0001	19.7	-
Macro-invertebrates	Grazers	-	189.88	11.843	0.0001	25.8	-
	Habitat	-	191.01	11.938	0.0001	25.9	-
	SASS	-	280.84	21.025	0.0001	38.2	-
	ASPT	-	273.06	20.098	0.0001	37.2	-
	Taxa	-	164.24	9.7837	0.0001	22.3	-
Sequential tests							
	+pH	0.4167	318.52	26.003	0.0001	43.3	43.3
	+Taxa	0.47945	55.735	5.0985	0.0001	7.58	50.9
	+ Benthic (Chl-a)	0.52611	42.289	4.2494	0.0001	5.7	56.6
	+Temp	0.57338	40.727	4.546	0.0001	5.54	62.2
	+Water (Chl-a)	0.61728	36.612	4.5553	0.0001	4.98	67.2
	$+NO_x$-N	0.66311	35.947	5.0811	0.0001	4.89	72.1
	+Water level	0.71066	35.04	5.7669	0.0002	4.76	76.9
	+TP	0.75777	32.787	6.4456	0.0132	4.46	81.3
	+Si	0.80665	31.775	7.8259	0.0001	4.32	85.6
	+Flow	0.85568	29.8	9.8328	0.0001	4.05	89.7
	+COD	0.88238	16.484	6.6736	0.0001	2.24	91.9
	+TN	0.9089	15.282	7.988	0.0001	2.07	94.01
	+Habitat	0.93371	13.377	9.6101	0.0001	1.82	95.8
	+Grazers	0.95633	11.365	12.393	0.0001	1.54	97.4
	+ASPT	0.97426	8.4485	15.631	0.0001	1.149	98.5
	$+Na^{+}$	0.9788	2.3501	5.2782	0.0002	3.197	98.8
	+Alkalinity	0.98246	1.8308	4.9713	0.0007	2.49	99.1
	$+Ca^{2+}$	0.98496	1.258	3.9817	0.0054	1.7	99.3
	$+Mg^{2+}$	0.98605	0.68462	2.3374	0.0503	9.31	99.4

Marginal tests show how much variation each variable explains when considered alone, ignoring other variables. Sequential tests explain the cumulative variation attributed to each variable fitted to the model in the order specified, taking previous variables into account.

Appendix 5.1 Presence/absence of species at the four sampling sites at all transects. Code refers to the first four letters of the species family name

Code		Site 1			Site 2			Site 3			Site 4		
	Species per study site	**1.1.**	**1.2**	**1.3**	**2.1**	**2.2**	**2.3**	**3.1**	**3.2**	**3.3**	**4.1**	**4.2**	**4.3**
Faba	*Acacia dealbata*				*								
Faba	*Acacia mearnsii*				*	*	*	*	*		*		
Faba	*Acacia melanoxylon*					*		*					
Faba	*Acacia saligna*												
Phyl	*Adrachne ovalis*		*	*		*							
Cyat	*Alsophila capensis*	*		*									
Malv	*Anisodontea scabrosa*												*
Icac	*Apadytes dimidiata*			*									
Irid	*Aristea ensifolia*	*	*		*		*					*	
Aspa	*Asparagus aethiopicus*												
Aspa	*Asparagus scandens*			*	*								
Aspa	*Asparagus setaceus*	*	*	*	*		*	*					
Aspl	*Asplenium gemmiferum*			*									
Blec	*Blechnum attenuatum*			*								*	
Blec	*Blechnum capense*	*		*	*	*							
Blec	*Blechnum punctulatum*		*		*		*						
Budd	*Buddleja salviifolia*								*			*	
Ruta	*Calodendrum capense*				*								
Ranu	*Calopsis paniculata*		*										
Rubi	*Canthium inerme*		*		*	*							
Rubi	*Canthium mundianum*	*	*										
Rubi	*Canthium ventosum*					*							
Rubi	*Galopina circaeoides*				*								
Cype	*Carex aethopica*			*				*	*			*	
Apoc	*Carissa bispinosa*			*		*	*						
Cype	*Carpha glomerata*				*	*		*	*				
Apoc	*Cassine peragua*										*	*	
Apia	*Centella asiatica*							*					
Apia	*Centella coriacea*				*								
Olea	*Chionanthus foveolata*	*											
Anth	*Chlorophytum comosum*			*	*	*							
Aste	*Cirsium vulgare*												*
Rose	*Cliffortia adorata*									*		*	
Euph	*Clutia pulchella*	*	*				*	*					
Comm	*Commelina*							*	*			*	
Aste	*Conyza scabrida*												*
Cras	*Crassula orbicularis*										*		
Cuno	*Cunonia capensis*	*											
Curt	*Curtisia dentate*			*									
Cyat	*Cyathea capensis*		*				*						
Cype	*Cyperaceae Isolepis*						*						
Cype	*Cyperaceae sp.*											*	
Dios	*Dioscorea mundaii*	*		*	*								
Eben	*Diospyros dichrophylla*					*							
Eben	*Diospyros glabra*		*										
Poac	*Ehrharta ramosa*	*						*	*				
Cela	*Elaeodendron croceum*	*	*										
Cype	*Ficinia sp*	*	*		*	*	*						
Cype	*Ficinia trispicata*							*					
Thym	*Gnidia denudata*	*											
Apoc	*Gonioma kamassi*	*		*	*	*							
Celas	*Gymnosporium remorosa*		*									*	
Stil	*Halleria lucida*							*					

Code		Site 1			Site 2			Site 3			Site 4		
	Species per study site	**1.1.**	**1.2**	**1.3**	**2.1**	**2.2**	**2.3**	**3.1**	**3.2**	**3.3**	**4.1**	**4.2**	**4.3**
Denn	*Histiopteris incisa*				*		*						
Denn	*Hypolepis sparsisora*	*											
Aqui	*Ilex mitis*	*											
Rest	*Ischiolepis*				*			*					
Junc	*Juncus capensis*							*	*				*
Junc	*Juncus effuses*			*			*		*				*
Junc	*Juncus lomatophyllus*				*	*	*						
Junc	*Juncus aethiopicus*											*	
Phyl	*Lachnostylis sp.*			*									
Urti	*Laportea peduncularis*											*	
Poac	*Microstegium nudum*			*									
Stil	*Nuxia floribunda*			*				*					
Ochn	*Ochna arborea*		*			*		*					
Laur	*Ocotea bullata*	*	*	*									
Olea	*Olea capensis*	*	*	*	*		*						
Olin	*Olinia ventosa*		*										
Poac	*Oplismenus hirtellus*	*	*	*	*							*	
Oxal	*Oxalis incarnata*	*	*	*	*	*	*	*	*				
Poac	*Panicum maximum*											*	
Gera	*Pelargonium zonale*												*
Poac	*Pennisetum*							*				*	
Pipe	*Peperomia retusa*		*										
Poly	*Persicaria decipiens*										*		
Cuno	*Platylopus trifoliatus*			*		*							
Aste	*Plecostachys*					*							
Lami	*Plectranthus fruticosus*	*	*		*	*	*					*	
Podo	*Podocarpus falcatus*		*										
Poda	*Podocarpus latifolius*	*		*		*							
Dryo	*Polystichum pungens*		*										
Pota	*Potamogetan sp*											*	*
Faba	*Psoralea pinnata*							*				*	
Denn	*Pteridium aquilinum*				*			*	*	*			
Cela	*Pterocelastrus rostratus*								*				
Cela	*Pterocelastrus*												
Myrs	*Rapanea melanophloes*	*			*	*			*				
Rham	*Rhamnus prinoides*												
Vita	*Rhoicissus tridentata*			*									
Vita	*Rhoicissus digitata*												*
Vita	*Rhoicissus tomentosa*											*	
Rubi	*Rothmarnia capensis*			*									
Rosa	*Rubus pinnatus*	*			*				*	*			
Dryo	*Rumohra adiantiformis*			*	*	*	*				*		
Apia	*Sanicula elata*										*		
Cype	*Schoenoxiphium altum*				*								
Flac	*Scolopia zeyheri*										*	*	
Rham	*Scutia Myrtira*	*		*									
Apoc	*Secamone alipini*								*				
Aste	*Senecio deltoideus*		*				*	*		*			
Aste	*Senecio quiquelobus*	*	*									*	*
Poac	*Setaria megaphylla*										*	*	
Sola	*Solanum mauritianum*									*			
Malv	*Sparmannia africana*	*							*				
Gesn	*Streptocarpus rexii*											*	
Aste	*Taraxacum afficinale*							*					
Osmu	*Todea barbara*			*									
Hamm	*Trichocladus crinitus*	*	*	*		*	*	*				*	
Sali	*Trimeria grandifolia*	*											
Urti	*Urtica urens*											*	
Urti	*Urtica dioica*												*

Code		Site 1			Site 2			Site 3			Site 4		
	Species per study site	**1.1.**	**1.2**	**1.3**	**2.1**	**2.2**	**2.3**	**3.1**	**3.2**	**3.3**	**4.1**	**4.2**	**4.3**
Aste	*Vernonia mespiliformis*			*					*				
Haem	*Wachendorfia*											*	*
Arac	*Zantedeschia aethiopica (L.) Spreng*				*			*					
Total		28	25	29	29	22	17	23	15	5	8	26	16
Grand total per setting			82			68			43			50	

Appendix 5.2 Descriptions of plant species in the study area. Four letter family code name in brackets as per Appendix 5.1.

Family and species name	Growth form	Life cycle	Indigenous or Exotic
Anthericaceae			
Chlorophytum comosum	Herb	Perennial	Indigenous
Apiaceae (Apia)			
Centella asiatica	Herb/Climber	Perennial	Indigenous
Centella coriacea	Herb	Perennial	Endemic
Sanicula elata	Herb	Perennial	Indigenous
Apocynaceae (Apoc)			
Carissa bispinosa	Shrub	Perennial	Indigenous
Gonioma kamassi	Shrub/small tree	Perennial	Indigenous
Secamone alipini	Climber	Perennial	Indigenous
Aquifoliaceae (Aqui)			
Ilex mitis	Tree/Shrub	Perennial	Indigenous
Araceaee (Arac)			
Zantedeschia aethiopica (L.) Spreng	Herb	Perennial	Indigenous
Asparagaceae (Aspa)			
Asparagus aethiopicus	Climber/herb	Perennial	Indigenous
Asparagus setaceus	Shrub	Perennial	Indigenous
Asparagus scandens	Herb/Climber	Perennial	Indigenous
Aspleniaceae			
Asplenium gemmiferum	Herb/Geophyte/Lithophyte	Perennial	Indigenous
Asteraceae (Aste)			
Senecio quiquelobus	Climber	Perennial	Indigenous
Cirsium vulgare	Herb	Annual/biennial	Exotic
Conyza scabrida	Shrub	Perennial	Exotic
Senecio deltoideus	Herb/scrambler	Perennial	Indigenous
Plecostachys serpillifolia	Shrub	Perennial	Indigenous
Taraxacum afficinale	Herb	Perennial	Naturalised
Vernonia mespiliformis	Shrub	Perennial	Indigenous
Blechnaceae (Blec)			
Blechnum capense	Herb	Perennial	Indigenous
Blechnum punctulatum	Herb	Perennial	Indigenous
Blechnum attenuatum	Herb/Geophyte/Epiphyte/Lithophyte	Perennial	Not endemic
Buddlejaceae (Budd)			
Buddleja salviifolia	Tree	Perennial	Indigenous
Celastraceae (Celas)			
Cassine peragua	Tree	Perennial	Indigenous
Elaeodendron croceum	Tree	Perennial	Indigenous
Gymnosporium remorosa	Shrub	Perennial	Indigenous
Crassulaceae (Crass)			
Crassula orbicularis	Succulent herb	Perennial	Indigenous
Cunoniaceae (Cuno)			
Platylopus trifoliatus	Tree	Perennial	Endemic
Cunonia capensis	Tree	Perennial	Indigenous
Curtisiaceae (Curt)			
Curtisia dentata	Tree	Perennial	Indigenous
Celastraceae (Cela)			
Elaeodendron croceum	Tree	Perennial	Indigenous
Pterocelastrus tricuspidatus	Tree	Perennial	Endemic
Commelinaceae Comm)			
Commelina benghalensis	Herb	Annual	Indigenous
Cyatheaceae (Cyat)			
Cyathea capensis	Tree	Perennial	Indigenous
Alsophila capensis	Tree	Perennial	Indigenous
Cyperaceae (Cype)			

Family and species name	Growth form	Life cycle	Indigenous or Exotic
Carpha glomerata	Herb/Emergent	Perennial	Indigenous
Cyperaceae sp.	Helophytic herb	Annual/Perennial	Indigenous
Cyperaceae Isolepis	Helophytic herb	Annual/Perennial	Indigenous
Alsophila capensis	Tree	Perennial	Indigenous
Cyathea capensis	Tree	Perennial	Indigenous
Carex aethopica	Sedge	Perennial	Indigenous
Ficinia sp.	Herb	Perennial	Indigenous
Ficinia trispicata	Sedge	Perennial	Indigenous
Schoenoxiphium altum	Herb/Cyperoid	Perennial	Endemic
Dennstaedtiaceae (Denn)			
Hypolepis sparsisora	Herb/Helophyte/Hydrophyte	Perennial	Indigenous
Histiopteris incisa	Herb/Geophyte/Hydrophyte	Perennial	Indigenous
Pteridium aquilinum	Herb/Geophyte	Perennial	Cosmopolitan
Dioscoreaceae (Dios)			
Dioscorea mundaii	Herb/climber	Perennial	Not endemic
Dryopteridaceae (Dryo)			
Polystichum pungens	Herb/geophyte	Perennial	Indigenous
Rumohra adiantiformis	Herb	Perennial	Not endemic
Ebenaceae (Eben)			
Diospyros glabra	Shrub	Perennial	Endemic
Diospyros dichrophylla	Tree	Perennial	Indigenous
Eriospermaceae (Erio)			
Eriospermum sp.	Geophytic herb	Perennial	Indigenous
Euphorbiaceae (Euph)			
Clutia pulchella	Shrub/Dwarf shrub/Herb	Perennial	Indigenous
Fabaceae (Faba)			
Psoralea pinnata	Tree/shrub	Perennial	Indigenous
Acacia melanoxylon	Tree	Perennial	Exotic
Acacia saligna	Tree	Perennial	Exotic
Acacia dealbata	Tree	Perennial	Exotic
Flacourtiaceae (Flac)			
Scolopia zeyheri	Tree	Perennial	Indigenous
Geraniaceae (Gera)			
Pelargonium zonale	Shrub	Perennial	Indigenous
Gesneriaceae (Gesn)			
Streptocarpus rexii	Herb	Perennial	Indigenous
Haemodoraceae (Haem)			
Wachendorfia thyrsiiflora	Geophyte	Perennial	Indigenous
Hamamelidaceae (Hamm)			
Trichocladus crinitus	Tree/Shrub	Perennial	Indigenous
Icacinaceae (Icac)			
Apadytes dimidiata	Tree	Perennial	Indigenous
Iridaceae (Irid)			
Aristea ensifolia	Herb	Perennial	Indigenous
Juncaceae (Junc)			
Juncus capansis	Herb	Perennial	Cosmopolitan
Juncus lomatophyllus	Herb/Hydrophyte/Hyperhydate	Perennial	Cosmopolitan
Juncus effusus	Herb/Helophyte	Perennial	Cosmopolitan
Juncus aethiopicus	Herb	Perennial	Cosmopolitan
Lamiaceae (Lami)			
Plectranthus fruticosus	Shrub/Herb	Perennial	Indigenous
Lauraceae (Laur)			
Ocotea bullata	Tree	Perennial	Indigenous
Malvaceae (Malv)			
Anisodontea scabrosa	Shrub	Perennial	Indigenous
Sparmannia africana	Shrub	Perennial	Indigenous
Myrsinaceae (Myrs)			
Rapanea melanophloes	Tree	Perennial	Indigenous
Ochnaceae (Ochn)			

Family and species name	Growth form	Life cycle	Indigenous or Exotic
Ochna arborea	Tree	Perennial	Indigenous
Oliniaceae (Olin)			
Olinia ventosa	Tree/Shrub	Perennial	Indigenous
Oleaceae (Olea)			
Chionanthus foveolata	Tree	Perennial	Indigenous
Olea capensis	Tree	Perennial	Indigenous
Osmundaceae (Osmu)			
Todea barbara	Herb	Perennial	Indigenous
Oxalidaceae (Oxal)			
Oxalis incarnata	Geophyte	Perennial	Indigenous
Piperaceae (Pipe)			
Peperomia retusa	Herb/succulent	Perennial	Indigenous
Poaceae (Poac)			
Ehrharta ramosa	Graminoid	Perennial	Indigenous
Setaria megaphylla	Graminoid	Annual/Perennial	Indigenous
Microstegium nudum	Graminoid	Annual	Indigenous
Oplismenus hirtellus	Graminoid/Scrambler	Perennial	Indigenous
Panicum maximum	Graminoid	Perennial	Indigenous
Pennisetum clandestinum	Graminoid	Perennial	Exotic
Podocarpaceae (Podo)			
Podocarpus latifolius	Tree	Perennial	Indigenous
Podocarpus falcatus	Tree	Perennial	Indigenous
Potamogetonaceae (Pota)			
Potamogetan sp	Hydrophytic herb	Annual	Indigenous
Phyllanthaceae (Phyl)			
Adrachne ovalis	Shrub/small tree	Perennial	Indigenous
Lachnostylis sp.	Tree	Perennial	Endemic
Polygonaceae (Poly)			
Persicaria decipiens	Helophytic herb	Annual	Indigenous
Restionaceae (Rest)			
Cheilanthes hirta	Herb	Perennial	Indigenous
Cheilanthes viridis	Herb	Perennial	Indigenous
Pellaea calomelanos	Herb	Perennial	Indigenous
Ischiolepis subverticullata	Reed	Perennial	Endemic
Ranunculaceae (Ranu)			
Calopsis paniculata	Shrub/restioid	Perennial	Indigenous
Rhamnaceae (Rham)			
Rhamnus prinoides	Tree	Perennial	Indigenous
Scutia Myrtira	Climber	Perennial	Indigenous
Rosaceae (Rosa)			
Cliffortia adorata	Shrub	Perennial	Indigenous
Rubus pinnatus	Shrub/Scrambler	Perennial	Indigenous
Rubiaceae (Rubi)			
Rothmarnia capensis	Tree	Perennial	Indigenous
Canthium mundianum	Tree	Perennial	Indigenous
Canthium inerme	Tree	Perennial	Indigenous
Canthium ventosum	Tree	Perennial	Indigenous
Galopina circaeoides	Herb	Perennial	Not endemic
Rutaceae (Ruta)			
Calodendrum capense	Tree	Perennial	Indigenous
Salicaceae (Sali)			
Thesium sp.1	Herb/shrub	Perennial	Indigenous
Sapindaceae (Sapi)			
Trimeria grandifolia	Tree/Shrub	Perennial	Indigenous
Solanaceae (Sola)			
Solanum mauritianum	Tree/Shrub	Perennial	Exotic
Stilbaceae (Stil)			
Halleria lucida	Tree/Shrub	Perennial	Indigenous

Family and species name	Growth form	Life cycle	Indigenous or Exotic
Nuxia floribunda	Tree/Shrub	Perennial	Indigenous
Thymelaeaceae (Thym)			
Gnidia denudata	Shrub	Perennial	Indigenous
Urticaceae (Urti)			
Urtica urens	Herb	Annual	Exotic
Urtica dioica	Herb	Perennial	Exotic
Laportea peduncularis	Herb	Annual	Indigenous
Vitaceae (Vita)			
Rhoicissus digitata	Climber	Perennial	Indigenous
Rhoicissus tomentosa	Climber	Perennial	Indigenous
Rhoicissus tridenta	Climber	Perennial	Indigenous

Appendix 5.3 Relationship between vegetation assemblages and environmental variables at all sites transects based on a Euclidean Distance matrix, using the multivariate F-statistic (i.e. Pseudo-F).

'% var' indicates the percentage of samples variation explained by a variable. 'Cum. % var.' is the cumulative percentage variation explained for each additional co-variate in the sequential tests.

Variable		Adjusted R^2	SS (trace)	Pseudo-F	P	% var	Cum % var
Marginal Tests							
Physico-chemical	Total Nitrogen (TN)	-	115.46	13.782	0.0001	17.49	-
	Total Phosphorus (TP)	-	171.02	22.734	0.0001	25.91	-
	Phosphate (PO_4-P)	-	151.92	19.436	0.0001	23.1	-
	pH	-	139.83	17.473	0.0001	21.18	-
	Organic carbon (C)	-	120.13	14.464	0.0001	18.20	-
	Elevation above (Elev)	-	143.83	18.112	0.0001	21.79	-
	Distance from active channel thalweg (Dist)	-	69.345	7.6312	0.0001	10.50	-
Grain sizes	Sand	-	130.63	16.04	0.0001	19.79	-
	Mud/silt	-	82.725	9.3146	0.0001	12.53	-
	Gravel	-	128.59	15.728	0.0001	19.48	-
Sequential tests							
	+TP	0.24773	171.02	22.734	0.0001	25.91	25.91
	+pH	0.42169	118.86	20.552	0.0001	18.01	43.92
	+Gravel	0.57709	106.69	24.517	0.0001	15.71	59.63
	+ Dist	0.6799	67.975	21.236	0.0001	10.29	69.93
	+Mud/silt	0.77064	58.552	25.529	0.0001	8.87	78.80
	+TN	0.83717	42.208	25.921	0.0001	6.39	85.19
	+C	0.89	32.802	29.82	0.0001	4.96	90.16
	+Elevation above (Elev)	0.93494	27.164	41.753	0.0002	4.11	94.28
	+Sand	0.97102	21.214	73.198	0.0132	3.21	97.49
	+PO_4-P	1	16.52	0	1	2.5	1

Marginal tests show how much variation each variable explains when considered alone, ignoring other variables. Sequential tests explain the cumulative variation attributed to each variable fitted to the model in the order specified, taking previous variables into account.

Appendix 5.4 Chemical parameters, rainfall and volume of runoff measured at runoff plots (2014-2017). Alk = Alkalinity, SS = Suspended solids. * Refers to plot located beneath predominantly alien trees at site K2

Station	Code	Season	Date	Alk	EC	pH	Na^+	NH_4^+	NO_x	TN	PO_4^{3-}	TP	DOC	Turbidity	SS	Volume (L)	Rainfall (mm)
Forest	K1	Wet	01/12/2014	4.2	8				2.4				15			0.04	139
Forest	K1	Wet	01/02/2015	-	-	-	-	-	-	57	-	4.2	-	-	-	0.02	34.6
Forest	K1	Dry	01/08/2015	-	-	-	-	-	-	83	8.5	-	-	-	-	0.1	253.6
Forest	K1	Dry	01/06/2016	0.5	-	4.4	-	8.6	33	-	2.9	3.1	-	-	-	0.1	132.6
Forest	K1	Wet	01/10/2016	27	18	7.3	8.3	0.08	6.5	5	32	3.7	15	14	89	1.8	87.8
Forest	K1	Wet	01/02/2017	0.3	54	4	21	5.2	44		5.6	7	35	44	-	0.2	29.8
Forest	K1	Wet	01/02/2017	0.3	54	4	20	13	30		3	3.1	22	9.7	-	0.175	29.8
Forest	K1	Dry	01/05/2017	27	28	7.4	16	0.4	7.4	6	1.2	1.5	27	20	25	0.6	42.8
Forest	K1	Dry	01/05/2017	4.9	20	6.1	12	1	7.9	4	0.87	0.96	20	7.5	-	1.6	70
			Average	**9.2**	**30.3**	**5.5**	**15.5**	**4.7**	**18.7**	**31.0**	**7.7**	**3.4**	**22.3**	**19.0**	**57.0**	**0.5**	**91.1**
			Std. dev	**12.3**	**19.4**	**1.6**	**5.4**	**5.3**	**16.5**	**36.8**	**11.0**	**2.0**	**7.7**	**14.7**	**45.3**	**0.7**	**74.1**
			Min	**0.3**	**8.0**	**4.0**	**8.3**	**0.1**	**2.4**	**4.0**	**0.9**	**1.0**	**15.0**	**7.5**	**25.0**	**0.02**	**29.8**
			Max	**27.0**	**54.0**	**7.4**	**21.0**	**13.0**	**44.0**	**83.0**	**32.0**	**7.0**	**35.0**	**44.0**	**89.0**	**1.8**	**253.6**
Semi	K2	Wet	01/10/2014	6.5	8	6.5	3.4	2.5	0.6	30	-	0.84	23	209	1044	1	93.6
Semi	K2	Wet	01/10/2014	6.6	5	6.5	2.8	0.56	0.1	11	-	0.44	12	225	643	1.3	93.6
Semi	K2	Wet	01/12/2014	4.6	3	-	-	0.9	0.1	-	-	-	9.8	-	-	3.5	112.4
Semi	K2	Wet	01/12/2014	2.3	6	-	-	2.2	4.6	-	-	-	14	-	-	2.3	112.4
Semi	K2	Dry	01/08/2015	3.9	11	5	5.1	4.1	4.2	11	0.92	1.2	16	-	56	0.45	72
Semi	K2	Wet	01/10/2015	4.7	3	5.4	2.3	0.72	1.3	2	0.19	0.29	5.2	6.9	-	0.4	127.2
Semi*	K2*	Wet	01/10/2015	2.1	64	4.7	40	30	11	-	25	0.05	-	-	-	0.1	127.2
Semi	K2	Wet	01/03/2016	0.5	56	3.9	17	24	27	-	4.6	7.5	33	115	133	0.35	108
Semi*	K2*	Wet	01/03/2016	-	98	-	-	31	53	-	12	-	89	-	-	0.05	270.2
Semi	K2	Wet	01/03/2016	-	-	-	-	-	-	-	17	-	-	-	-	8.2	108
Semi	K2	Dry	01/06/2016	-	-	-	-	16	19	-	5.8	-	20	-	-	0.2	24.6

Station	Code	Season	Date	Alk	EC	pH	Na^+	NH_4^+	NO_x	TN	PO_4^{3-}	TP	DOC	Turbidity	SS	Volume (L)	Rainfall (mm)
Semi	K2	Dry	01/06/2016	-	12	-	5.5	6.9	4.6	-	1.4	1.4	8.5	43	-	0.5	24.6
Semi	K2	Wet	01/02/2017	0.5	16	4.5	10	2.7	5.7	-	0.64	0.91	12	24	34	0.4	29.8
Semi	K2	Wet	01/02/2017	0.5	17	4.5	11	2.7	6.3	5	0.64	0.78	12	8.5	121	1.4	29.8
Semi	K2	Wet	01/02/2017	0.5	74	3.7	23	38	49	-	8	9.8	54	25	-	0.1	29.8
Semi	K2	Dry	01/05/2017	14	70	6.2	59	22	14	39	9.3	10	93	13	62	0.8	70
Semi	K2	Dry	01/05/2017	2.6	25	5.1	15	3.2	5.6	10	0.67	0.89	24	5.2	77	1	51.4
Semi	K2	Dry	01/05/2017	3.6	25	5.3	15	4.5	8.1	10	0.85	0.97	34	6.1	85	1.2	51.4
			Average	**3.8**	**30.8**	**5.1**	**16.1**	**11.3**	**12.6**	**14.8**	**6.2**	**2.7**	**28.7**	**61.9**	**250.6**	**1.3**	**85.3**
			Std. dev	**3.6**	**30.8**	**0.9**	**16.6**	**12.7**	**16.1**	**12.8**	**7.5**	**3.7**	**27.2**	**83.0**	**352.1**	**1.9**	**59.2**
			Min	**0.5**	**3.0**	**3.7**	**2.3**	**0.6**	**0.1**	**2.0**	**0.2**	**0.1**	**5.2**	**5.2**	**34.0**	**0.1**	**24.6**
			Max	**14.0**	**98.0**	**6.5**	**59.0**	**38.0**	**53.0**	**39.0**	**25.0**	**10.0**	**93.0**	**225.0**	**1044.0**	**8.2**	**270.2**
Degraded	K3	Wet	01/02/2015	-	-	-	-	-	-	36	-	8.8	-	-	790	0.03	104.6
Degraded	K3	Dry	01/05/2015	-	72	-	-	0.48	23	-	6.4		-	-	-	0.02	98.4
Degraded	K3	Wet	01/10/2015	2.5	11	4.9	9.7	0.14	2.2	6	0.05	0.45	18	23	45	0.75	127.2
Degraded	K3	Dry	01/05/2017	4	40	5.2	33	1.7	10	9	1.5	1.7	46	18	47	3.2	51.4
Degraded	K3	Wet	01/03/2016	23	110	8.4	34	54	0.4	-	0.26	4.8	77	-	433	0.4	108
Degraded	K3	Dry	01/06/2016	-	-	-	-	1	0.1	-	0.15	-	-	-	-	0.05	24.6
Degraded	K3	Dry	01/06/2016	-	-	-	-	1.2	0.1	-	4	-	-	-	-	0.1	24.6
Degraded	K3	Dry	01/06/2016	41	47	7.7	28	0.69	14		4.1	7.3	40	113	-	0.72	24.6
Degraded	K3	Wet	01/10/2016	23	62	6.9	55	12	7.3	18	3.3	3.3	50	6.4		0.5	87.8
Degraded	K3	Wet	01/10/2016	-	-	-	-	-	-	-	-	-	-	-	183	0.05	87.8
Degraded	K3	Wet	01/10/2016	-	48	4.3	38	4.4	11	14	2.1	3	55	33	97	1	87.8
Degraded	K3	Wet	01/02/2017	3.6	27	5.3	23	2.2	2.9	6	0.69	1.3	30	7.6	168	7.3	29.8
Degraded	K3	Wet	01/02/2017	0.3	30	4.1	25	4.3	7	8	1.9	2.3	37	15	26	3.7	29.8
Degraded	K3	Wet	01/02/2017	0.3	84	4.5	76	6.1	21	-	4.8	5.7	39	55	26	0.3	29.8
			Average	**12.2**	**53.1**	**5.7**	**35.7**	**7.4**	**8.3**	**13.9**	**2.4**	**3.9**	**43.6**	**33.9**	**201.7**	**1.3**	**65.4**

Station	Code	Season	Date	Alk	EC	pH	Na^+	NH_4^+	NO_x	TN	PO_4^{3-}	TP	DOC	Turbidity	SS	Volume (L)	Rainfall (mm)
			Std. dev	**15.0**	**29.5**	**1.6**	**19.4**	**15.1**	**7.9**	**10.7**	**2.1**	**2.7**	**16.6**	**35.6**	**255.5**	**2.1**	**38.0**
			Min	**0.3**	**11.0**	**4.1**	**9.7**	**0.1**	**0.1**	**6.0**	**0.1**	**0.5**	**18.0**	**6.4**	**26.0**	**0.02**	**24.6**
			Max	**41.0**	**110.0**	**8.4**	**76.0**	**54.0**	**23.0**	**36.0**	**6.4**	**8.8**	**77.0**	**113.0**	**790.0**	**7.3**	**127.2**
Pasture	K2	Wet	01/10/2014	12	7	7.3	3.9	1.8	0.3	4		0.25	12	14	377	1.5	366.2
Pasture	K2	Dry	01/05/2015	-	18	-		4.7	5.1	-	0.65	-	-	-	-	0.03	98.4
Pasture	K2	Wet	01/10/2015	0.5	8	4.4	5.8	0.05	0.1	-	1.7	3.3	24	70	-	0.35	386.6
Pasture	K2	Wet	01/10/2015	57	16	7.5	6.7	0.13	0.1	-	1.7	1.3	19	26	-	0.15	386.6
Pasture	K2	Wet	01/02/2017	16.5	7.5	9.35	6.75	0.315	0.35	6.5	0.43	0.965	23	28.5	95.5	5.7	231
Pasture	K2	Wet	01/03/2016	22	96	6.1	70	12	38	-	-	23	90	-	149	0.2	270.2
Pasture	K2	Wet	01/03/2016	1948	485	9	230	231	2.1	-	4.4	6.1	107	-	177	0.015	270.2
Pasture	K2	Dry	01/06/2016	-	520	9.1	-	114	321		383	7.6	118	88	-	0.5	83.6
Pasture	K2	Dry	01/06/2016	-	-	-	-	109	46	-	8.3	-	-	-	-	0.05	83.6
Pasture	K2	Dry	01/06/2016	-	-	-	-	25	95	-	37	-	-	-	-	0.05	83.6
Pasture	K2	Dry	01/06/2016	2118	-	-	169	280	0.1	-	7.6	-	-	-	-	0.01	83.6
Pasture	K2	Wet	01/10/2016	3.9	7	5.9	4.8	0.35	3.4	2	0.77	1	5.1	2.3	9	8	231.8
Pasture	K2	Wet	01/10/2016	1.5	16	4.8	6.7	4.1	9.2	7	3.5	3.6	11	6.7	34	5.5	231.8
Pasture	K2	Wet	01/10/2016	2.5	11	5.2	6.4	2.5	5	4	3.3	3.3	8	1.6	5	6.8	231.8
Pasture	K2	Dry	01/05/2017	-	-	-	8.2	0.09	0.09	-	0.82	-	36	-	39	0.1	70
Pasture	K2	Dry	01/05/2017	50	29	7.5	29	0.39	0.09	-	0.33	8.1	41	136	-	0.4	70
Pasture	K2	Dry	01/05/2017	31	32	7.7	16	0.28	15	3	1.6	1.9	14	5.8	-	2	70
			Average	**355.2**	**96.3**	**7.0**	**43.3**	**46.2**	**31.8**	**4.4**	**30.3**	**5.0**	**39.1**	**37.9**	**110.7**	**1.8**	**191.1**
			Std. dev	**784.7**	**181.9**	**1.7**	**72.7**	**87.0**	**78.6**	**2.0**	**98.0**	**6.2**	**39.4**	**45.4**	**125.1**	**2.8**	**118.6**
			Min	**0.5**	**7.0**	**4.4**	**3.9**	**0.1**	**0.1**	**2.0**	**0.3**	**0.3**	**5.1**	**1.6**	**5.0**	**0.0**	**70.0**
			Max	**2118.0**	**520.0**	**9.4**	**230.0**	**280.0**	**321.0**	**7.0**	**383.0**	**23.0**	**118.0**	**136.0**	**377.0**	**8.0**	**386.6**
Pasture	K3	Wet	01/12/2014	1.5	8	-	-	-	2.2	-	-	-	15	-	-	1.7	195
Pasture	K3	Wet	01/10/2014	0.4	48	4.3	38	8.7	7.9	18	-	2.1	49	138	33	3	366.2

Station	Code	Season	Date	Alk	EC	pH	Na^{+}	NH_4^{+}	NO_x	TN	PO_4^{3-}	TP	DOC	Turbidity	SS	Volume (L)	Rainfall (mm)
Pasture	K3	Dry	01/05/2015	0.5	21	4.4	15	4.3	4.7	7	1.5	1.8	14	27	31	1.7	349.2
Pasture	K3	Wet	01/10/2015	6	11	5.4	11	0.07	0.1	21	4.2	6	41	130	233	0.9	386.6
Pasture	K3	Wet	01/03/2016	80	21	7.3	12	0.08	0.1	-	0.07	4.4	55	-	-	0.5	270.2
Pasture	K3	Wet	01/03/2016	45	25	7.4	27	0.05	0.1	-	0.17	4.6	78	-	-	0.35	270.2
Pasture	K3	Dry	01/06/2016	-	-	-	-	70	68	-	28	11	146	-	-	0.1	83.6
Pasture	K3	Dry	01/06/2016	-	76	-	-	21	35	-	2.6	-	125	-	-	0.05	83.6
Pasture	K3	Dry	01/05/2017	-	-	-	160	7.1	107	-	39	-	200	-	-	0.1	70
			Average	**22.2**	**30.0**	**5.8**	**43.8**	**13.9**	**25.0**	**15.3**	**10.8**	**5.0**	**80.3**	**98.3**	**99.0**	**0.9**	**230.5**
			Std. dev	**33.2**	**24.1**	**1.5**	**57.9**	**23.7**	**38.3**	**7.4**	**15.9**	**3.4**	**63.7**	**61.9**	**116.1**	**1.0**	**127.6**
			Min	**0.4**	**8.0**	**4.3**	**11.0**	**0.1**	**0.1**	**7.0**	**0.1**	**1.8**	**14.0**	**27.0**	**31.0**	**0.1**	**70.0**
			Max	**80.0**	**76.0**	**7.4**	**160.0**	**70.0**	**107.0**	**21.0**	**39.0**	**11.0**	**200.0**	**138.0**	**233.0**	**3.0**	**386.6**