MINING, AGRICULTURE AND WETLAND ECOLOGICAL INFRASTRUCTURE IN THE UPPER KOMATI CATCHMENT (SOUTH AFRICA): CONTESTATIONS IN A COMPLEX SOCIAL-ECOLOGICAL SYSTEM

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This thesis is submitted in fulfilment of the requirements for the degree of

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By

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Abstract

Wetlands provide a wide variety of natural benefits (ecosystem services) from the natural environment to human populations, making them key examples of ecological infrastructure. However, the use of wetlands and their associated catchments is often contested by different users, making them nodes of conflict. Thus, there is a range of pressures on many wetlands which can ultimately lead to degradation or destruction. This study investigated the X11B quaternary catchment in the Upper Inkomati basin, Mpumalanga, South Africa. This catchment is characterised by a network of wetlands and streams that provide catchment residents with water. The sub-catchment is heavily used, dominated by the agricultural sector and coal mining.

To understand the contestation, a contextual analysis was carried out. Selected wetland conditions and ecosystem services, along with user perceptions and the value of wetland-use, were assessed. Wetlands were observably in a relatively healthy condition. The resilience of wetlands and the efficiency of the ecosystem services they offer, especially in mediating water quality, were clear. The early results indicated a healthy landscape despite multiple-user impact from human activity. The health scores and provision of ecosystem services, along with the identified National Freshwater Ecosystem Priority Areas (NFEPAs) and red-listed fauna and flora, provide a substantial grounding for advocating the conservation of the wetlands of the contested X11B catchment.

When water quality measures were added to the wetland health and ecosystem service assessment, low pH levels and high electrical conductivity were recorded. Both measures indicate coal mining impacts, more specifically Acid Mine Drainage (AMD) impacts, since AMD typically has sulphate as the dominant salt ion, and high concentrations of trace elements and metal ions. Concentrations breaching the recommended resource quality objectives (RQOs) of trace elements and ions, found in fertilizers and pesticides, were recorded in most sites, suggesting agricultural impacts on the landscape's hydrology. Further, these agricultural impacts would add to the compromising effect of the wetlands' capacity to remove pollutants from the water body. Livestock farming on all sites were also near wetlands which may have limited the vegetation cover of grazed land, so increasing runoff and the volume of water entering wetlands and compromising their ecosystem services.

Poor water quality has implications for biophysical processes, which play an important role in the functioning of wetlands, for the benefit of users. Without the water quality measures, ecosystem health and ecosystem service methodology used suggested a healthy catchment. However, simple field water quality measures indicated past and present mining impacts. Therefore, the mandatory use of water chemistry is recommended in the assessment of wetlands in catchments with past and present mining activity taking place. Without this, repercussions would include wetland loss, and a more thorough investigation into the water quality and its effects on the wetland ecosystems is suggested.

Further ecological investigation of water chemistry (heavy metals, ions, nutrients and trace elements) and macroinvertebrate assemblages identified links to water chemistry impacts on macroinvertebrate abundance and diversity. Abundance results based on the presence, absence and abundance of macroinvertebrates at the different sites did not reveal any clear patterns associated with different landscape users. Diversity, on the other hand, was related to land-use, where sites with high mining use had lower macroinvertebrate diversity than other sites.

Related, concurrent, hydro-pedology research produced a more comprehensive understanding of the impact of mining on hydro-connectivity that clearly indicates mining as the cause of long-term deterioration of functional wetland health in a way that is practically impossible to restore.

This study suggests that wetlands provide a strong ecosystem service of intermittent resetting of the wetland sediment adsorptive capacity for toxic metal and other salt ions. The hypothesis arising from the work is that, in the case of another heavy rainfall event, the town of Carolina risks another AMD crisis. As sediments are likely to be accumulating and saturated with toxic metal ions. Further AMD-related changes in acidity will increase the mobilisation of adsorbed ions. Future flooding and flushing of wetlands will therefore once again move toxic metal ions through the system, and possibly re-contaminate the Boesmanspruit dam.

The value of the study is in delivering specific evidence on the impacts of mining (and to a lesser extent agriculture) on wetland quality. Overall, this study, combined with additional research, indicates that in the X11B catchment, mining impacts are long-term and more serious than agriculture. In terms of contestation the research indicates that reliance on bio-physical data and knowledge is inadequate in resolving conflict between coal mining and other land- and water-users. The study demonstrates the necessity of insight into the social system and the value of a transdisciplinary approach in addressing land-use conflicts and wetland protection.

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Table of Contents

Table of figures

Table of tables

List of abbreviations and acronyms:

AEV: Acute Effect Values

AMD: Acid Mine Drainage

ASPT: Average Score Per Taxon

BFAP: The Bureau for Food and Agricultural Policy

CCA: Canonical Correspondence Analyses

CEV: Chronic Effect Values

CMA: Catchment Management Agency

CMS: Catchment Management Strategy

C-SES: Complex Social-Ecological Systems

DCA: Detrended Correspondence Analysis

DMR: Department of Mineral Resources

DO: Dissolved Oxygen

DWAF: Department of Water Affairs and Forestry

DWA: Department of Water Affairs

DWS: Department of Water and Sanitation

EA: Environmental Authorisation

EC: Electrical Conductivity

EIA: Environmental Impact Assessment

EMPR: Environmental Management Programme

GIS: Geographical Information System

HGM: hydrogeomorphic

IUCMA: Inkomati-Usuthu Catchment Management Agency

IWRM: Integrated Water Resource Management

NDNR: Nooitgedacht Dam Nature Reserve

NEMA: National Environmental Management Act

NFEPA: National Freshwater Ecosystem Priority Area

NWA: National Water Act

PCA: Principal Components Analyses

PES: Present Ecological State

RDM: Resource Directed Measure

RQOs: Resource Quality Objectives

SAM: Strategic Adaptive Management

SANBI: South African National Biodiversity Institute

SASS: South African Scoring System

SDC: Source Directed Control

SES: Social-ecological system

TDS: Total Dissolved Solids

TWQR: Target Quality Ranges

UKCF: Upper Komati Catchment Forum

WMA: Water Management Areas

WRC: Water Research Commission

Glossary of terms:

Adsorption: The process of molecules forming a surface coating on a particle.

Assimilation: The process whereby plants and organisms absorb nutrients and/or toxicants from their surrounding environment.

Bio-physical: The immediate biotic and abiotic environment of an organism or population.

Catchment: The area of land where precipitation drains into a single outlet point.

Ecosystem: The physical environment and interconnected biological community of interacting organisms.

Multidisciplinary: Different types of discipline-specific concepts and methods used to conduct research, whereby the disciplines may not necessarily share a mutual aim and or objective.

Sedimentation: The process of settling or deposition of sediment.

Sequestered: The situation where chemical compounds are trapped and isolated in a storage area, so limiting the possibility of reactions occurring.

Stakeholders: Affected and or interested people, such as researchers, local residents, local establishments, government, corporations, and industries.

Transdisciplinary: Different types of concepts, experiences, knowledge and methods used to conduct research, whereby the information is derived from different actors sharing a common aim and or objective.

Declaration

The following thesis has not been submitted to any university other than Rhodes University, Grahamstown, South Africa. The work presented here is that of the author and section 4.3.3 draws on research authorised by JH van der Waals.

Chapter 1: Complex social-ecological systems and management

In South Africa, the Eastern highveld vegetation type (Mucina and Rutherford, 2006) extends over several important water catchment areas (Department of Water and Sanitation, 2016) made up of a network of streams and wetlands. The area supports extensive livestock production and contains large coal reserves. Coal mining has escalated in the past decade, giving rise to contested views on the value to people of alternative land and water protection and use in the short, medium and long term (Tempelhoff et al., 2012; Munnik et al., 2016). This research addresses social and ecological issues arising from this contestation, and presents evidence of the impact of the various land-uses on the biodiversity of wetlands, which are key and vulnerable ecosystems.

1.1 Research context

This research is part of a larger project funded by the South African Water Research Commission (WRC) from a fine payment for severe damages to a wetland as a consequence of unlawful mining activities (Golfview court case: Munnik et al., 2014). Mining damage included watercourse diversions, inadequate pollution control, and the transformation of indigenous vegetation (Munnik et al., 2016). The court specified that the fine should be used to achieve the following objectives:

1. To analyse available resource- and catchment-based tools aimed at sustainable development of water resources and management;
2. To investigate and evaluate the decision-making processes followed in issuing mining authorisation;
3. To determine the relationship between licensing processes and ecological infrastructure from a landscape and connectivity perspective;
4. To develop a proposal for an integrative decision-making process and institutional arrangement required to support licensing for sustainable use of natural capital;
5. To develop guidelines necessary to understand the socio-economic value of selected wetlands by demonstrating their importance to society;
6. To develop and test a multi-sectoral integrative monitoring framework linked to a decision-support system that will cater for bio-physical, economic and societal needs;
7. To develop the appropriate capacity for officials involved in licensing, business,

and affected communities.

This research only addresses Objectives 1 and 4 by evaluating the ecological health of and ecosystem service provision offered by selected wetlands (key ecological infrastructure), and seeking to demonstrate the impacts of agricultural, mining and subsistence users. The history of Acid Mine Drainage (AMD) and land-use contestation in the X11B quaternary catchment (including the town of Carolina), made it an appropriate study area (McCarthy and Humphries, 2013; Munnik et al., 2016). Agriculture was historically the main land- and water-user sector, and contributed significantly to the catchment's history and income. As mining activities steadily escalated, disputes between the mining and agricultural land-users, and especially their claims on wetlands and other ecological infrastructure, also escalated.

Using a transdisciplinary approach, this investigation focused on:

- Conducting a qualitative description of social perspectives about wetlands and their value (chapter 2);
- Using and evaluating a rapid, commonly used qualitative assessment of wetland health and ecosystem service provision, together with a rapid assessment of wetland macro-invertebrates and water quality variables (chapter 2);
- Deepening the bio-physical assessment through a quantitative assessment of wetland macroinvertebrate community structure in relation to quantified concentrations of a variety of water quality variables (chapter 3);
- Making a recommendation for clear sustainable water resource use and protection (chapter 4).

The results are discussed in the context of results from the wider project. The research theory and conceptual framing, which guides methodology and methods, is provided in Figure 1.1.

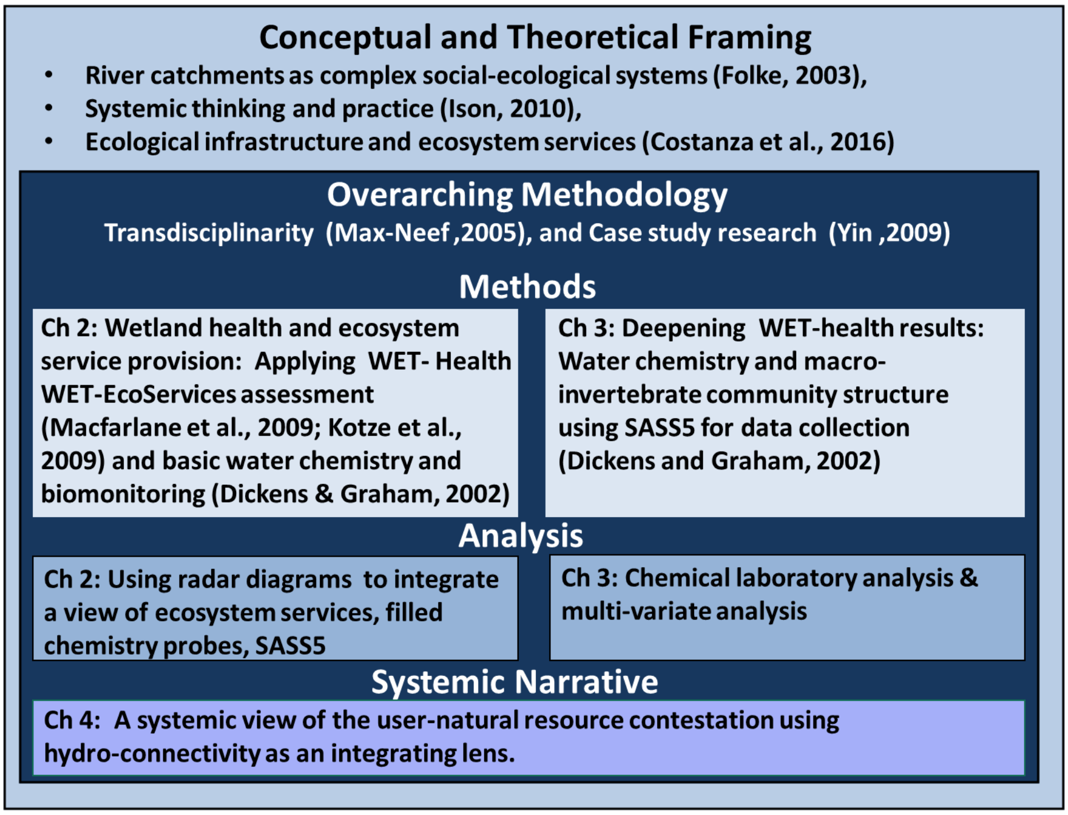

Figure 1.1: River catchments as complex social-ecological systems and the advance of systemic thinking and practice, focusing on ecological infrastructure and ecosystem services, were used as the conceptual and theoretical framing of this thesis.

The dynamic social-ecological co-evolution of river catchments is beginning to diminish the planet's capacity to absorb and support social development, resulting in environmental degradation and emergent uncertainty (Folke, 2003; Ison, 2010). In order to sustain and strengthen the planet's capacity, complexity and social-ecological well-being, a shift to a greater systemic way of thinking and practice is necessary (Ison, 2010). Such an approach entails resilience, adaptability, learning, and holistically viewing and understanding a system- i.e., complex systems thinking (Folke, 2003), and understanding that the planet's capacity and human well-being is largely related to the resilience and adaptability of naturally occurring provisions and services that ecosystems (ecological infrastructure) provide (Folke, 2003; Costanza et al., 2016). Understanding social-ecological systems involves integrating multiple disciplines within their contexts into the system and recognizing interlinked relationships (Max-Neef, 2005; Yin, 2009). Hence transdisciplinary and case study approaches are used. For the exploration of land-use contestation, this thesis uses a combination of chemical and ecological methodology and analysis (Figure 1.1) (Dickens and Graham, 2002; ter Braak and Smilauer, 2002; Macfarlane et al., 2009; Kotze et al., 2009). A systemic narrative was also included, using hydro-connectivity as an integrating lens, which helped to understand the complexity found deeper within the social-ecological system (SES) (Figure 1.1) (Folke, 2003; Ison, 2010).

1.2 Social-ecological context

The study area consists of the quaternary catchment X11B in the upper reaches of Inkomati River catchment, Mpumalanga (Figure 2.1). The three main tributary streams of the quaternary catchment are: the Boesmanspruit, Witrandspruit and Droogvaleispruit, which rise in highland grasslands. The Boesmanspruit and Witrandspruit join at a confluence 1 km before the Boesmanspruit dam where the combined flow supplies the main water storage and supply dam for the town of Carolina. The Droogvaleispruit also forms another junction north of the storage dam. From the supply dam, the water flows to the Nooitgedacht dam which feeds into the Komati River, which in turn, forms the upper part of the Inkomati international watercourse, shared between South Africa, Mozambique and Swaziland. The X11B catchment is dominated by agricultural and mining land-uses which pose threats to the hydrology of the catchment and the town's drinking water, as well as threatening the state of an international watercourse.

This study is founded on the recognition that catchments comprise a wide range of interacting elements: the ecological elements of land and water, their related ecosystems, as well as people based in social, economic and political settings (Ostrom, 2010). System elements include ecological and social components. Ecological components range from cells, individual species of fauna and flora, to soil, water and ecosystems (Biggs et al, 2012); social components include landscape users and governance systems (Holland, 1994; Biggs et al, 2012). At the individual level, each ecological (biotic and abiotic) and social element has its own properties and functions (Odum and Barrett, 1971; Pianka, 2011; Biggs et al, 2015). At a catchment level, the properties and functions of the catchment are a reflection of individual components, within and between different levels of ecological and social hierarchy, interacting with each other where important interactions include feedbacks (Noss, 1990; Jewitt, 2002; Biggs et al, 2015). Interactions within and between components drive cause and effect relationships, and a network of integrated and interdependent parts emerge as a system (Pollard et al., 2011; Ison 2007).

Berkes and Folke (1998) term an "integrated system of ecosystems and human society", such as a catchment, a social-ecological system (SES). Such systems are characterised by the phenomenon of emergence, for example, the emergence of direct and indirect services humans derive from an ecosystem. These services are the result of the cumulative outcomes of interactions throughout the system, beginning from selection processes and interactions at lower levels of ecological hierarchy (Pollard et al., 2014; Biggs et al, 2015). The concept of SESs gives a "humans-in-nature perspective", recognizing social and ecological components as interdependent elements rather than the traditional understanding of all components being separate entities of a system (Walker et al., 2004; Audouin et al., 2013; Biggs et al, 2015). Historically, components of a system were reduced to isolated units and it was assumed that changes regarding components were governed by linear laws of cause and effect, thus, by knowing the components initial positions and velocities, and the forces acting on the system, the system's outcomes were predictable and reversible (Cilliers et al., 2013; Audouin et al., 2013; Rogers et al., 2013). This reductionist way of thinking does not take into account the interdependencies and feedbacks within a system.

The functions and structures of a SES change continuously because of the heterogeneity of interactions among social and ecological components, whereas the physical nature of individual components may have simpler and more direct causal relationships (Walker and Salt, 2012). The explicit links between components of a SES, where changes in one component will result in inevitable changes elsewhere in the system, are made evident by system feedbacks.

Feedbacks are the reactions to *in situ* spatial and temporal interactions of predominant drivers within a system. Feedbacks are described as balancing or reinforcing where interactions between different scales have outcomes based on reactions that induce further (positive or negative) change in linked components (Pollard et al., 2008). Outcomes are generally a consequence of dominant drivers (human values, social, technical, economic, ecological, and political) that have greater influences within a system (Pollard et al., 2011). The relationships between the many elements and drivers of a system have different histories of use, impact, and influence and may change over space and time (Cilliers et al., 2013; Rogers et al., 2013). The nature of interactions among elements has therefore been moulded according to interactions and reactions of past and present events, and their environments, which have subsequently evolved the system to its present state (stable or changing) and overall behaviour (Holland, 1994; Pollard et al., 2011; Cilliers et al, 2013).

The overall behaviour of a SES is the result of combinations of processes occurring within a system. However, the many feedbacks within systems generate unpredictable outcomes, where new properties of the system emerge that are different from the original components that make up the system (Holland 1994; Holland, 1999; Folke et al., 2016). Feedback also plays a vital role in the self-organizing structure of a system, where reactions to changes can restructure a system (without the influence of external pressures) in order to keep functioning (Pollard et al., 2011). Feedbacks therefore have a major role in outcomes and in the functioning of SESs (Walker et al., 2004; Pollard et al., 2011; Walker and Salt, 2012).

1.3 Complexity in SES

The many outcomes that can emerge within a SES are unpredictable and uncertain. Such systems can therefore be termed "complex social-ecological systems" (C-SES) (Folke, 2006; Audouin et al., 2013). Characteristics of a complex system include: 1) multiple drivers and many interacting elements, 2) with differing contexts and histories, 3) that function in a state of flux and interact dynamically, and are 4) linked by multiple non-linear processes, 5) with feedbacks between elements and processes, 6) creating emergent, unexpected outcomes, 7) and are inter-linked and adaptive in nature (Cilliers, 2000; Pollard and du Toit, 2008). These characteristics are representative of most SESs and have implications for management and policy initiatives (Rogers et al., 2013; Cilliers et al., 2013). In C-SESs, functions and structures have co-evolved in a dynamic fashion, where both human development and nature have played integral parts in shaping one another and the associated observed characteristics of C-SESs (Folke, 2003; Ison et al., 2007). More recently, humans have accelerated the co-evolution to a point where the planet's capacity to support development is strained (Folke, 2003).

Humans have always depended on the natural environment for food, fibre and fuel, and on the sustained generation of ecosystems from which goods and services are derived (Millennium Ecosystem Assessment, 2005). However, with the evolution of society, human practices and living circumstances have become more complex, focusing on growth and efficiency, and overlooking recovery of ecological components which humans exploit (Fisher et al., 2009; Rockstrom et al., 2009). Improved human circumstances opened opportunities for population growth which was accompanied by an increase in land-use changes, deforestation, urbanisation, burning of fossil fuels, and pollution (Crutzen, 2006; Walker and Salt, 2012). These anthropogenic-induced impacts have resulted in the destruction and loss of natural diversity and ecosystem services through the direct loss of species, genes and ecosystems (Walker and Salt, 2012). One result is that present-day society has an increased level of poverty, which through desperation, lack of resources and understanding results in the unsustainable use of resources (Walker and Salt, 2012). However, a main cause is due to perceptions that human well-being and success depends on consumption (based on ignorance, misunderstanding, greed and perverse incentives (i.e., economic advances)), and where personal aims and values are motivated by optimising profits in the short-term, with little regard for the natural environment (Ostrom, 2010; Walker and Salt, 2012; Pollard et al., 2014).

Industrial and individual competitiveness are founded on increasing productivity in the most efficient way (minimum effort or expense wasted) while maximizing benefits (Latruffe, 2010). In terms of C-SESs, humans benefitting from ecological outputs generally means simplifying and tightly controlling ecological elements in order to gain and maintain maximum yields in minimal time at the lowest cost (Folke, 2003; Walker and Salt, 2012). Maximising efficiency requires a reductionist way of thinking, such as keeping an optimal state requires reducing elements into smaller parts (often reducing diversity and eliminating redundancy), understanding how each part functions individually and then deciding what inputs will yield the greatest outputs (Walker and Salt, 2012; Audouin et al., 2013). Trade-offs to gain efficiency tend to overlook the importance and complexity of the natural environment, with the result that industries and individuals ignore the negative consequences of their actions (Walker and Salt, 2012). This approach has repercussions for human well-being as both social and ecological research evidences the interdependence between the condition of the natural environment and human well-being (Palmer et al., 2015).

1.4 Undermining complexity

Undermining complexity in a C-SES, as in the case of efficiency, diminishes interactions and relationships. By removing components and reducing diversity, the system loses the range and capacity "to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks" (Walker et al., 2004:2) and renders the system vulnerable (Biggs et al., 2015). Without redundancy, changes may cause the system to cross a threshold and shift into a different state (i.e., regime) (Holland, 1994; Walker and Salt, 2012; Biggs et al., 2012). Ecological thresholds are reached when the quality and/or quantity of a system's drivers' (components or phenomena) are altered (abruptly, or even small directional changes). The nature of the ecosystem responds by changing (Holland, 1994; Groffman et al., 2006).

Social factors, such as perverse agendas, selective admittance and limitation of stakeholders, can limit knowledge diversity and exclude practical experience. (Cilliers et al., 2013). In this way, social thresholds within a C-SES can be exceeded. Multiple-use of resources can result in over-use, and the exclusion of less powerful users (Pollard et al, 2011; Tempelhoff et al., 2012). Conflicts then emerge among different stakeholders (Ison et al., 2007; Pollard et al, 2011; Tempelhoff et al., 2012), leading to despondency, animosity and trust issues (Ison et al., 2007).

Ison et al. (2007), describe the social and environmental stress arising from a mismatch of human use and ecosystem services as a "resource dilemma" that highlights the conflict, controversy, inter-dependence, and multiple perspectives that emerge from such situations:

> "Resource dilemmas do not lend themselves easily to scientific analysis and solutions. In fact, they are complex in that a great many factors, bio-physical, social, economic and political, interact in processes that are only partially path-dependent and usually unpredictable." (Ison et al., 2007, p10)

Resource dilemmas therefore have the following characteristics: 1) there are imbalances of use resulting in conflict and controversy (competing claims), and 2) inter-dependence between components where users' objectives are based on other users achieving their objectives – this is difficult for stakeholders to accept, 3) as there are multiple user perspectives (own optimisation strategies, theories and understandings) which makes resource dilemmas, 4) complex, 5) and result in high levels of uncertainty (Ison et al., 2007).

1.5 Managing for complexity

In order to manage a C-SES, managers need to be alert to bio-physical and social thresholds, seeking to shift so as to remain in a zone of relative stability, by using strong leadership, laws and regulations in order to support good governance within the social system. The current state, drivers that can cause a shift (Rogers and Luton, 2011; Kingsford and Biggs, 2012), function, feedbacks, and structure of the system need to be understood in order to understand thresholds, and to support diversity in order to maintain the system's resilience (Walker et al., 2004; Cilliers et al, 2013; Biggs et al., 2015). However, understanding these many aspects of a C-SES is complex and it is often easier to resort to a reductionist way of thinking (Audouin et al., 2013). Management protocols are seldom designed to encompass the complexity of dealing with multiple drivers (Jewitt, 2002). For example, linear models are used to predict outcomes of cause and effect based on average conditions of a system, overlooking the fact that systems are subject to and sometimes depend on extreme events (Walker and Salt, 2012; Cilliers et al., 2013). Many institutions and their governance and management initiatives are still in the process deepening understandings, and incorporating reductionist approaches into strategies for coping with impacts (Ison et al., 2007; Audouin et al., 2013). Thinking in terms of complexity is difficult; for instance, maintaining resilience can be costly and counterintuitive of efficiency and therefore unappealing to industries (Walker and Salt, 2012). Managing complex SESs as an integrated whole is therefore difficult, but the integration is imperative for sustainable management (Biggs et al, 2012).

The main focus of this thesis is the strain that social-ecological interactions have on water resources and users in catchments. With economic development, catchments have shifted from systems that were once ecologically constructed to systems that are now primarily socially constructed (Folke, 2003; Folke et al., 2016). Water interactions are premised on the fact that the resource provides a number of services: the basic needs for a human life (drinking, cooking, bathing and cleaning); important aspects of economic and political development (transport and water use); and are a vital factor in the shaping and survival of elements of the bio-physical world on which humans depend (Ison et al., 2007; Pollard et al., 2011). These interactions are based on human, often corporate, short-term agendas and the agendas are then lived out in the catchment, frequently negatively influencing the quality and quantity of water (Folke, 2003; Pollard et al., 2011). Furthermore, there are indirect water quality implications in the complex connection water resources have with land-use activities, as well as the downstream impacts that water has on users and the environment (Jewitt, 2002; Cilliers et al, 2013).

At a catchment scale, managing social and ecological systems individually can be a difficult task (Pollard and du Toit, 2008). For example, Pollard and du Toit (2008) emphasise the complexity of freshwater as an ecological system, where individual aspects of water use management include: water demand and supply; waste discharge; surface and ground water interactions; rate of runoff, infiltration and evaporation; interactions of water quantity and quality, as well as links to land-use and biotic responses, each of which operates differently over various scales in time and space (Pollard et al., 2011). As explained above, managing each individual aspect separately, with the expectation of resolving issues, is considered a linear and reductionist way of solving problems that emerge at the scale of the whole C-SES (Jewitt, 2002). Effective management calls for the use of frameworks that incorporate social and ecological aspects, and the inclusive complexity of both, to understand and resolve issues and impacts within the system as a whole.

In comparing different frameworks for analysing C-SESs, Binder et al. (2013) concluded that a single framework alone cannot be used to address all research issues in C-SESs (Ostrom, 2010). Instead, integrated concepts, theories, and more flexible and transdisciplinary approaches are necessary for scientists and managers in the domain of institutional arrangements and governance, and when dealing with and analysing SESs and their complexity (Audouin et al., 2013; Biggs et al., 2015). Scientists and managers need to adopt a systems-thinking approach (Ison et al, 2007; Biggs et al., 2015).

A major concept that has been closely recognised in relation to C-SES thinking is resilience thinking. Several researchers and managers have begun to focus on the use of systemic resilience as a lens to think about the dynamics and complexity of environments (Folke, 2003; Pollard et al., 2011; Biggs et al., 2015). Resilience thinking involves viewing a C-SES as a whole, interlinked system that operates over many spatial and temporal scales within a certain regime. This holistic approach supports management strategies that understand social and ecological thresholds (emergence, feedbacks); that manage for resilience (diversity, feedbacks); that encourage collective action and engagement/co-operation of all stakeholders, making sure people in the system have the capacity to manage appropriately (social capital, innovation, polycentric and overlapping governance); that assist in system recovery subsequent to disturbances; that increase sustainability, and promote socially, ecologically, and economically possible relationships between social and ecological components (e.g., maximum sustainable yields) (Ison et al., 2007; Kingsford and Biggs, 2012; Pollard et al., 2014). The resilience approach highlights the secondary effects of actions within a C-SES, so identifying unrecognized ecological benefits, and making consequences of greed harder to conceal (Walker and Salt, 2012).

A broader theme of C-SESs and resilience thinking is a concept termed “complex adaptive systems”. Resilience thinking is based on cycles of system adaptation that arise from feedbacks of selection processes (competition and cooperation) among agents, where variation and novelty are constantly being added to the system (Holland, 1994; Walker and Salt, 2012; Biggs et al., 2015). Structures of systems are therefore formed by infinite adaptive cycles occurring at different temporal and spatial scales within the system (Walker and Salt, 2012; Pollard et al., 2014). Scales of adaptive cycles include:

- Reconstruction (connections break, regulatory controls weaken),
- Reorganisation,
- Growth (new opportunities and available resources are exploited),
- Conservation (connections between actors increase, thus increasing resilience. Actors change, and growth and development rates continue to grow. However, growth rates eventually weaken; in a social context, development leads to increased efficiency, so decreasing resilience. Internal states become more regulated and thresholds become closer).

This concept can be viewed together with the resilience thinking framework where all components that confer resilience are better understood in their positions of the adaptive cycle in order for better management (Walker and Salt, 2012; Pollard et al., 2014; Biggs et al., 2015). The shifting dynamics of complex adaptive systems therefore need management schemes that are able to also strategically learn from shifts in regime and adapt appropriately (Kingsford and Biggs, 2012; Cilliers et al., 2013).

A management framework created to incorporate the uncertainty associated with complex systems is Strategic Adaptive Management (SAM) which focuses on facilitating stakeholder engagement in order to create shared and collective understanding and future-based visions and compliance (Pollard and du Toit, 2011; Kingsford and Biggs, 2012; Rogers et al., 2013). Different understanding of similar issues builds a diversity of knowledge and the system's social resilience. Cases of management success depend on the equal and sustainable participation of all stakeholders involved, for example, the way water is used upstream impacts on downstream stakeholders. Where there is high social resilience, stakeholders are made aware of such impacts and are able to reach consensus and contribute towards understanding and managing feedbacks (Rogers and Luton, 2011; Cilliers et al., 2013). In South Africa, SAM has been used in conservation planning (Kruger National Park, and Inkomati-Usuthu Catchment Management Agency area (Rogers and Luton, 2011) and is well understood and developed (Munnik et al., 2016). Emphasising stakeholder engagement builds new behaviours, and a greater intellectual acceptance of complex systems. It is regarded as important for Integrated Water Resource Management (Rogers et al., 2013; Ison et al., 2013).

1.5.1 Integrated Water Resource Management (IWRM)

South Africa's National Water Act (NWA) (No. 36 of 1998) aligns well with an understanding of catchment SESs and therefore the governance structures exist for using a

SAM approach, and enables both scientifically robust land-water use regulation methods. Practical implementation of Integrated Water Resource Management (IWRM) is a real possibility. The Department of Water Affairs (DWA) (2013) defines IWRM as "a philosophy, a process and a management strategy to achieve sustainable use of resources by all stakeholders at catchment, regional, national and international levels, while maintaining the characteristics and integrity of water resources at the catchment scale within agreed limits". IWRM was acknowledged by the United Nations World Water Development Report (2006) as a "holistic, ecosystem-based approach which, at both strategic and local levels, is the best management approach to address growing water management challenges and is seen as the best approach for meeting equity and sustainability". The approach seeks to embrace and adopt context-appropriate frameworks and approaches such as systemic resilience, complex adaptive cycles, transdisciplinarity (broadening multiple disciplinary understanding), recovery, flexibility, regulation, learning, action research, and to challenge the notions of linear thinking and optimisation (Holland, 1999, Rodgers et al., 2013; Biggs et al., 2015; Palmer et al., 2015). Using this approach, together with efficient capacity and governance should theoretically result in what Palmer et al. (2015) term "social and ecological justice", in order to promote increased human well-being in conjunction with healthy structural and functional ecosystem biodiversity.

Two main policy-based mechanisms are used in order to balance water resource protection and water resource use, as stated in the latest National Water Resource Strategy (DWS, 2013): the first mechanism is Resource Directed Measures (RDMs) which include three responsibilities: 1) classifying water resources, 2) determining the Reserve (amount of water set aside to provide for basic human needs and environmental flow), 3) deriving quantified resource quality objectives. The second mechanism, Source Directed Control (SDC), deals with the licensed allocation and authorisation of water resource use (Palmer et al., 2004; DWS, 2013). In order to implement these measures, various approaches are necessary to grasp the complexity behind impacted catchments. Data used for RDM and SDC water quality processes include analysing chemical and physical variables; collecting information on the presence, absence and abundance of biota (biomonitoring), and responses of biota to different chemical concentrations of water (ecotoxicology) (Palmer et al., 2004). By integrating RDMs and SDCs in water resource management, and incorporating SAM, water resource management initiatives are more likely to support the ecological and social sustainability of C-SESs.

South Africa has established institutions that support the key principles of equity and sustainability underpinning water policy in order to effect IWRM (Pollard et al., 2014). The primary national government department is the Department of Water and Sanitation (DWS) which, together with appointed management agencies, is responsible for ensuring the protection, development, management, use and control of South Africa's water resources on a local government and catchment level. This is done by supporting and regulating sufficient water supply, water quality and sanitation. There are nine Water Management Areas (WMAs), each of which has an associated Catchment Management Agency (CMA). A first task of a CMA is to draft a Catchment Management Strategy (CMS). There is the opportunity for the CMA to view the WMA catchment/s as C-SES/s (Cilliers et al., 2013). To date, only one CMS has been developed in this way (the Inkomati WMA), while other CMAs are still in the early stages of development. The Inkomati-Usuthu CMS guides water resource management in the study area, and this study serves as an indicator of current governance and IWRM implementation.

1.6 Carolina

This study is located in the X11B quaternary catchment within the upper Inkomati River catchment, which is part of the Inkomati-Usuthu Water Management Area (Figure 2.1). The catchment is managed by the Inkomati-Usuthu Catchment Management Agency (IUCMA), the most advanced CMA, explicitly committed to strategic adaptive management (Pollard et al., 2014; Inkomati-Usuthu Catchment Management Agency, 2016; Munnik et al., 2016).

Carolina, the town situated in X11B, has a population of around 23 000 and is part of the Albert Luthuli local municipality within the Gert Sibande district municipality (Munnik et al., 2014). The region's biome is made up of threatened Eastern highveld grassland and there is a high level of hydrological connectivity (streams, wetlands, springs, and groundwater), including wetlands identified as National Freshwater Ecosystem Priority Areas (NFEPAs) (Nel et al, 2011; Mpumalanga Tourism and Parks Agency, 2012). The hydrological connectivity is due to the sandstone and plinthic clay layers of rock below the soil surface that prevent water penetrating deeper into the ground, and causing it to travel horizontally, feeding streams and wetlands (van der Waals, 2016b).

The X11B catchment feeds into the Inkomati international water course shared by South Africa, Mozambique and Swaziland (Pollard et al., 2014). It is a heavily used landscape supporting agricultural, (dryland crop farming: potatoes, maize, wheat, sunflowers, beans, soya beans), cattle grazing, and mining (especially coal mining) (Figure 2.2) (Golder Associates, 2014). The “resource dilemma” (Ison et al., 2007) phenomenon was clearly illustrated in the 2012 Carolina water crisis, when the town’s water supply became acidic (Tempelhof et al., 2012; McCarthy and Humphries, 2013). This was ascribed to multiple converging causes, including an accumulation of mining decant and runoff sites; spillages from mine pollution control dam sites; changes in volumes of water pumped to nearby power stations, and an unusually heavy flooding event, which led to the runoff of contaminated water from various land-uses in the catchment (Tempelhof et al., 2012; McCarthy and Humphries, 2013). The Boesmanspruit wetland, a channelled valley-bottom type wetland, was the worst affected ecosystem; as the main X11B tributary flowing into the Carolina municipal dam contaminated the town’s drinking water.

1.7 Nexus of contestation

In the X11B catchment, there are strong hydrological links between aquatic ecosystems and water users that influence wetlands, agriculture and coal mining (van der Waals, 2016a). Groundwater and water tables within landscapes are fed by precipitation infiltrating the ground. Sub-surface water can account for 60% of the water in the landscape, important for wetland support. Wetlands offer vital resource-based services to various users, and their protection is a long-term goal for a sustainable landscape, but they are fragile (Dixon and Wood, 2003) and become a source of dispute between users. For instance, the impact of open-pit coal mining on wetlands is more severe than that of agriculture. Mining impacts on water and landscape also have consequences for the agricultural sector, whereas agriculture has minimal effect on mining (van der Waals, 2016a). Because perceived values of water resources vary among users, these impacts have further implications (see section 2.4). For example, agriculture depends more heavily on wetlands as a water resource to support livestock/crop agriculture, and therefore values the quality and quantity of the water. Open-pit coal mining, however, does not directly use the water source, but shallow water zone features are ideal for shallower open-pit coal mining, destroying the wetland ecosystem and its connectivity to the landscape (White, 2003; van der Waals, 2016a). Further, mining activities drain channel streams that feed into some wetland types, hence limiting water supply to the wetlands- however this research is explicitly focused on the wetland units and not stream channels. The wetland ecosystem and surrounding natural environment, as well as local and downstream domestic and bio-physical users depend on wetlands for support and provision of goods and services. Hence use and impact balances can become unequal, making wetlands a clear nexus of water resource contestation.

1.8 Wetlands and their importance

Wetlands are identified by shallow, extensive water, creating saturated and anaerobic (oxygen-depleted) soil conditions (Ollis et al., 2013; Ellery et al., 2009). These unique features form part of the complex biogeochemical functions, processes and structures that occur within a wetland (Pollard et al., 2013). The gentle slopes, slow flow and close inter-connections to stream channels and groundwater create the ideal landscape for prolonged water logging (Ellery et al., 2009; Ollis et al., 2013). Over time, microorganisms and plant roots consume the oxygen present in the soil, creating anaerobic conditions, as oxygen is unable to replenish effectively because of the saturated soil. The lack of oxygen in the soil:

- inhibits decomposition in the wetland, leading to high content of soil organic matter;
- modifies physical and chemical (ionic) functions and properties of wetland soils;
- provides stressful environments for biota (such as the lack of oxygen, fluctuating water levels and ionic changes in soil composition).

Specific wetland plant species (hydrophytes) and bacteria are adapted to these conditions. The changes in soil chemistry and associated fauna and flora that occur in anaerobic conditions play a major role in the unique services wetlands provide, including purifying and storing water, and storing sediment (Ellery et al., 2009).

1.8.1 Favourable chemical transformations

The depletion of oxygen through decomposition of organic matter by microorganisms in wetland soil causes an oxidation reaction which releases electrons, changing the natural charge of wetland sediment particles (Collins, 2005; Reddy and DeLaune, 2008). Available electrons cause charged minerals in the soil to become reduced as ions are exchanged (Collins, 2005; Reddy and DeLaune, 2008). These conditions produce a wide range of favourable chemical transformations, removing and storing in the long-term contaminants in the wetland's water column (Reddy and DeLaune, 2008; Ellery et al., 2009). Such transformations include the removal of nutrients (nitrogen and phosphorus), trace metals, transition elements, and heavy metals from the water body. Soluble forms of these contaminants, as well as their particulate forms, enter a wetland as runoff and discharge from manufacturing, mining, agricultural and urban land-uses. In the wetland, contaminants either settle into the sediment, or are adsorbed onto particulate matter. Contaminants settle between the wetland sediment particles, and are sequestered into the sediment, or are directly adsorbed on to wetland sediment (Hemond and Benoit, 1988; Reddy and DeLaune, 2008). Minerals (usually metals) within soils can then become soluble and leach out of the sediment, producing grey colouring, and becoming bioavailable for organism uptake (Macfarlane et al., 2009; Hogsden and Harding, 2011).

Sheoran and Sheoran (2006) refer to sediments aiding in removing contaminants from the water solution as “removal sites”. The magnitude of metal and ion exchange depends on the available sediment volume to accumulate precipitates, the types of sediments (with clay being the most reactive), and the amount of organic matter (higher adsorption capacity) present in the wetland (Hemond and Benoit, 1988; Horowitz, 1991; Reddy and DeLaune, 2008). It is assumed that deeper, larger wetlands with their greater volumes of removal sites would have a greater overall capacity to remove contaminants. However, removal sites can become saturated and ion exchange capacity can be reached (Nichols, 1983; Sheoran and Sheoran, 2006). In this situation, coupled with incidences of high water acidity, toxicants are released back into the water solution, and become bio-available (Sheoran and Sheoran, 2006). When toxicants become bio-available in the wetland they can affect other ecosystem components, or can be moved through the system, contaminating downstream human and natural users (Nichols, 1983; Ellery et al., 2009).

Plants make up a large portion of a wetland’s biodiversity and, in most cases, cover a large percentage of a wetland area. Wetlands have a range of flora because they are transitional ecosystems and are also associated with terrestrial vegetation. Collins (2005) classifies wetland vegetation as emergent plants, floating plants, floating-leaved plants, and rooted and submerged plants (Hemond and Benoit, 1988). Hydrophytes, specifically, have adapted by employing strategies to deal with dry periods and flooding events, fluctuating water levels, and the lack of oxygen and nutrients. Strategies include accelerated stem growth, C_4 photosynthesis, flexibility in germination, oxidised rhizosphere, development of air spaces in roots and stems for diffusion of oxygen from exposed portions of the plant, shallow root systems, and hollow stems (Collins, 2005). Wetland vegetation plays a major role in the removal of water contaminants and in the wetland’s capacity to do so (Sheoran and Sheoran, 2006). Wetland vegetation slows down the velocity of water entering and within the wetland, facilitating deposition of suspended solids (increasing chances the containment of pollutants by wetland anaerobic wetland soils) and providing short-term uptake of nutrients from sediment and directly from the water (Sheoran and Sheoran, 2006).

The actual removal of contaminants is a physical phenomenon. In the event of flooding or an increase in water velocity, removal sites (sediment) are suspended in the water body and flushed out and replaced with new sediment particles in the wetland (Sheoran and Sheoran, 2006). Similarly, contaminants taken up by biota are also only removed once the biota are removed from the system (Collins, 2005; Garcia et al, 2010). These removal mechanisms have proved convenient for treating chemical anthropogenic wastes and have led to the use of constructed wetlands to mitigate land-use impacts (Garcia et al, 2010). Research on the removal of nutrients in water by wetlands indicates that the combination of sediment characteristics and vegetation uptake in temporal regions can reach maximum potential removal rates of phosphorus (P) and nitrogen (N), ranging between 60 to 100 kg P ha^{-1} y^{-1} and 1000 to 3000 kg N ha^{-1} y^{-1} (Verhoeven et al., 2006). This is considered high in relation to fertiliser application in intensively farmed landscapes (Verhoeven et al., 2006).

1.8.2 Wetland hydrogeomorphic units

In theory, wetlands in South Africa should be rare, as the country's high elevation, low rainfall, high potential evapotranspiration, and absence of geologically recent glaciation do not typically support wetland formation (Ellery et al., 2009). Wetlands are generally associated and linked to streams, as is the case with the wetlands of the Mpumalanga highveld (Joubert and Ellery, 2013). South African wetlands are fed by surface and groundwater inflow (sub-surface lateral drainage (> 60%), recharge (4%)), and rainfall) (van der Waals, 2016a). Water leaves via evapotranspiration and surface and groundwater outflows (van der Waals, 2016a) (Section 1.10.3). The manner in which water moves though a landscape (arrives at, flows through, and leaves a wetland) is a result of the landscape's different hydrological and geomorphic settings (hydrogeomorphic units: HGM units) (Kotze et al., 2009; Ollis et al., 2013). A single wetland can be made up of numerous types of HGM units that individually contribute to the functionality of the whole wetland (Ellery et al., 2009; Kotze et al., 2009; Ollis et al., 2013).

Hydrogeomorphic units vary, depending on the stream channel, slope, positioning (bottom of a valley, or hillslopes), water source inflow and outflow. The six HGM unit types include floodplains, channelled valley-bottoms, unchanneled valley-bottoms, hillslope seeps feeding a stream, hillslope seeps not feeding a stream, and depressions (See appendix A) (Ellery et al., 2009). The characteristics of each type of HGM unit provides for associated ecosystem services and the extent of delivery/provision. Further, wetland size in relation to the wetland's catchment also determines a wetland's ability depending on the service and locality, where the greater the wetland's size the great the greater the provision of benefits-especially for assimilating and regulating services (Kotze et al., 2009). Therefore, altering hydrological and geomorphic aspects of an HGM units and their sizes has implications for the wetland's condition, but further implications for the ecosystem's underlying services and the goods and services from which people benefit (Hemond and Benoit, 1988; Ellery et al., 2009). For example, the assimilation and adsorption of toxicant contaminants from water entering the wetland are examples of toxicant removal benefits. The extent of delivery is based on the characteristics of the wetland HGM unit that encourages slower water velocity and greater contact and retention time of water in the ecosystem, so creating opportunities for chemical transformation (Macfarlane et al., 2009; Kotze et al., 2009). These naturally derived benefits therefore depend on the condition of the wetland components, and ultimately, the condition (physical conditions and physico-chemical composition of the water) of the wetland's HGM units.

1.8.3 Value of wetlands

Wetlands are arguably among the world's most important, yet threatened, ecosystems (Van Vuuren, 2014). Their valuable ecosystem services are the results of interactions, composition, structure and ecological functioning of the ecosystem (Millennium Ecosystem Assessment, 2005; South African National Biodiversity Institute, 2014). These services supply a range of valued benefits that humans and the natural environment use, which benefits are based on functional outcomes encompassing numerous tangible goods/benefits and intangible provisions (Millennium Ecosystem Assessment, 2005; Pollard et al., 2013).

The Millennium Ecosystem Assessment (2005) categorises ecosystem services as provisioning, cultural, supporting and regulating. Wetland provisioning services are the tangible goods provided by the ecosystem, for example, livestock grazing, fibre, cultivation, water supply, food, and fuel. These goods and services are considered direct benefits to humans and the bio-physical world (Millennium Ecosystem Assessment, 2005). The intangible cultural services are, for example, the aesthetic appreciation, spiritual connection, educational gains, and recreational values that people obtain from the presence of a wetland ecosystem (Millennium Ecosystem Assessment, 2005). The regulating services are the services that well-functioning ecosystems indirectly provide by regulating the natural environment. For example, regulation of floods, groundwater recharge, streamflow regulation, erosion control, maintenance of biodiversity, water quality regulation. However, the supporting services of all three of these services contribute to the enhanced functioning of the ecosystem by providing the fundamental structures and processes, such as soil formation, photosynthesis, respiration, and water and nutrient cycling (Millennium Ecosystem Assessment, 2005). Wetland ecosystems play a vital role in regulating the natural environment and supplying surrounding communities with food, fuel and fibre, and so increasing human well-being, and ultimately protecting local economies (Van Vuuren, 2014).

Wetland ecosystems and their services are especially important to people living in semi-arid regions and for human well-being that depends on a water supply of appropriate quality and quantity. However, many people are not aware of the services that ecosystems provide, nor are they aware of their finite capacity. This ignorance leads to degradation which highlights the importance of greater awareness (Jewitt, 2002; Pollard et al., 2013; SANBI, 2014). Ecosystem diversity must be conserved and restored in order to preserve a wetland's integrity and ability to deliver natural services (Munnik et al., 2016). The connection between ecology and society that ecosystem services provide requires greater awareness and protection efforts, as ecosystem benefits are valuable to humans (de Groot et al., 2010; Pollard et al., 2013). Conservation frameworks therefore focus on using approaches that engage with stakeholders.

The concept of ecosystem service valuation can play an important role in assisting in negotiated outcomes over competing land-use and the needs of the environment. There are several methods of valuing ecosystems: monetary (tools of resource economics and replacement costs), aesthetic, cultural/spiritual, moral, and bequest values. Research into valuing ecosystems shows that valuation efforts are complex and tend to underestimate and inadequately allocate monetary values, to portray nature as a product with infinite intersubstitutability, and to downplay intangible values (de Groot et al., 2010). Despite the complexity, scientists are still able to provide economic and moral arguments for conservation (e.g., informed viability of land-use, estimates of loss due to land degradation) (de Groot et al., 2010). The entry of ecosystems into the social-ecological arena allows for conservation efforts to incorporate both societal and ecological needs in socio-political decisions (Norton, 2003; Pollard et al., 2013).

1.9 Wetlands as ecological infrastructure

To promote more social awareness and understanding of the social, economic, and ecological importance of wetlands and other service-providing ecosystems, is the recent adoption and implementation of the term "ecological infrastructure" by conservation biologists and the South African National Biodiversity Institute (SANBI) (SANBI, 2014). Defined by SANBI (2014) as "the nature-based equivalent of hard infrastructure, and can be just as important for providing services and underpinning socio-economic development. Ecological infrastructure does this by providing cost-effective, long-term solutions to service delivery that can supplement, and sometimes even substitute, built infrastructure solutions." The term infrastructure is more easily understood by investing stakeholders, as the word emphasises the necessity of ecology and ecosystems for the long-term, functional operation of structures of society (SANBI, 2014).

The term, 'ecological infrastructure', is based on the idea that people are well aware that built infrastructure is a capital investment that requires on-going investments in upgrades and maintenance (as is the case with the persistence and efficiency of natural services). With an infrastructure-based understanding, values can be placed on ecosystems and provide a means of protection and prolonged continuation. Valuing ecological infrastructure "makes visible [the] transition from socially owned wealth into private wealth, and enables a form of accounting for the production of negative and positive values, for example pollution, depletion of resources, as well as financial profits and social investments" (Munnik et al., 2014: 18). This is vital in a country such as South Africa when, in 2012, 65% of wetlands were categorised as threatened (Driver et al., 2012).

Wetland ecosystems in the X11B catchment form the ecological infrastructure focus of this study, and are viewed as the nexus between ecological infrastructure, water systems, and water users. Wetlands are therefore ideal systems in which to explore the issues of contestation between resource uses and protection.

1.10 Contestation in the Carolina context

The many claims on the X11B catchment's natural resources create a complex situation regarding liability for social vulnerability and bio-physical degradation. Water was the main driver of both the social and ecological systems; hence the changes in water quantity and quality caused a breach in the system's thresholds (Tempelhoff et al., 2012). In this landscape of numerous stakeholders, farming, and mining industries, understanding interactions between land, water, and use are complex. Some of the competing claims by and various perspectives of different users of the X11B catchment's natural resources are presented in the next sections.

1.10.1 Local communities

Local communities (civil society organisations, groups of residents) depend on the raw water supply in the storage dam, fed by the wetlands, streams and tributaries of the catchment for basic needs and local industry (Tempelhof et al., 2012). The rural communities claim the use of grasslands and wetlands, streams and tributaries for livestock, subsistence farming, and the collection of natural products such as thatching grass. The catchment community therefore depends on government departments and municipalities to ensure and maintain sufficient amounts of quality water (Tempelhoff et al., 2012). However, Carolina has a history of poor municipal services, such as inconsistent electrical power and water supply, issues with water sanitation, and poor upkeep of infrastructure which contributed to the town's water purification plant's incapacity to maintain quality water after heavy rains (Tempelhoff et al., 2012). There was already distrust between the community and the town's governance structures when the 2012 AMD crisis occurred. The crisis impacted the entire catchment, and the lack of mitigation resources available to the municipality left the town without safe municipal water for seven months (Tempelhoff et al., 2012).

The community was reassured of numerous things over the crisis period: that interventions were taking place in order to mitigate the AMD impacts on the town's water; that the water purification plant's settling tanks and de-sludging processors were being fixed; that investigation indicated that the mining companies were to blame and they were to be convicted, and that water tanks around the town would be a temporary supply solution (Tempelhoff et al., 2012). However, with the continued poor and unreliable alternative water supply, poor communication, and lack of conviction of the mining industry culprits, further issues of trust emerged (McCarthy and Humphries, 2013). Stakeholders accused the municipality of being ill-equipped to deal with AMD situations (Tempelhoff et al., 2012). For example, the town's water purification plant's settling tanks and de-sludging processes were not functioning properly because they were not able to deal with the high metal content in the water as a result of AMD. Further, the municipality was blamed for the crisis as investigation of mining activities showed that two of the four accused mining companies did not have mining licences (Tempelhoff et al., 2012). The municipality was therefore at fault as it is responsible for ensuring licences are issued before operations begin and that adequate risk management and general infrastructure are implemented and maintained. This revelation added to the community stakeholders' doubts about the municipality's capabilities and competence in dealing with the situation (Tempelhoff et al., 2012; McCarthy and Humphries, 2013). The overall animosity that emerged resulted in protests of civil unrest, anger, and a court case against the local municipality (Chief Albert Luthuli) for not supplying acceptable water to residents of Carolina (McCarthy and Humphries, 2013).

Distrust of coal mining companies in the catchment also materialised. Community members conveyed to Tempelhoff's team (2012) that the coal mining sector was not actively involved in the Carolina community, even though it is seen as an option for employment. The mining companies denied responsibility for the crisis that unfolded in 2012, even though evidence suggested mining impacts were responsible for the water contamination (Tempelhoff et al., 2012; McCarthy and Humphries, 2013). One coal mining company did, however, take the initiative to commission an environmental consultation firm to assist in identifying possible impacts of coal mining operations in the catchment.

1.10.2 Agricultural natural resource-users

Agriculture is a main driver of the Mpumalanga economy and is therefore an unavoidable component of the X11B catchment (Stats SA, 2016). An estimated 22% of South Africa's maize is produced in the province. In dry seasons, the province's contribution to the country's maize supply could be as high as 55% because of the higher than average rainfall in parts of Mpumalanga (Tempelhoff et al., 2012). Carolina is historically an agricultural town, hence much of the X11B catchment is agricultural land and 40% of households of the catchment derive their income from agriculture (Tempelhoff et al., 2012; Stats SA, 2016). The agricultural sector claims use of grasslands and rangelands with their component wetlands, streams and tributaries for crop and livestock agriculture. Generally, human settlement has been strongly linked to water access for basic needs and farming (Folke, 2003). Water resources provide agricultural practices with services and benefits that include arable land, water provision for growth of crops and irrigation use, grazing and drinking sources for livestock, especially in periods of drought and dry seasons (Scoones, 1991; Swanepoel and Barnard, 2007).

These ecosystem services, and more specifically wetland services, are described by Scoones (1991) as being vital in any mainstream and subsistence agricultural context (Egoh et al., 2012). However, farming activities have detrimental impacts on water resources, on both the hydrogeomorphic and the natural functions of wetlands. Practices often lead to (Kotze et al., 2009; Walker and Salt, 2012):

- removal of native vegetation,
- overgrazing and trampling in and around wetlands by livestock,
- building infrastructure (e.g., embankments and roads) within, and alterations of water channels (dredging in order to control speed flow through the wetland and to prevent flooding) on farms,
- construction leading to the fragmentation of natural biodiversity, sedimentation and erosion from bare soils within a wetland's catchment,
- increased toxicants, nitrate, nitrite and phosphate levels from fertilisers and pesticides, leading eutrophication, and
- drainage of wetlands for irrigation purposes.

Maize and livestock farmers in the catchment have serious concerns involving coal mining and future agricultural practices. The Bureau for Food and Agricultural Policy (BFAP) (2012) reported that only 1.5% of South Africa is covered in soils best suited for cash crop production and 46.4% of the total area is located in Mpumalanga. Mpumalanga is also a heavily mined province and mining activities pose threats to the arability of the soils (BFAP, 2012). Prospective coal mining alone is estimated to transform 12% of Mpumalanga's arable land and a further 14% once prospecting rights are administered (BFAP, 2012). These land transformations not only impact agricultural practices, but also maize meal prices (approximately 5%), thus impacting a large population of the country's food supply (BFAP, 2012).

1.10.3 Coal mining

Physical impacts

South Africa is the sixth largest exporter of coal in the world and the sector plays a strong role in the country's economy. The Mpumalanga highveld geology is made up of the Karoo supergroup which consists of layers of sedimentary rocks layered with coal, resulting in a heavily mined landscape (McCarthy and Humphries, 2013). Mining is also one of South Africa's most contentious water-users and polluters and, in many cases, has proved to be destructive and environmentally unsustainable, for example in relation to AMD (Hallowes and Munnik, 2016). Wetlands have been targeted by coal mines as they provide a shallower geology in the landscape, making it easier to access coal seams, but resulting in the destruction of the wetland (White, 2003). Coal mining claims in grasslands are often in close proximity to wetlands, and mining operations require extensive built infrastructure on and around claimed landscapes.

Open-pit coal mining impacts form the focus of this thesis. In 2014, four operational mines, seven defunct mines, along with three functioning coal sidings and a beneficiation plant were identified within the X11B catchment (Golder Associates, 2014).

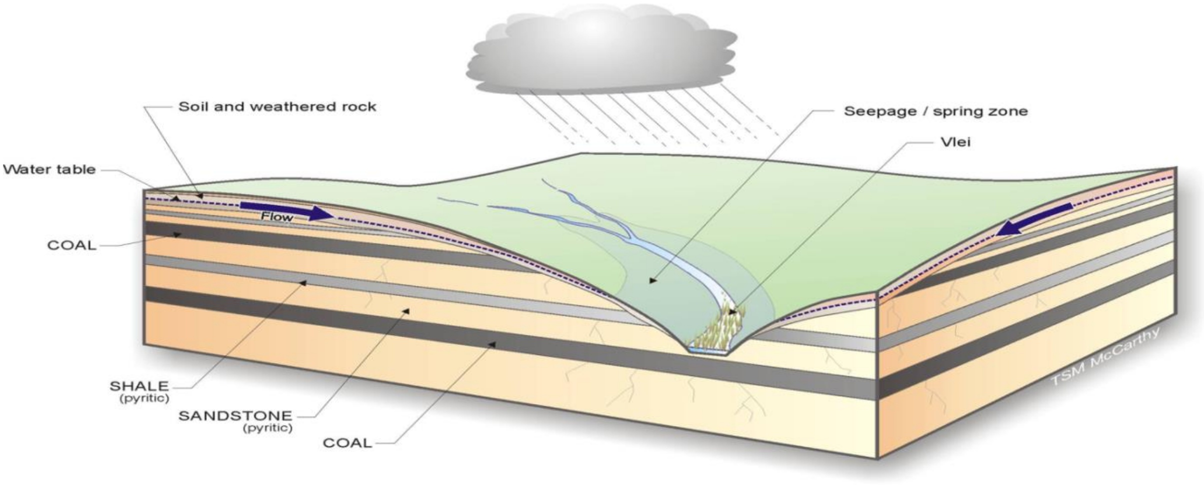

Figure 1.2: Relationship between surface water sources, soil water and coal seams (Source: Hallowes and Munnik, 2016).

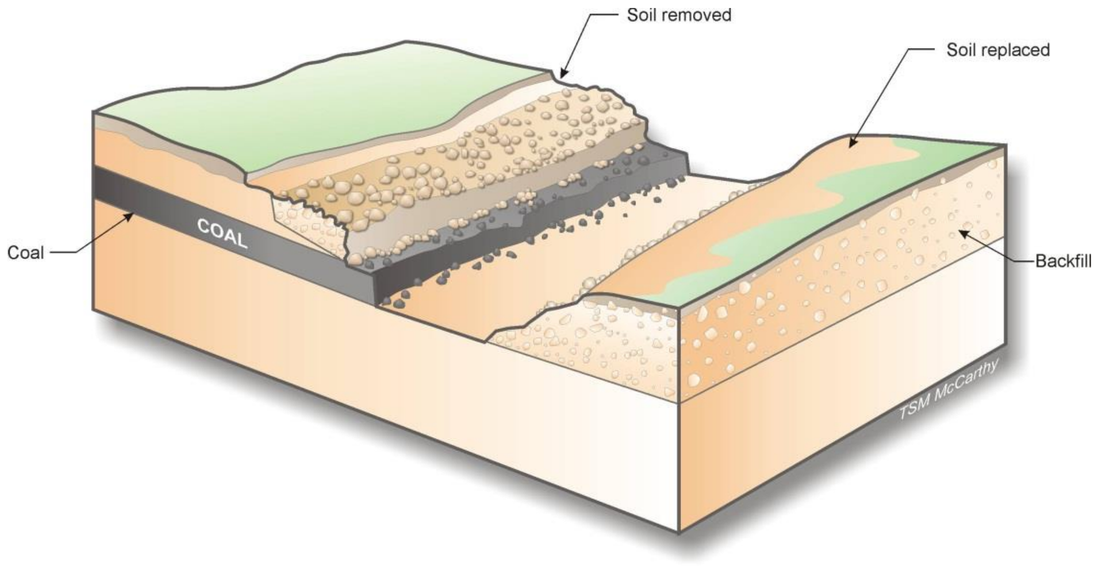

Figure 1.3: Removal of layers of soil, shale and sandstone from the landscape to expose coal seams in open-pit coal mining (Source: Hallowes and Munnik, 2016).

Open-pit mining requires all layers of soil, shale and sandstone (including pyrite) to be removed from the landscape to expose coal seams (Figure 1.2 and Figure 1.3). The ground is then purged, creating a void that intercepts the landscape's sub-surface lateral drainage (that feeds the water table), so intercepting flows through soil profiles that feed wetlands and biota, drying out ecosystems, and restricting and contaminating water supply to other parts of the landscape (Figure 1.4) (van der Waals, 2016b). Purging the landscape also can create new mine drainage wetlands in close proximity to the mining site as water may accumulate in the mine profile and discharge at the lowest point of the mine pit (van der Waals, 2016b). The implications are the establishment of a decanting point directly from the mine and the possible decantation of polluted water (Figure 1.4). Coal mining also has impacts of increased runoff due to surface hardening at coal mining related sites. Runoff containing fine coal deposits and possible AMD, along with coal dust generated at sites, also enters water systems within catchments (White, 2003; Munnik et al., in prep).

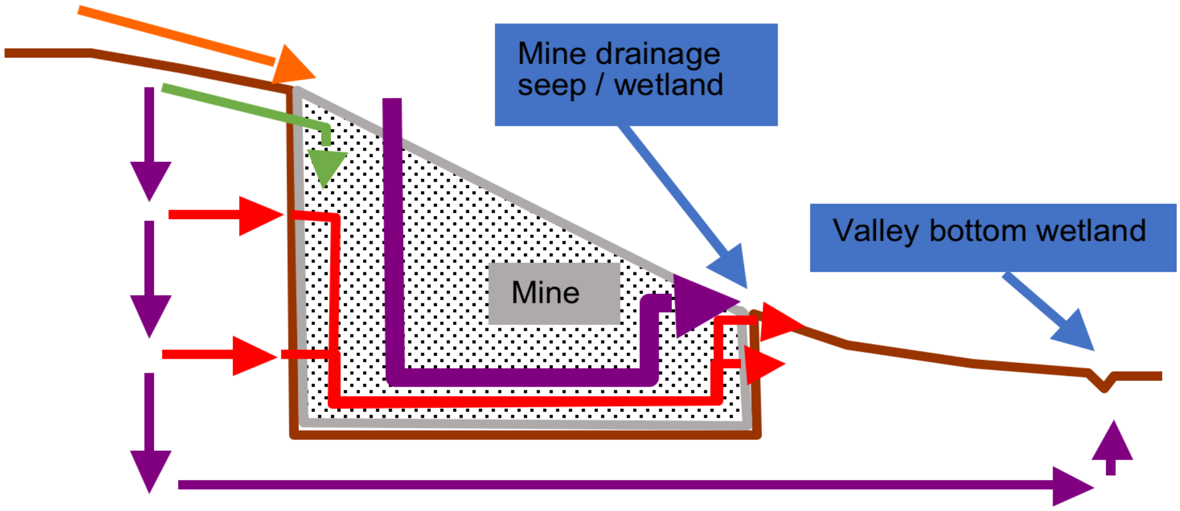

Figure 1.4: Underground impact of open-pit mining: surface runoff (orange arrow), sub-surface shallow lateral drainage (green arrow) and sub-surface deep lateral drainage/seepage (red arrows) are intercepted by open-pit mining, with only deeper groundwater recharge (purple arrow) feeding naturally established wetlands. Intercepted sub-surface deep lateral drainage/seepage (red arrows) and groundwater recharge accumulates in the mine profile and discharges at the lowest point of the mine pit, usually creating new mine drainage wetlands in the mining site (Source: van der Waals, 2016a).

Implications for closure of mines and rehabilitation is that, after the mine has been repacked and filled, the intercepted sub-surface water paths (sandstone and plinthic layers) are not reconstructed, thus compromising the hydro-connectivity within the landscape (van der Waals, 2016b; Figure 1.4). Furthermore, pits are usually loosely repacked with previously exposed pyritic rock/coal. Water accumulates and discharges in the same manner as during coal extraction (Figure 1.4); however, percolation and build-up of water makes contact with repacked pyritic rock/coal, creating AMD issues in established mine drainage wetlands after closure (White, 2003; van der Waals, 2016b).

Acid Mine Drainage from mining activities is a dominant concern. Rock containing coal contains the mineral, pyrite. When the mineral is exposed to oxygen and water, the pyrite oxidises, creating sulphuric acid (McCarthy and Humphries, 2013). The oxidation is a naturally occurring process, but the acidity is typically neutralised by other minerals (carbonates and by hydrolysis of aluminosilicate minerals) occurring in the ground (McCarthy and Humphries, 2013; van der Waals, 2016b). However, the extent of the rock-type that coal mining exposes, produces acid beyond natural neutralising capabilities. Not only is the sulphuric acid problematic, but it also enhances the solubility of iron (Fe), aluminium (Al), and other heavy metals (McCarthy and Humphries, 2013). Thus, once oxidation has taken place, water typical of pyrite exposure has low pH levels, and high levels of sulphate concentrations, Fe, Al, and other heavy metals. Contaminated water leaches into water courses, usually wetlands (White, 2003; Hallowes and Munnik, 2016).

Once AMD enters a water system it has adverse effects on the fauna, flora, humans that use the water resource, and specifically, on the toxicant removal capacity of the wetland ecosystems. Heavy metals can become bio-available under acid conditions and harm aquatic organisms, where, depending on specific tolerances, ecosystem damage can occur (Dabrowski et al., 2015). The complex wetland processes that remove excess nutrients and store metals involve a range of biological, chemical (sedimentation and adsorption of particles and oxidation of contaminants) and physical extraction (Sheoran and Sheoran, 2006). When a wetland's adsorption capacity has been reached and water is not filtered by the wetland, it moves through the wetland and flows into connecting water resources, impacting downstream users (including the agricultural sector, who use water for livestock and irrigation) (White, 2003; Mpumalanga Tourism and Parks Agency, 2012; McCarthy and Humphries, 2013). Secondary implications may include the impact of such effects on food webs and other ecosystem services. Hence, coal mining's far-reaching and negative impact on the biodiversity of the country and the associated ecosystems is a major concern.

Restoration has become part of mining responsibilities (Hallowes and Munnik, 2016), but the layered sub-surface geology means restoration has to include layering materials that are differentially permeable (Figure 1.2). This is expensive and not routinely undertaken.

Another issue that emerges with restoration of mining sites is the stripping of soil profiles. Healthy nutrient-rich topsoil is removed and mixed with other rubble from the digging of mine pits. When mines repack mining pits, the healthy topsoil, important for sustained plant growth, is not sufficiently replaced and, at best, the land can be used only for pastures (van der Waals, 2016b).

Legacy issues and social impacts

Past investigations indicate that open-pit coal mining taking place in the X11B catchment is a destructive land-use activity for both ecology and stakeholders (Tempelhoff et al., 2012; McCarthy and Humphries, 2013). This, in conjunction with mining's apparent legacy issues, was one of the main causes of the 2012 AMD crisis and contestation.

According to South African law (Humby et al., 2015; Munnik et al., in prep), mining companies need to go through an elaborate process in order to gain the right to mine, and the right to mine includes the whole mining process, from prospecting to closure and rehabilitation of the land post-mining (van der Waals, 2016b; Humby et al., 2015). In terms of the environment, mining companies under the National Environmental Management Act (NEMA) (National Environmental Management Act, 1998) need to use the best available decision support tools, for example:

- The Mining and Biodiversity Guidelines (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, South African Mining and Biodiversity Forum, and South African National Biodiversity Institute, 2013),
- High Risk Wetlands Atlas and Users' Guide (Holness et al., 2016),
- Wetland offsets: A best practice guideline (Macfarlane et al., 2016a),
- Wetland Rehabilitation in Mining Landscapes: An Introductory Guide (Macfarlane et al., 2016b)

and submit an application for Environmental Authorisation (EA) to the Department of Mineral Resources (DMR) (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, South African Mining and Biodiversity Forum, and South African National Biodiversity Institute, 2013). This application requires scoping and Environmental Impact Assessment (EIA) processes to be followed. Only a scoping report is needed for the application of the mining right and, if accepted by the DMR, then mining companies are meant to provide an Environmental Management Programme Report (EMPR) and EIA report (Humby et al., 2015). The EIA report is an assessment of environmental impacts the mine will have, along with considerations of risk scenarios and mitigation plans, whereas the EMPR is a report, based on the EIA, which details how the mining company will manage and limit impacts on the environment during construction and operation, and how the mine plans to rehabilitate the land after closure (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, South African Mining and Biodiversity Forum, and South African National Biodiversity Institute, 2013). Only once these reports, along with other socially focused reports, are accepted by the DMR can a company initiate mining activities.

If carried out appropriately and thoroughly, Environmental Impact Assessments and EMPRs are powerful assessments in terms of giving an intricate picture of impacts, risks and their associated management and mitigation (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, South African Mining and Biodiversity Forum, and South African National Biodiversity Institute, 2013). These assessments and reports, along with adequate enforcement of NEMA, ensure the most sustainable means in terms of current mining in South Africa. However, cases of AMD and other mining-related crises occurring in the country (Ochieng et al., 2010), suggest that appropriate impact mitigation and sustainability protocols are not being adequately carried out or enforced by mining companies.

Furthermore, mining companies are known to make use of reductionist, command-and-control styles of resource management as opposed to more complexity-based approaches (Pollard and Du Toit, 2011; Cilliers et al., 2013; Houdet and Chikozho, 2014). A South African case study by Cilliers et al. (2013) on the mining sectors in the Mpumalanga region highlighted the fact that mining companies do little to embrace diversity and have a linear, standardised modelling approach to management. Instead, mining companies design and predict future occurrences with little unexpected emergence and there is little need for flexibility in constructing rules and goals (Deloitte and Touche 2009). Therefore, it is more likely that secondary effects (such as limiting ecosystem services) and unpredicted extreme events are not considered in management programmes and reports.

In terms of the X11B catchment, open-pit coal mining impacts were evident. At the time of the AMD crisis, McCarthy and Humphries (2013) recorded that that the pH of the Carolina water supply was at 3.7 and concentrations of sulphate, Al, Fe and other heavy metals (including magnesium) were above acceptable water standard limits (Department of Water Affairs and Forestry, 1996). It was also recorded that fish and plant life in the Boesmanspruit dam were dying, and there were clear signs linking the deaths to AMD. The AMD limited the people of Carolina’s basic right to a safe water supply, and impacted fauna and flora of the catchment. The presence of AMD within the X11B catchment alone, suggests that mining companies are not carrying out adequate impact mitigation and sustainability protocols, adding to the conflict and the poor mining legacy issues found in South Africa (Munnik et al., 2016).

Mining companies took little responsibility for the situation in 2012 (Tempelhoff et al., 2012). Two of the four mining companies accused of the town's contamination did not have water-use licences and the mines were publicly accused of being primarily responsible for the contamination. However, legal measures against the companies were not taken, nor did mining companies assist with mitigation, except for commissioning an environmental consultation identifying possible mining impacts (Tempelhoff et al., 2012; McCarthy and Humphries, 2013; Golder Associates, 2014). Communication with mining companies, officials of the municipality and political local leaders was unsuccessful. Animosity towards the catchment's mining companies and doubt in government authorities' ability to restore the water supply emerged among stakeholders (Tempelhoff et al., 2012). Farmers, specifically, still perceive coal mining as a destructive activity that threatens the future viability of the agricultural sector in Mpumalanga.

Acid Mine Drainage has been an environmental hazard for decades (Akcil and Koldas, 2006), highlighting the past and continued apparent South African mining legacy issue. Despite the inadequacy of mining companies, legacy issues also include South African mining-associated governmental departments. Issues of uncontrolled mining, inappropriate sign-off of prospecting sites, and lack of enforcement by municipalities continue (Tempelhoff et al., 2012; Humby et al., 2015). Studies of the 2012 incident revealed that it was the local CMA and local government (DWA) that were ultimately to blame (Tempelhoff et al., 2012).

1.10.4 Governance structures

The accusation levelled against the local CMA and the DWA was based on the perception of the IUCMA and local government's poor capabilities to mitigate the AMD situation, due to the lack of appropriate infrastructure, experience and knowledge, despite the presence of mining in the catchment for the last decade (Tempelhoff et al., 2012; McCarthy and Humphries, 2013). In addition to these shortcomings, two of the four mining companies did not have licences to begin with, and even with this knowledge, government agencies were unable to prosecute mining companies (Tempelhoff et al., 2012). The mistrust that emerged resulted in protests, some of which were violent, and a court case against the government (Tempelhoff et al., 2012).

Furthermore, communication between national, provincial/catchment, local government, mining companies and local residents was minimal. Trust issues emerged not only between local stakeholders and government, but the DWS also began to question the IUCMA’s functional capabilities because, instead of dealing with the IUCMA over the issue directly, the legal advisors of the mining companies communicated directly to the DWS that it was the CMA that did not follow through with pre-directive water licensing processes (Tempelhoff et al., 2012).

1.10.5 Eskom

In the past, the X11B catchment was part of an Eskom water transfer scheme to provide extra water to service power stations located around the X11B catchment (Tempelhoff et al., 2012). The transfer was from a dam located in an adjacent catchment and flowed through the X11B Witrandspruit tributary and into the Boesmanspruit dam, where it continued into the Nooitgedacht dam located north of the catchment (Tempelhoff et al., 2012). In 2011 the scheme in the catchment was stopped and the surface water inputs of the wetlands and the dam changed. Less water in the dam meant less dilution of contaminants, making the Eskom scheme and its implementation an exacerbating factor in the 2012 AMD crisis (Tempelhoff et al., 2012).

1.11 Aims and objectives

The 2012 AMD incident is a good example of stakeholders in the catchment overlooking secondary effects and ignoring long-term consequences of their actions (Tempelhoff et al., 2012). The AMD, the incompatibility of infrastructure in dealing with the situation, the contamination, the shortages of water supply, all highlight the contestation within the catchment. However, the many levels of conflict that emerged from the situation can be directed at issues of governance and land-use, especially the issues involving mining legacy. Therefore, an avenue to ensure the long-term sustainable and natural flow of benefits from the landscape, in relation to mining, together with an understanding of conflicts, opinions, and of the extent of knowledge of the different land-users and uses, is required. A deeper understanding of the integrity of the ecology of the wetland is also necessary. With this knowledge, future planning can consider careful location and management options for greater sustainable land-use and conservation of biodiversity and ecological infrastructure. The aim of this study is therefore to explore mining, agriculture and wetland ecological infrastructure in the Upper Komati River Catchment, taking account of the contestations in this C-SES.

Objectives and related thesis chapters:

1) Chapter 2: Provides a contextual analysis of the case study, focusing on:
    a. Evaluation of wetland health and ecosystem services using WET-Health and WET-EcoServices (Macfarlane et al., 2009; Kotze et al., 2009).
    b. Perceptions of wetland value.
2) Chapter 3: Addresses perceived gaps in WET-Health and WET-EcoServices (Macfarlane et al., 2009; Kotze et al., 2009) methodology:
    a. Water chemistry.
    b. Macroinvertebrate community structure.
3) Chapter 4: Discusses natural resource contestation in the context of wetland health, ecosystem services and user perceptions.

Chapter 2: Contextual analysis of natural resource use, focusing on wetlands in the Upper Komati River catchment: a complex social-ecological system (C-SES)

The aim of this section is to recount the use of the WET-Health and WET-EcoServices (Macfarlane et al., 2009; Kotze et al., 2009) wetland assessments to provide an initial understanding of wetland health, and therefore, of signs of resiliency, of feedbacks and system drivers, and of potential ecosystem service provision. WET-Health and WET-EcoServices are the two resource- and catchment-based tools aimed at sustainable development of water resources and management selected for this study. They were selected because of the nexus role of wetlands in the contestation between coal mining and other resource-users. The WET-Health and WET-EcoServices assessments are presented together with water-resource user perceptions because of the importance of users in the contestation. User perceptions also indicate wetland importance (value and conservation worth) and show to what extent intervention with stakeholders is necessary, as is the land-use context. The chapter provides a basis for identifying gaps in the WET-Health and WET-EcoServices (Macfarlane et al., 2009; Kotze et al., 2009) methodology. These gaps are addressed in Section 4.4.

2.1 Bio-physical characteristics of quaternary catchment X11B, in the Carolina area

The highland grasslands of the X11B catchment are characterised by a network of aquatic ecosystems, including wetlands throughout the grasslands. The system supports a range of resource-users, from large-scale use for agriculture and coal mining to smaller-scale use by local communities. In this study, wetlands are viewed as nexus ecological infrastructural elements – where the contestation for resources between users is clearly evident. For this reason, an assessment of wetland health in relation to these different uses provides insights for the long-term sustainability of the grassland social-ecological system. In the case of the Carolina AMD crisis, contestation between agriculture and mining emerged clearly as a result of the contamination of the Boesmanspruit wetland that feeds into the Boesmanspruit dam, supplying Carolina with drinking water. The prolonged poor water conditions impinged on the human rights of townsfolk, caused issues regarding farming practices, and exposed governing institutions as being ill-equipped to respond (Tempelhof et al., 2012). It is for these reasons, wetlands have been selected as nodes of conflict manifestation for the purpose of this research.

Recently, the value of wetlands has become more widely acknowledged socially and ecologically and recognising ecosystem goods and service values can be a useful way of weighing land-use and development decisions (Pollard et al., 2013; Cools et al., 2013). Human values can be expressed as an awareness of ecosystem benefits and of the costs and disbenefits of ecosystem impairment (Houdet and Chikozho, 2014), for which reasons, natural resource management has become more concerned with and focused on ecosystem service valuation, especially in a water-scarce country like South Africa (Kotze et al., 2009). Water resource protection entails creating awareness of the value of ecosystems by promoting wetland ecosystems as ecological infrastructure, as well as understanding ecosystem feedbacks and appreciating what is involved in protecting and managing wetland ecosystem structures and functions (de Groot et al., 2010; Cilliers et al, 2013; Pollard et al., 2013).

The South African National Water Act (National Water Act, No. 36 of 1998) defines wetlands as "land which is transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface, or the land is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soil". This highlights the strong connectivity that wetlands have within the catchment landscape. That said, the main driver of wetland formation is hydrology. Wetland hydrology determines the timing and duration of flooding, chemistry of soils, the types of plants and animals, the chemistry of the water (nutrients and detrimental solutes) and the overall productivity of the wetland

Wetland ecological health is sensitive to anthropogenic activities that take place in catchments, especially in catchments highly utilised by anthropogenic land-use activities, as in the X11B quaternary catchment (Pollard et al., 2013; Cools et al., 2013). Activities can pollute water sources and exceed sediment loading capacities, change water inputs, change hydrological characteristics (water supply, timing, moderation and distribution patterns), cause erosion and sedimentation, and transform wetland and catchment vegetation (Macfarlane et al., 2009; Pollard et al., 2013), so altering hydrogeomorphic units (HGM) units, disrupting connections and feedbacks of the ecosystem and associated services, threatening biodiversity, and limiting the services the unit provides (de Groot et al., 2010; Pollard et al., 2013). This was the case in Carolina's crisis, which included flood retention and toxicant assimilation and the removal from the wetland feeding into the town's water supply dam (Tempelhof et al., 2012).

2.2 Assessing wetland health and ecosystem services

Assessment of wetland health and ecosystem service provision in South Africa is most commonly undertaken using the wetland management series "WET-Management" (Ellery et al., 2009; Kotze et al., 2009). WET-Health, an assessment tool of WET-Management, is conducted by a specialist who assesses the integrity of a wetland (the average condition of the wetland's HGM units) in relation to an unimpacted "natural reference condition" (Macfarlane et al., 2009). Designed specifically for use on South African wetlands, the tool is based on an impacts-based, risk assessment approach (Macfarlane et al., 2009).

WET-Health consists of two levels of assessment, the first is a desktop assessment based on HGM settings. The second is a rapid, field-based, qualitative, expert or specialist assessment of indicators of degradation, also based on HGM settings, which includes elements of the desktop analysis (Macfarlane et al., 2009). The level 2 analysis was undertaken in this study. Sets of processes, interactions, and interventions of three WET-Health modules were assessed individually:

1) the hydrology and the movement of water through the soil: an assessment based on activity impacts;
2) the geomorphology of the wetland units, based on indicators of loss or gain of sediment;
3) the vegetation structure for each HGM unit of the wetland, based on indicators of vegetation composition and structure change (Macfarlane et al., 2009).

By assessing WET-Health modules individually physical indicators (vegetation and geomorphology) of impacts, wetland conditions are related to the sum of magnitudes of different impact categories of human activity, which also provide insights into the causes of degradation (Macfarlane et al., 2009). Magnitudes are estimated using the extent of the impact, assessed as a percentage, multiplied by the intensity of impact, assessed as the degree of alteration caused by impact (scored on a scale of 0 (wetland unimpacted and close to the natural reference condition) to 10 (wetland critically transformed such that it has few or no wetland characteristics)). Human activities include: reduced inflow quantity, potentially altered flow patterns, canalised and modified streams. The score also takes into account impeding/obstructing features, alterations of surface roughness which may account for direct water loss, and assesses the impact of recently created depositions, infilling or excavation.

The WET-Health assessment also allows for specialists to assess the Present Ecological State (PES) of the wetland by allocating a corresponding present state category score (A–F, "A" representing unmodified and "F" representing a critical modification level) to the associated impact scores (see Table 2.1) (Macfarlane et al., 2009). The PES determination forms part of the processes for determining the ecological Reserve (Palmer et al., 2004; Kleynhans and Louw, 2007) which contributes towards creating the Resource Quality Objectives (RQOs) of a WMA in line with the mechanisms for implementing Integrated Water Resource Management (IWRM) (Palmer et al., 2004).

The WET-Health tool also includes a water quality module, which is described as a simple, coarse guideline and "allows users to superficially assess wetland health" (Macfarlane et al., 2009: 19). The water quality approach also assesses the affected wetland area using the extent (location of potential sources) and impact intensity of pollutants (sensitivity of wetland to pollution increases, type and amount of pollution). The water quality module is an indirect assessment that provides only a general indication of water quality risk, and, since a qualitative assessment of water quality risk from mining was the basis for selecting the study wetland sites, the water quality module of WET-Health was not used.

Table 2.1: Impact scores and categories of Present State used by WET-Health for the description of the integrity of wetlands (Macfarlane et al., 2009)

Impact category	**Description**	**Impact score range**	**Present State category**
None	Unmodified, natural.	0–0.9	A
Small	Largely natural with few modifications. A slight change in ecosystem processes is discernible and a small loss of natural habitats and biota may have taken place.	1–1.9	B
Moderate	Moderately modified. A moderate change in ecosystem processes and loss of natural habitats has taken place, but the natural habitat remains predominantly intact.	2–3.9	C
Large	Largely modified. A large change in ecosystem processes and loss of natural habitat and biota has occurred.	4–5.9	D
Serious	The change in ecosystem processes and loss of natural habitat and biota is great, but some remaining natural habitat features are still recognisable.	6–7.9	E
Critical	Modifications have reached a critical level and the ecosystem processes have been modified completely with an almost complete loss of natural habitat and biota.	8–10	F

Closely related to the WET-Health tool, is the WET-EcoServices tool for rapidly assessing possible wetland ecosystem service provision (Kotze et al., 2009). WET-EcoServices is a procedure for a relatively quick, specialist/expert assessment of ecosystem services supplied by wetlands. Highlighting the services that a wetland performs and can provide increased awareness among resource-users of the value of the wetland. The assessment uses descriptions of key wetland traits which serve as indicators of value (Kotze et al., 2009). WET-EcoServices has two levels of assessment, based on features that affect the extent of flow modification and alteration, and biochemical processes by the wetland. Level 1 is a desktop-based assessment on HGM settings. Where-as a level 2 assessment (used for this study) is a rapid field assessment based on HGM setting, describing key features that serve as indicators relevant to a specific service (Kotze et al., 2009). These qualitative assessments do not require quantified measures of any wetland elements such as vegetation, water chemistry, or aquatic macroinvertebrates.

By using the WET-EcoServices and WET-Health tools, this chapter provides a scoping study of the selected wetlands' ecosystem health and services in relation to land-use influences, with a specific focus on mining and agricultural activities. Applying the WET-EcoServices and WET-Health tools in an impacted system also allows limitations to be identified, and the tools to be evaluated. Further, the chapter presents the perspectives of land- and water-users in the sub-catchment in relation to contestation for ecosystem services. The objective for this chapter was to provide a contextual analysis of the case study, focusing on: 1) wetlands health and ecosystem services and 2) user perceptions.

2.3 Study area and methods

The X11B catchment experiences dry, cool to warm winters, and warm, wet summers. The mean annual rainfall falls within the range of 700–800 mm per annum and evaporation ranges between 1,650–1,900 mm per annum. The landscapes shift between dry and mesic seasons (Golder Associates, 2014). The highveld region of Mpumalanga encompasses a large number of South Africa's Fresh Water Priority Areas (FEPAs) (Nel et al., 2011). Wetlands of all types are common across this highland grassland catchment, including non-channelled valley-bottom wetlands, channelled valley-bottom wetlands, and hillslope seeps and depressions (i.e., pans) (Nel et al., 2011). These wetlands, along with the region's threatened highveld grassland biome, host an array of significant species worth conserving (Mbona et al., 2015).

2.3.1 Desktop preparation for WET-Health and WET-EcoServices: Wetland delineations

Wetlands and their immediate catchments (the highest-lying land immediately around the wetland) were delineated prior to the field assessment using desktop programs and satellite images, such as Geographical Information System (GIS) tools (ESRI, ArcGIS 10.2), Google Earth (Google Earth Pro), and 1:10 000 orthophotographs and 1:50 000 topographic photographs. From such programs and maps, the following topographic indicators were identified: low-lying areas situated in between elevated land, depressions or concave features of hill slopes, morphological and vegetation features (such as floodplains, and changes in vegetation identifiable by the changes in colour and shape) (Kotze et al., 2009; Macfarlane et al., 2009). Using these distinguishing features, the wetlands in this study, specifically those with connected channels and located at the bottom of valleys, their HGM units, and their "immediate catchment" boundaries were identified and selected.

In order to align with the Carolina 2012 incident, channelled valley-bottom units formed the focus of the study (Tempelhoff et al., 2012) because the wetland of concern in 2012 (the Boesmanspruit wetland) was a channelled valley-bottom wetland. Therefore, all wetlands selected from the identified wetlands were selected from a main channelled valley-bottom unit (Tempelhoff et al., 2012). These wetland delineations were cross-referenced with the South African National Biodiversity Institute's (SANBI) National Freshwater Ecosystem Priority Areas (NFEPA) wetland delineations, and with delineations of the SANBI's Mpumalanga highveld Wetlands project (Mbona et al., 2015) before mapping them using ArcGIS tools (ESRI). Delineations were also ground-truthed later in assessments carried out in the field (see section 2.3.2).

During the wetland identification and selection process, land cover, infrastructure, and dams within the quaternary catchment (X11B) were also mapped, and erosional features and drains were noted (Ellery et al., 2009; Kotze et al., 2009; Macfarlane et al., 2009) (Figure 2.2). Maps were constructed using spatial data (shape files) curated by SANBI, the Department of Environmental Affairs (Environment GIS), the South African National Land Cover Dataset (2013/2014) (© GEOTERRAIMAGE-2014) and polygons constructed using Google Earth Pro (Figure 2.2 and Figure 2.3).

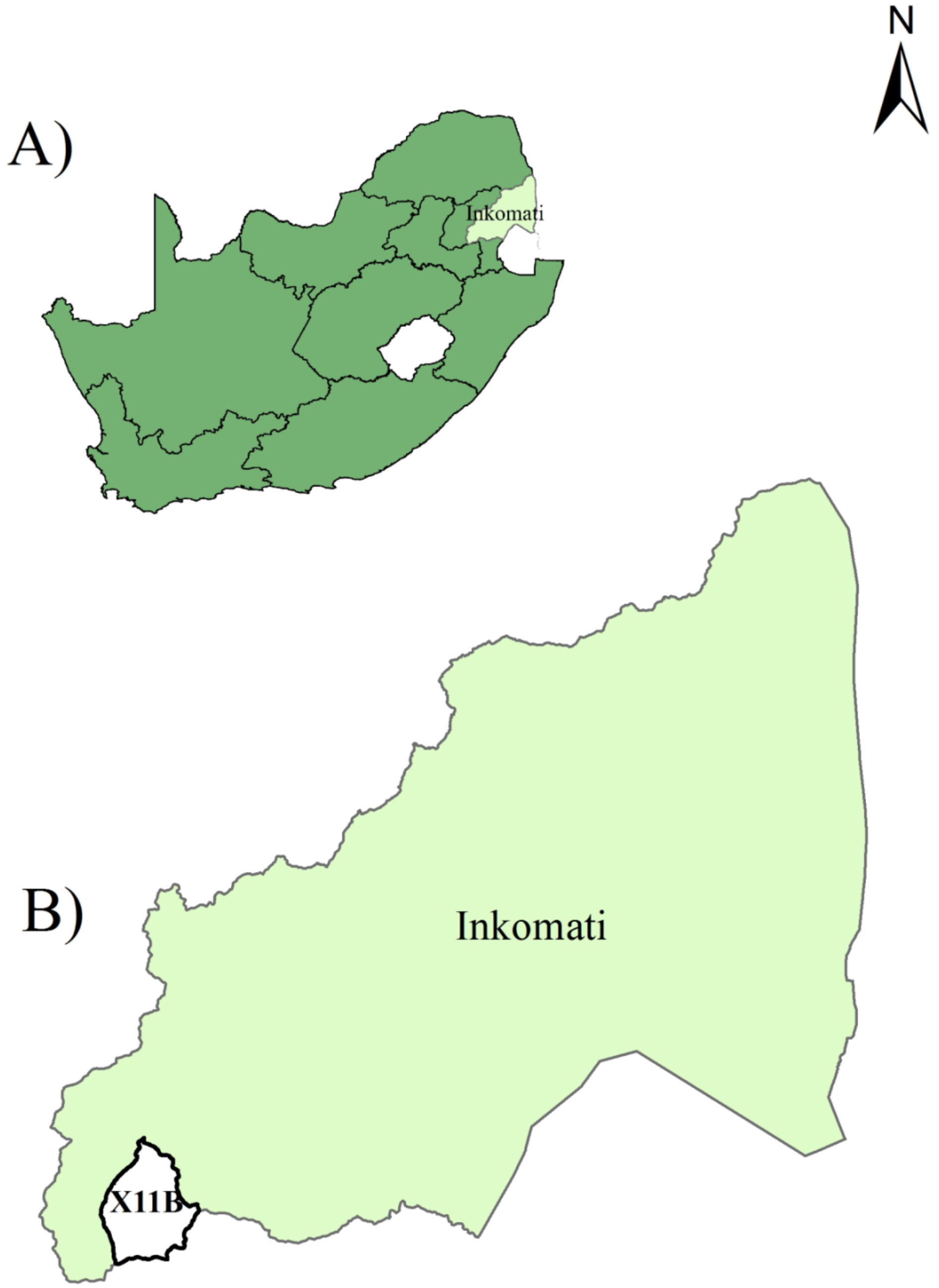

Figure 2.1: Map of South Africa showing the location of the Inkomati River catchment and the X11B quaternary catchment A) South Africa B) Inkomati catchment showing the location of the X11B quaternary catchment located in Mpumalanga.

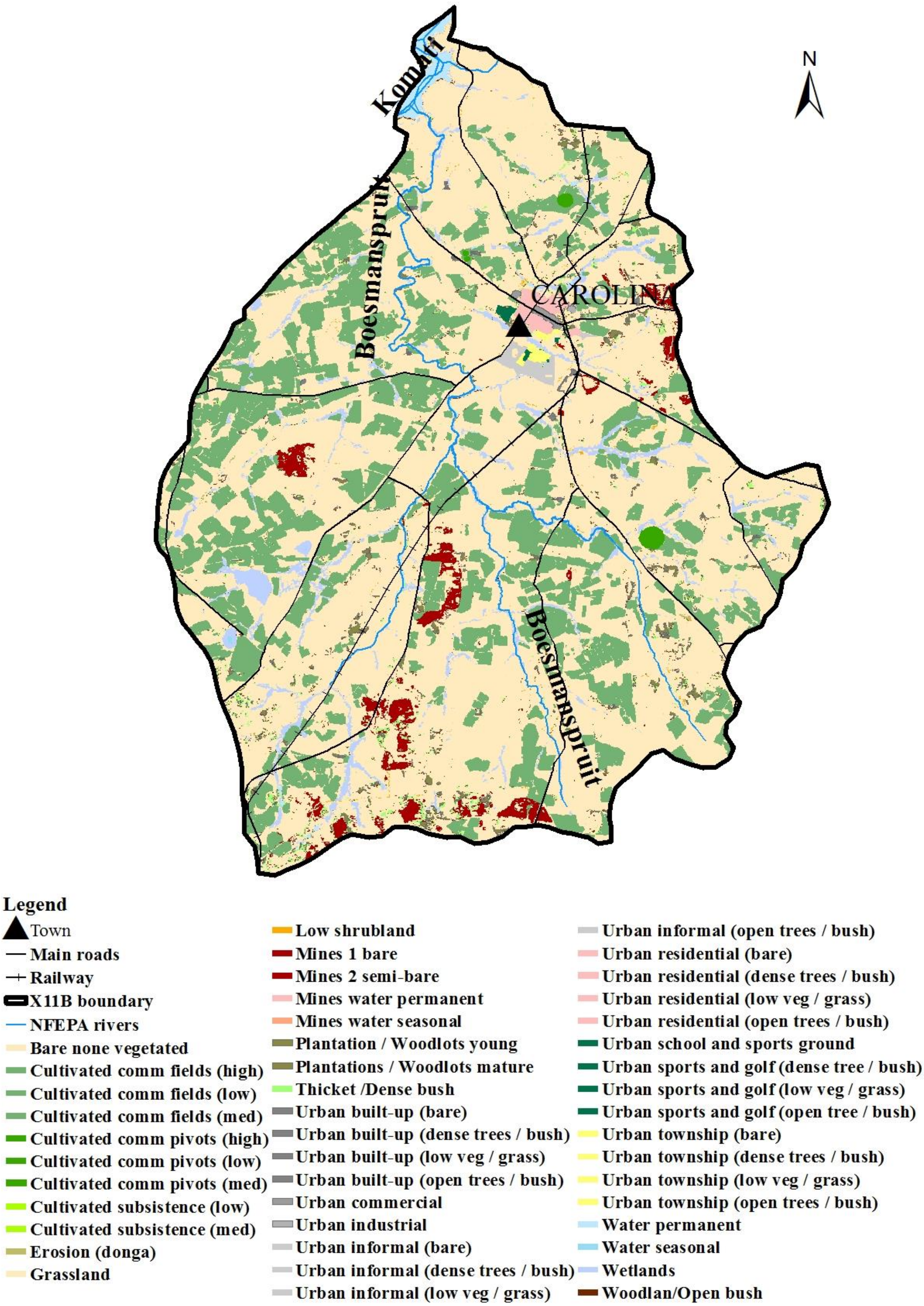

Figure 2.2: Land cover of X11B (Figure 2.1) catchment.

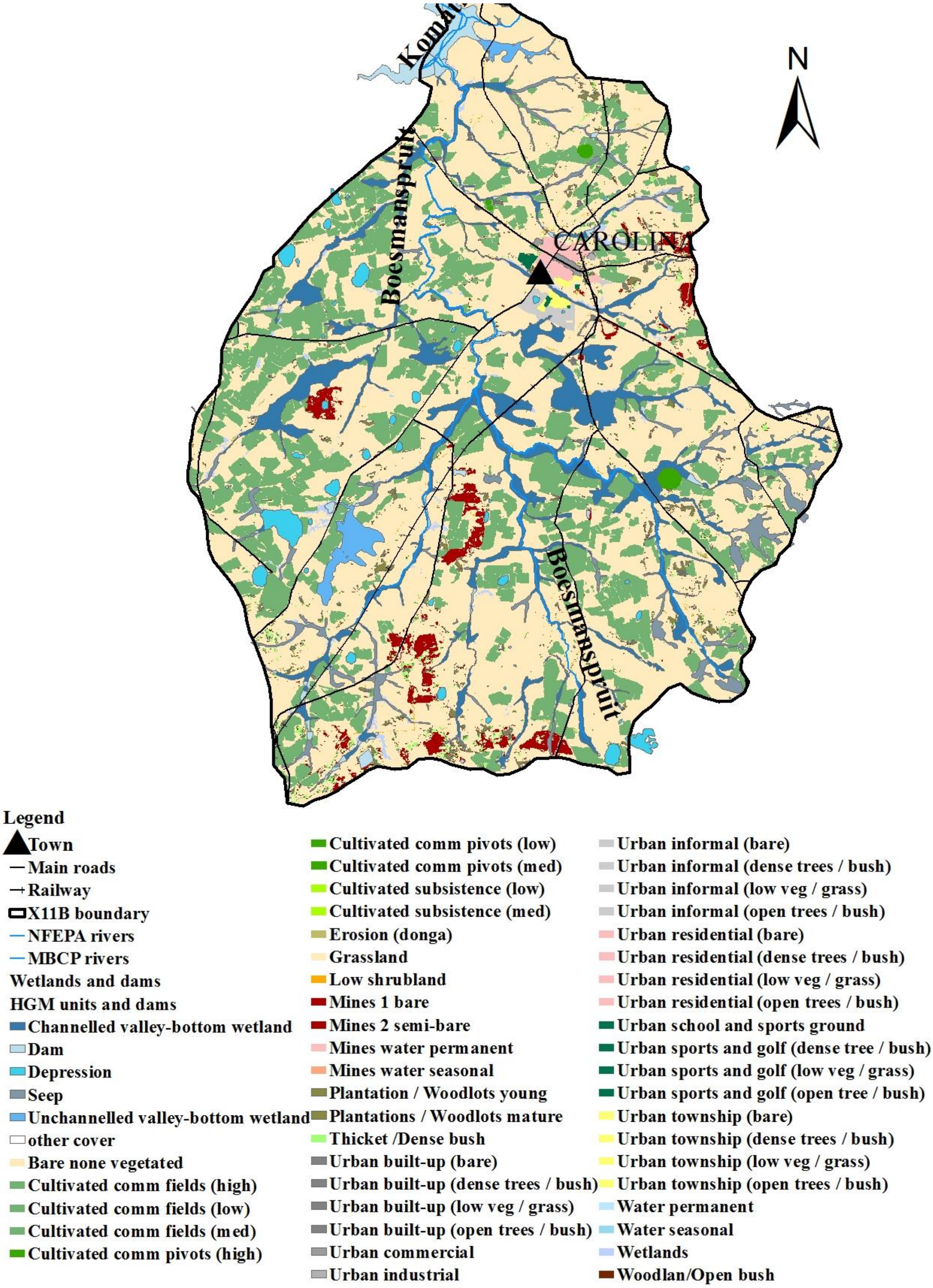

Figure 2.3: Delineated wetlands in the X11B catchment are shown darker blue areas connected with rivers.

Six channelled valley-bottoms and their connected HGM units were chosen in different contexts of land-use activity (Table 2.2, Figure 2.4, Figure 2.5) of which:

- three are situated in a combined mining and agricultural context,
- one is a historical mining decant site with some grazing,
- one is situated in a community/grazing context influenced by runoff, used by the adjacent town and community, and indirectly, by mining (sidings runoff) (Table 2.2),
- one is situated in an agricultural context without mining (grazing and some croplands), and therefore chosen as the reference site for the study.

The selected wetlands were identified according to the farm name that the larger part of the wetland was located on, i.e., Roodepoort, Boesmanspruit, Jagtlust, Witbank historical decant, Witbank, Droogvalei (Figure 2.4).

Table 2.2: Context of each wetland chosen for the study in relation to land-use activities (x = slight activity present, xx = intermediate activity present, xxx = strong activity present)

Wetland	Wetland area (ha)	In relation to Mining	In relation to Commercial Agriculture		In relation to Community Agriculture
			Crops (maize fields)	Grazing (Grassland)	
Roodepoort	21.54	No mining	x	xxx	
Boesmanspruit	206.84	xxx The main river channel on the X11B catchment is situated adjacent to and in close proximity to a coal washing plant, coal railway sidings, and an abandoned site of exposed heaps of coal. The channel is also connected to other tributary wetlands that are contaminated from active and defunct mining activities upstream	xx	x	
Jagtlust	47	x Threatened by an active mining quarry located at the top of the south-west side of the wetland's immediate catchment	xx	xx	
Witbank historical decant	153	xxx A decant outlet from a defunct mine flows into the wetland's main channelled valley-bottom unit		x	
Witbank	94.68	xx Threatened by an active mining quarry located towards the west of the wetland's immediate catchment. Rehabilitated mining activity situated at the top of the south-west end of the wetland's immediate catchment is connected to the main channelled valley-bottom wetland. Rehabilitated mining activity in the south of the catchment is showing signs of decanting into the wetland	x	xx	
Droogvalei	788.67	x Experiences runoff from the coal railway sidings adjacent to the wetland	x		xxx Grazing

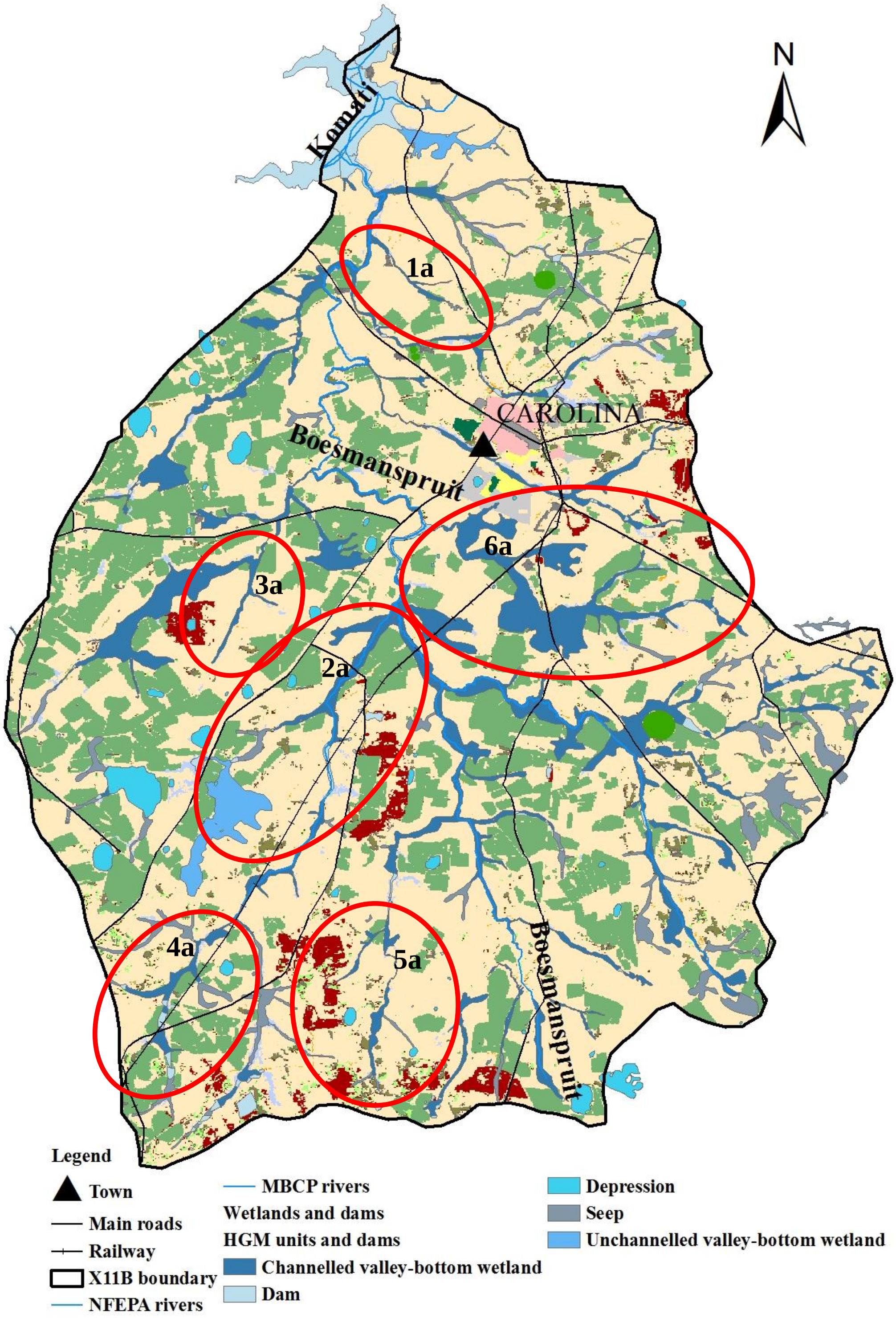

Figure 2.4: Selected wetlands and their associated hydrogeomorphic units (HGM). Wetlands are indicated by circles: 1a) Roodepoort, 2a) Boesmanspruit, 3a) Jagtlust, 4a) Witbank historical decant, 5a) Witbank, 6a) Droogvalei. These are shown at a finer scale in Figure 2.5.

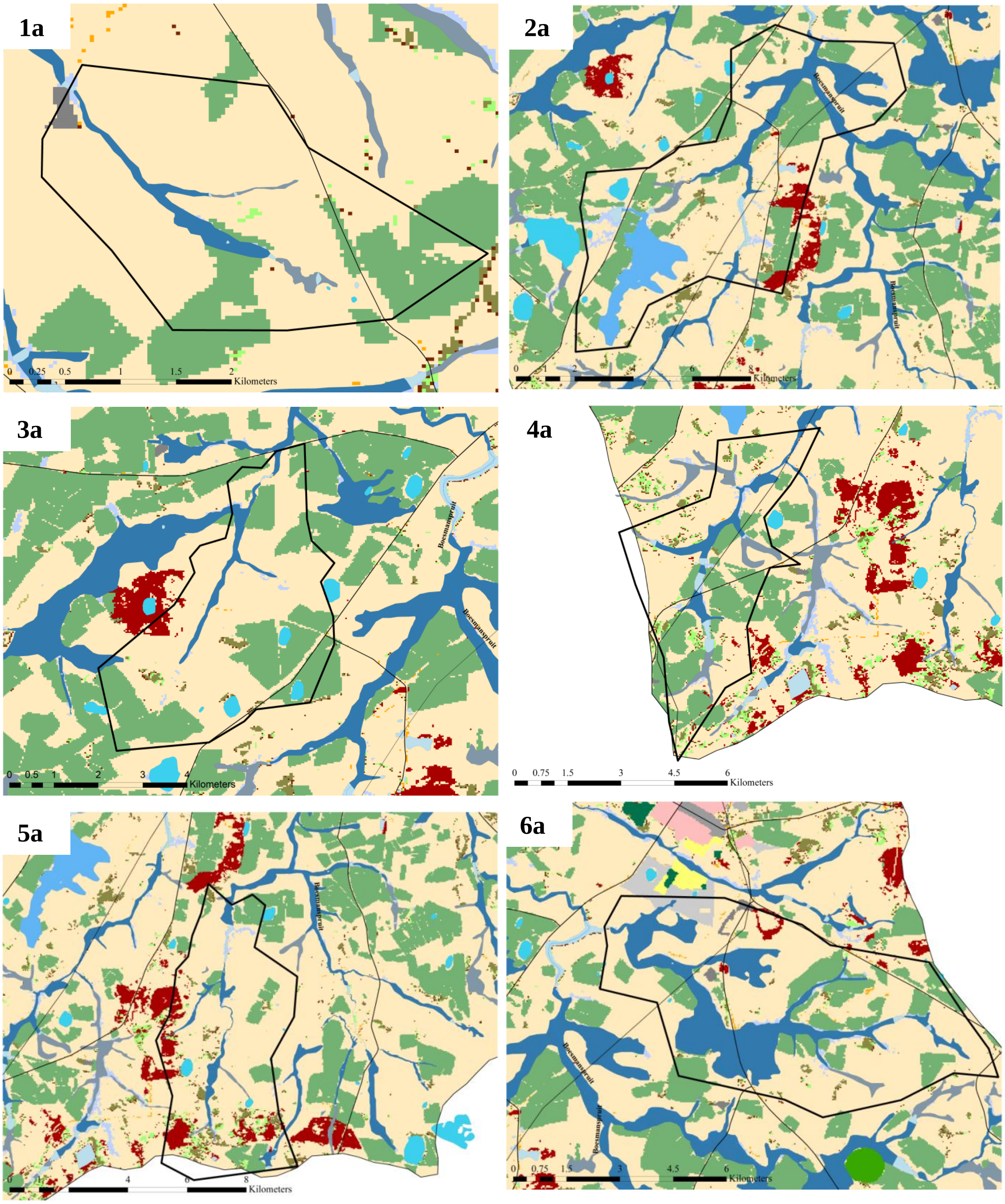

Figure 2.5: Studied wetlands showing the estimated immediate catchment boundary of the wetland itself: 1a) Roodepoort, 2a) Boesmanspruit, 3a) Jagtlust, 4a) Witbank historical decant, 5a) Witbank, 6a) Droogvalei.

2.3.2 Field assessments

Field assessments were carried out to validate the desktop delineations and apply the relevant WET-EcoServices and WET-Health methods to assess the integrity of the current wetlands and the ecosystem services present at the selected sites (Kotze et al., 2009; Macfarlane et al., 2009). Water quality measurements (pH, dissolved oxygen, electrical conductivity) were also taken at each site. Site field assessments (WET-EcoServices and WET-Health and water quality measurements) were carried out in August (dry season) on the following dates: 1a-27/08/2015, 2a-22/08/2015, 3a-23/08/2015, 4a-22/08/2015, 5a- 4/08/2015, 6a-24/08/2015.

Wetland delineations

The wetland HGM units were the units of assessment for the study. Delineations of the main channelled valley-bottom HGM unit of each wetland were confirmed by identifying specialised characteristics of soil morphology and vegetation types (hydrophytes) in the field (Collins, 2005; Van Ginkel et al., 2011; Ollis et al., 2013). Soil samples were collected using a soil auger at two depths: 0–10 cm and 40–50 cm (Department of Water Affairs and Forestry, 2005). This provided data in terms of soil mottling contrast and abundance (indicating oxygen presence in the soil). Soil mottles of rich colours (red, orange and yellow) are the result of the presence of metal precipitation and are created in areas of wetlands where flooding is more seasonal (Figure 2.6). Higher levels of oxygen result in more mottles, thus areas further away from the permanently saturated soils will be more mottled and have more colour (Ollis et al., 2013). The Munsell soil chart was used to assess the soil matrix hue and chroma, which indicated the amount of iron oxide reduction and therefore soil saturation (greyer soils indicate saturation over relatively long periods of time) (Collins, 2005; DWAF, 2005). As soils with greater saturation and less oxygen become more prevalent permanent zones of the wetland, plant species will gradually adapt to anaerobic growing conditions (hydrophytes), so indicating the wetland boundary (DWAF, 2005). Random soil samples were collected at observed vegetation composition boundaries to confirm wetland and ecological zones (i.e., permanent, seasonal and temporary zones, where the outer boundary of a wetland is defined by the outer margin of the temporary zone) (Collins, 2005). This method was repeated from an area outside the wetland, progressively moving into the wetland.

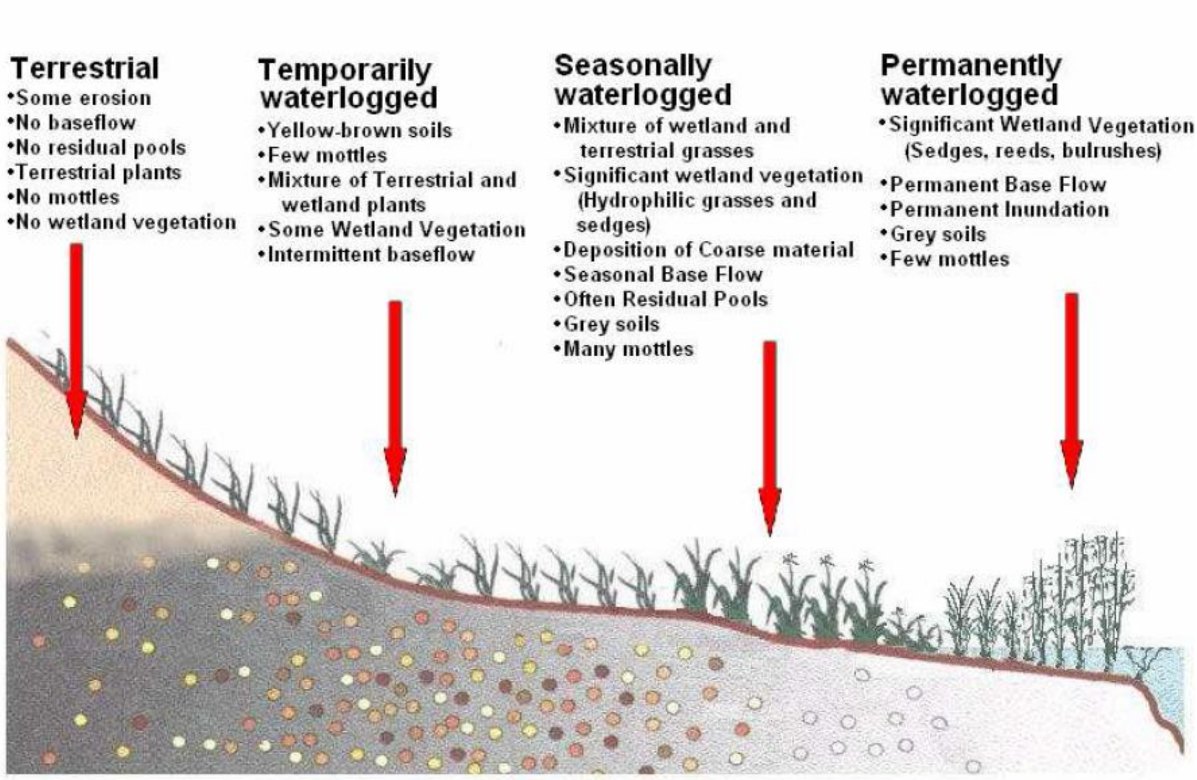

Figure 2.6: Schematic of a cross-section through a valley-bottom wetland, indicating changes in the soil wetness and vegetation indicators, as one moves along a gradient of increasing wetness, from a terrestrial zone to the permanently wet hydrological zone (from: Collins, 2005).

Ecosystem services assessments

WET-EcoServices were used to assess individual ecosystem services, applying a scoring system based on a rating devised by a group of specialists An aquatic ecologist Professor CG Palmer, Rhodes University, a grassland ecologist was Dr AR Palmer, Agricultural Research Council, assisted by three MSc and one PhD post-graduate students in water resource science from Rhodes University (as indicated in Appendix B), to gauge each service provided by the wetland HGM units (Kotze et al., 2009). Specific check-sheets were used to assess the possible service provision by the various wetland units. Scores were based on a range of between zero (delivering an ecosystem service to a limited extent) and four (delivering an ecosystem service to the maximum extent), allocated according to the specialist’s judgment. Individual scores from the field check-sheets were then entered into Microsoft Excel datasheet templates, part of the WET-EcoServices electronic tool kit. The electronic tool kit formulated, calculated and presented overall scores of the services provided based on the average service score allocated to each service, also calculated as part of the WET-EcoServices electronic tool kit. Overall scores were determined by the calculation of a combined average of effectiveness (of the wetland unit in supplying the service) and the opportunity (wetland unit supplying the ecosystem service) of services. The overall scores were then rated according to the likely extent of the benefit supplied (Table 2.3) and the scores were translated into radar diagrams for each HGM unit of each wetland.

For the purposes of this study the following services formed the basis of a level 2 assessment analysis (Kotze et al., 2009):

- flood attenuation
- regulation of stream flow
- sediment trapping
- phosphate assimilation
- nitrate assimilation
- toxicant assimilation (include heavy metals and biocides)
- erosion control
- carbon trapping
- maintenance of biodiversity
- water supply
- provision of natural resources
- provision of cultivated food
- cultural significance
- tourism and recreation
- education and research

Table 2.3: Categories for determining the likely extent to which a benefit is being supplied based on the overall score for that benefit

Score	**<0.5**	**0.5–1.2**	**1.3–2.0**	**2.1–2.8**	**>2.8**
Rating of the likely extent to which a benefit is being supplied	Low	Moderately low	Intermediate	Moderately high	High

Wetland health assessment in current state

For each module (hydrology, geomorphology and vegetation), a table and guideline were provided as part of the WET-Health tool set, with each module having its own formatted Excel document. Local knowledge from conversations with stakeholders, and judgements made by the specialist team provided the following information which was used to complete the Excel documents and formulate the impact of each category: the extent (proportion affected by an activity), intensity (degree of alteration resulting from given activity), and the magnitude (product of extent and intensity) of the impact (Macfarlane et al., 2009). The overall summary of each module's scores calculated in the Excel datasheets, determined the final Present State score (Based on the criteria outlined in Table 2.1).

Using WET-Health, specialists (Appendix B) were able to determine an intermediate ecological Reserve. From these health category results, an average area, weighted score of each module was calculated for the whole wetland. The overall health score was then calculated using each of the whole wetland's module health scores by adding and multiplying weighted factors according to the contribution of individual modules to the health of the wetland (Macfarlane et al., 2009): Health = ((Hydrology score) x 3 + (Geomorphology score) x 2 + (Vegetation score) x 2) ÷ 7. These scores were then allocated corresponding health categories in the same manner as the previous individual scores.

WET-Health and WET-EcoServices rely on specialist knowledge, based on theoretical understandings, and do not include empirical procedures, nor do they give an indication of the chemical impact of activities. Therefore, in order to gain a more detailed understanding of the wetlands, simple water quality measures were taken (Macfarlane et al., 2009). In rivers, macroinvertebrate taxa have an assigned water-quality sensitivity scale that is used in a biomonitoring, river health assessment method called the South African Scoring System (version 5) (SASS5) (Dickens and Graham, 2002). SASS5 has been criticised as an unreliable assessment method for wetlands (Bird and Day, 2010), but was used at this scoping level with the justification that the wetlands selected are channelled and have flowing water. In Chapter 3, macroinvertebrate sampling and analysis is extended, and additional methods used are described. Macroinvertebrate community structure as indicated by SASS5 was therefore added to the WET-Health and WET-EcoServices methods of wetland assessment.

Water quality sampling

At all sites the following were usually recorded: Water pH (standard units), dissolved oxygen (DO; mg/l), electrical conductivity (EC; μS/cm), total dissolved solids (TDS; mg/l), and temperature (°C), using a Hanna multi-parameter meter (HI 9829).

Aquatic biota

At all wetland sites (Figure 2.4), a 1 m length of marginal vegetation was selected from the main channelled valley-bottom wetland HGM units, following the steps from the SASS5 method of macroinvertebrate collection (Dickens and Graham, 2002). Using a 1 mm^2 mesh on a 30 cm square-shaped frame with a sturdy handle, vegetation was pushed robustly, moving backwards and forwards in the same area (Dickens and Graham, 2002). The net was then emptied, by first washing samples down to the bottom of the net, carefully inverting and flushing (with water) the net out into a flat-bottomed tray (approximately 30 x 45 cm and 10 cm deep). Any specimens remaining in the net were added to the tray, using forceps. The sample was then fully immersed by adding clean water, and debris was removed after checking for any attached macroinvertebrates. Using forceps, soft plastic wide-mouth pipettes, and a magnifying glass, biota were then identified to family level, for 15 minutes, using a field identification book; data was captured in SASS scoring sheets (Dickens and Graham, 2002). SASS5 results were compared among all sites, with the Roodepoort wetland – the reference site for the qualitative assessment – being the least impacted.

Informal conversations

Informal conversations were conducted with various participants, including the owners of Jagtlust, Witbank, and Roodepoort farms, three other local residents, and the manager of a coal washing plant. Each participant was asked pre-formulated questions, aimed at capturing reflections on the past and present activities that took place in each wetland's immediate catchment, what impacts there were and they were aware of, what they depended on the wetland for (any ecosystem services and benefits they were aware of), and any other relevant local knowledge they had. Conversations were a hour long and were conducted on the same dates as field assessments of each site. The insights acquired were then used in the contextual introduction of each wetland adding vital local knowledge to and understanding of each wetland.

Several participants were asked about the Boesmanspruit wetland because it is a relatively big wetland which spreads over a large area and feeds the Carolina water supply dam, and was directly affected by the Carolina AMD crisis. The Droogvalei wetland forms part of a land claim initiative, so the community elder was interviewed about that wetland.

2.4 Results

Use of land and water resources is keenly contested in the X11B quaternary catchment (Figure 2.2). Agricultural use includes dryland maize cultivation, and the use of natural grassland for grazing. The mining is generally coal extraction. Wetlands were identified as key nexus elements of the landscape: they are the ecological infrastructure elements that provide essential ecosystem services to both agricultural and mining land- and water-users (Collins, 2005). Wetland conditions, and the relationship people have with wetlands, were therefore used as indices of the state of the sustainability relationship between people and the landscape (DWAF, 2007).

2.4.1 Wetland overviews

This section provides a contextual social-ecological overview of six selected wetlands and the people who use them. Each of the selected wetlands had a different combination of use for agriculture and mining (Table 2.2). The sites were selected to enable recognition of individual and combined impacts of agriculture and mining.

There are limitations to this initial contextual assessment. Conversations were only with a small set of resource-users because they were immediately available; the conversations are therefore used to provide general narrative context. Wetland health and ecosystem services were assessed in a dry winter period, dictated by the time-frame of the study. Summer macroinvertebrate and water quality assessments are reported in Chapter 3.

Roodepoort wetland

The Roodepoort wetland (21.54 ha) is a main channelled valley-bottom wetland (wetland HGM unit one) that flows throughout the year. The middle of this main channel veers to the east into a small dam that is fed by a hillslope seep (wetland HGM unit two). The main channel flows from the south of the catchment and is fed by another hillslope seep (wetland HGM unit three) (Figure 2.5). The immediate wetland catchment is situated in an agricultural setting with no visible mining influence. A meeting with the Roodepoort landowner indicated that she has been an active member of the Carolina community for more than 20 years. She is a passionate crop, sheep, cattle and horse farmer, and owns thousands of hectares of farm land in the X11B catchment. Her farms are located north of Carolina. She is aware that there is an array of natural freshwater ecological infrastructure on the farms and practises sustainable irrigation and farming techniques; however, she acknowledged that her cattle do cross through the wetlands located in the farm landscape – causing trampling and erosion.

She has noticed that winter season grazing of the wetland is more intense, as the cattle seek and select palatable grasses. About ten years ago she built a road crossing in the middle region of the wetland. She has built strategic dams in order to supply three houses located on her land. Two of these houses are currently vacant so no water is being used from the dam. One house at present does use the water, but only in small amounts.

Roodepoort wetland WET-EcoServices assessment

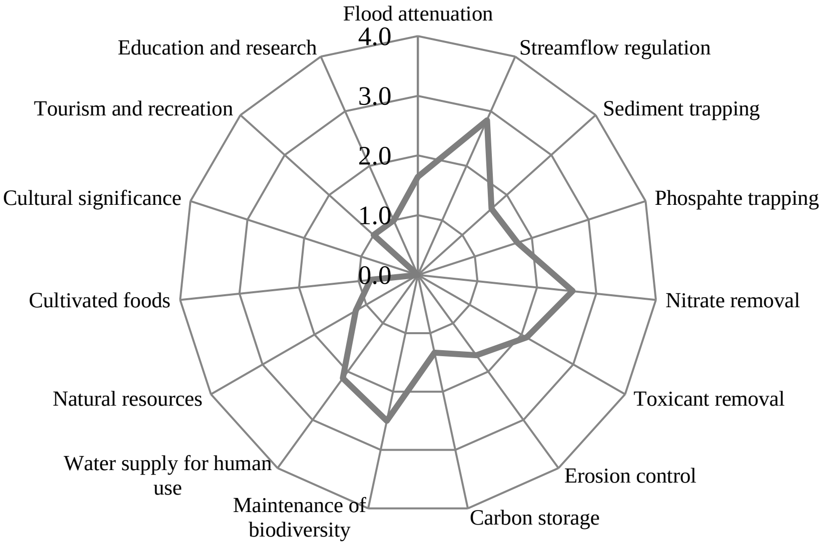

Figure 2.7: WET-EcoServices (Kotze et al., 2009) scores for Roodepoort wetland HGM unit one (channelled valley-bottom wetland).

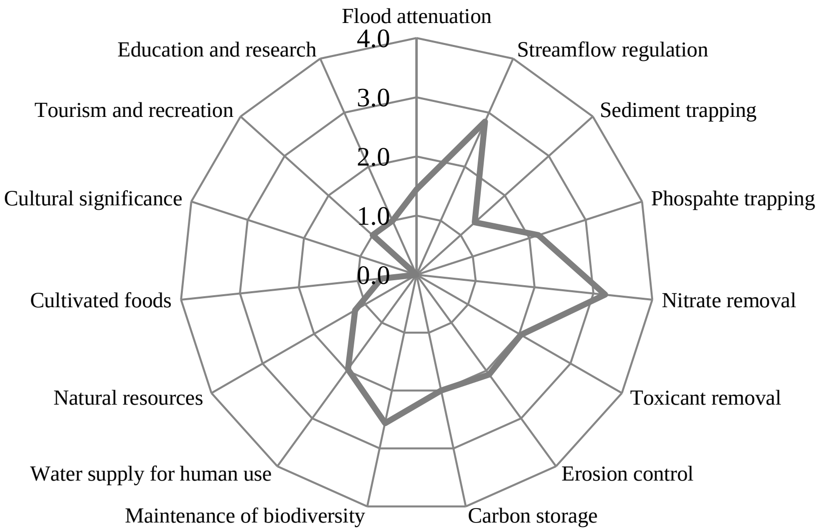

Figure 2.8: WET-EcoServices (Kotze et al., 2009) scores for the Roodepoort wetland HGM unit two (hillslope seep linked to a stream channel).

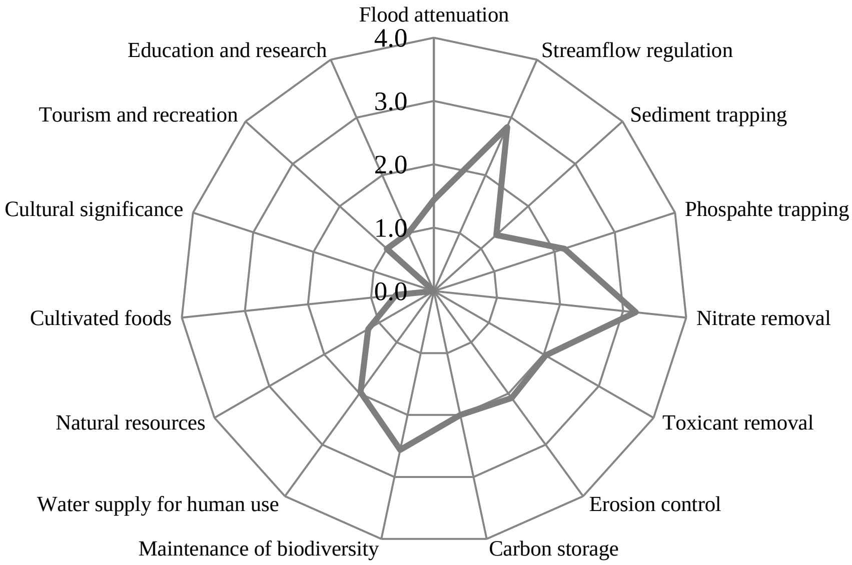

Figure 2.9: WET-EcoServices (Kotze et al., 2009) scores for the Roodepoort wetland HGM unit three (hillslope seep linked to a stream channel).

Wetland HGM unit one evidenced heavy grazing on the banks of the wetland, so reducing vegetation cover, increasing erosion and destabilising the sides of the channel. Investigation suggested that the impacts have changed the features of the wetland, resulting in reduced ecosystem services being provided by the main channel wetland.

Roodepoort wetlands WET-Health assessment

Table 2.4: Present State category for each module and the overall Present State of Roodepoort wetland

Wetland	Present State category and likely			
	Hydrology	**Geomorphology**	**Vegetation**	**Overall Present State**
Roodepoort (1a)	A	A	B	A

Observation of the Roodepoort wetland HGM unit one indicated trampling by cattle, and therefore removal of sediment from the unit, resulting in channel widening and deepening. The cattle grazing in and around the main channel has resulted in a variety of issues, including instream channel erosion that has led to sediment removal, encouraging further erosion. These effects may have created impacts that differ from the other wetlands investigated; however, their magnitudes were judged as not being great enough to reflect modification in the wetland (Table 2.4). The overall health of the wetland was an “A” and the water quality indicated the good condition of the water (Table 2.12). The site is not recognised as a NFEPA, however the landowner depends on the landscape for an agricultural livelihood.

Boesmanspruit wetland

The Boesmanspruit wetland (206.84 ha) consists of a main channelled valley-bottom wetland (wetland HGM unit one) that flows throughout the year, and forks to form two arms upstream towards the south end of the catchment. The east arm terminates at a relatively large earth dam and the west arm ends at a small earth dam and is followed by a change in HGM unit type. The HGM unit (wetland HGM unit two) is a hillslope seep that continues south, towards the top of the catchment where, after another dam, the wetland changes into another HGM unit (not included in this assessment) (Figure 2.5).

The wetland is situated in a small sub-catchment within the X11B catchment, referred to as the Witrandspruit. The wetland is relatively large, and is situated in an agricultural setting with direct and indirect mining impacts and is one of three catchment systems that feeds directly into the Boesmanspruit dam, supplying 31% of the inflow to the dam (McCarthy and Humphries, 2013). The wetland is of significant importance in the catchment, as it is the main tributary of the catchment and flows into the Boesmanspruit dam. The major contaminants recorded in the 2012 incident were pollutants associated with AMD from the Boesmanspruit wetland and the connected Witrandspruit catchment.

Current mining-associated sites in the Boesmanspruit wetland catchment include: 1) a railway coal siding, where the site is flattened and compacted and coal is left exposed to the elements before being loaded on a train; remnants of finer coal remain after the coal is loaded, 2) the coal washing plant, where coal is piled and left exposed, and 3) an historic coal mining site with small discarded coal heaps. Sites one and two are situated in close proximity east of the main channel unit, towards the middle of the catchment, and site three is located at the top of the west side of the wetland’s immediate catchment, opposite the coal siding. The catchment is also connected further south of the primary X11B catchment through a riparian channel that is influenced by upstream mining.

Contestation among resource-users of the Boesmanspruit site was identified between the coal washing plant manager (considered an environmentally conscious member of the Carolina community) and a farm owner within the catchment. The manager's washing plant has, in the past, been accused of contributing to the high acidity and salinity of an adjacent pan/depression and the Boesmanspruit wetland, via acid mine decant and coal-contaminated runoff. The manager made it clear that all the appropriate rules and regulations had been followed, and Environmental Impact Assessment (EIA) studies had all been carried out and accepted, hence he did not take any responsibility for the contamination of either wetland. The farm landowner felt strongly about the accusation, as he claimed to have seen the impact in dead vegetation where runoff from the coal washing plant channels into the Boesmanspruit wetland.

Boesmanspruit wetland WET-EcoServices assessment

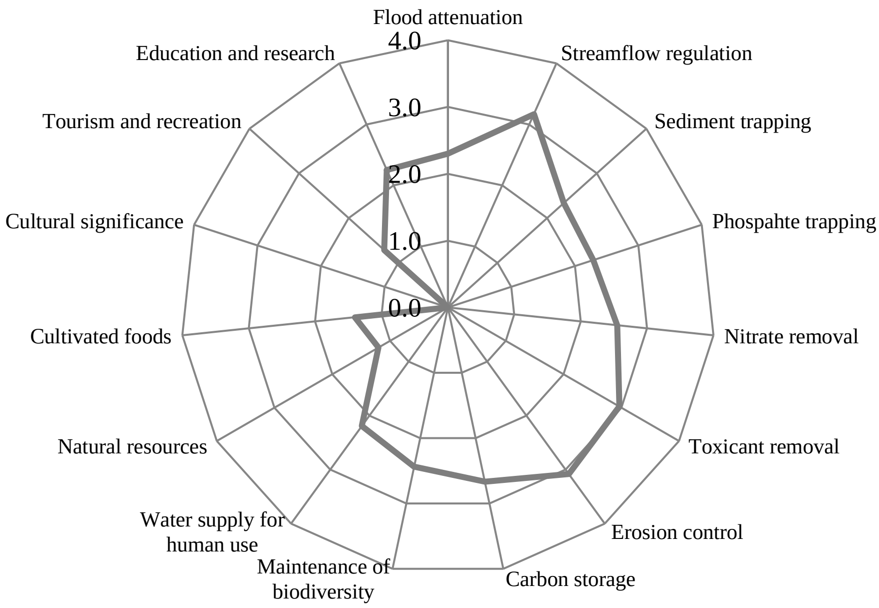

Figure 2.10: WET-EcoServices (Kotze et al., 2009) scores for Boesmanspruit wetland, HGM unit one (channelled valley-bottom wetland).

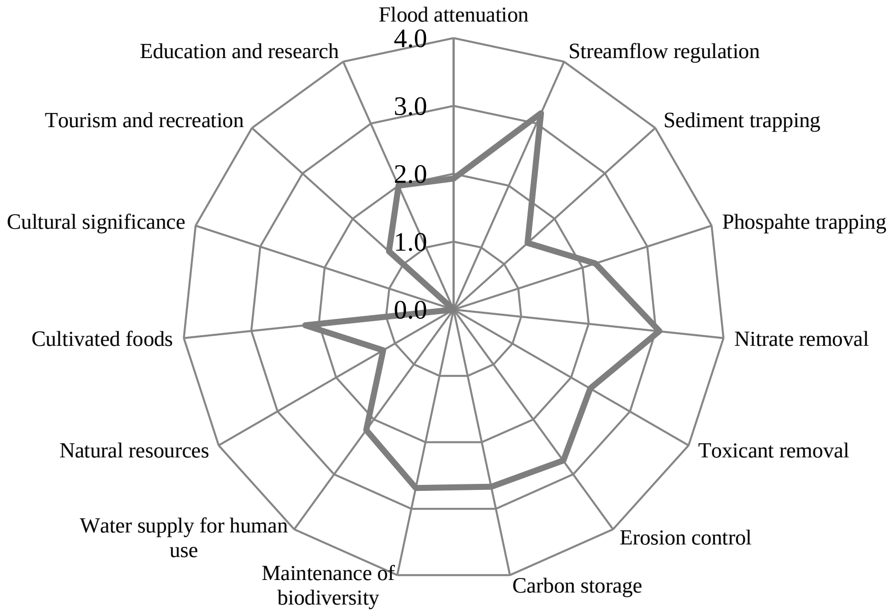

Figure 2.11: WET-EcoServices (Kotze et al., 2009) scores for Boesmanspruit wetland HGM unit two (hillslope seep linked to a stream channel).

Boesmanspruit wetland WET-Health assessment

Table 2.5: Present State category for each module and the overall Present State of Boesmanspruit wetland

Wetland	**Present State category**			
	Hydrology	**Geomorphology**	**Vegetation**	**Overall Present State**
Boesmanspruit (2a)	C	C	B	C

Most of the impacts in the Boesmanspruit associated with wetland integrity were impeding features/infilling and recent deposition and slight gully impact in the wetland. These affected vegetation and flow. Scores indicated the overall influence of the immediate catchment's activities has been a negative overall impact on the wetland's health, with a Present State of a "C" category (Table 2.5). Water quality was also poor, with a low pH (3.86) and high EC (423 µS/cm) (Table 2.12). The wetland health was estimated as being poorer than the ecosystem service provision; this indicated wetland resilience, with wetlands providing ecosystem services even at lower levels of wetland health.

The SANBI NFEPA project has identified this wetland as a site of conservation importance (reviewed at the NFEPA National Stakeholder Review Workshop, July 2010) because most of the wetland area is within “a sub-quaternary catchment that has sightings or breeding areas for threatened Wattled Cranes, Grey Crowned Cranes and Blue Cranes” (Nel, et al., 2011). As a result, the poor water quality of the wetland site has biodiversity conservation implications.

Jagtlust wetland

The Jagtlust wetland (47 ha) (Figure 2.4 and Figure 2.5) is a channelled valley-bottom wetland that feeds the main Boesmanspruit tributary stream after the Boesmanspruit dam, leading to the Nooitgedacht reserve. Although there was no visible surface flow, a channel in the wetland was present and the ground was noticeably saturated. The wetland is located in a mainly agricultural setting with the least proximity to defunct mining sites, south of Carolina. There is still potential for impact from new open-pit mining currently taking place at the top of the west slope on the catchment. After desk-top scrutiny, this wetland was considered least impacted by mining. Mining operation regulations require the mines to address environmental liabilities, pollution and ecological practices through a risk assessment process (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, African Mining and Biodiversity Forum, and South African National Biodiversity Institute, 2013). Therefore, the mining activity has not yet shown significant impact on the wetland, and the threat is likely to emerge only after closure if adequate closure precautions are not taken.

The Jagtlust landowner’s family has been part of the Carolina community for five generations; his family owns more than ten thousand hectares of farmland in the X11B catchment. The farm owner was aware of the high hydrological connectivity on his land and uses the ecosystem service – provision of water – to his advantage. He is aware of a particular spring that is relatively full even in the dry season, both historically and currently, which feeds a channel that enters a dam downstream of the Jagtlust wetland site. The dam is used to pipe water to other areas on his farms in times of drought and/or in the dry season. The owner is concerned about the contamination of water between his farms, due to hydrological connectivity and he is also troubled by the prospective mining plans that were said to be starting in 2016 in regions close to his land.

Jagtlust wetland WET-EcoServices assessment

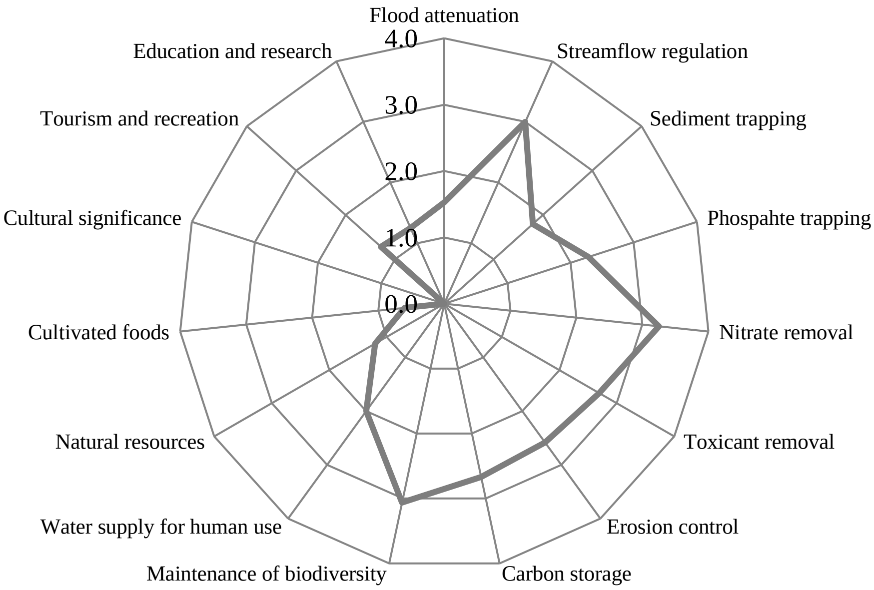

Figure 2.12: WET-EcoServices (Kotze et al., 2009) scores for Jagtlust wetland, HGM unit one (channelled valley-bottom wetland).

The wetland ecosystem services assessment (Figure 2.12) indicated high potential for biodiversity maintenance, nitrate removal and streamflow regulation, and moderately high toxicant removal. This is important as the wetland could be threatened by the catchment's mining activity after closure (expected in 2016/2017). The effectiveness of the toxicant removal service was initially judged as high; however, the active status of the mine lowered the likelihood scores, resulting in a moderately high potential toxicant removal score.

Jagtlust wetland WET-Health assessment

Table 2.6: Present State category for each module and the overall Present State of Jagtlust wetland

Wetland	**Present State category**			
Jagtlust (3a)	**Hydrology**	**Geomorphology**	**Vegetation**	**Overall Present State**
	A	A	A	A

The Jagtlust wetland site had low EC and TDS (Table 2.12), indicating better wetland health, but pH was low (4.5), suggesting AMD infiltration (Table 2.12), and it was therefore necessary to find a site north of Carolina, unaffected by mining. Although the wetland has been categorised as “A”, there is still a pending threat from the mining activity, which has resulted in concerns expressed by the farming landowner. The wetland is a breeding site for threatened Wattled Cranes, Grey Crowned Cranes and Blue Cranes, hence is classified as a NFEPA site (reviewed at the NFEPA National Stakeholder Review Workshop, July 2010). Not only is the wetland significant for biodiversity, but the landowner recognises the importance of the wetland for good water quality and high hydrological connectivity.

Witbank historical decant wetland

The Witbank historical decant wetland (153 ha) is made up of a main channelled valley-bottom (wetland HGM unit one). This main channel includes a tributary fed by a hillslope seep (wetland HGM unit two) (Figure 2.4 and Figure 2.5). Further upstream of the main channel (south of the catchment) the channel forks, and towards the west, the main channel continues and the wetland ends. Towards the east the channel is dammed and fed by two small channelled valley-bottom wetlands separated by dams, which are ultimately fed by another hillslope seep (wetland HGM unit three) (Figure 2.5). The hillslope seep ends at the southern-most boundary within the Witrandspruit sub-catchment and indirectly feeds into the Boesmanspruit wetland via a stream.

In the dry season, the main channelled valley-bottom wetland (unit one) was saturated with large pools of water and some flow. Historically, the wetland was the site of mining activities in and adjacent to the channelled valley-bottom wetland, HGM unit one. The historical mining activity has been rehabilitated. However, there is evidence of AMD decanting into the wetland and the decant tributary is being treated for low acidity. The third unit of the wetland may also experience decant from the now largely rehabilitated open-pit colliery situated at a higher elevation upstream (Figure 2.4).

Witbank historical decant wetland WET-EcoServices assessment

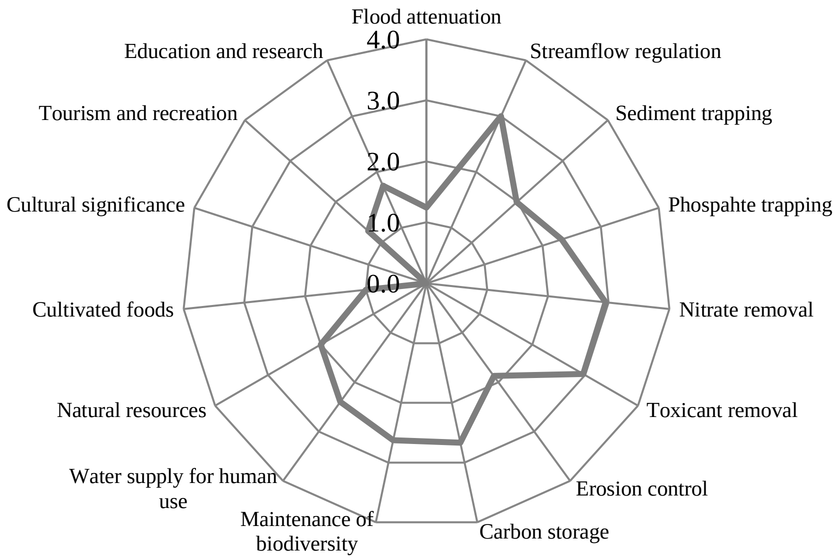

Figure 2.13: WET-EcoServices (Kotze et al., 2009) scores for the Witbank historical decant wetland HGM unit one (channelled valley-bottom wetland).

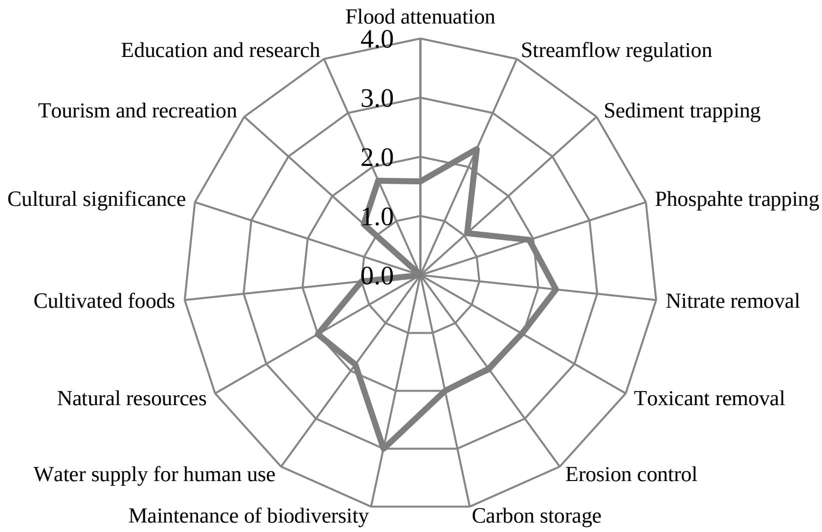

Figure 2.14: WET-EcoServices (Kotze et al., 2009) scores for the Witbank historical decant wetland HGM unit two (hillslope seep linked to a stream channel).

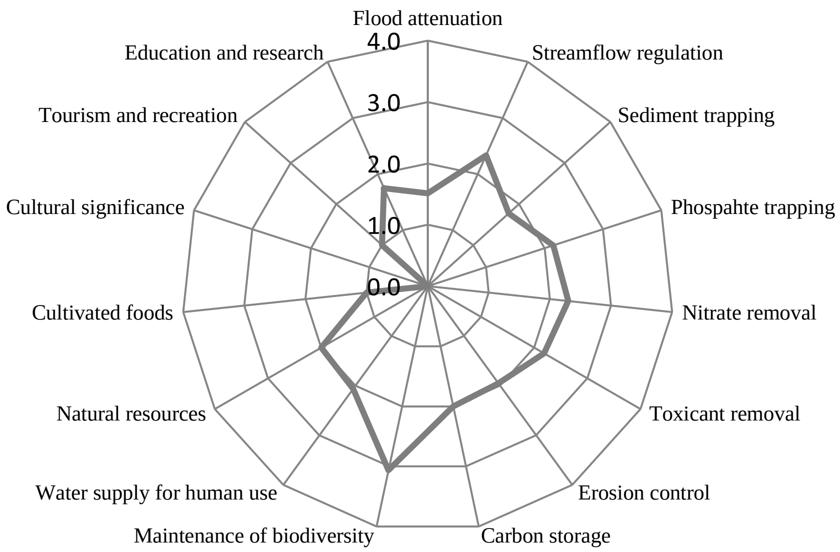

Figure 2.15: WET-EcoServices (Kotze et al., 2009) scores for the Witbank historical decant wetland HGM unit three (hillslope seep linked to a stream channel).

Witbank historical decant wetland WET-Health assessment

Table 2.7: Present State category for each module and the overall Present State of Witbank historical decant wetland

Wetland	**Present State category**			
Witbank historical decant (4a)	**Hydrology**	**Geomorphology**	**Vegetation**	**Overall Present State**
	A	B	B	A

The overall integrity was categorised as an "A", indicating a near-to-natural wetland (Table 2.7). However, the water quality revealed a low pH (3.24) and high EC (457.8 µS/cm) and TDS (Table 2.12), indicating an unhealthy aquatic system. Little information is available on this wetland's immediate catchment, but implications of the poor water condition include possible impacts on breeding areas of threatened crane species (Nel et al., 2011). These contradictory lines of evidence indicate the need for further study (Chapter 3).

Witbank wetland

The Witbank wetland (94.68 ha) is made up of a main channelled valley-bottom wetland (wetland HGM unit one). Towards the north of the main wetland, a valley-bottom wetland tributary feeds into the channel. The tributary flows from a rehabilitated mining site located west of the immediate catchment and forks in the main channelled valley-bottom wetland. The forked tributary is regarded as a separate valley-bottom wetland (wetland HGM unit six). The Witbank wetland is situated at the southern-most point and occupies a relatively small portion of the Boesmanspruit X11B sub-catchment (McCarthy and Humphries, 2013). This sub-catchment is one of the three sub-catchments that feed into the Boesmanspruit dam, supplying 60% of the dam’s inflow (McCarthy and Humphries, 2013). The Witbank wetland is connected to the dam via the Boesmanspruit River and other connected wetlands (McCarthy and Humphries, 2013).

The main channel of the Witbank wetland flows from the south of the catchment and is fed by three hillslope seeps (wetland HGM units two, four, and five) (Figure 2.4 and Figure 2.5). Wetland HGM unit two is connected to a depression (wetland HGM unit three). In the winter season, the main channel contained only small pools of water, mainly in the upper catchment; downstream of the wetland, the soil remained saturated with no pools or flow. The north tributary (Wetland HGM Unit one) originates from a defunct, rehabilitated mining site, increasing the chances of AMD and runoff contamination. Historically, relatively large mining activities took place upstream of the catchment, posing possible impacts on the three seeps that feed the main channel wetland (Figure 2.4). Presently these mines are largely rehabilitated, although there are some reports of concern about the groundwater and mine decant of the sites (Golder Associates, 2014). The wetland is also threatened by an active mining quarry located towards the west of the wetland’s immediate catchment.

Witbank wetland WET-EcoServices assessment

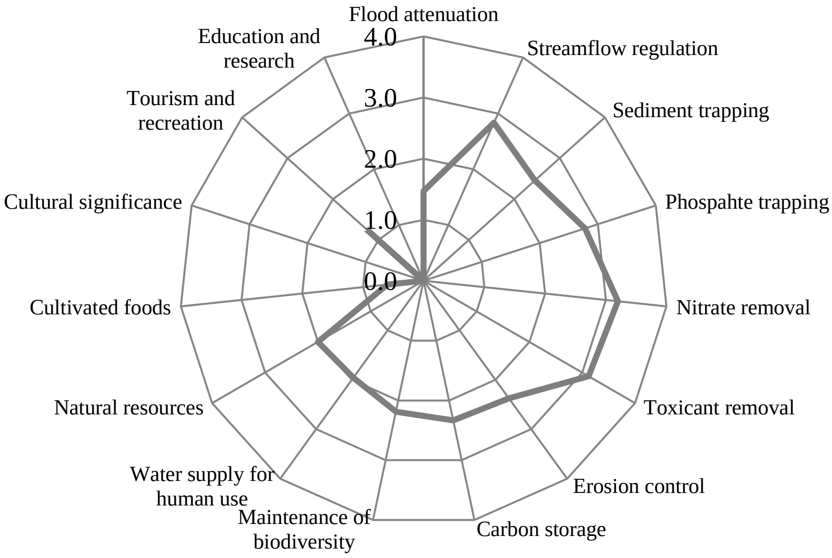

Figure 2.16: WET-EcoServices (Kotze et al., 2009) scores for the Witbank wetland HGM unit one (channelled valley-bottom wetland).

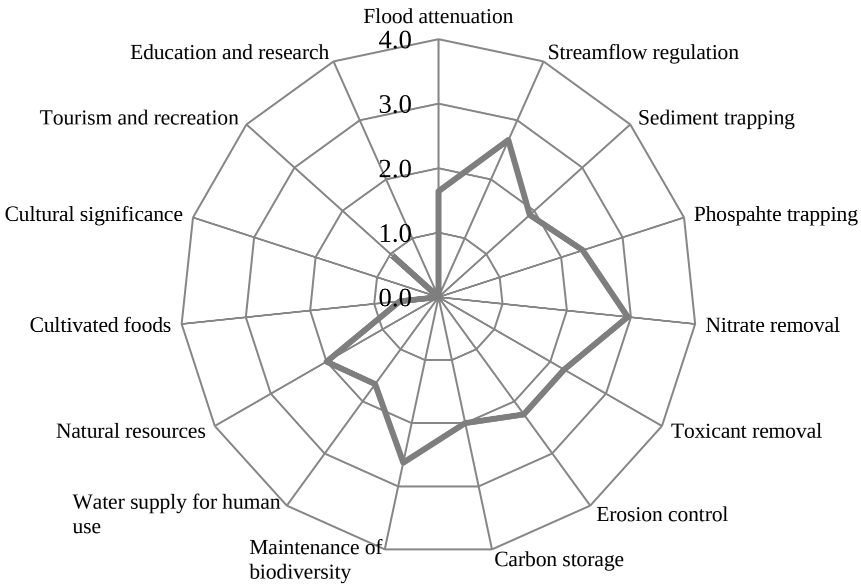

Figure 2.17: WET-EcoServices (Kotze et al., 2009) scores for the Witbank wetland HGM unit two (hillslope seep linked to a stream channel) and three (depression).

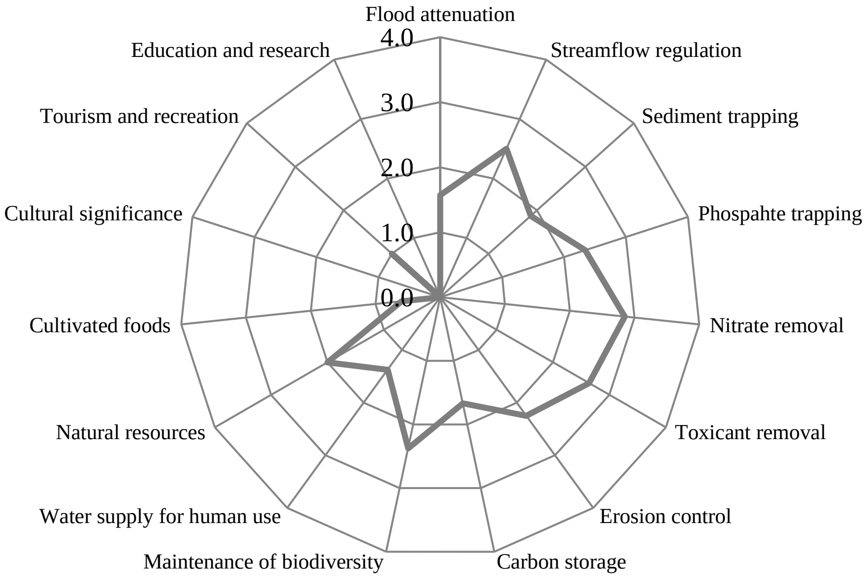

Figure 2.18: WET-EcoServices (Kotze et al., 2009) scores for the Witbank wetland HGM unit four (hillslope seep linked to a stream channel).

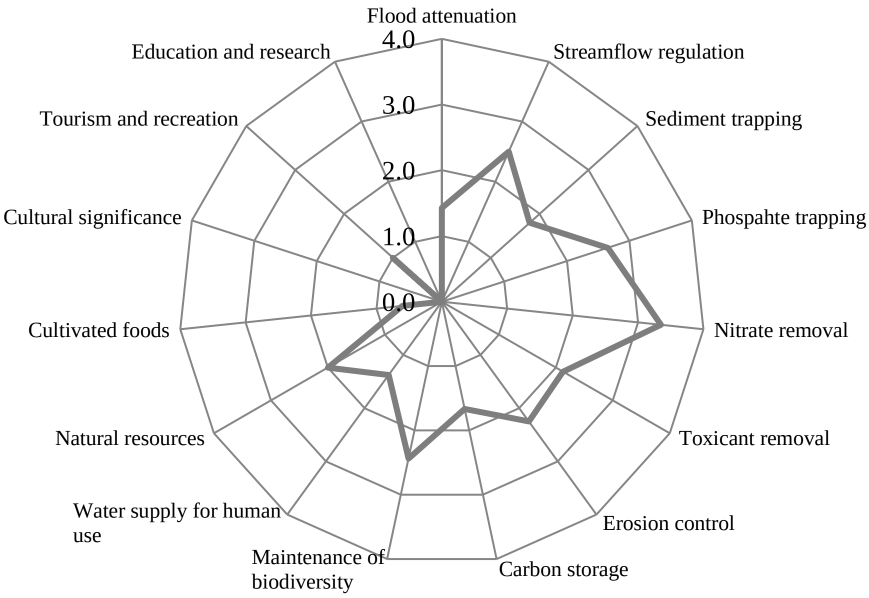

Figure 2.19: WET-EcoServices (Kotze et al., 2009) scores for the Witbank wetland HGM unit five (hillslope seep linked to a stream channel).

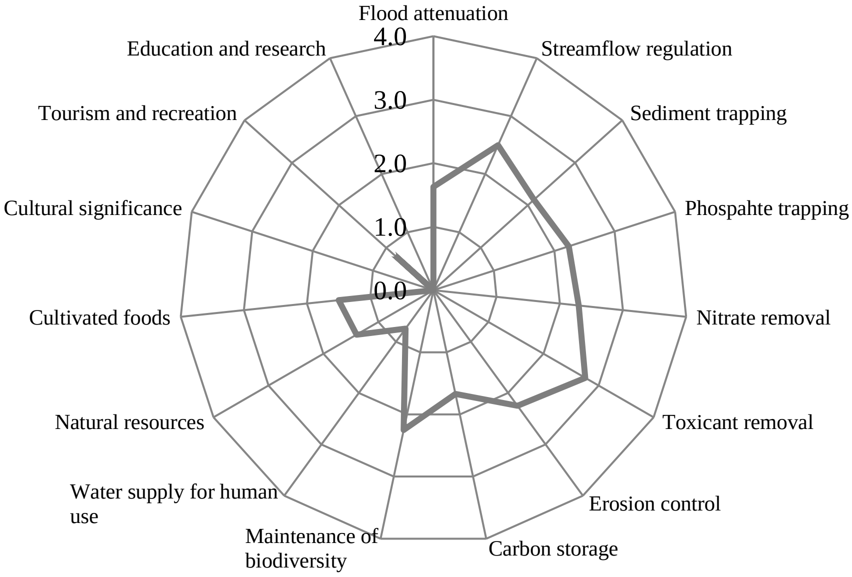

Figure 2.20: WET-EcoServices (Kotze et al., 2009) scores for the Witbank wetland HGM unit six (channelled valley-bottom wetland).

Witbank wetland WET-Health assessment

Table 2.8: Present State category for each module and the overall Present State of Witbank wetland

Wetland	**Present State category**			
	Hydrology	**Geomorphology**	**Vegetation**	**Overall Present State**
Witbank (5a)	B	A	A	A

The wetland's overall integrity/health impact score was categorised by specialists as an "A", indicating a near-to-natural wetland (Table 2.8). The water sample results indicated a relatively good quality of water (Table 2.12). However, the water was sampled in the middle of the main channel and did not include the possible peripheral impact of the current mining activity north of the catchment. The wetland is not identified as a NFEPA wetland, so there are few implications for conservation. The land is used for sheep grazing. Boreholes that supply water to surrounding households are located in the catchment. Again, a present state of "A" is uncertain in combination with indications of water quality impact. These results motivated the quantitative investigation.

Droogvalei wetland

The Droogvalei wetland (788.67 ha) consists of a relatively large, main channelled valley-bottom (wetland HGM unit one) and seep connected to the north-east (wetland HGM unit two) (Figure 2.4 and Figure 2.5). The wetland's immediate catchment is the Droogvaleispruit sub-catchment (one of three) of the X11B catchment feeding directly into the Boesmanspruit dam, supplying 3.9% of the inflow (McCarthy and Humphries, 2013). The catchment is situated adjacent to the Silobela Township and a portion of the wetland catchment is part of a 2010 land claim. East of the catchment (upstream), a large portion consists of agricultural crop farming, plantations and cattle grazing. The township and the confluence with the Boesmanspruit dam are at the north-western part of the wetland. The middle of the catchment (north to south) is largely agricultural commonage, with some runoff-influence from the Droogvalei siding in the north. In this dry-season investigation, the wetland had saturated soil with only scattered pools of water in the channel and flooding evident at impeding features.

Conversation revealed that the land claim community use the wetland only for grazing purposes. Community members were not aware of the importance and possible benefits of wetland services that could be sustained if the wetland was protected. The apparent lack of value attached to the wetland was evident from talking with a community representative who is a commonage-user. He shared a farm plot with seven households. In the commonage, members grow small patches of subsistence crops and get their water from a borehole pump. They have cattle that graze in and around the Droogvalei wetland HGM unit one which is close to their homes. This grazing is the only way the wetland was used. It may be that that, with the movement of people in context of white farms and land claims, cultural knowledge of wetlands and their use has been lost.

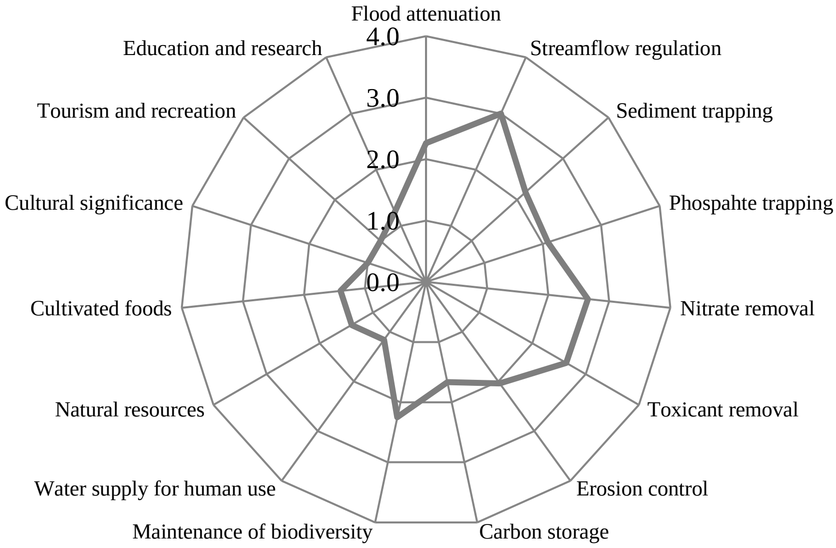

Figure 2.21: WET-EcoServices (Kotze et al., 2009) scores for the Droogvalei wetland HGM unit one (channelled valley-bottom wetland).

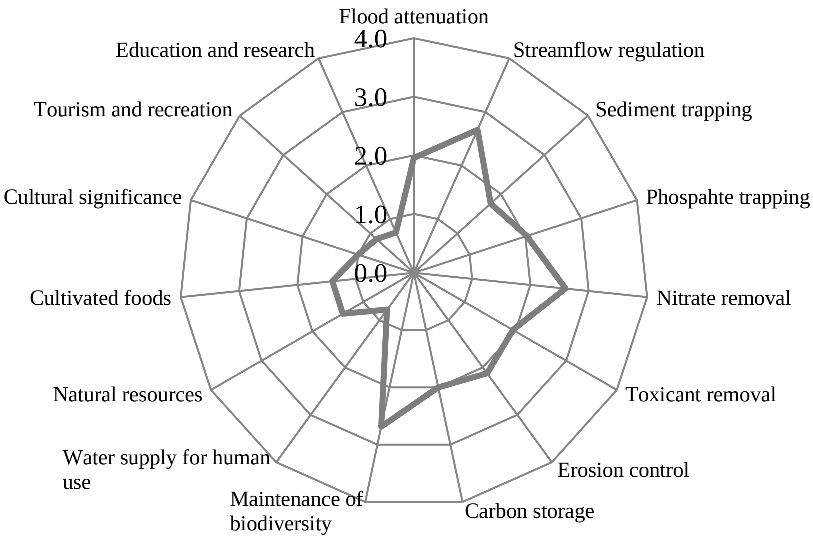

Figure 2.22: WET-EcoServices (Kotze et al., 2009) scores for the Droogvalei wetland HGM unit two (hillslope seep linked to a stream channel).

The wetland ecosystem was assessed as providing a range of services, however none were particularly high (Figure 2.21 and Figure 2.22) possibly indicating that with heavy use in the immediate wetland catchment there is a lower provision of services.

Droogvalei wetland WET-Health assessment

Table 2.9: Present State category for each module and the overall Present State of Droogvalei wetland				
Wetland	**Present State category and likely**			
Droogvalei (6a)	**Hydrology**	**Geomorphology**	**Vegetation**	**Overall Present State**
	C	B	A	B

The health of the wetland had an overall score of "B" (Table 2.9) indicating that the wetland is largely natural with few modifications. Water samples given in Table 2.12 indicated relatively good water quality from the middle of the wetland, where the surrounding activity is largely cattle grazing. The implications of these results are a cause of concern not only for the people who depend on the wetland, but also for NFEPAs located within this wetland area (SANBI, 2013). It is increasingly evident that the WET-Health assessment is not sensitive to water quality.

Table 2.10: Summary of winter contextual analysis results (see text for full description)

Wetlands and HGM units	HGM type	Area (ha)	Overall health score	Activity impact score			NFEPA	Ecosystem service scores														
				Mining	Grazing	Crop		Flood attenuation	Streamflow regulation	Sediment trapping	Phosphate trapping	Nitrate removal	Toxicant removal	Erosion control	Carbon storage	Biodiversity maintenance	Water supply	Provision of natural resources	Provision of cultivated foods	Cultural significance	Tourism and recreation	Education and research
Jaglust	CVB	47	A	1	2	1	Yes	1.5	3.0	1.8	2.3	3.2	2.6	2.6	2.7	3.1	2.0	1.2	0.6	0.0	1.3	1.3
Witbank unit one	CVB	94.68	A	2	2	1	No	1.5	2.8	2.5	2.8	3.2	3.1	2.4	2.3	2.2	2.0	2.0	0.6	0.0	1.3	0.0
Unit 2 and unit 3	HS and D	14		0	2	1		1.6	2.7	1.9	2.4	3.0	2.2	2.3	2.0	2.6	1.8	2.0	0.6	0.0	1.0	0.0
Unit 4	HS	4.84		1	1	1		1.6	2.5	1.9	2.3	2.9	2.6	2.3	1.7	2.4	1.6	2.0	0.6	0.0	1.0	0.0
Unit 5	HS	2.76		1	1	1		1.4	2.5	1.8	2.7	3.4	2.1	2.3	1.7	2.4	1.6	1.4	0.6	0.0	1.0	0.0
Unit 6	HS	11.4		0	3	1		1.6	2.5	2.1	2.2	2.3	2.8	2.3	2.0	2.3	0.8	1.4	1.5	0.0	0.9	0.0
Roodepoort Unit 1	CVB	21.54	A	3	0	0	No	1.6	2.8	2.2	1.9	2.6	2.2	1.7	1.3	2.5	1.8	1.2	0.8	0.0	1.0	1.0
Unit 2	HS	1.6		0	2	0		1.4	2.8	1.3	2.2	3.2	2.0	2.1	2.0	2.6	1.6	1.2	0.6	0.0	1.0	1.0
Unit 3	HS	4.71		0	2	0		1.4	2.8	1.3	2.2	3.2	2.0	2.1	2.0	2.6	1.6	1.2	0.6	0.0	1.0	1.0
decant unit 1	CVB	153	A	3	1	1	Yes	1.2	3.0	1.6	2.3	3.0	2.9	1.8	2.7	2.4	2.4	2.0	0.8	0.0	1.3	1.8
Unit 2	HS	35		0	1	1		1.6	2.3	1.1	1.9	2.3	2.0	1.8	1.7	3.0	1.9	2.0	0.8	0.0	1.3	1.8
Unit 3	HS	30		0	1	1		1.5	2.3	1.8	2.2	2.3	2.2	2.0	2.0	3.1	1.9	2.0	0.8	0.0	1.0	1.8
Droogvellei unit 1	CVB	788.67	B	1	3	2	Yes	2.3	3.0	2.2	2.1	2.7	2.6	2.0	1.7	2.3	1.2	1.4	1.4	1.0	1.0	1.3
Unit 2	HS	19.15		0	2	2		2.0	2.7	1.8	2.0	2.6	2.0	2.1	2.0	2.7	0.8	1.4	1.4	1.0	0.9	0.8
Boesmanspruit unit 1	CVB	206.84	C	3	1	1	Yes	2.2	3.2	2.3	2.3	2.6	3.0	3.0	2.7	2.4	2.2	1.2	1.4	0.0	1.3	2.3
Unit 2	HS	38.92		0	1	2		1.9	3.2	1.6	2.4	3.1	2.5	2.8	2.7	2.7	2.2	1.2	1.4	0.0	1.4	2.0

High

Moderately high

Intermediate

Moderately low

Low

Table 2.10 summary of winter contextual analysis results displaying wetlands and their hydrogeomorphic (HGM) units, HGM types including: channelled valley-bottom (CVB), hillslope seep (HS), and depression (D) wetlands and their associated areas (ha). The overall WET-Health scores, appointed activity impact scores, and each HGM unit's ecosystem service scores: high service (orange), moderately high service (green), intermediate service (blue), moderately low service (pink), low service (purple)) are also summarised.

2.4.2 Aquatic biota

Table 2.11: 2015 South African Scoring System 5 (SASS5) and Average Score Per Taxon (ASPT) scores for each sampled wetland of the X11B catchment

Site	Roodepoort (1a)	Boesmanspruit (2a)	Jagtlust (3a)	Witbank historical decant (4a)	Witbank (5a)	Droogvalei (6a)
SASS	91	18	41	26	50	45
No. Taxa	14	5	9	6	10	11
ASPT	6.5	3.6	4.56	4.33	5	4.09

2.4.3 Water physico-chemistry

Table 2.12: 2015 Physico-chemical water quality results at each wetland

Site	**pH**	**DO (%)**	**EC (µS/cm)**	**TDS (ppm)**	**Temp (°C)**
Roodepoort (1a)	6.53	58.4	58.2	42.6	18.2
Boesmanspruit (2a)	3.86	57.4	423	212	16.2
Jagtlust (3a)	4.55	77.9	25	12	21.7
Witbank historical decant (4a)	3.24	41.8	457.8	289	16.1
Witbank (5a)	5.01	59.4	186	93	15.7
Droogvalei (6a)	5.15	43.3	251	124	15

Table 2.11 and Table 2.12 provide summaries of all the wetlands and their biotic and chemical characteristics.

2.5 Discussion

2.5.1 Ecosystem services

The objective was to provide a contextual analysis of the case-study landscape, focusing on wetland health and ecosystem services and the perceived value of the wetlands to users. The ecosystem services and benefits that inferred a "moderately high" to a "high" degree of likelihood (i.e., extent scores of 2.1–4 of each ecosystem service radar diagram) included: streamflow regulation (naturally sustaining streamflow during periods of low flow), phosphate assimilation (removal of phosphates carried by runoff), nitrate assimilation (removal of nitrates carried by runoff), toxicant assimilation (removal of toxicants, e.g., metals, biocides), and biodiversity maintenance (by the provision of habitat and maintenance of natural process) (Table 2.10).

However, the indication of services provided is influenced by the opportunities presented to the wetlands by anthropogenic activities within the immediate catchment of each wetland. For example, the Witbank wetland reflected a higher degree of probable toxicant removal in wetland units most closely associated with mining use. The inferred presence of toxicants that could be mobilised by AMD increased the likelihood and opportunity of their adsorption in wetland soils and vegetation. Therefore, wetlands can still have a high potential to reduce toxicants, but if they are not exposed to any toxicant sources they will not display high degrees of likelihood in terms of providing the service/benefit. Similarly, the high likelihood of nitrate removal was prevalent in all wetlands, as the catchment is strongly associated with agricultural activities.

Specialists judged services mainly related to enhanced water quality, a typical wetland service. Miners and farmers alike acknowledge that wetlands improve the water quality within the X11B catchment and many studies have concluded that wetlands (constructed and natural) result in lower concentrations of agricultural nutrients and AMD-based heavy metals and ions downstream of an impacted wetland catchment (Sheoran and Sheoran, 2006; Kovacic et al., 2006). Water quality enhancement and water provision were particularly important in the X11B catchment. All the wetlands investigated in this study feed into the main water supply dam that serves grazing livestock on farms through which the water flows. The water quality service is likely in all the wetlands of this study and therefore motivates for a deeper investigation of nutrients, ions and heavy metals (Chapter 3).

Streamflow regulation was interpreted as a “likely” service in all the chosen wetlands. This service is inferred from the high connectivity of the catchment’s hydrological processes, particularly the connections with sub-surface discharge by the many naturally occurring springs in the area. This hydrological network was recognised as highly valuable and important by all of the farm owners in the study, and could be as valuable in sustaining the hydrological integrity and presence of the wetlands in the catchment (Kotze et al., 2009). The defining feature of any wetland is its hydrology which it strongly influences the flora and fauna inhabiting the ecosystem, thus contributing towards its functioning (Kotze et al., 2009).

Some services were assessed as moderately low and low depending on numerous factors. Flood attenuation services were assessed as low, possibly because the areas of the wetlands of this study were relatively small in comparison to the areas of the wetlands’ immediate catchments, and the wetlands fall within a ‘moderately low’ category of sinuosity. It must be noted that for this study size was not taken into account for this study, as wetlands had similar scale dimensions. The low level of provision of natural resources, cultivated food, services of cultural significance, tourism and recreation, and education and research could be because five of the six wetlands are located on privately owned agricultural farms, where accessibility is limited. Landowners on those wetlands benefit primarily from water provision, as opposed to wetland use for fibre, food, fuel or for any culturally significant uses.

The methods used to assess the ecosystem services offered by the selected wetlands have been used for catchment planning and management, and for “determining the relative [conservation] importance of individual wetlands in the catchment” (Kotze et al., 2009). In the case of this study, the wetlands all reflected relatively similar ecosystem service provision, probably because of their structural and scale similarities. Provisioning and cultural services were assessed as low for all the wetlands, whereas regulating and supporting services were more common. This has important implications for the conservation of wetlands, as supporting and regulating services underpin the supply of provisioning and cultural services, thereby influencing both direct and indirect effects on human well-being (Raudsepp-Hearne et al., 2010). One such supporting service is the maintenance of biodiversity.

The maintenance of biodiversity provision was not considered a service as such, but research carried out by de Groot (2011) suggests that habitat provision can justifiably fall under supporting services because, while the role of biodiversity in ecosystem provision is not well understood, studies on the general understanding of ecosystem services suggest that all service provision depends on biodiversity (de Groot, 2011). The diversity of biota and their natural behaviour and interactions within the system maintain the complexity and processes that create a healthy ecosystem (de Groot, 2010). All the wetlands were assessed as providing high levels of biodiversity. Thus, supporting and regulating services were judged as particularly important, supporting observations by de Groot (2011) and Pollard et al. (2013). Further, four out of the six wetlands provide habitats for threatened crane species, and have been identified as NFEPAs.

The outcomes of applying the WET-EcoServices methodology indicate the wetlands are inherently highly important to the X11B catchment users and downstream users, and that the natural environment should be managed and protected.

2.5.2 Wetland health

The aim of this contextual study was to establish a social-ecological overview of six wetlands and the people that use them, on which to base a more detailed assessment of particular ecosystem characteristics (water chemistry and macroinvertebrate community structure (Chapter 3)). The site-specific user-influence factors which guided the selection of the wetlands were: mining, dryland agriculture (crops and grazing), and community activities (Table 2.2). These land-use activities are associated with built infrastructure (e.g., tarred, gravel, and dirt roads, with and without culverts, and open-cast mine sites) which could intercept wetlands. Such activities compact the ground and impede wetlands, impacting on hydrology and geomorphology (Macfarlane et al., 2009). In the winter season, the extensive dryland crop and livestock agricultural lands also showed the consequences of runoff, erosion and sedimentation during later rainfall events and flood peaks because of the bare soil, trampling, and overgrazing in and around the wetlands. However, despite these land-uses, the overall health of four out of the six wetlands was classified as close to natural and largely unmodified (Table 2.10)

These results were influenced by the overall relatively low extent and intensity scores of given activities occurring in the wetlands and their immediate catchments (Macfarlane et al., 2009).

Impacts of land-use did have more adverse effects on the Boesmanspruit and Droogvalei wetlands. The two wetlands are both exposed to greater human influence than the other wetlands. The Boesmanspruit wetland scored an overall Present State score of “C”, a category which describes a moderate impact score, implying a moderate loss of habitat and a change in ecosystem processes; however, the natural habitat remains still intact. This seems fitting as, historically and presently, the Boesmanspruit wetland has been and is impacted by agriculture and coal mining. Historic mining had left an impact on the wetland, as abandoned coal piles and mining-associated built infrastructure remain in the wetland. Present anthropogenic activity has impacted hydrology and geomorphology to the point where historic and present impacts of both modules (hydrology and geomorphology) of assessment indicate moderate impacts on the wetland.

The Droogvalei wetland was the largest wetland assessed. The size and locality were the main factors of its Present State category scoring a “B” (largely natural with few modifications). The modifications are caused by built infrastructure intercepting the wetland, including small gravel roads, a railway, and a tarred main road with culverts redirecting and channelling flow. The road structures impede the flow and have impacts associated with drying, channelling, erosion and sedimentation. Hence impact on the wetland’s hydrology was a significant negative weighting factor for the wetland’s poorer health outcome.

2.5.3 Stakeholder perspectives

In line with the transdisciplinary approach to research concerning the maintenance and management of C-SESs, engagement with stakeholders was encouraged in the WET-EcoServices and WET-Health methodology (Kotze et al., 2009; Macfarlane et al., 2009). Landowners and local residents shared their local and historical knowledge and personal values based on each of the wetlands. This provided vital information on the land-use context of the catchment and gave a clear and interpreted picture of importance, dependency and appreciation of the wetland services and the hydrological connectivity they provide.

2.5.4 Water quality and aquatic biota

Simple water quality samples collected in the dry, low-flow season indicated the presence of chemical properties typical of AMD. With little rain and flooding, wetlands sampled were more likely to evidence contamination, as little dilution and flushing had occurred. All sites except Roodepoort (the only wetland with no evident potential mining influence) were acidic, with pH values less than 6 (Table 2.12) (Sheoran and Sheoran, 2006). Although soils in the X11B catchment are naturally acidic and may have influenced the pH at some sites (Golder

Associates, 2014), mining is a pervasive influence in the catchment. In addition, the Boesmanspruit, Droogvalei, and the Witbank historical decant wetlands had high salinities, reaching values beyond the recommended Resource Quality Objectives of the catchment (Department of Water and Sanitation, 2016). Low pH and high salinity are both characteristics of AMD (Garcia-Criado et al., 1999; Sheoran and Sheoran, 2006).

The SASS5 scores at all wetland sites were lower in SASS, Taxa and Average Score Per Taxon (ASPT) values than in the Roodepoort reference site. There were also fewer sensitive species at mining-influenced sites, with seven sensitive species found at the reference site and a maximum of two sensitive taxa found at potential mining-influenced sites (Appendix F), suggesting that pollutant concentration at low-water flow in the mining, urban and agricultural landscape impacts wetland health. The Boesmanspruit had no sensitive species present, had the lowest ASPT score, was a wetland with one of the lowest pH values (3.86), and one of the highest EC (423) and TDS (212) readings. The Witbank historical decant site had the lowest pH (3.24) and the highest EC (457.8) and TDS (289) readings, typical of AMD (Sheoran and Sheoran, 2006; Hogsden and Harding, 2011). The higher ASPT score of the historical decant site indicated that the presence of a Naucoridae was the reason for the subsequent increase in the score (Ojija and Laier, 2016). Naucoridae, a relatively sensitive family, can be pollution-tolerant; therefore, the finding corresponds with the typical effects of AMD on the presence of macroinvertebrates. Other sites were less impacted by AMD and had higher ASPT scores than the Boesmanspruit and Witbank historical decant wetlands. The water quality at the Droogvalei wetland could most likely be explained by its proximity to the urban land-use and the urban runoff experienced at the site (Haidary et al., 2013).

Investigations in 2012 and 2013 which focused on the Boesmanspruit wetland recorded EC values of 137 μS/cm and 114 μS/cm, and pH values of 7.5 and 6.4 for the immediate two years prior to the AMD event (Tate and Husted, 2015). This study, conducted in 2015, recorded EC levels almost four times higher and pH values considerably lower in the low water flow period (Table 2.12). SASS5 score comparisons also indicated that fewer taxa were present, and within two years, the site's ASPT score increased only slightly (Table 2.11). Comparisons with Tate and Husted's (2015) study indicate declining water quality conditions. The decline in values could possibly indicate that: 1) pollution of the wetlands is still occurring, despite interventions (established Acid Mine Drainage task team, revived the Upper Komati Catchment Forum, and a number of monitoring points established within the catchment) that took place after the 2012 incident; 2) the Boesmanspruit dam may be vulnerable to another AMD incident in the case of future high flow events, and 3) wetland biodiversity, and therefore wetlands (and other associated ecosystems), may be threatened by contaminants (Tate and Husted, 2015).

The catchment's land-uses, more specifically coal mining, were identified as the key actors in wetland contamination (Tempelhof et al., 2012; McCarthy and Humphries, 2013). Therefore, it would be in the best interests of catchment management initiatives to implement impact mitigation policies that address the complexity of different users and their impacts, especially cumulative impacts (Hemond and Benoit, 1988). Catchment managers should gain and communicate a better understanding of the value of the wetlands in terms of which natural benefits they provide, and to whom (Macfarlane et al., 2009). Assessing the health of wetlands would give an idea of what is worth conserving, thus providing a more informed basis for conservation planning and implementation.

2.5.5 Critique of methodology

Wetlands are complex systems with many interlinked functions and components, and most assessment tools tend to simplify this complexity (Cools et al., 2013). This point has been acknowledged by the WET-EcoServices and WET-Health developers (Kotze et al., 2009; Macfarlane et al., 2009). For this study in particular, the tools fail to expose mining impacts sufficiently (Day and Malan, 2010). The open-pit coal mining activity that takes place in the X11B catchment removes natural vegetation, compacts earth, promotes built infrastructure and intercepts groundwater flows (Golder Associates, 2014). WET-Health assesses the runoff/increased flood peaks, consequential erosion and gully formation, presence of alien vegetation that is associated with the removal of natural vegetation, impeding features, and soil compaction. However, the implication of intercepted groundwater flow, hydro-connectivity, and associated coal mining implications are not included in assessments. Defunct open-pit mines are rehabilitated by filling the mine void and re-establishing the natural vegetation of the site, as was the case with this study. Most wetlands in this study had rehabilitated mine sites within their immediate catchments, superficially resembling healthy catchment landscapes. The water quality results of the wetlands suggest that the impact of mining-related pollution was still present, especially after mine closure (van der Waals, 2016b).

WET-EcoServices and WET-Health represented broad-scale, high biophysical assessments, for which reason it is suggested that, in the case of wetlands associated with coal mining, a quantitative water quality module, using relatively simple probes, as in this study, should be used for wetland health assessment, in order to indicate unobservable impacts in the landscape. It is also suggested that a more detailed vegetation module should be included, using quantitative vegetation assessments (for example, Lopez and Fennessy (2002) and Miller and Wardrop (2006)). The vegetation diversity was noticeably lower at mining impacted sites than at non-impacted sites, possibly making vegetation diversity an observable indication of water quality implications associated with mining, as suggested by Lopez and Fennessy (2002).

2.6 Conclusion

The concept of valuing ecosystem services is a key approach to facilitating negotiated outcomes of competition between water abstraction and waste disposal by users on one hand, and the needs of the environment on the other (de Groot, 2011, Pollard et al., 2013). This contextual analysis provides clear information about the extensive value of the catchment's wetlands. The Mpumalanga Biodiversity Conservation Plan Handbook (Ferrar and Lötter, 2007) states that, in this heavily used mining and agricultural catchment, the wetlands of the Mpumalanga Komati region are probably the last sustained biodiversity centres of the region. This, together with many of the catchment's wetlands categorised as NFEPAs (Nel et al., 2011) containing red data fauna and flora, indicates how valuable this area is in conserving biodiversity.

By using wetland integrity as a proxy for ecological function, (Pollard et al., 2013) the results showed that the wetlands were intact and relatively healthy. The results also showed non-linear patterns of relationship between different levels and kinds of use, and wetland health and the ecosystem services deemed to be offered. However, impact scores of wetlands with the poorest health were not at the "F" level, described as "Modifications have reached a critical level and ecosystem processes have been modified completely with an almost complete loss of natural habitat and biota". From the results of this study, it can be concluded that wetlands studied have been resilient in the face of the severe 2012 impact, and are in relatively good condition, and consequently providing clear ecosystem services.

It is important to note that the poor water quality results and impacted macroinvertebrate structure highlight the concerns of an inadequate warning of impaired ecological integrity using only WET-Health and WET-EcoServices qualitative assessment. This is particularly true for water quality. Research indicates that poor water quality ultimately impacts the biodiversity of ecosystems, thus impacting the entire ecological system and its feedbacks that maintain functionality (Allan, 2004; Dudgeon et al., 2006; Verhoeven et al., 2006). It became evident in this study that a deeper investigation of land-use impacts – specifically mining – was necessary. The simple level of water quality in the contextual study indicates that the X11B's human and ecological Reserve for water quality may be compromised if another flooding event were to occur after pollutants had accumulated over a period of time. More detailed water quality and related macroinvertebrate community structure are the focus of a deeper investigation to provide evidence of coal mining impacts and threats, presented in Chapter 3.

Chapter 3: Wetland ecological infrastructure and ecosystem services

The contextual observations and local knowledge-based findings of the selected wetlands previously described in Chapter 2 indicated that the wetlands were in relatively good condition and were supplying important regulatory services. There were some observations that indicated the impacts of coal mining on the ecological status of the wetlands, for which reason, water quality and biomonitoring techniques were used to further investigate the wetland conditions.

3.1 Wetlands and water chemistry

The water chemistry of wetlands varies in different types of hydrogeomorphic (HGM) settings, and according to the surrounding land-uses (Ollis et al., 2013). Typical water chemical variables that are measured include nutrients, trace elements, heavy metals, ions, and system variables (Dallas and Day, 2004). Runoff from catchments and mining decant have a complex effect on the combination of dissolved contents entering a wetland and reacting with vegetation, microbes and sediments (Pinetown et al., 2007). Different land-uses can have distinct chemistry associations, making it possible to identify sources of pollution. For example, coal mining is strongly associated with heavy metals and ions from the elements found in coal (organic: C, H; inorganic: Al, Fe, Ca, Na, Mg, K, S; trace elements: Be, Cd, Co, Pb, Cr, Hg, Mn, Ni, Sb, As, Se) and the chemical reactions occurring during the oxidation of pyrite (see Chapter 1) found in both functioning and defunct mines (Harding, 2005; Pond et al., 2008). Coal mining decant is also high in sulphates (SO_4^{2-}) and generally has a low pH. On the other hand, intensive dryland agriculture is generally associated with higher levels of nitrates (NO_3^{1-}), nitrites (NO_2^{2-}), and phosphates (PO_4^{3-}) from the application of fertilisers, pesticides and animal manure (Bizzi et al., 2013). Wetland ecosystems are naturally adapted to the fluctuating amounts of chemical variables available in the water column, playing a vital role in the adsorption of ions on sediment particles, trapping of sediment, and therefore filtration of inflowing water for downstream users. Healthy wetlands can adapt to natural disturbances and remain healthy, but sustained human impacts often cause stressors that exceed the resilience of the wetland (Gunderson, 2000).

With regard to coal mining, pressures causing a breach in wetland adsorption capacity can have detrimental impacts on the biodiversity and functioning of the system. Biodiversity is threatened when coal mining-associated acidity leaches metals from sediments and soluble toxicants become biologically available and abundant in the water solution (Sheoran and

Sheoran, 2006). Trace elements are found naturally in the tissue of living organisms and are necessary nutrients for growth and metabolism (Na, K, Mg, Fe, Co, Zn, Mo), but concentrations of any variable that are above organism tolerance limits result in toxic effects (Harding, 2005) and the integration and bioaccumulation of contaminants in food webs pose a threat to different organisms of different trophic levels both within the wetland (Hogsden and Harding, 2011) and in areas that provide similar niches where organisms inhabit or to which they migrate (Harding, 2005).

The Department of Water and Sanitation (DWS) acknowledged water quality threats to complex social-ecological systems (C-SESs) and created a series of water quality guidelines (Department of Water Affairs and Forestry, 1996) which are a set of criteria designed to protect freshwater ecosystems in South Africa, and which are used for water quality management by the Department of Water and Sanitation (DWAF, 1996). Resource protection is related to resource classification and toxicity-derived Resource Quality Objectives (Department of Water and Sanitation, 2016). The guidelines provide a source of primary reference information and support in water management decisions with regard to organic and inorganic toxic and non-toxic constituents and system variables. Constituents represented in the guideline are based on international guidelines and specialist recommendations. Testing the range of harmful variables and their impacts has yet to be completed, but the variables represented in this guide are a range considered most harmful to freshwater ecosystems (DWAF, 1996). The guideline used for protecting aquatic ecosystems employs Target Water Quality Ranges (TWQR) as management objectives that assume life-long exposure to toxicants and these TWQR are within measures assure there are no adverse effects on aquatic ecosystem health. These ranges are used by the DWS as water Resource Quality Objectives for toxic component limits within the Komati River System (DWAF, 1996; DWS, 2016). The aquatic ecosystem concentrations or levels of toxic constituents and system variables that are expected to have measurable effects are categorised. Measurable effects on up to (chronic) or above (acute) 5% of species in aquatic communities are defined by either Chronic Effect Values (CEV) or Acute Effect Values (AEV) (DWAF, 1996). AEVs are measures of constituents that, if they occur for a short time or more, can quickly cause the death and disappearance of species. Aquatic environments experiencing AEVs usually require urgent management attention. CEVs, on the other hand, are measures of constituents that, if they occur for a longer period of time, can lead to the eventual death and disappearance of species (DWAF, 1996).

3.2 Wetlands and biomonitoring

As outlined in Chapters 1 and 2, diversity plays an important role in the function of wetland systems and in maintaining their resilience, thus making the biology of wetlands appropriate drivers of wetland condition (Dugan, 1990; Hansson et al., 2005). Many wetland plants and animals are local endemics, as well as being valued globally by communities that depend on their presence and provisions, thus encouraging further initiatives to protect them (Dugan, 1990). Biomonitoring is therefore used as a measure of ecological health, with the most useful aquatic biota being aquatic invertebrates (Palmer et al., 2004).

Important advantages of using invertebrates as indicators of the health of a water body are a diverse range of tolerances, short life cycles, rapid response times and that they are mainly sedentary (Palmer et al., 2004). More specifically, macroinvertebrates are visible with the naked eye, easy to identify, and have season-based life cycles (Dickens and Graham, 2002). These attributes allow for relatively easy sample collection which may be used in conjunction with chemical and physical analysis to give a clearer indication of causal links of pollution sources (Palmer et al., 2004). The South African Scoring System (SASS), which is one of the most successful and widely adopted riverine biomonitoring methods in South Africa (Dickens and Graham, 2005), assesses the ecological status of aquatic ecosystems based on the absence, presence and abundance of macroinvertebrate taxa that range in sensitivity and tolerance to changes in water quality.

Macroinvertebrates play an important role in nutrient cycles, food webs, assimilation of metals associated with inorganic and organic matter, material decomposition, and primary productivity (Wallace and Webster, 1996; Harding, 2005). The general consensus is that macroinvertebrates respond with different degrees of sensitivity to unnatural and natural wetland conditions and water quality parameters (Bizzi et al., 2013). Tolerances vary between taxa, and the invertebrates genetically adapt to, die, or leave the contaminated sites in response to stress (Harding, 2005; Petrin et al., 2007). However, using index-based biomonitoring protocols like SASS5, developed for aquatic systems with higher flow, may not be appropriate because the index scores were not developed for wetland taxa (Dickens and Graham, 2005). Different wetland types contain different macroinvertebrate communities, making it difficult to set response standards related to impacts or natural settings (Monitoring your Wetland, 2011). Bird and Day (2010) questioned the applicability of SASS for wetland bioassessments according to wetland types by testing the "correspondence between observed impairment and SASS scoring" and concluded that the

SASS scoring system would be unreliable in inferring wetland conditions. For this reason, macroinvertebrate samples collected using the SASS5 method, were only used as a list of aquatic taxa. Macroinvertebrates were collected, counted and identified to family level, and subjected to multivariate analysis, that related biotic presence and abundance to the measured concentrations of water quality concentrations.

The wet-season sampling in this study enabled a more nuanced perspective of wetland ecosystem health and ecosystem services in relation to the primary land-uses, mining and agriculture. This wet-season assessment represented an opportunity to reassess and compare the wetlands' ecosystem health and services for the two seasons, building the argument whether ecological infrastructure can be experienced deeper in summer. It also provides a more detailed picture of the biota in the wetlands and their surrounding catchments for comparing the effects of different land-uses. The assessment aims to understand conditions of the selected wetlands within the X11B catchment better, addressing perceived gaps in the WET-EcoServices and WET-Health methodology reported on in the contextual analysis that formed part of Chapter 2. In order to do this, the 1) water chemistry of the wetlands, and 2) the macroinvertebrate community structures were investigated further.

3.3 Study area

The study area is reported on in Chapter 2 and illustrated in Figure 2.2. In the summer, wet-season assessment, an additional wetland was investigated (the "Nooitgedacht" wetland). The wetland is located within the Nooitgedacht Dam Nature Reserve (NDNR) and is characteristic of the wetland and grassland region of Mpumalanga. The NDNR's protection of wetland and grassland ecosystems, and associated fauna and flora is a priority of the Mpumalanga Tourism and Parks Agency Board for conservation and tourism purposes. The Nooitgedacht dam was originally built for, and continues to provide water to, the Komati River Government scheme, supplying water to the power stations located in the upper Olifants River Catchment (AfriDev, 2006). The dam is hydrologically connected to the Komati River, supplying the Komati with the bulk tributary drainage from the X11B catchment (on which this investigation is based). Mining and agricultural practices upstream of the dam in the X11B catchment reportedly impact and contaminate the dam's water quality, thus contributing to further contamination of the Komati river system. This highlights the importance of conserving the NDNR's wetlands as ecological infrastructure in order to limit pollution to the Komati River. The Nooitgedacht wetland is a rehabilitated wetland that is monitored and maintained by Working for Wetlands (Working for Wetlands, 2008). The site was originally impacted by erosion, caused primarily by the site's steep gradient. After rehabilitation, the site has recovered, showing dense vegetation and a constant, healthy flow and water saturation. Un-impacted wetland reference sites are rare, especially in heavily used landscapes (Brown et al.,1997; Macfarlane et al., 2009), but the healthy conditions of the Nooitgedacht wetland and the lack of agriculture or mining impacts make the wetland a reasonable reference site for this study.

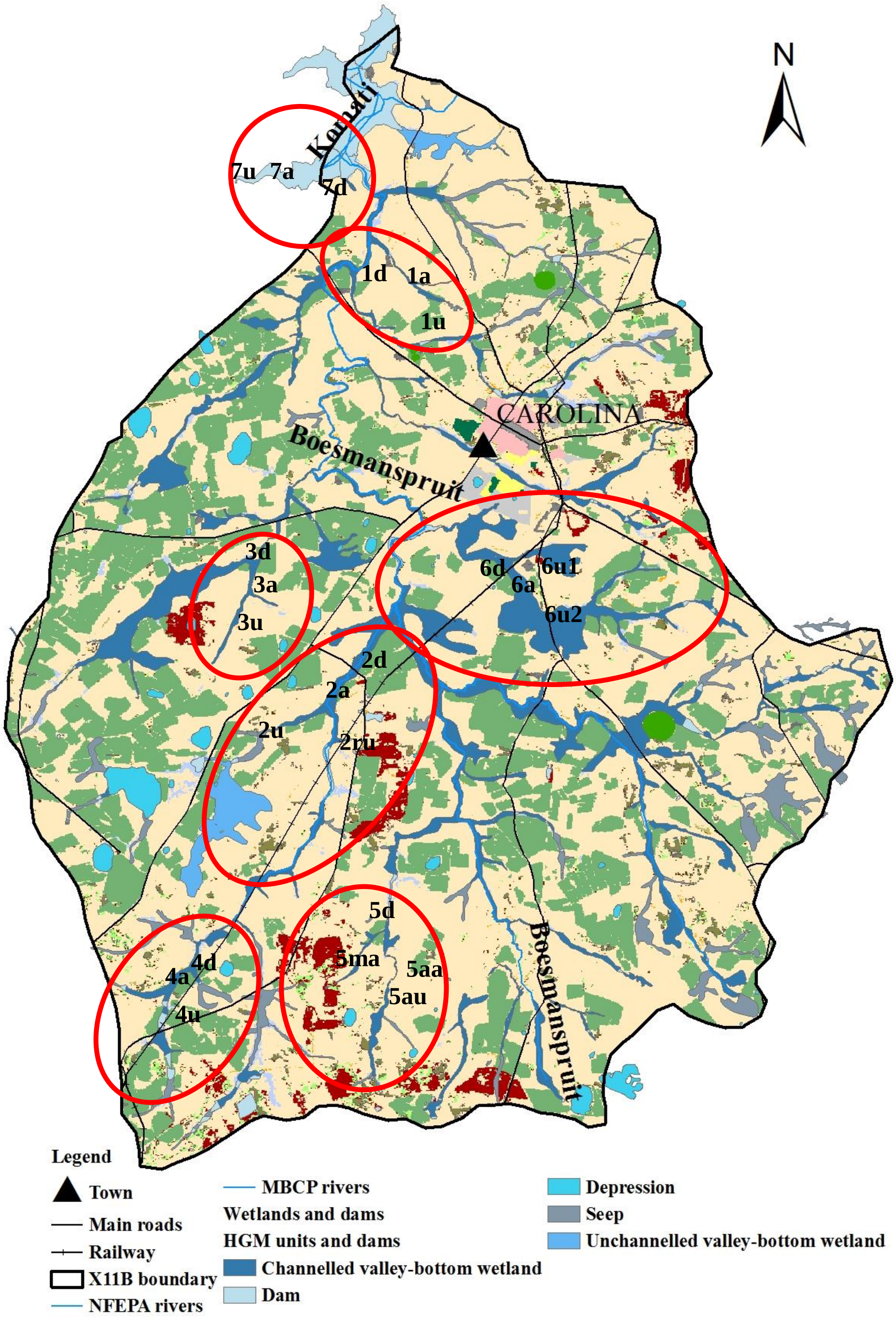

Figure 3.1: Locations of selected wetlands (indicated by red circles) and sampling points up (u), at (a) and downstream (d) of the wetlands 1) Roodepoort, 2) Boesmanspruit, 3) Jagtlust, 4) Witbank historical decant, 5) Witbank, 6) Droogvalei, 7) Nooitgedacht (reference site).

Table 3.1. Descriptions of selected wetland sampling points	
Sampling points	**Description**
1u	Upstream of Roodepoort wetland
1a	At Roodepoort wetland
1d	Downstream of Roodepoort wetland
2u	Upstream of Boesmanspruit wetland
2ru	Railway bridge upstream of Boesmanspruit wetland
2a	At Boesmanspruit wetland
2d	Downstream of Boesmanspruit wetland
3u	Upstream of Jagtlust wetland
3a	At Jagtlust wetland
3d	Downstream of Jagtlust wetland
4u	Upstream of Witbank historical decant wetland
4a	At Witbank historical decant wetland
4d	Downstream of Witbank historical decant wetland
5au	Upstream of Witbank wetland agricultural tributary
5aa	At Witbank wetland agricultural tributary
5ma	At Witbank wetland mining tributary
5d	Downstream of Witbank wetland
6u1	Upstream of Droogvalei wetland site 1
6u2	Upstream of Droogvalei wetland site 2
6a	At Droogvalei wetland
6d	Downstream of Droogvalei wetland
7u	Upstream of Nooitgedacht wetland
7a	At Nooitgedacht wetland (reference site)
7d	Downstream of Nooitgedacht wetland

3.4 Methods

Sample collections were planned for mid-March 2016 in order to coincide with the region's reported wet season. Site field assessments (WET-EcoServices and WET-Health, water quality measurements, and macroinvertebrate sampling) were carried out in March (wet season) on the following dates: 1- 09/03/2016, 2- 14/03/2016, 3- 12/03/2016, 4- 10/03/2016, 5- 13/03/2016, 6- 11/03/2016, 7- 11/03/2016. This would have allowed macroinvertebrates to re-establish in the wetland sites during the summer rainy season and experience limited water quality perturbations and disturbances, so providing a more accurate assessment (Dickens and Graham, 2002). However, as the region was experiencing a nation-wide drought, the rains were delayed and occurred during the sampling period. In order to avoid altered stream conditions, sampling was only carried out on days of little or no precipitation.

3.4.1 Nooitgedacht wetland delineation

The Nooitgedacht wetland delineations were carried out using the same desktop preparation and field methodology used for the other wetlands described in Chapter 2 (Sections 2.1.2 and 2.1.3).

3.4.2 Ecosystem services and wetland health

WET-EcoServices and WET-Health (Macfarlane et al., 2009; Kotze et al., 2009) were used for assessing wetland conditions and service provisions at each of the wetland HGM sites (Refer to section 1.3.2).

3.4.3 Aquatic biota

The South African Scoring System (version 5) (SASS5) was used to collect macroinvertebrate data (Dickens and Graham, 2002). After SASS data were recorded, samples were emptied into 350 ml jars containing 70% ethanol for preservation and labelled with the site, the replicate number and the date of collection. Three replicate samples were collected at each wetland site and transported back to the laboratory for identification to the family level. The presence and abundance of all taxa were identified to family level using SASS identification guides (Department of Water Affairs and Sanitation, 2002). SASS5 results were compared among all sites, with the Nooitgedacht wetland being the chosen reference site for the study.

3.4.4 Water chemistry

At the same time as biota sampling, water samples were collected using standard South African sampling protocols (Dickens and Graham, 2002; DWA, 2004). Water sample sites were selected based on their location relative to the location of the seven main wetlands, with water sampling sites located upstream ('u'), at ('a') and downstream ('d') of the main wetland (Figure 3.1 and Table 3.1). At site 2, one of the upstream tributaries is labelled "ru" (railway bridge). At site 5, tributaries are labelled "ma" (mining at site) and "aa" (agricultural at site). Twenty-four sites were sampled in total (Figure 3.1 and Table 3.1). Water samples were collected in acid-washed 500 ml HDPE plastic bottles for laboratory analysis. Water samples were kept cool, frozen, and delivered to the Agricultural Research Council: Institute for Soil Climate and Water (ARC.LNR) certified laboratory for analysis of **major ions** (Anions (mg/l): fluoride, nitrite, nitrate, chloride, sulphate, phosphate, carbonate, bicarbonate, sodium carbonate, sodium bicarbonate. Cations (mg/l): sodium, potassium, calcium, magnesium, boron), **trace elements** (lithium (Li), beryllium (Be), titanium (Ti), vanadium (V), chromium (Cr), manganese (Mn), cobalt (Co), nickel (Ni), copper (Cu), zinc (Zn), arsenic (As), selenium (Se), rubidium (Rb), strontium (Sr), molybdenum (Mo), cadmium (Cd), tin (Sn), antimony (Sb), tellurium (Te), caesium (Cs), barium (Ba), lanthanum (La), tungsten (W), platinum (Pt), mercury (Hg), lead (Pb), bismuth (Bi), uranium (U)) and **system variables** pH, EC (mS/m), Total Dissolved Solids (TDS), alkalinity, hardness, Permanent hardness and temperature (°C)).

Trace elements, major ions, and system variables were also measured against the Komati River System's Resource Quality Objectives (RQOs) and Water Quality Guidelines for freshwater ecosystems (DWAF, 1996; DWS, 2016). Toxic constituents and system variables that exceeded the RQOs and Water Quality Guidelines were used to extend the understanding of impacts to aquatic biota and therefore the health of the wetland systems.

3.4.5 Data analysis

Aquatic biota

In order to identify the strength of relationships between the macroinvertebrate family abundance and environmental gradients (**trace elements, major ions and system variables**), one Detrended Correspondence Analysis (DCA) and two Canonical Correspondence Analyses (CCAs) were carried out using CANOCO v4.55 (ter Braak and Smilauer, 2002). Analysis ordinations included: 1) sites by the presence and abundance data of macroinvertebrate families (DCA), 2) sites by the presence and abundance data of macroinvertebrate families and the **trace elements** of each site (CCA), 3) sites by the presence and abundance data of macroinvertebrate families and the **major ions** and **system variables** of each site (CCA) (Garcia-Criado et al., 1999). These analyses were designed to determine which water quality variables could best explain the spatial distribution of macroinvertebrates (Faith and Norris, 1989; Kilonzo et al., 2014).

Canonical correspondence analysis is a multi-variate analytical technique that prepares an initial ordination of the taxa by sample matrix. The analysis applies regression equations to these gradients to further adjust the position of samples using environmental variables collected at each site. The position of a sample in the ordination space is determined by the cumulative contribution of each taxon in the sample (ter Braak and Smilauer, 2002). A measure of family diversity was also calculated for each of the wetland sites using the Shannon Wiener diversity index (Spellerberg and Fedor, 2003). Statistical significances of the CCA models were determined by applying 1000 permutations in a Monte Carlo permutation test to each ordination (Faith and Norris, 1989).

Water chemistry

Principal Components Analyses (PCA) (Stewart et al., 2000; Bizzi et al., 2013; Kilonzo et al., 2014) were carried out using CANOCO v4.55 for the ordination of sites based only on 1) **trace elements** and 2) **major ions** and **system variables**. Principal Component Analysis is a multi-variate analytical technique that prepares a linear combination of a set of variables using a correlation or covariance matrix. The technique produces a set of components each able to explain a certain amount of variance in the overall dataset, thus simplifying a large dataset by grouping variables so that it can they can be explained by a few components (Janžekovič and Novak, 2012). Water quality data from each site sampled are given in Appendix C.

3.5 Results

3.5.1 Ecosystem services and wetland health

Analysis of samples collected in the wet season showed that provision of ecosystem services increased in comparison with the dry season results (section 2.4) although some service magnitudes decreased slightly. Conditions of wetlands all increased to category “A” with the exception of the Boesmanspruit and Droogvalei wetlands which increased, to a “B” category (Table 3.2 and Table 2.4 to Table 2.9 – refer to previous chapter).

Wetland WET-Health assessments

Table 3.2: Present State category for each module and the overall Present State of all wetlands assessed in the X11B catchment

Wetland	Present Ecological State category			
	Hydrology	**Geomorphology**	**Vegetation**	**Overall Present State**
Roodepoort (1a)	A	A	A	A
Boesmanspruit (2a)	B	B	A	B
Jagtlust (3a)	A	A	A	A
Witbank historical decant (4a)	A	B	A	A
Witbank (5aa)	A	A	A	A
Droogvalei (6a)	B	B	A	B
Nooitgedacht (7a)	A	A	A	A

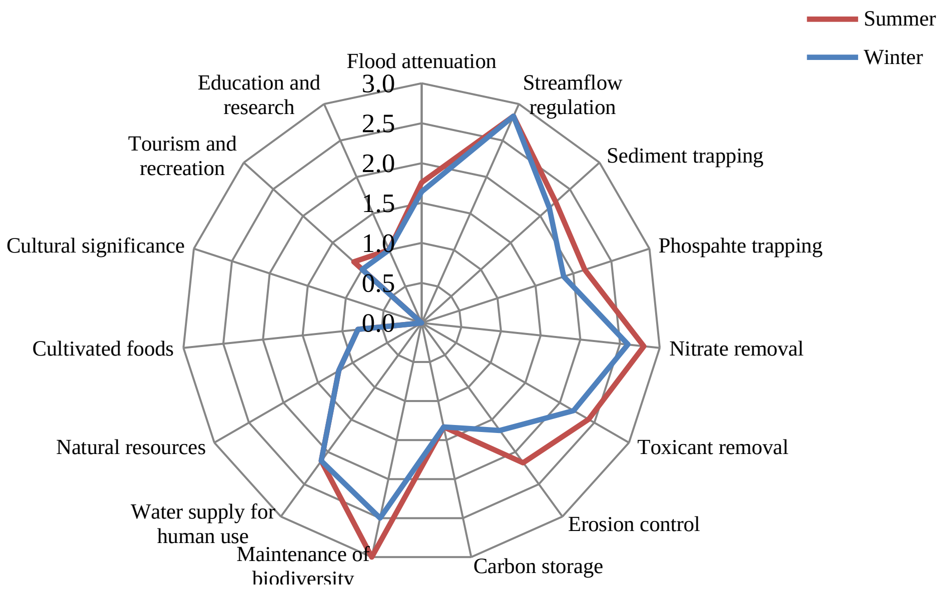

Figure 3.2: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Roodepoort wetland HGM unit one (channelled valley-bottom wetland).

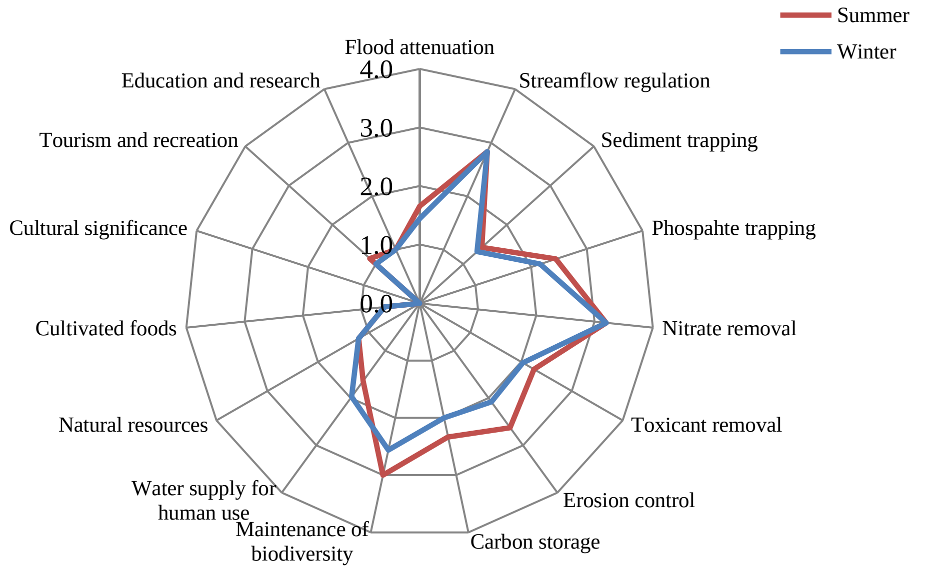

Figure 3.3: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Roodepoort wetland HGM unit two (hillslope seep linked to a stream channel).

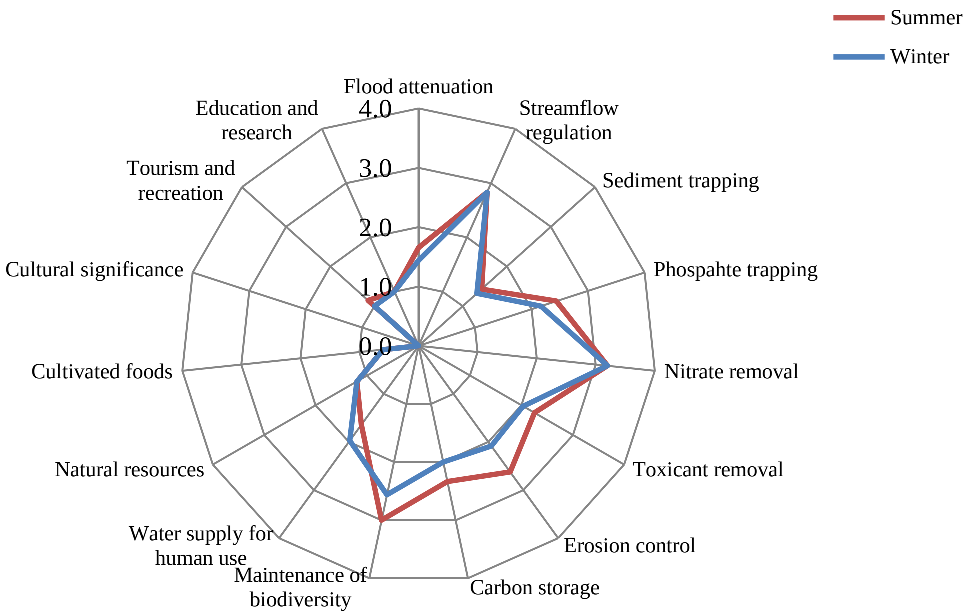

Figure 3.4: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Roodepoort wetland HGM unit three (hillslope seep linked to a stream channel).

Boesmanspruit wetland WET-EcoServices assessment

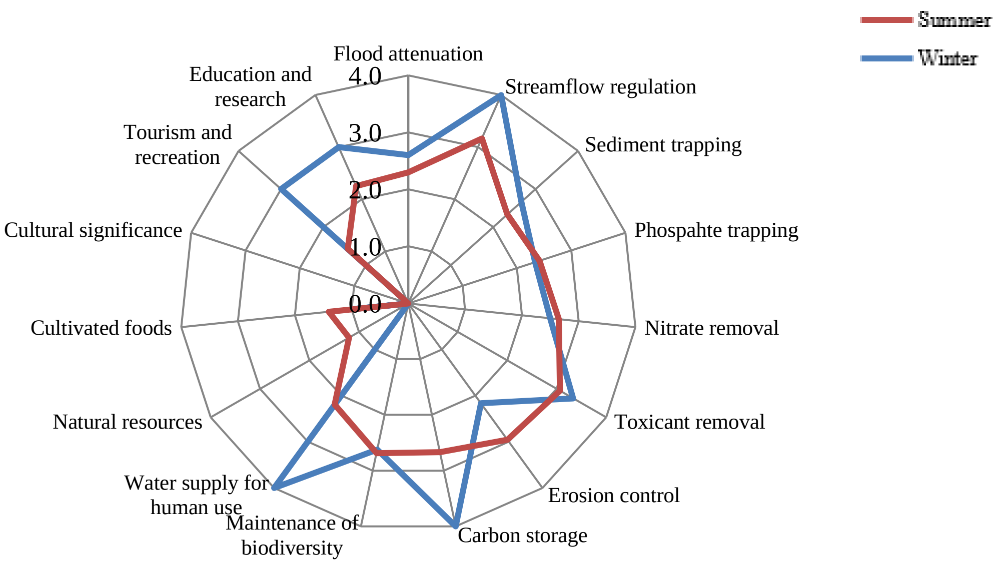

Figure 3.5: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Boesmanspruit wetland HGM unit one (channelled valley-bottom wetland).

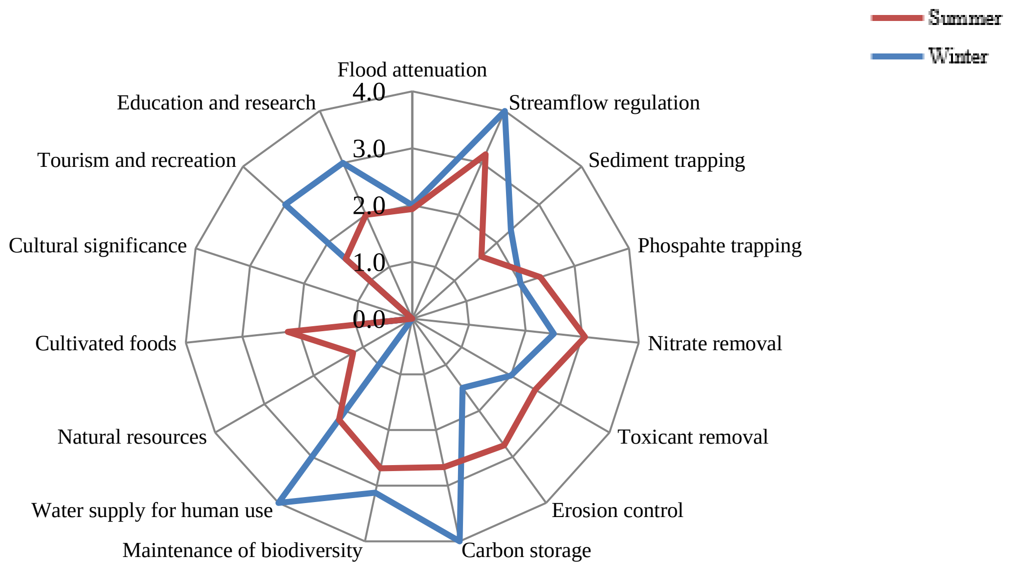

Figure 3.6: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Boesmanspruit wetland HGM unit two (hillslope seep linked to a stream channel).

Jagtlust wetland WET-EcoServices assessment

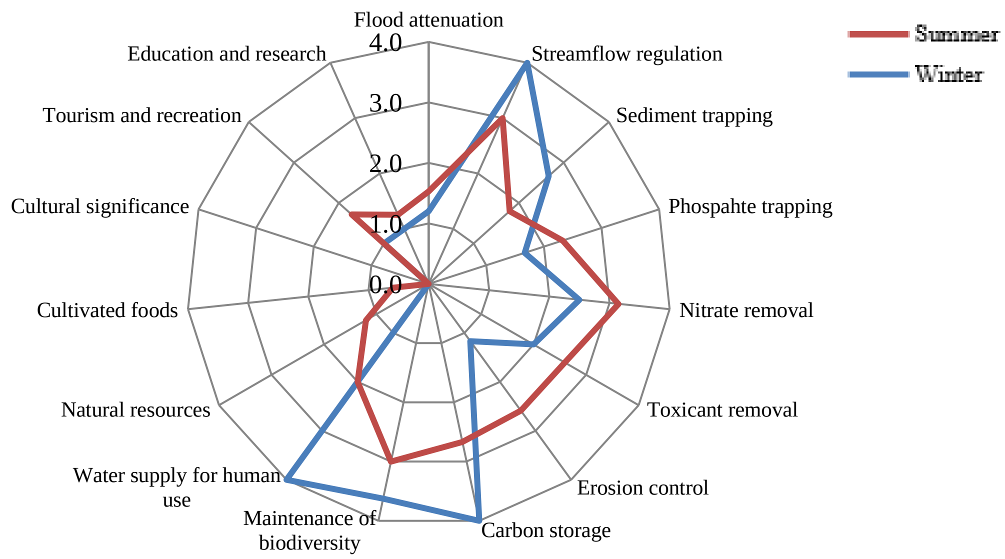

Figure 3.7: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Jagtlust wetland HGM unit one (channelled valley-bottom wetland).

Witbank historical decant wetland WET-EcoServices assessment

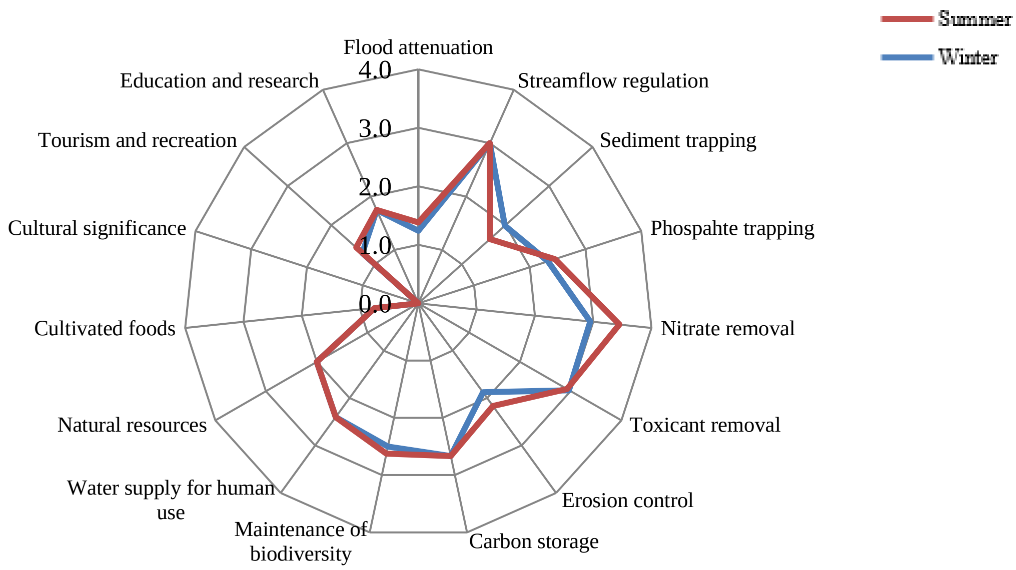

Figure 3.8: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank historical decant wetland HGM unit one (channelled valley-bottom wetland).

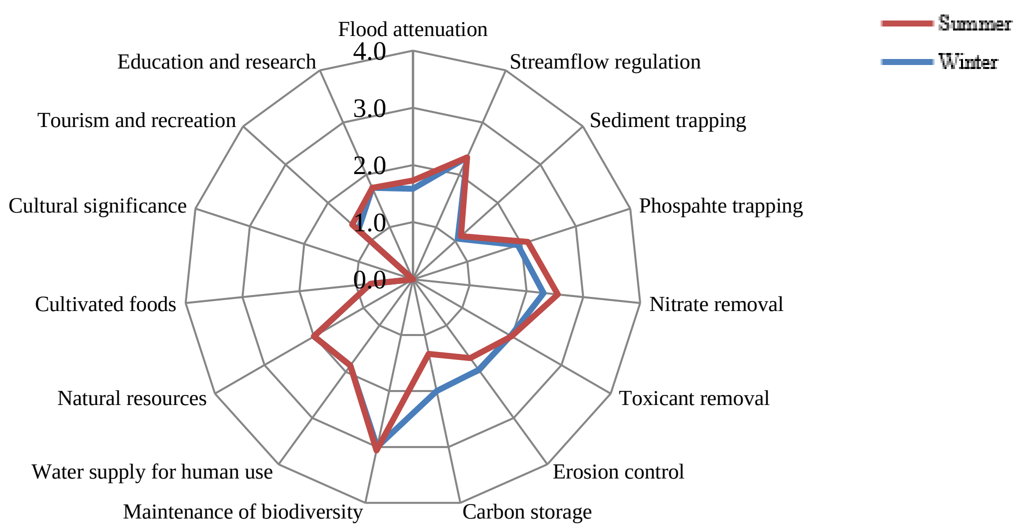

Figure 3.9: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank historical decant wetland HGM unit two (hillslope seep linked to a stream channel).

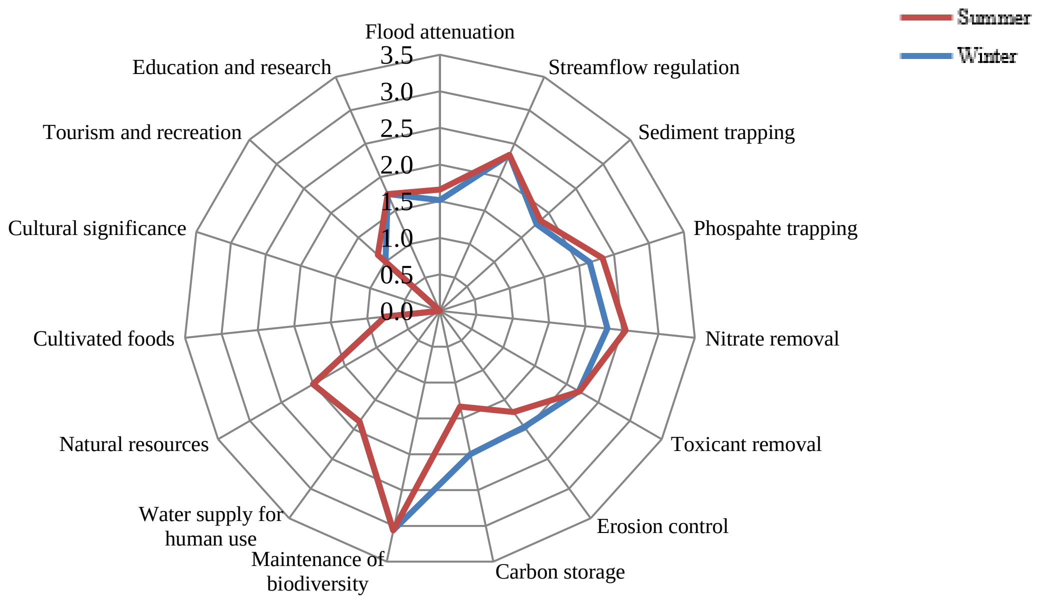

Figure 3.10: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank historical decant wetland HGM unit three (hillslope seep linked to a stream channel).

Witbank wetland WET-EcoServices assessment

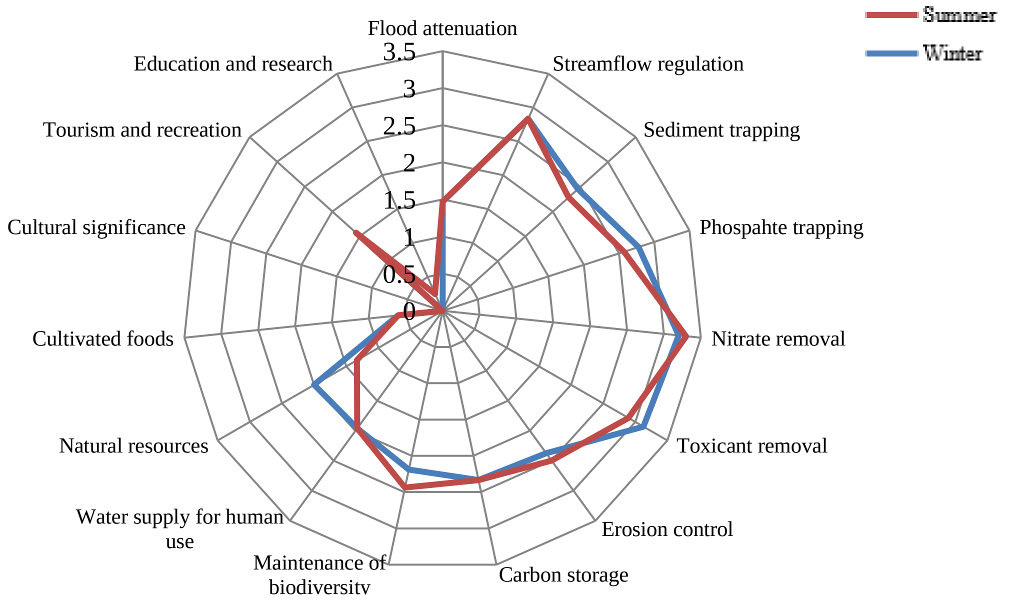

Figure 3.11: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank wetland HGM unit one (channelled valley-bottom wetland).

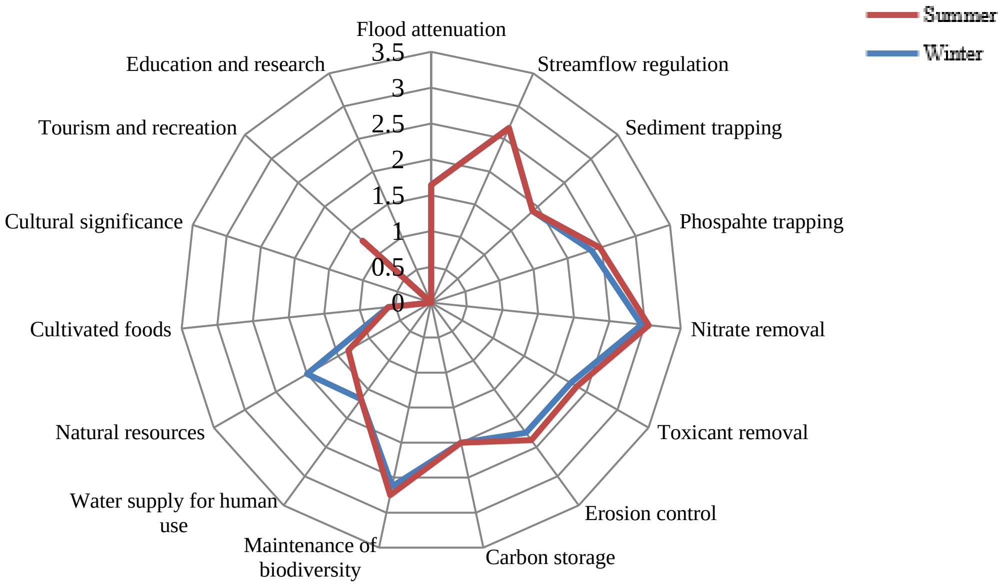

Figure 3.12: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank wetland HGM unit two (hillslope seep linked to a stream channel) and three (depression).

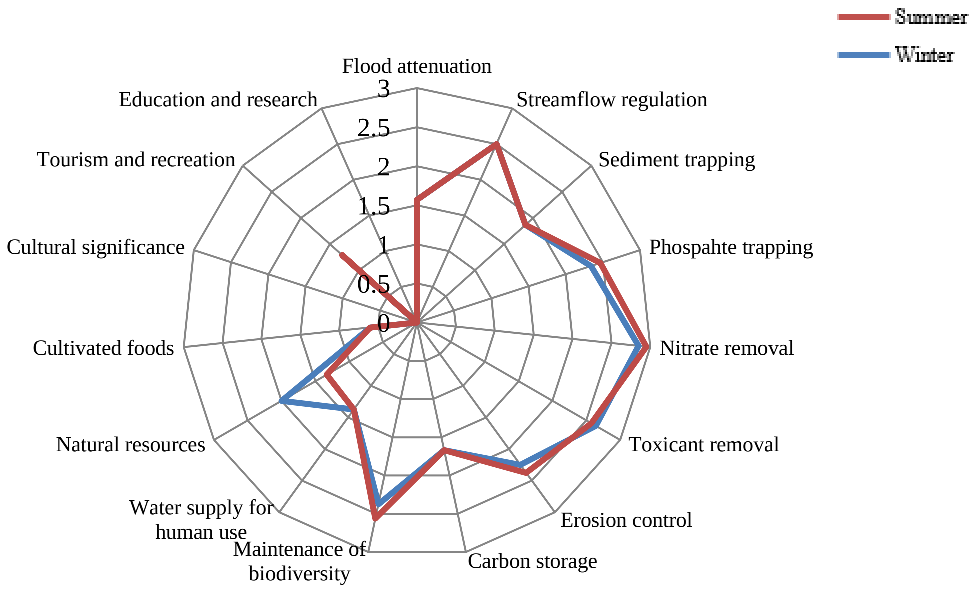

Figure 3.13: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank wetland HGM unit four (hillslope seep linked to a stream channel).

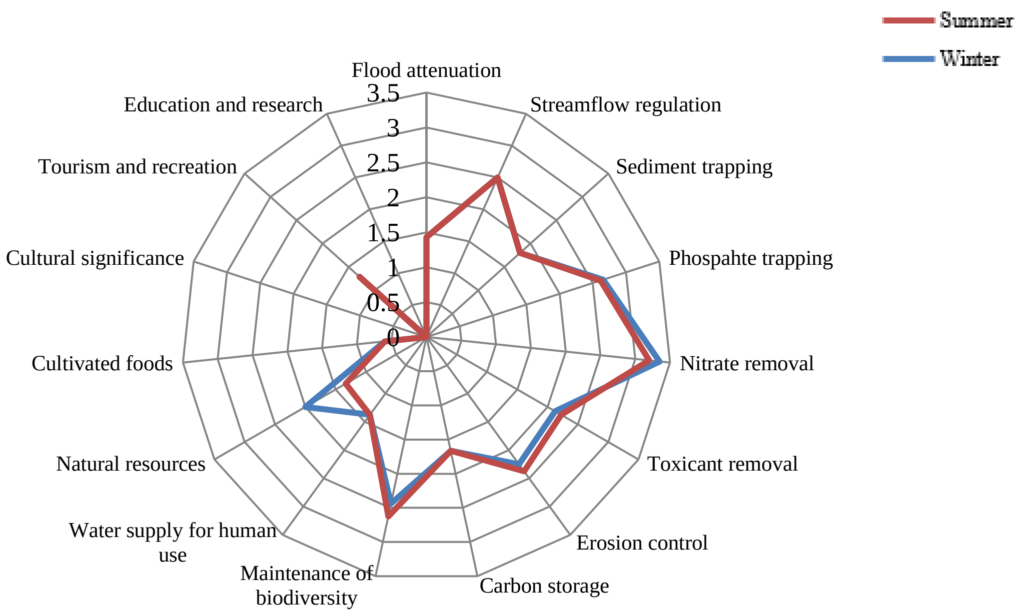

Figure 3.14: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank wetland HGM unit five (hillslope seep linked to a stream channel).

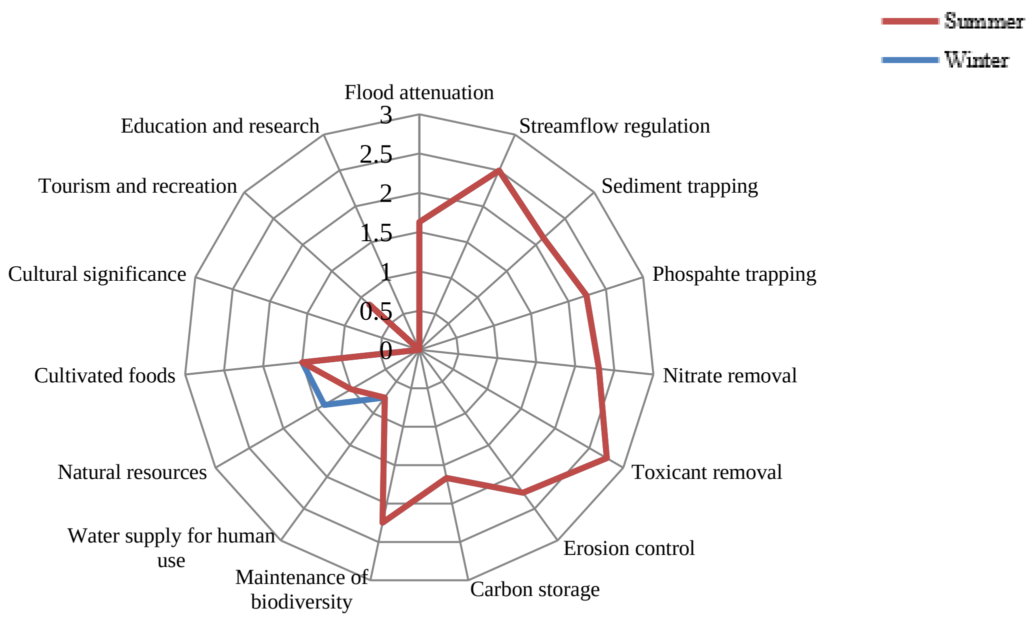

Figure 3.15: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Witbank wetland HGM unit six (channelled valley-bottom wetland).

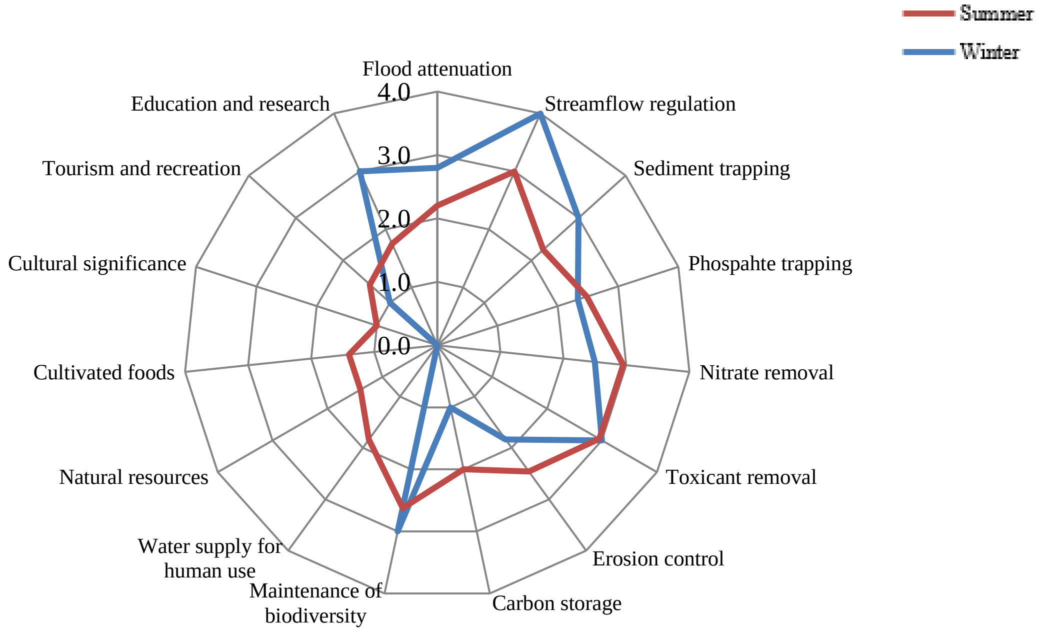

Figure 3.16: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Droogvalei wetland HGM unit one (channelled valley-bottom wetland).

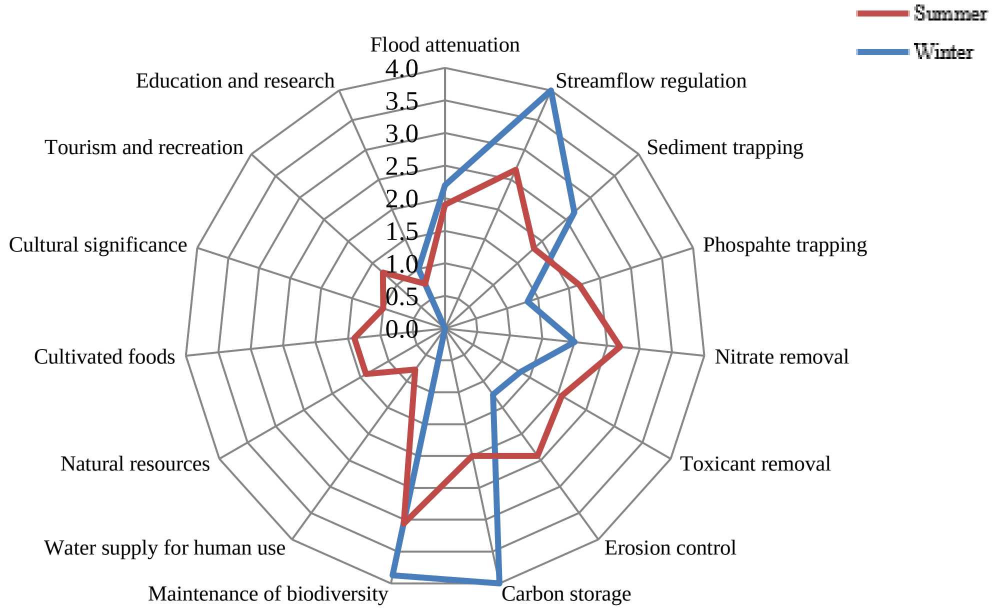

Figure 3.17: WET-EcoServices (Kotze et al., 2009) overall summer and winter season scores for Droogvalei wetland HGM unit two (hillslope seep linked to a stream channel).

Nooitgedacht wetland WET-EcoServices assessment

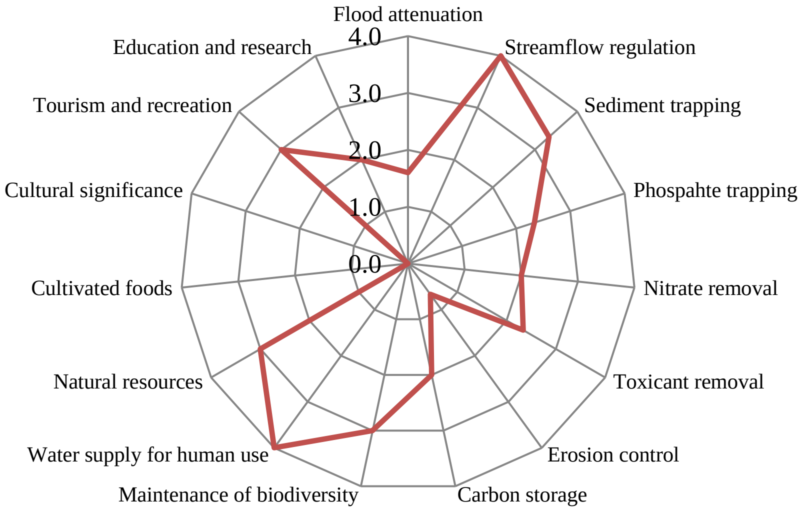

Figure 3.18: WET-EcoServices (Kotze et al., 2009) overall summer season scores for Nooitgedacht wetland HGM unit two (channelled valley-bottom wetland).

3.5.2 Aquatic biota

Canonical correspondence analysis

The initial Detrended Correspondence Analysis (DCA) was carried out using macroinvertebrate families, their abundance and sites (with no environmental variables). Following the addition of both sets of environmental data (firstly **major ions,** and secondly **trace elements and system variables**) to the analysis and the application of CCA, there was no appreciable change in the position of either families or samples from the DCA bi-plot. This indicated that the addition of environmental data did not materially affect the final position of families and sites in the bi-plot. The first CCA bi-plot (family x site x major ions) is presented for all further interpretation (Figure 3.19). Environmental gradients did not contribute further to the elucidation of the gradients, and macroinvertebrate families had dominant effects on the position of samples (ter Braak, 1988).

Macroinvertebrate associations with water chemistry parameters are consistent with each wetland (Figure 3.19, Figure 3.20 and Figure 3.21). Trace elements, major ions and system variables are consistently linked with particular suites of macroinvertebrate families. Results of the CCA, using the presence of macroinvertebrate taxa and major ions, showed 53.1% of total variation explained by the first two axes (33.0% on the first and 20.1% on the second axis) (Figure 3.19). There is a strong correlation between the water quality parameters and the species axes. The families Gerridae, Sphaeriidae, Ceratopogonidae, Gomphidae, Corbiculidae, Psychodidae are in greater abundance in the site 3a wetland. Sites 2a and 6a had high abundance of Simuliidae, Planorbinae, Physidae, Thiaridae, Hydracarina families. Sites 4a, 5ma and 5aa had a high abundance of Tabanidae, Veliidae, Pisulidae, Nepidae, corduliidae, Platycnemidae families, and sites 1a and 7a (reference site) had a high abundance of Oligochaeta, Culicidae, Chironomidae, Potamonautidae, Belostomatidae, Corixidae, Coenagrionidae, Dytiscidae, Gyrinidae, Hydrophilidae, Tipulidae, Naucoridae, Lestidae, Protoneuridae, Aeshnidae families.

Although four distinct groups could be identified, no associations can be attributed to a particular land-use, or combination of land-uses. There are some specific organism associations, but these do not follow patterns where more sensitive organisms are absent from land associated with mining, agriculture, or conservation. Therefore, one may argue that there is no clear pattern of macroinvertebrate family distribution based on the direct impact from either agriculture or mining. It is likely that this classification of the eight wetland sites reflects more generally the broad ecological attributes of each catchment and its associated soils, lithology and water chemistry.

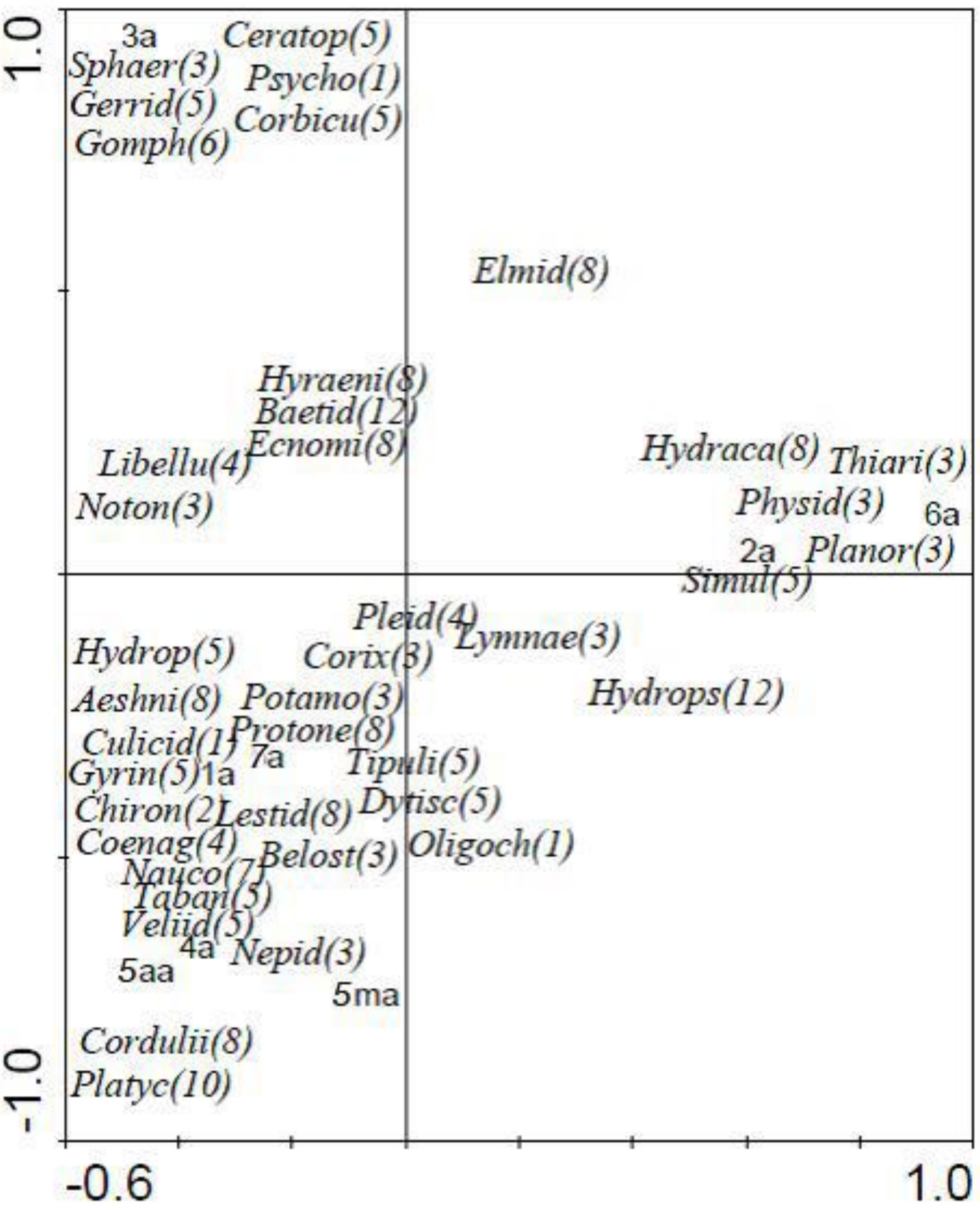

Figure 3.19: CCA ordination bi-plot for macroinvertebrate families by samples by major ions at the eight wetland sites sampled in the X11B catchment. Samples used in this ordination are 1a, 2a, 3a, 4a, 5aa, 5ma, 6a and 7a (Table 3.1). Macroinvertebrate families (Table 3.3) are presented with their associated sensitivities to polluted water (Dickens and Graham, 2002)

Table 3.3: Macroinvertebrate family abbreviations	
Macroinvertebrate family abbreviations	**Macroinvertebrate Families**
Oligoch	Oligochaeta
Culicid	Culicidae
Psycho	Psychodidae
Chiron	Chironomidae
Potamo	Potamonautidae
Belost	Belostomatidae
Corix	Corixidae
Nepid	Nepidae
Noton	Notonectidae
Lymnae	Lymnaeidae
Physid	Physidae
Planor	Planorbinae
Thiari	Thiaridae
Sphae	Sphaeriidae
Baetid	Baetidae
Coenag	Coenagrionidae
Libellu	Libellulidae
Pleid	Pleidae
Hydrops	Hydropsychidae
Gerrid	Gerridae
Veliid	Veliidae
Dytisc	Dytiscidae
Gyrin	Gyrinidae
Hydrop	Hydrophilidae
Ceratop	Ceratopogonidae
Corbicu	Corbiculidae
Gomph	Gomphidae
Nauco	Naucoridae
Hydra	Hydracarina
Lestid	Lestidae
Protone	Protoneuridae
Aeshni	Aeshnidae
Ecnomi	Ecnomidae
Elmid	Elmidae
Hyraeni	Hyraenidae
Platye	Platycnemidae
Simul	Simuliidae

Results of the CCA of macroinvertebrate taxa presence and the major ions and system variables showed 54.5% of total variation explained by the first two axes (33.4% on the first and 21.1% on the second axis). Site 3 had high concentrations of nitrate, potassium, bromine and a high sodium adsorption rate (SAR). Concentrations of sulphate, fluorine, and TDS were high in sites 2 and 6, and phosphate concentrations in these sites were lower than the other sites (Figure 3.20). Sites 1, 4, 5aa, 5ma and 7 had high concentrations of nitrite (Figure 3.20).

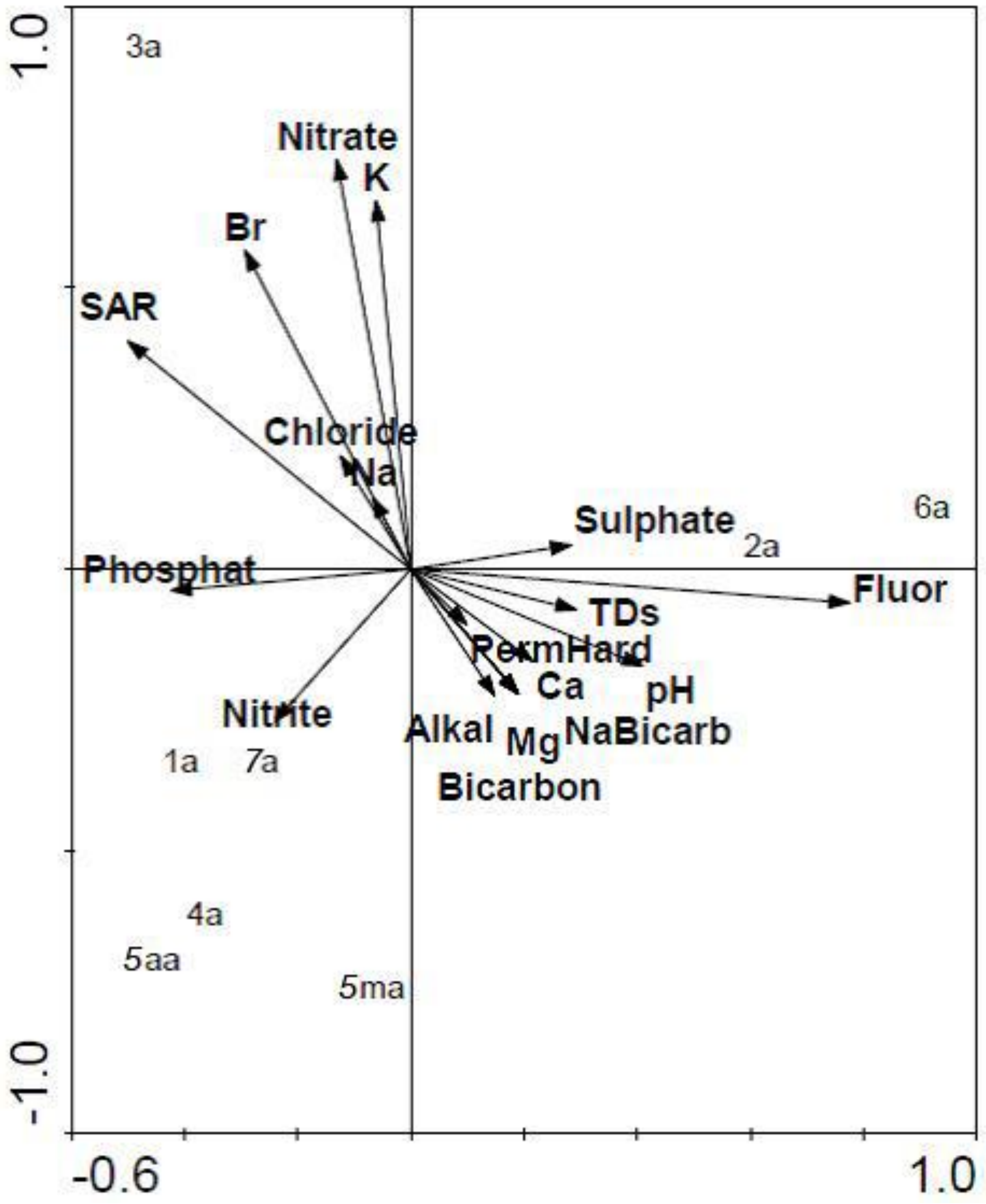

Figure 3.20: CCA ordination bi-plot for macroinvertebrate family abundance and the major ions and system variables at the eight wetland sites (Table 3.1) sampled in the X11B catchment. Arrow length indicates the strength of the relationships between taxa abundance and the major ions and system variables.

The CCA results of macroinvertebrate taxa presence and the trace elements showed 53.1% of total variation explained by the first two axes (33.0% on the first and 20.1% on the second axis). Site 3 had concentrations of Li, Ba, Rb, Te, whereas the sites with the most human activity (2a and 6a), had higher concentrations of Cs, Sr, Hg, but low trace elements overall (Figure 3.21). Ti, V, La, Be, were higher in sites 1 and 7 and Cu, As, Se, Mo, Sn, Sb, Pt concentrations were higher in sites 4, 5ma and 5aa (Figure 3.21).

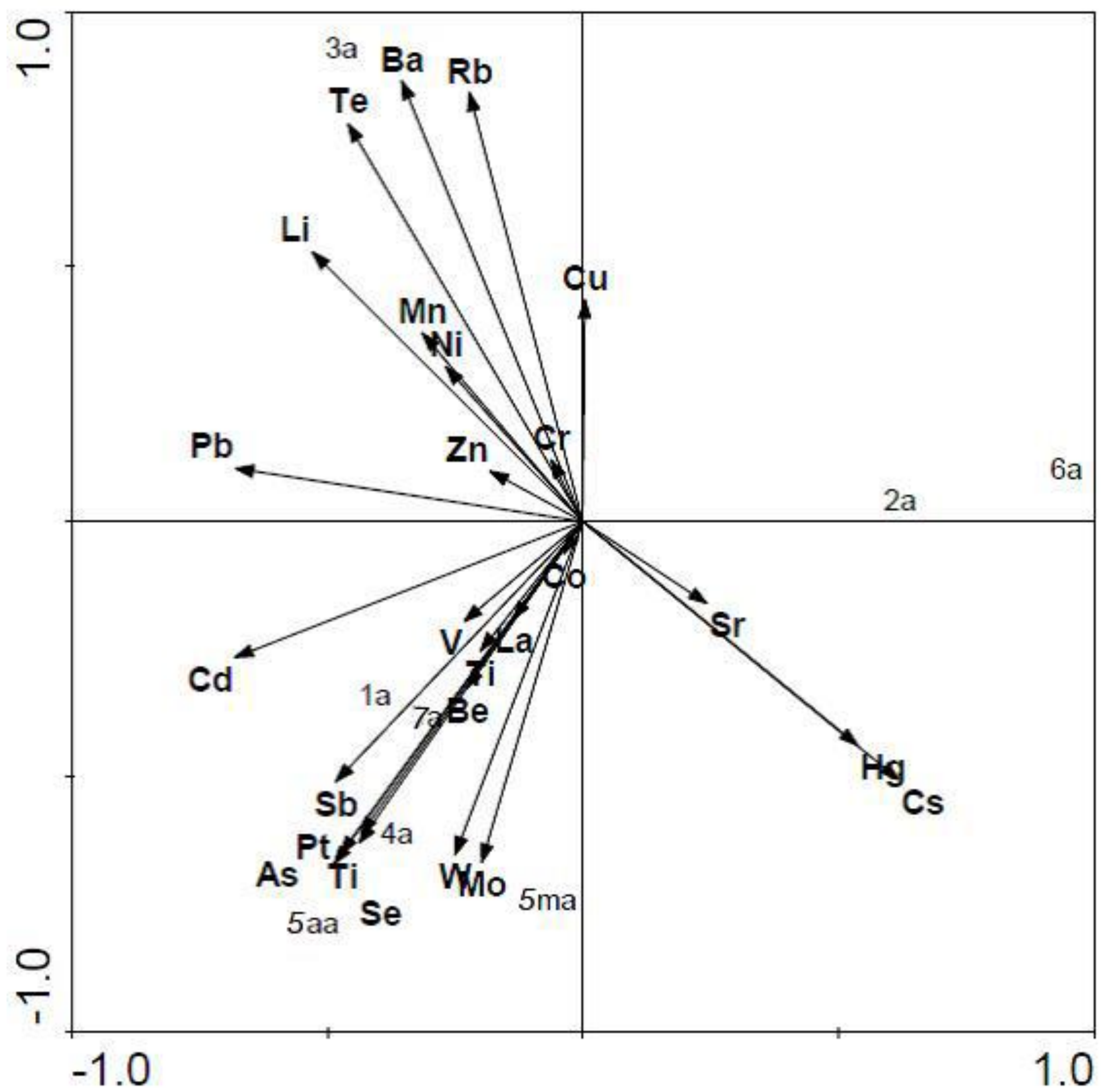

Figure 3.21: CCA ordination bi-plot for macroinvertebrate family abundance and the trace elements at the eight wetland sites (Table 3.1) sampled in the X11B catchment. Arrow length indicates the strength of the relationships between taxa abundance and the major ions and system variables.

Since the ordination analyses did not reveal a clear correlation between land-use or water chemistry and macroinvertebrate sensitivities, a Shannon Weiner analysis was undertaken. The sites impacted by mining had the lowest Shannon Weiner diversity indices (Figure 3.22). The Witbank mining tributary (site 5ma) and the Witbank historical decant site (site 4a) both had lower index scores than other sites. The wetlands of two tributaries flowing into wetland 5 (5aa in an agricultural landscape and 5ma in a mining landscape) had distinct differences in macroinvertebrate family diversity, insinuating mining influence.

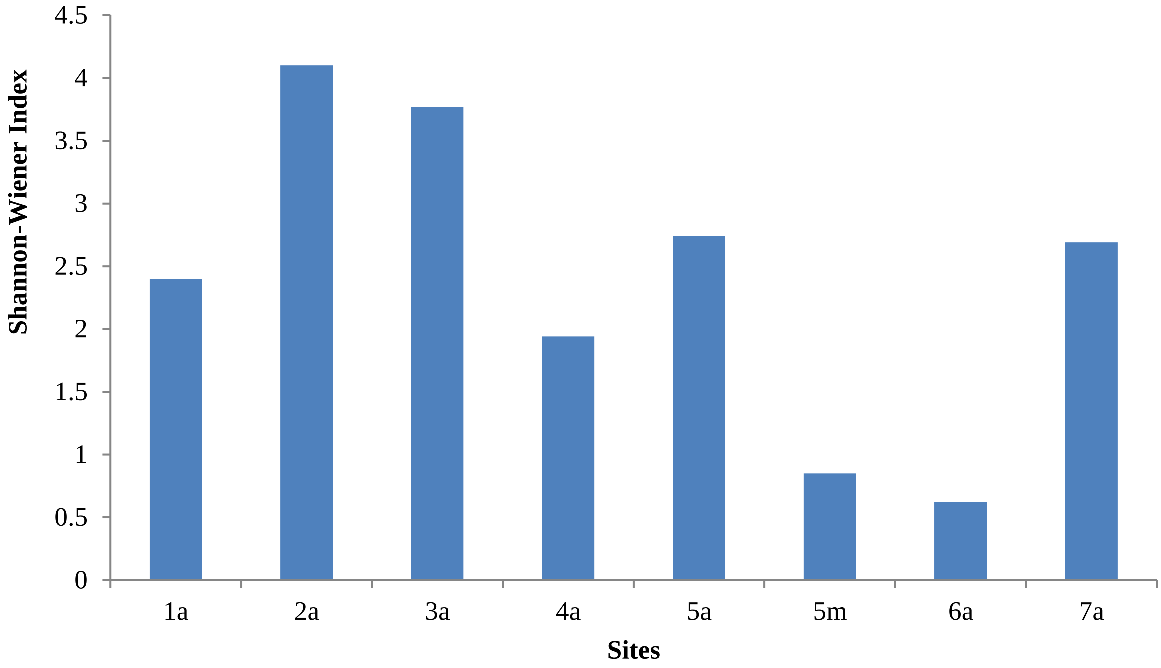

Figure 3.22: Shannon Wiener diversity index scores for macroinvertebrates at the eight wetland sites (Table 3.1) sampled in the X11B catchment.

Of all the wetlands, the 7a reference wetland had the greatest number of taxa as well as the most sensitive taxa.

Table 3.4: The 2016 South African Scoring System (SASS5) and Average Score Per Taxon (ASPT) scores for each sampled wetland of the X11B catchment

Site	**1a**	**2a**	**3a**	**4a**	**5aa**	**5ma**	**6a**	**7a**
SASS	23	25	35	10	19	10	34	66
No. Taxa	5	6	9	3	4	2	4	13
ASPT	4.6	4.17	3.89	3.33	4.75	5	4.8	5.08

3.5.3 Water chemistry

Mining-impacted sites have more distinct chemistry parameters, and hence they are clearly apparent in Figure 3.23 and Figure 3.24 PCAs. For both, trace elements and major ions at sites 4a and 5ma are positioned away from groups formed from sites not impacted by mining. Based on system variables, Site 5ma is more dissimilar than all sites, including 4a, which is subject to rehabilitation efforts, hence the strong calcium correlation in Figure 3.24 (Matlock et al., 2002). Impacts on other sites are difficult to distinguish as sources of pollution range and add to a complex integration of water chemistry parameters.

PCA results of trace metals at each site showed 96.2% of total variation explained by the first two axes (59.2% on the first and 37.0% on the second axis). The PCA of trace metals for all water samples (Figure 3.23) shows three distinct groupings, namely, large numbers of sites near the centroid, with sites 4 (4a, 4d, 4u) and 5 (5ma, 5d, 5u) forming two separate clusters. This reflects the same mining impact pattern previously identified within the macroinvertebrate analysis (Figure 3.22). The sites around the centroid probably represent characteristics of relatively normal wetland trace element measures, implying relatively normal wetland trace element concentrations (Figure 3.23). Site 4 (4u, 4a, 4d) contained high concentrations of Sr, and site 5ma, 5d contained the highest concentrations for Ni, Li, La, Zn, Be, Co, and Mn (Figure 3.23).

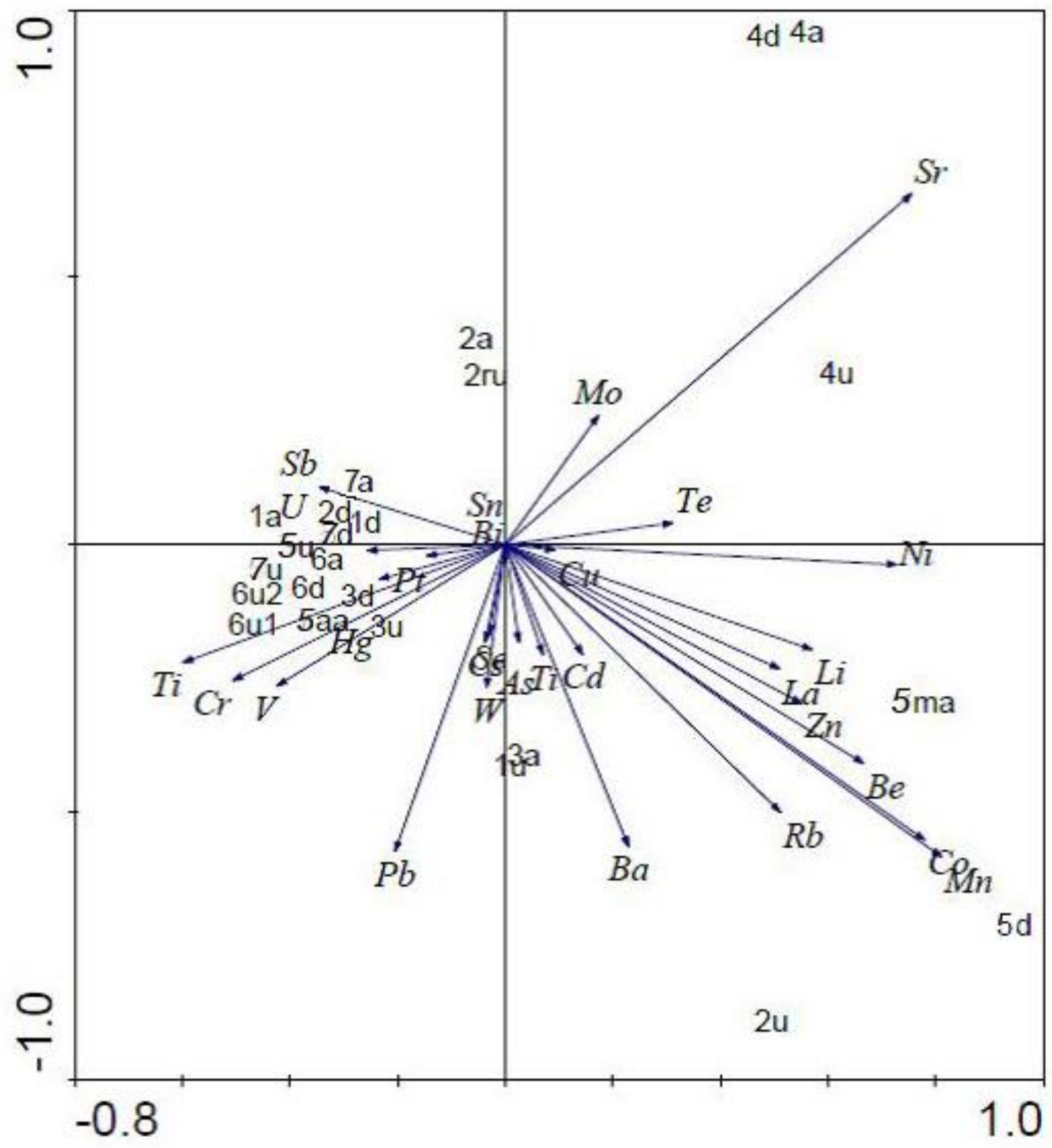

Figure 3.23: PCA ordination bi-plot for trace metals at 24 water samples sites (Table 3.1) in the X11B catchment in comparison to the other sample sites. Arrow length indicates the strength of the relationships between taxa abundance and the major ions and system variables.

The PCA results of major ions and system variables at each site showed 92.2% of total variation explained by the first two axes (59.0% on the first and 33.2% on the second axis). Principal Component Analysis ordination of showing **major ions** and **environmental variables** (Figure 3.24) shows the same pattern: sites 5 (5ma and 5d) and 4 (4a, 4u, 4d) deviated from the centroid where all the other sites are located. Site 5d contained high concentrations of sodium bicarbonate, fluoride, and sodium and had high EC, whereas sites 4 (4a, 4u, 4d) and site 5ma contained high concentrations magnesium, sulphate, calcium, and bromine (contributors to the high TDS), as well as high nitrate and nitrate and phosphate (nutrients).

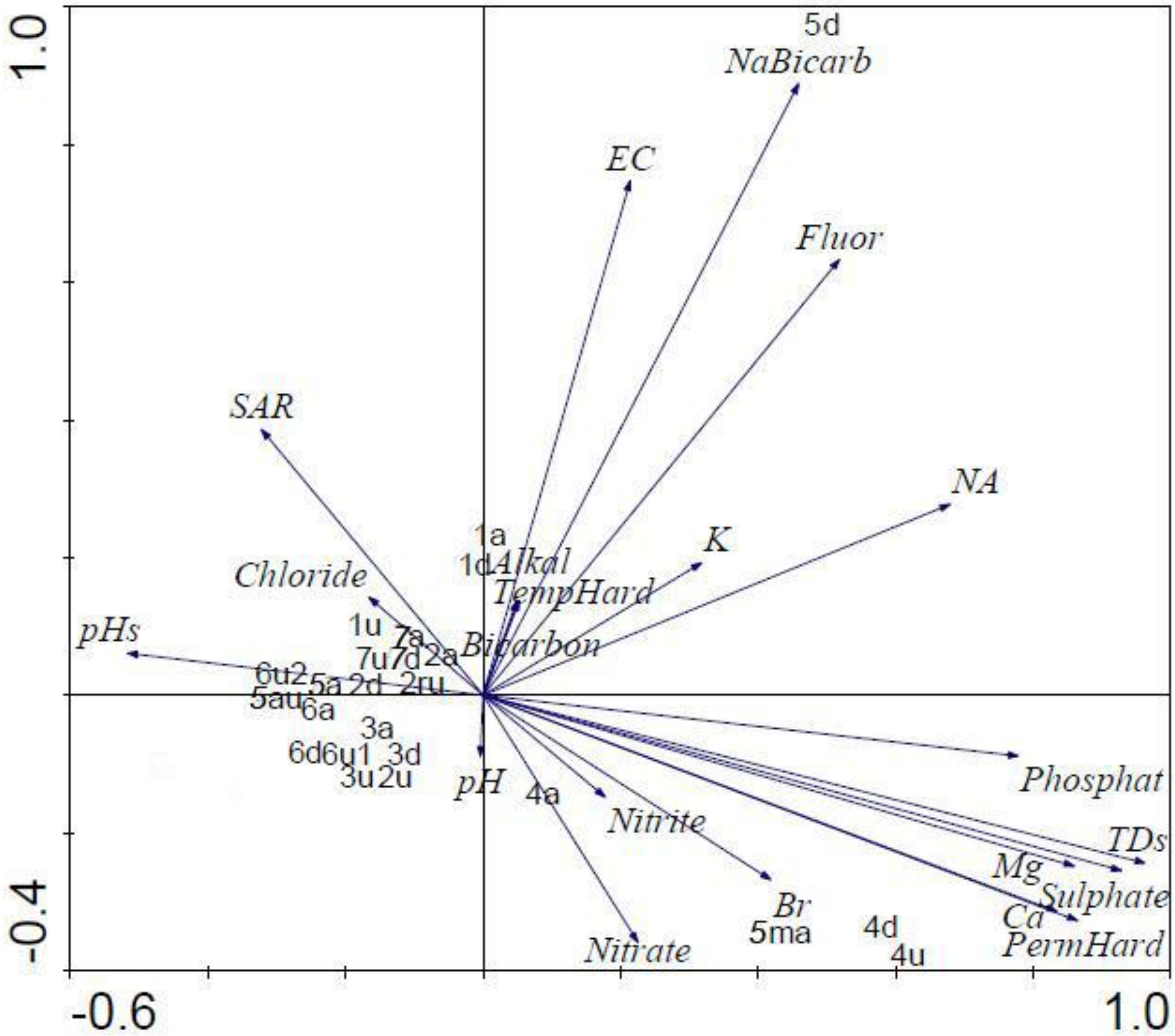

Figure 3.24: PCA ordination bi-plot for major ions and system variables at 24 sites (Table 3.1) in the X11B catchment. Arrow length indicates the strength of the relationships between taxa abundance and the major ions and system variables.

Table 3.5: Comparisons of water quality results at each wetland sampled in the X11B catchment and Water Quality Guidelines (DWAF, 1996) and Resource Quality Objectives of the Komati River System (DWS, 2016). Water quality concentrations and levels exceeding RQO limits are represented in bold											
	SA aquatic acute	**SA aquatic chronic**	**Inkomati -Usuthu RQOs and SA aquatic TWQR**	**1a**	**2a**	**3a**	**4a**	**5ma**	**5aa**	**6a**	**7a**
Mn (mg/l)	1.3	0.37	0.18	**4.06**	**0.76**	**227.10**	**34.45**	**906.13**	**5.26**	**1.17**	**0.45**
Cu (mg/l)	0.0016	0.00053	0.0003	**3.48**	**1.8**	**2.85**	**2.74**	**2.9**	**1.3**	**3.95**	**2.51**
Zn (mg/l)	0.036	0.0036	0.002	**0.82**	**0.025**	**11.88**	**13.42**	**73.74**	**2.95**	**12.48**	**10.47**
As (mg/l)	0.13	0.02	0.01	**0.33**	**0.19**	**0.21**	**0.32**	**0.16**	**0.43**	**0.31**	**0.44**
Se (mg/l)	0.03	0.005	0.002	**0.01**	**0.01**	**0.01**	**0.01**	**0.01**	**3.75**	**0.01**	**2.63**
Cd (mg/l)	0.003	0.0003	0.00015	**0.01**	**0.005**	**0.10**	**0.03**	**0.13**	**0.2**	**0.03**	**0.05**
Hg (mg/l)	0.0017	0.00008	0.00004	**0.50**	**1.12**	**0.21**	**0.04**	**0.4**	**0.93**	**0.46**	**0.04**
Pb (mg/l)	0.004	0.0005	0.0002	**0.05**	**0.01**	**0.06**	**0.01**	**0.04**	**0.06**	**0.06**	**0.03**
pH			5.9-6.5	**7.4**	**6.84**	**5.4**	**6.67**	**4.1**	5.46	5.54	**6.75**
EC (mS/m)			30	**40**	**31**	13	9	8	6	8	25
Fluoride (mg/l)	2.54	1.5	0.75	0.13	**0.8**	0.1	0.32	0.33	0.15	0.17	0.24
Sulphate (mg/l)			80	30.28	**90.91**	53.58	1.06	**436.42**	12.22	19.51	57.28
Phosphate (mg/l)			0.025	**0.04**	**0.06**	**0.69**	**2.57**	**3.33**	**0.5**	**0.03**	**0.23**

3.6 Discussion

Using macroinvertebrate community structure and more elaborate water quality samples, the aim of his chapter was to extend the WET-Health methodology used in Chapter 2. Increases in features that infer wetland ecosystem service and benefit provisions and health evidently increased in the summer season (Table 3.2). The 2016 summer season results also indicated higher pH values and lower EC values than the 2015 sample (Table 2.12). Many concentrations and levels of constituents exceeded RQOs and fell within the CEVs and AEVs of the aquatic ecosystem Water Quality Guidelines.

Macroinvertebrate taxa abundance was used as a proxy for wetland health. Strength of relationships between macroinvertebrate family abundance and environmental gradients occurring at sites with different land-uses were identified without clearly detecting mining sites. However, SASS5 results indicated lower scores in wetlands than in the reference wetland, indicating agricultural, urban and mining land-uses in the catchment have an impact on wetlands. The lowest taxa numbers and SASS scores were recorded at mining sites, suggesting mining impact. Comparisons in 2016 (Table 3.4) with SASS5 2015 scores (Table 2.11) were lower, indicating continued impact and limited recovery. Results also indicated that mining had an impact on the diversity of taxa.

3.6.1 Wetland ecosystem services and health

The WET-EcoServices tool is based on the opportunity provided by and effectiveness of the wetlands' features, both of which can increase or decrease, depending on the combinations of features that are associated with the service and the influence of a change in season. During the winter seasons in the X11B catchment when rainfall is low and temperatures are cold, flow through wetlands is minimal and crop farming is discontinued, leaving the soil bare (Ometo et al., 2000). Livestock depend on the availability of water from wetlands for grazing as winter seasons too (Scoones, 1991). The result is trampling and overgrazing of the wetland banks and surroundings leading to disturbance and erosion of soil, and exposure and hardening of the surrounding landscape surface. These seasonal characteristics contribute towards lower vegetation cover and hydrological flow, which can ultimately lead to changes in geomorphology of a wetland (Macfarlane et al., 2009), so therefore, lowering health scores and the likelihood of ecosystem services provided by wetlands, distinctive of these features (Scoones, 1991). The opposite is found with the change to the summer season when warm temperatures and increased rainfall result in an increase in hydrological flow, vegetation cover, and crop growth; an overall increase in health modules, scores and the likelihood of services provided by the wetlands. The positive hydrological and vegetation influence that the summer season holds suggests that ecological infrastructure in the X11B catchment is more effective at servicing and benefiting people and the natural environment, according to the season.

3.6.2 Water quality and aquatic biota

Many studies have identified that heavy metals and system variables associated with mining-impacted sites reduce freshwater macroinvertebrate. Sensitive species, abundance, and diversity are reduced when compared to other impacted and reference sites (Cherry et al., 2001; Pond et al., 2008) (Appendix G). These were also the expected outcomes of this study. However, using a CCA, most of the wetlands seem to have a similar characteristic set of wetland biota (Figure 3.19). Results do not follow a pattern in which more sensitive organisms are absent from land associated with mining, agriculture, or conservation. The Nooitgedacht wetland (the reference site) had greater family abundance and numbers of sensitive biota than other sites, as expected, but it should be noted that the Nooitgedacht wetland site has gabion headcut protection structures built into the wetland as a rehabilitation effort (Working for Wetlands, 2008). Research by Brown et al. (1997) shows that a characteristic of rehabilitated wetlands is significantly lower macroinvertebrate family abundance. The abundance, however, depends on different characteristics of the restored wetland, features such as the ecology of the macroinvertebrate family, (for example, flightless families will not be able to establish, whereas aerial macroinvertebrates can colonise quickly), and the habitat structure (the presence of vegetation and open water) (Brown et al. 1997). Therefore, had the wetland not needed rehabilitation, the abundance, diversity and presence of sensitive species would have been greater.

During the 2016 sampling of water, higher pH values and lower EC values than the 2015 sampling were observed. At the time of sampling, the wetlands were in high flow resulting in flushing and dilution within the wetlands. SASS5 indicated SASS scores, number of Taxa and Average Score Per Taxon (ASPT) scores were lower than the selected reference wetland. The same was found in the low-flow 2015 SASS5 sample, further suggesting that agricultural, urban and mining land-use in the catchment had an impact on wetland integrity (Ometo et al., 2000). More specifically low numbers of taxa and SASS scores were recorded for mining sites with Witbank mining and the Witbank historical decant wetlands having the lowest macroinvertebrate taxa numbers and no evidence of highly sensitive families. Mining sites could therefore be impacting macroinvertebrates more than other land-uses (Tate and Husted, 2015).

Research suggests that SASS5 is minimally influenced by seasonality in the highveld region (Fourie et al., 2014). Therefore, lower SASS5 scores found when comparing the 2015 and 2016 SASS5 scores, indicate that there is limited recovery of macroinvertebrates (Table 2.11 and Table 3.4). Comparisons with high-flow data collected by Tate and Husted (2015) suggest a similar situation. Comparisons of SASS5 scores at the Boesmanspruit wetland suggested that biota diversity and therefore wetland integrity had not improved. The 2016 Boesmanspruit SASS5 scores all reflect worse values than the high-flow data collected in Tate and Husted’s November 2013 study. The low numbers of taxa and sensitive species present in the data collected for this thesis may indicate that the Boesmanspruit has not recovered from the 2012 Acid Mine Drainage (AMD) event, despite the contaminants being flushed out and wetland sediments being replaced (Refer to section 1.8.1).

Mining impacts on macroinvertebrate families were also indicated by the Shannon Wiener diversity index (Figure 3.22). The results indicated that the diversity of families was less at the two sites with extensive mining. Figure 3.24 shows the higher correlations of sulphate and TDS associated with mining sites. Both parameters are known to reduce macroinvertebrate presence and therefore may justify the low level of macroinvertebrate family diversity (Harding, 2005). Mining influence was particularly evident for wetlands on two tributary arms flowing into a downstream Witbank wetland (Site 5); the wetland on the tributary influenced by mining (5ma) has a lower biodiversity score than the one influenced by agriculture (5aa).

Trace elements, major ions, and system variables occur naturally and are essential for the biological functions of biota within aquatic ecosystems. Trace elements such as, Mn, Zn, and Se are essential nutrients for aquatic biota (DWAF, 1996). However, at higher concentrations, trace elements, as well as major ions and system variables, can become extremely toxic. The increased concentrations are generally a result of anthropogenic activity within a catchment (DWAF, 1996). All overlapping trace elements tested for in the water quality guidelines and the water quality analysis of this thesis (Mn, CU, Zn, As, Se, Cd, Hg, Pb) were recorded at concentrations above RQOs and AEVs for all wetland sites (Table 3.5). The Boesmanspruit and Witbank mining wetlands both had sulphate concentrations higher than the RQOs. The Boesmanspruit and Roodepoort wetlands both had EC readings over recommended RQOs, and the Witbank mining wetland had pH values lower than the RQOs range (Table 3.5).

The X11B catchment is dominated by dryland crop agriculture, a farming practice that uses a range of fertilisers and pesticides. Trace elements and ions such as: manganese, phosphate and zinc are used in fertilisers and zinc, copper and selenium in pesticides (DWAF, 1996). These trace elements and ions accumulate in aquatic systems from runoff from the surrounding landscape. Most sites sampled were influenced by agricultural activities occurring adjacent to and upstream of the wetlands, thus explaining the breach of RQOs at these sites. Farming-related constituents usually contain a mixture of ions which tends to increase the salinity of aquatic systems, thus increasing the EC at the Roodepoort wetland (DWAF, 1996). High EC readings can also be explained by mining activities, hence the high readings at the Boesmanspruit wetland, a wetland that receives water from most wetlands within X11B. Mining activities can also be explained by high Mn and sulphate concentrations, and low pH values, such as those found in the Witbank mining tributary wetland (DWAF, 1996). The Boesmanspruit site also had high sulphate readings, further indicating the presence of mining (AMD) contamination in the catchment.

The high concentrations and readings exceeding RQOs and falling beyond the CEVs and AEVs call for urgent management intervention. Biota that absorb toxicants directly have biological implications (Lange et al., 2014; López-López and Sedeño-Díaz, 2015). Excessive trace metals, major ions and system variables influences may have implications for toxicant assimilative services, adding to the bioavailability of toxicants within the wetland as well as downstream of the ecosystem (Lange et al., 2014; López-López and Sedeño-Díaz, 2015). The sampled wetlands and water resources are threatened by a loss in biodiversity and ultimately a change in ecological feedbacks, and the loss of wetland functions. The readings may also explain the low SASS5 scores presented in Table 2.11.

The results presented interesting correlations of macroinvertebrate community structure responses with chemical attributes, but these did not correlate either with site-related land-use, or the sensitivity of the biota. This means that the more detailed quantitative studies corroborated the qualitative assessments in showing little evidence of 2015/6 impacts after the 2012 AMD incident. However, macroinvertebrates can be sensitive to changes in water flow (Bunn and Arthington, 2002), where, depending on the organisms' ecology and physiology, flow can wash away macroinvertebrates, moving them downstream, or even transporting them to other wetlands (Tangen et al., 2003). Rain may also cause more mobile and terrestrial biota to physically leave the wetland. Therefore, a limitation to the summer samples would include the collection of biota during a rainfall period, and the possibility of biota displacement influencing the results (Tangen et al., 2003). The results are also limited to single data values and not average-based values of water chemistry and macroinvertebrate family abundance and diversity; however, results are similar to studies that have collected data on a monthly basis (Pond et al., 2008).

The geomorphology of the site contributes directly to the wetland's chemistry in the form of natural minerals and sediments (Sheoran and Sheoran, 2006). For instance, the Roodepoort and newly added Nooitgedacht wetland sites are part of the volcanic Transvaal sequence, made up of basalt rock rich in magnesium oxide (MgO) and lime/calcium oxide (CaO), and low in silica (SiO_2) and alkali oxides (Na_2O and K_2O) (McCarthy and Rubidge, 2005).

Jagtlust, Droogvalei, Boesmanspruit and the historical decant wetlands are all part of the open volcanic Karoo sequence, ECCA subgroup, and Vryheid formation, typically consisting of grit, sandstone, shale and coal seams. The Witbank wetland site is part of the open volcanic Pongola sequence, which is made up of intrusive rock sub-outcrop (slightly porphyritic) (Appendix D and E) (Bell et al., 2001). These different sediment characteristics, based on the geology of the wetlands, add to the complexity of wetland analysis and, in order to better understand their effect on biota, further investigation is needed.

3.7 Conclusion

Although CCAs indicated no clear pattern based on the types of macroinvertebrates present at each site, results indicate that land-use activities, especially mining, could be impacting wetland integrity. Water quality and macroinvertebrate structures infer poor wetland conditions, despite good wetland health and high provision of services indicated by WET-Health and WET-EcoServices. It is therefore recommended that WET-Health be re-evaluated. The issue is that wetlands are considered to be in good functional condition. However, with continued exposure to pollutants, there is the possibility that the wetland capacity to absorb toxins will be exceeded and the functionality of the wetlands impaired, potentially threatening the overall functioning of impacted wetlands, downstream users (humans and natural resource-users), and the ecological infrastructure services. The results do, however, indicate the resilience of wetlands: presently sites are in good condition and there is a strong basis to argue for systematic wetland protection by primary land-users, especially mining-impacted wetland sites during the winter season.

Constituent water quality values exceeded RQOs and chronic and acute limits, indicating continued contamination by land-users. This, along with the limited recovery of the Boesmanspruit wetland, calls for urgent management intervention in order to protect and conserve these wetlands of the Mpumalanga highveld region.

The 2012 AMD event would have flushed the sediment of the wetlands and reset the adsorptive capacity of the ecosystem. However, the toxic constituents and macroinvertebrate family structures indicated that the wetlands may have reached maximum adsorptive capacity and therefore it is hypothesised that, in the case of another heavy rainfall event, the town of Carolina risks another AMD crisis. As sediments are already perceived to be saturated with toxins and available in the water body, further AMD-related changes in acidity will increase the mobilisation of adsorbed toxins. Flooding and flushing of wetlands will therefore move toxins through the system and re-contaminate the Boesmanspruit dam.

Chapter 4: A systemic view of the user-natural resource contestation using hydro-connectivity as an integrating lens

The overall aim of this project was to explore the contestation between mining and agriculture, using wetlands as indicators, a nexus of ecological infrastructure. In order to achieve this aim, the previous chapters presented the results of a contextual assessment of wetland-use and impacts on wetlands in the quaternary catchment X11B of the Komati River, in the area around the town of Carolina. Results from previous chapters indicate that wetland water quality of the X11B catchment is still contaminated with traces of Acid Mine Drainage (AMD) despite management interventions after the 2012 crisis. As a result, the ecological and interdependent social components of the social-ecological system (SES) are under threat, adding to continued contestation. In order to fully understand the past and possible contestation, a greater systemic understanding of the SES is needed, highlighting the importance of transdisciplinary research.

4.1 Understanding resilience in Carolina

The X11B catchment is comprised of many feedbacks that keep the system within the boundaries of its evolved regime. The structures and functions of the ecological components maintain the threatened highveld grassland biome and support vulnerable, yet healthy, river and wetlands ecosystems (Van Vuuren, 2014). Naturally occurring ecosystems and their components are appreciated and used by people residing and working in the catchment (Section 2.5.3). The ecological infrastructure, such as wetlands, provides services like flood attenuation and water filtration (Section 2.5.1). These naturally occurring systems are also sustained through governance and sustainable water resource and land-use, but the consequences of unsustainable actions and poor decision making became apparent as causes of the slow alteration of drivers and feedbacks in the X11B catchment system, leading to the AMD crisis (Tempelhoff et al., 2012).

One of the main ecological and social drivers of the X11B catchment is water (Tempelhoff et al., 2012). The change in the quality and quantity of water resources in the catchment influences the naturally occurring feedbacks within the system. Important feedbacks that were subjected to changes included: balance of water chemistry constituents that undermined the capacity of wetland services to mediate water quality; changes in volumes of water which altered the flushing frequency of wetland sediment and the dilution of wastes, and the interactions of water resource-users with the catchment's water, and their responses to the catchment's governance institutions. Changes to feedbacks ultimately resulted in the town's resource dilemma, implicating different water-resource users, resulting in conflicts, jeopardising food webs and other ecosystems, and threatening the international watercourse shared between South Africa, Mozambique and Swaziland. Figure 4.1 depicts a systemic view of system components and their actions that had consequences on feedbacks influencing the ecological infrastructure of the X11B complex social-ecological system (C-SES) prior to the heavy rainfall event in 2012.

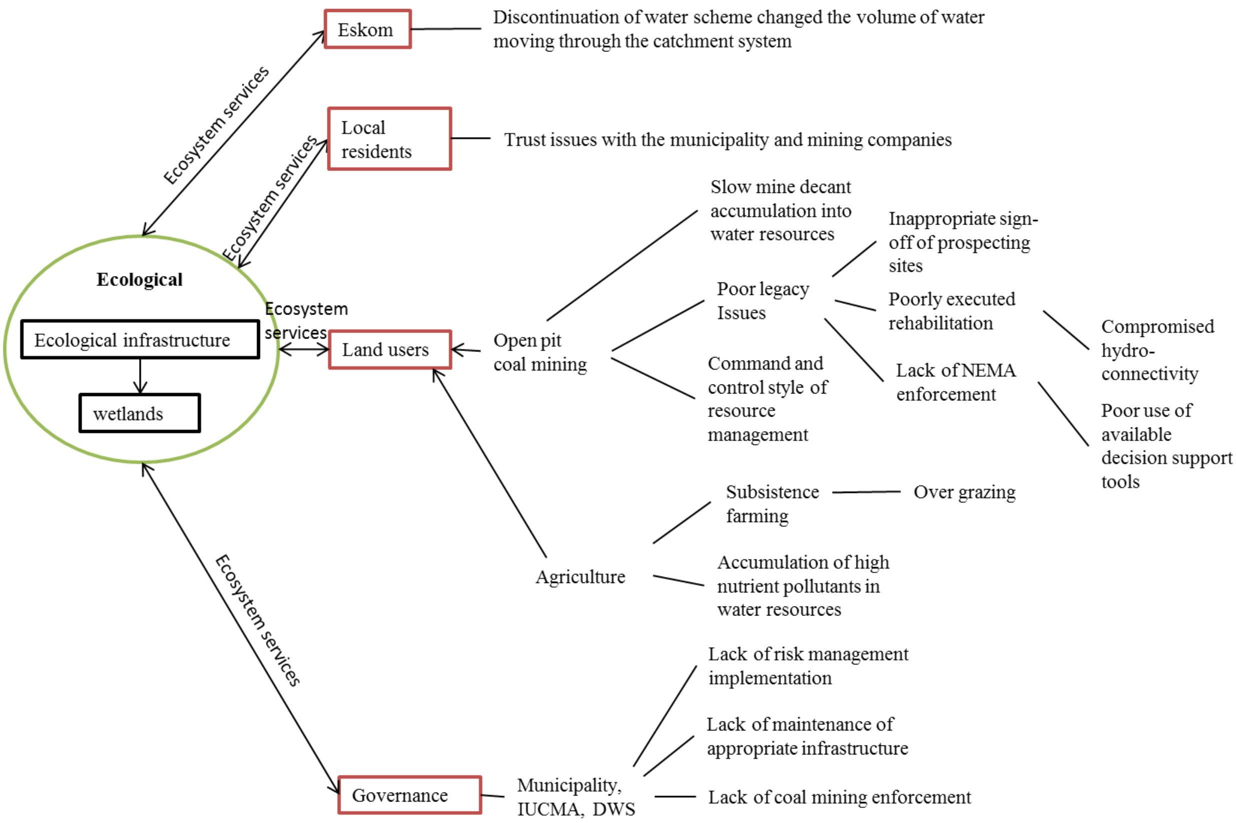

Figure 4.1: Systemic diagram of the X11B C-SES prior to the 2012 heavy rainfall event. Each social component (Eskom, local residents, land-users (subsistence farmers, open-pit coal mining, and agriculture), Governance (municipality, Inkomati-Usuthu Catchment Management agency (IUCMA), and Department of Water and Sanitation (DWS)) contributed to the disruption of feedbacks that led to the contamination of the Boesmanspruit dam and overall 2012 crisis.

Prior to the 2012 rainfall event, open-pit coal mining's poor legacy issues, and command-and-control style of resource management became evident. The high presence of heavy metals, sulphate, and low pH levels measured at sites during 2012 (McCarthy and Humphries, 2013) indicated the lack of enforcement of the National Environmental Management Act (NEMA) and the poor execution of rehabilitation of mining sites in the past. In addition to mining companies' poor law enforcement, two of the four mining companies found guilty of the 2012 contamination had proceeded without mining licences (Tempelhoff et al., 2012). The lack of law enforcement and rehabilitation execution would have ultimately contributed to the hydro-connectivity in the catchment being compromised and the slow accumulation of mine decant into water resources (Section 1.10.3), so changing the water quality and quantity, and compromising the wetland ecosystem's capacity to remove toxicants.

The capacity of wetlands to remove pollutants from the water body would also have been compromised by the accumulation of high nutrient pollution in water resources stemming from agriculture land-users in the catchment. Furthermore, livestock farming in close proximity to wetlands tends to limit the vegetation cover of grazed land, so increasing runoff and the volume of water entering wetlands (Scoones, 1991) and compromising their ecosystem services (Egoh et al., 2012).

The discontinuation of the Eskom water scheme also changed the volume of water that flowed through the sub-catchment's wetlands and streams, and the Boesmanspruit dam (Tempelhoff et al., 2012). The change in the volume of water would have altered the hydrological and geomorphic aspects of the wetlands and therefore the conditions and underlying services of the wetlands. Volume changes would have also reduced the amount of pollutant and sediment being flushed out and replaced within the system (Tempelhoff et al., 2012). This was evident when the scheme was reinstated in 2012 in an attempt to improve water quality, an attempt that raised the pH of the Boesmanspruit dam to 7 and diluted the dam's heavy metals and major ions. However, since the reinstatement in 2012, the transfer scheme has been idle, thus possibly creating a similar build-up of concentrated toxicants in the system, as hypothesised by McCarthy and Humphries (2013).

The extent and prolonged time frame of the AMD crisis that occurred in 2012 implied that the governance of the X11B catchment was ill-equipped and lacked the capacity to deal with the event (Section: 1.10.4). The municipality in charge had failed to maintain and implement infrastructure, leading to the further contamination of the town's drinking water. The DWS and the Inkomati-Usuthu Catchment Management Agency (IUCMA) failed to implement the appropriate risk management required in catchments where there is coal mining activity. Further, the DWS and IUCMA had not adequately monitored the coal-mining activity as mining was occurring without mining licences. Therefore, the inadequacy of the catchment's governance contributed to the contamination created by the open-pit coal mining.

Civil unrest and protests by local residents were an outcome of the poor governance of the catchment, as communication was limited and the right to safe water was impinged. However, the negative attitudes on the part of local residents towards catchment governance had emerged before the 2012 event and contributed to further animosity during the AMD crisis. The lack of conviction of mining companies responsible for the contamination as well as the little aid in crisis mitigation carried out by mining companies during 2012 further contributed to the civil unrest (Tempelhoff et al., 2012).

4.2 After the 2012 crisis

Since the Carolina 2012 incident, the local community has become more interested and aware of their catchment's environmental situation. The interest, according to Tempelhoff et al. (2012), was sparked by the revived civil society activism that emerged from the crisis. The IUCMA has since established a number of monitoring points within the catchment, established an Acid Mine Drainage task team, and revived the Upper Komati Catchment Forum (UKCF) made up of different stakeholders, including mining companies. This is viewed as part of the solution to restoring the capacity of the CMA in dealing with AMD related situations. The forum increases communication efforts shared between different stakeholders in an integrated and collective fashion, where social learning and the adaptation of understanding takes place (Ison et al., 2007). Once adaptive understanding is accomplished among all stakeholders, including researchers, practising and implementing appropriate management schemes is more likely to become an accomplishable reality (Rogers et al., 2013).

An example of the effects of improved collaborative efforts can be seen in the Komati's neighbouring catchment, the Crocodile catchment (Pollard and Du Toit, 2011), where the Inkomati CMA and irrigation farmers have worked together to form a catchment management strategy, focusing on implementing Integrated Water Resource Management (IWRM) and the equitable and sustainable sharing of water resources (Pollard and Du Toit, 2011). There is also an emphasis on strengthening feedback loops in order to keep "the management process responsive to contextual changes" (Pollard and Du Toit, 2011).

Tempelhoff et al. (2012) identified the system as experiencing emergent growth to conservation phase in the system's adaptive cycle. In this transition phase, new opportunities and available resources are exploited and connections between actors increase, thus increasing resilience. In order to maintain this phase, it is recommended that the CMA and

Department of Water Affairs (DWA) adopt a more holistic approach (i.e., Integrated Water Resource Management) and are encouraged to learn from the past incident and use new and shared knowledge to direct their decisions (i.e., Strategic Adaptive Management) (Walker and Salt, 2012; Pollard et al., 2014).

4.3 Transdisciplinary research

4.3.1 Evaluation of WET-EcoServices and WET-Health Methodology

Chapter 2 of this thesis provided a contextual analysis of the case study focusing on wetland health and ecosystem services, with user perceptions of wetland value. Results indicated that the wetlands were in a relatively healthy condition, delivering important and high levels of ecosystem services. These results indicate a healthy landscape with little impact from human activity. Health scores and the provision of ecosystem services, along with the identified National Freshwater Ecosystem Priority Areas (NFEPAs) and red-listed fauna and flora, can therefore provide a substantial foundation for the conservation and preservation of the wetlands and associated sub-catchments of the X11B catchment.

However, simple water quality measures demonstrated low pH levels and high Electrical Conductivity occurring in the wetland channels. Both measures indicate coal mining impacts, more specifically AMD impacts. Acid mine drainage typically has high concentrations of trace elements and metal ions available in a water solution, which are often biologically available to biota in the wetlands (Sheoran and Sheoran, 2006). Implications of this are that biota play an important role in the functioning of wetlands and, without them, the diversity of the system can become vulnerable to state shifts (Lange et al., 2014; López-López and Sedeño-Díaz, 2015). Repercussions of a shift in state include the loss of wetland functionality and eventual wetland loss, therefore a deeper investigation into the water quality and its effects on the wetland ecosystems was suggested.

Without adding water quality measures, the WET-Health methodology (Macfarlane et al., 2009) used would have indicated a healthy catchment, and any remediation needed for the provision of the wetland protection would have been based on observable land-use impacts opposed to the less obvious and detrimental chemical effects of mining. Therefore chapter 2 also suggested the mandatory use of water chemistry and a more elaborate use of wetland macroinvertebrate biota diversity in the assessment of catchments where past and present mining activity has taken place.

4.3.2 Stakeholder influence

A vital component in understanding the historical and present context and contestation of the X11B was the interaction and involvement of local stakeholders. However, the focus of this project is more ecologically based. A more thorough understanding of contestation would involve a deeper understanding of the coal and agricultural activity system where learning could be explored using Cultural Historical Analysis Theory (CHAT) (Roth and Lee, 2007).

4.3.3 Importance of transdisciplinary research

Chapter 3 gave a deeper understanding of the ecological aspects of wetland contestation in the X11B C-SES. Based on McCarthy and Humphries's (2013) claim of coal mining being the main culprit of wetland contamination in the X11B quaternary catchment, in Chapter 3, water chemistry variables were reported: heavy metals, ions, nutrients and trace elements. Macroinvertebrates were also identified to a family level and statistically analysed in order to identify any links to water chemistry impacts on abundance and diversity. Although distinct abundance and diversity measures according to different land-uses were expected, the abundance results did not resemble any associated clear patterns based on the types of macroinvertebrates present at the different sites. South African Scoring System 5 (SASS5) scores and diversity, on the other hand, did reflect a difference, where sites associated with high mining use had lower macroinvertebrate scores and diversity than other sites.

Deeper contextual reading revealed that geology plays a critical role in the chemical composition of wetlands and macroinvertebrate community structure, and more specifically, the hydrological flows of groundwater on which most wetlands depend (Jolly et al., 2008). A related study (which, in conjunction with this study, forms a greater transdisciplinary project) indicated the severe overall impact open-pit coal mining has on the landscape's hydrology, especially after poor rehabilitation of defunct mines (van der Waals, 2016b). In his study, van der Waals (2016b) elaborates the implications and importance of soil stripping, stockpiling and placement execution pre-mining and post-mining, as well as the near impossible rehabilitation of the landscape's hydrology functioning (i.e., flow regimes and wetland feeding processes after puncturing the landscape).

van der Waals's (2016b) study gives an in-depth critique of the definitions (e.g., the National Water Act's definition of wetlands) and tools of wetland assessment and characterisation (e.g., Department of Water Affairs and Forestry (2005) manual "A practical field procedure for identification and delineation of wetland and riparian areas" and the Present Ecological State (PES) method of the "Resource Directed Measures for protection of water resources: volume 4: wetland ecosystems") used for wetland protection based on wetland boundaries and measurable properties. van der Waals (2016b) argues that existing tools are mostly restricted to the: delineation of the outer boundary, the assessment of the ecological status, and the assessment of related services of wetlands, thus neglecting the more complex hydrological parameters that ultimately sustain wetlands. He argues that wetlands should be viewed more holistically within their supporting landscape, and hydrological processes such as headwaters and feeding areas throughout the landscape that supply water to the ecosystems should receive more attention. He uses these points to highlight the flaws of wetland delineation in the mining authorisation process and the need for new policy on defining and dealing with wetlands, especially with regard to land-use.

van der Waals' (2016b) study, in combination with this thesis, highlights the limitations of conventional disciplinary approaches. A transdisciplinary approach offers concurrent knowledge-building across disciplines (Audouin et al., 2013; Biggs et al., 2015). The ecological approach used in this thesis provides insights into wetland health, service provision, and macroinvertebrate assemblages. However, the thesis gives a convincing example of how one cannot rely only on observable wetland traits and ecologically specific information in order to resolve conflict with regard to coal. Although ecological understanding is required in order to understand the full complexity of the catchment's contestation, investigation and engagement-based knowledge is also required (Audouin et al., 2013).

Understanding the interconnectedness of drivers and feedbacks of all components is vital for full protection and better mitigation efforts of the ecology and relationship of C-SESs components. This project highlights the importance of a transdisciplinary approach when dealing with C-SESs and land-use. Specific assessments are needed to identify and pinpoint impacts. As catchment systems are largely influenced by social elements, the need for data on social (law, economics, political ecology and social engagement) and ecological (ecology, chemistry and geology) components to understand and conserve ecological elements is important for the balance of coal mining and wetland protection (Ison et al., 2007; Audouin et al., 2013). The results of this thesis looked at the core ecological knowledge placed in a rich context of social and geological understanding, and provided contextual background and some indication of ecological and system integrity status. However, without a more elaborate understanding of how people, land-uses, and ecosystem services are connected by water (the social-ecological hydro-connectivity), supplied by van der Waals's hydro-pedology study, a more in-depth social level of understanding would not be possible. Basically the "geological knowledge" is a framework or background knowledge that allows stakeholders to predict, discuss, contest, and manage (make decisions together) coal mining, on the basis of how the water flows are connected in the landscape. Overall, studies concluded that, in the long term, mining impacts are worse than agriculture, and there is a lack of streamlined licensing and other regulatory procedures, a lack of fair negotiation, and of participation in decision-making by catchment residents.

4.4 Conclusion

With the conclusions made by van der Waals' study (2016b) along with findings of this thesis, it is clear that there is limited capacity among stakeholder to engage with the complexity and resilience characteristic of C-SESs. Thus, it is recommended that, in order to protect wetlands and ecological infrastructure, conservation governmental departments need to use platforms such as established forums as a means to enhance: the complexity; resilience and resilience thinking within social components of the C-SES. By collaboratively focusing on how the resource dilemma experienced in the catchment arose, understanding the thresholds present in the system, evaluating the driving factors, strengthening links between social hierarchy scales, and knowing what feedbacks are likely, researchers and stakeholders are more likely able to revive a system in the case of external shocks. In conjunction with this, conservation authorities should map the hydro-connectivity of wetlands that remain intact and use that for the delineation for aquatic ecosystem health classification. The present mapping of wetlands based on defined boundaries that do not consider the full connectivity of a landscape will not offer protection of wetland ecosystems. In terms of the X11B catchment, the hydrology is highly connected and most mine sites are located at the sources/headwaters of the wetland, springs and streams, thus having major implications for the long-term functioning of wetlands, should restoration not be carried out effectively.

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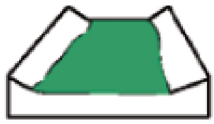
Appendices

Appendix A: Wetland hydrogeomorphic (HGM) types typically supporting inland wetlands in South Africa (Kotze et al., 2009)

Hydrogeomorphic types		Description	Source of water maintaining the wetland[1]	
			Surface	Sub-surface
Floodplain		Valley bottom areas with a well defined stream channel, gently sloped and characterized by floodplain features such as oxbow depressions and natural levees and the alluvial (by water) transport and deposition of sediment, usually leading to a net accumulation of sediment. Water inputs from main channel (when channel banks overspill) and from adjacent slopes.	***	*
Valley bottom with a channel		Valley bottom areas with a well defined stream channel but lacking characteristic floodplain features. May be gently sloped and characterized by the net accumulation of alluvial deposits or may have steeper slopes and be characterized by the net loss of sediment. Water inputs from main channel (when channel banks overspill) and from adjacent slopes.	***	*/ ***
Valley bottom without a channel		Valley bottom areas with no clearly defined stream channel, usually gently sloped and characterized by alluvial sediment deposition, generally leading to a net accumulation of sediment. Water inputs mainly from channel entering the wetland and also from adjacent slopes.	***	*/ ***
Hillslope seepage linked to a stream channel		Slopes on hillsides, which are characterized by the colluvial (transported by gravity) movement of materials. Water inputs are mainly from sub-surface flow and outflow is usually via a well defined stream channel connecting the area directly to a stream channel.	*	***
Isolated Hillslope seepage		Slopes on hillsides, which are characterized by the colluvial (transported by gravity) movement of materials. Water inputs mainly from sub-surface flow and outflow either very limited or through diffuse sub-surface and/or surface flow but with no direct surface water connection to a stream channel.	*	***
Depression (includes Pans)		A basin shaped area with a closed elevation contour that allows for the accumulation of surface water (i.e. it is inward draining). It may also receive sub-surface water. An outlet is usually absent, and therefore this type is usually isolated from the stream channel network.	*/ ***	*/ ***

[1] Precipitation is an important water source and evapotranspiration an important output in all of the above settings

Water source: * Contribution usually small

*** Contribution usually large — Wetland

*/ *** Contribution may be small or important depending on the local circumstances

*/ *** Contribution may be small or important depending on the local circumstances.

Appendix B: Table of expert team information

Specialist	Qualification	Experience of related use
Dr Anthony Palmer	Ph.D. Botany	• Nature Conservation Scientist, Cape Department of Nature and Environmental Conservation • Senior Researcher, South African National Botanical Institute • Specialist Scientist, Agricultural Research Council-Animal Production Institute, Grahamstown, South Africa • Research Co-ordinator, Department of Environmental Sciences, University of Technology Sydney, Sydney, NSW, Australia • Senior Specialist Scientist, ARC-Animal Production Institute, Grahamstown, South Africa
Prof. Tally Palmer	Ph.D. Ecotoxicology	• Professor and Director at Unilever Centre for Environmental Water Quality, Institute for Water Research, Rhodes University, Grahamstown, South Africa • Executive Director or Applied Research and Innovation at National Research Foundation, South Africa • Professor of Water Resources; Director, Institute for Water and Environmental Resource Management (IWERM); and Director, Centre for Ecotoxicology, University of Technology, Sydney (UTS)
Notiswa Libala	M.SC agriculture (Rangeland science)	Training in agricultural Rangeland vegetation identification
Tia Keighley	B.SC. (Hons.) Environmental Sciences	Training in WET-Health and WET-EcoServices methodology

Appendix C: Major ions, environmental variables and trace elements collected at each wetland including upstream and downstream sites

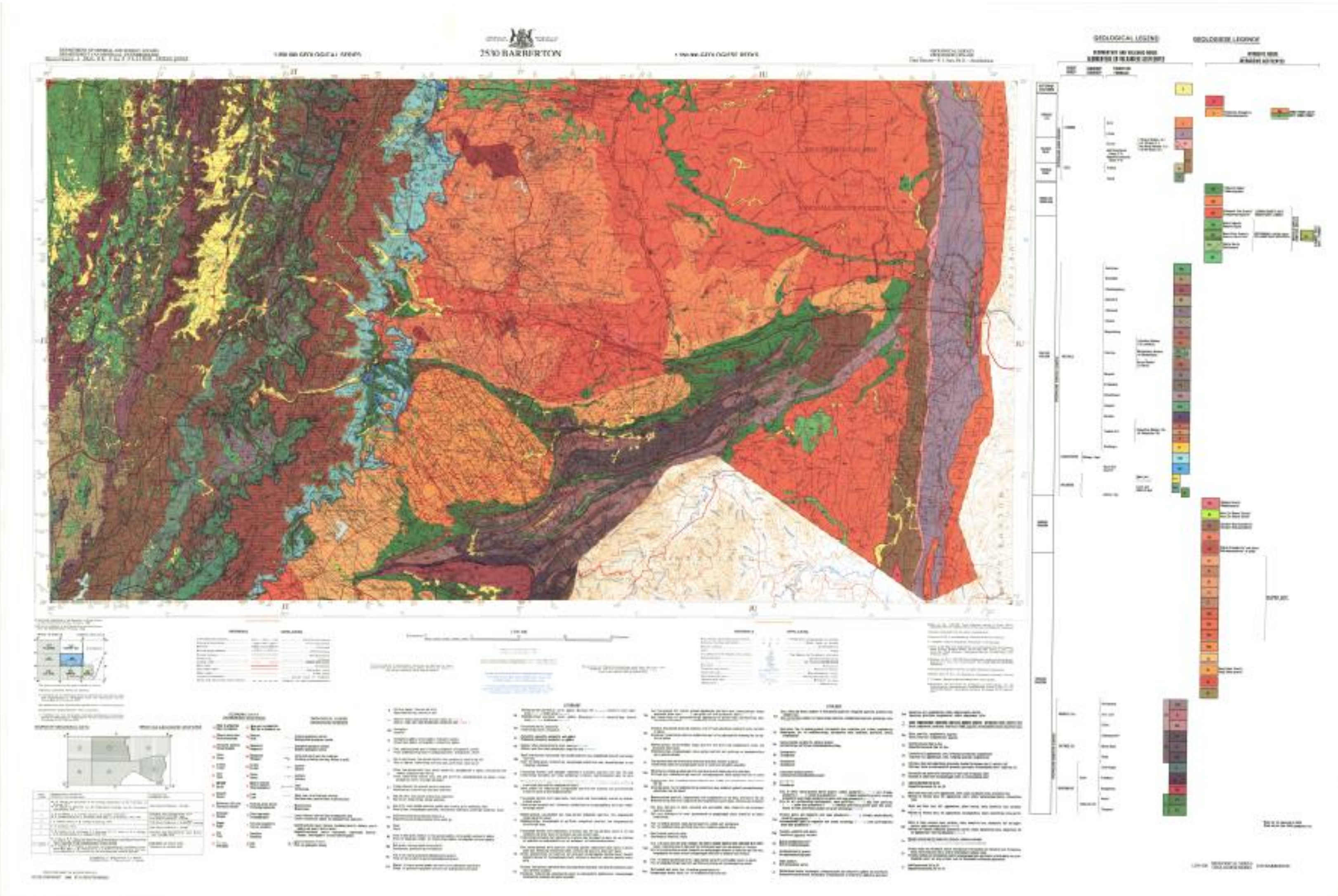
Water quality variable	**Sampling sites**																						
Sites	1u	1a	1d	2ru	2u	2a	2d	3u	3a	3d	4u	4a	4d	5u	5a	5d	6u1	6u2	6a	6d	7ma	7au	7a	7d
pH	6.72	7.4	7.34	6.61	5.04	6.84	6.57	5.27	5.4	5.4	6.47	6.67	6.74	6.37	6.75	6.74	5.2	5.28	5.54	5.24	4.1	5.27	5.46	4.55
pHs	8.44	7.62	7.74	8.26	9.38	8.14	8.79	9.07	9.36	9.33	7.85	7.78	7.82	8.58	8.41	8.56	9.63	9.69	9.39	9.76	8.83	9.71	9.68	8.96
SAR	0.43	0.42	0.44	0.31	0.7	0.34	0.41	0.6	0.59	0.6	0.24	0.34	0.33	0.75	0.72	0.75	0.63	0.57	0.54	0.56	0.41	0.34	0.44	0.7
Electric Conductivity (mS/m)	15	40	40	28	19	31	14	14	13	14	10	9	9	24	25	24	10	6	8	6	8	6	6	52
Anions (mg/l)																								
Fluoride	0.14	0.13	0.15	0.21	0.15	0.8	0.19	0.12	0.1	0.08	0.25	0.32	0.28	0.41	0.24	0.22	0.1	0.15	0.17	0.1	0.33	0.11	0.15	1.21
Nitrite	0	0.2	0	0.65	0	0	0	0	0	0.09	0.06	0.82	0.53	0.35	0.36	0.7	0	0	0	0	0	0	0	0
Nitrate	0.77	0.56	0.99	1.25	1	1.47	0.53	1.73	2.33	3.49	1.27	0.76	1.92	0.21	0.5	0.58	0.83	0.97	0.55	0.76	2.94	1.09	0.93	0.69
Chloride	10.9	15.69	18.25	5.55	23.25	7.09	7.39	9.82	7.96	9.25	6.69	6.3	5.67	7.58	7.92	7.54	9.05	6.71	6.18	4.99	2.8	3.6	4.36	6.48
Sulphate	7.01	30.28	32.96	84.26	83.84	90.91	35.92	50.63	53.58	58.7	507.12	1.06	517.36	62.56	57.28	64.13	43.56	13.48	19.51	18.1	436.42	14.25	12.22	274.62
Phosphate	0	0.04	0.34	0.42	0.52	0.06	0	0.52	0.69	0	3.06	2.57	1.27	0	0.23	0.07	0	0.35	0.03	0.91	3.33	0.91	0.5	2.49
Carbonate	0	0	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Bicarbonate	76.25	197.64	164.7	67.71	10.37	82.35	38.43	29.79	12.2	13.42	41.48	56.73	57.95	62.22	77.47	62.83	7.93	13.42	17.69	9.76	9.76	12.81	13.42	10.98
Sodium Carbonate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sodium Bicarbonate	105	272.16	226.8	93.224	14.28	113.4	52.92	0	16.8	18.48	57.12	78.12	79.8	85.68	106.68	86.52	10.92	18.48	24.36	13.44	13.44	17.64	18.48	1171.38
Alkalinity	62.5	162	135	55.5	8.5	67.5	31.5	19.5	10	11	34	46.5	47.5	51	63.5	51.5	6.5	11	14.5	8	8	10.5	11	9
Temp. Hardness	62.5	162	135	55.5	8.5	67.5	31.5	19.5	10	11	34	46.5	47.5	51	63.5	51.5	6.5	11	14.5	8	8	10.5	11	9
Perm. Hardness	15.41	45.48	47	70.31	54.17	71.54	30.92	32.16	42.48	40.53	514.07	450.3	446.64	42.78	40.35	41.83	34.77	9.19	15.03	13.53	330.98	9.94	8.74	193.75
Cations (mg/l)																								
Sodium	8.72	14.04	13.71	8.02	12.79	9.3	7.47	9.91	9.81	9.95	13.17	17.39	17.05	16.8	16.84	16.61	9.36	5.87	6.69	5.98	17.4	3.53	4.46	22.84
Potassium	1	0.86	1.32	2.22	9.05	2.52	2.11	3.75	3.77	3.81	2.05	2.96	2.94	1.34	1.551	1.03	2.41	2.14	1.96	1.72	5.65	2.29	1.83	7.07
Calcium	13.97	39.34	35.73	25.73	12.59	27.76	12.51	10.64	10.72	10.54	122.56	94.36	93.85	12.95	15.43	13.46	8.59	4.14	6.38	4.92	54.28	4.09	4.24	34.18
Magnesium	10.42	26.54	22.53	14.92	7.52	16.87	7.54	6.06	6.01	6.09	58.74	62.81	62.44	14.9	15.84	14.48	4.78	2.36	3.27	2.21	49.37	2.45	2.19	28.46
Boron	0.01	0	0.01	0.01	0.02	0.02	0	0.01	0.07	0.01	0.01	0.19	0.2	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.01	0.01	0.02
TDs	90.88	226.18	207.65	176.15	155.82	197.83	92.79	104.11	100.9	108.29	735.26	217.64	732.03	147.89	154.27	149.59	82.04	42.29	53.16	44.12	577.12	38.6	37.29	383.51

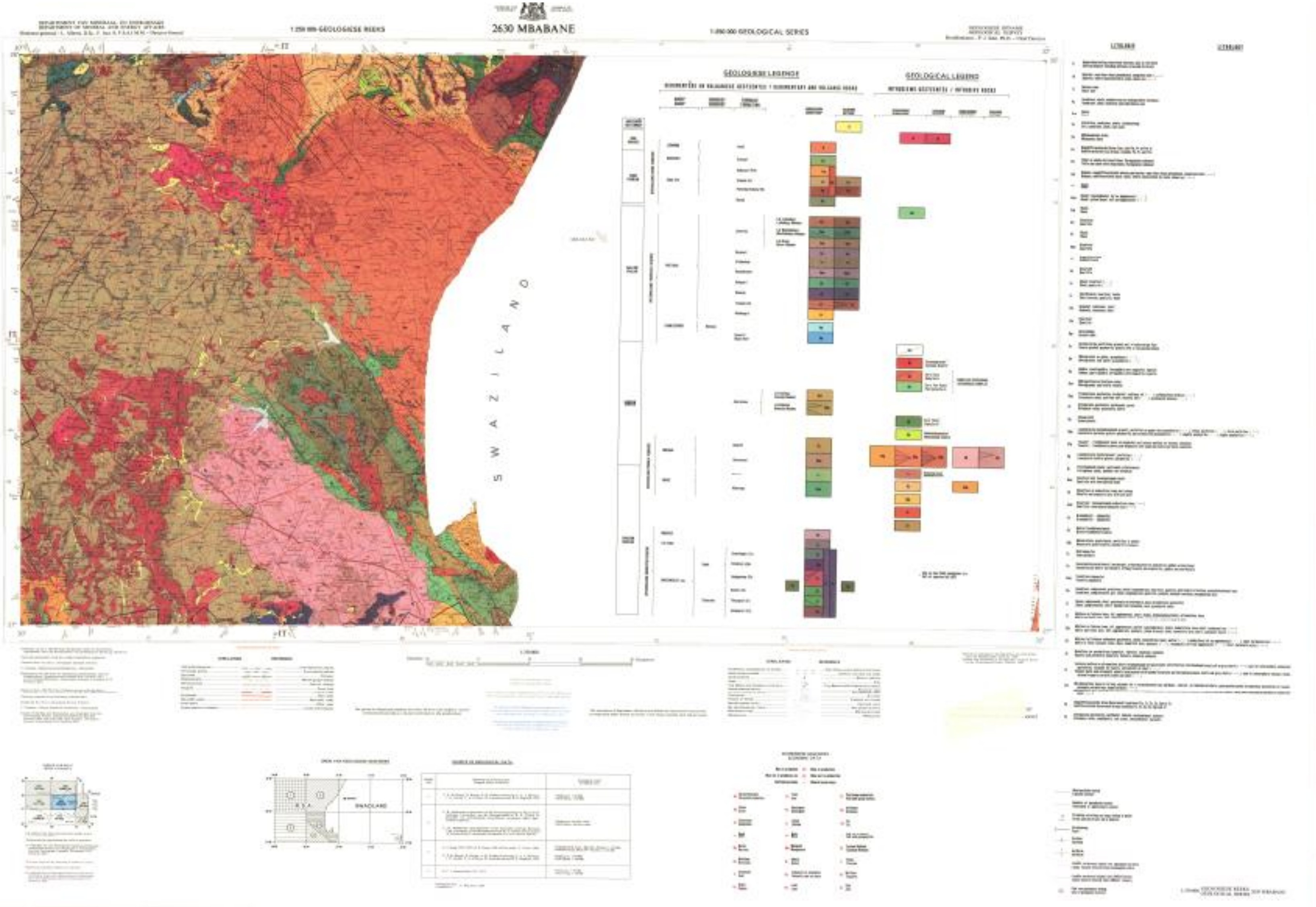
Trace element	Sampling sites																						
	1u	2ru	2u	3u	4u	5u	6u1	6u2	7u	1d	2d	3d	4d	5d	6d	7d	1a	2a	3a	4a	5a	6a	7ma	7aa
7 Li	0.36	0.20	0.88	2.12	3.98	1.15	0.77	0.52	0.69	0.26	0.65	1.80	0.78	0.93	0.65	2.95	0.26	0.44	2.04	1.07	0.78	0.98	2.78	1.03
9 Be	0.06	0.02	0.18	0.10	0.34	0.03	0.13	0.05	0.06	<0.01	0.09	0.07	0.02	0.03	0.06	0.68	0.01	0.02	0.07	0.05	0.01	0.05	1.23	0.2
Ti	35.85	<0.35	1.21	5.29	3.54	29.89	9.23	16.31	2.04	16.07	37.34	4.72	0.72	14.90	19.81	3.54	31.18	0.35	5.11	0.62	11.17	24.97	0.35	10.09
V	2.74	0.03	0.25	0.14	0.08	0.45	0.34	0.30	<0.01	2.04	0.52	0.08	<0.01	0.48	0.42	0.08	3.04	0.02	0.15	0.01	0.53	0.58	0.01	0.19
Cr	1.48	0.06	0.13	0.27	0.07	1.46	0.59	0.47	0.06	0.35	0.55	0.22	0.09	0.68	0.72	0.21	0.34	0.08	0.31	0.10	0.29	0.81	0.05	0.27
Mn	243.64	5.20	1305.84	27.22	295.69	0.64	5.65	1.28	1.23	1.37	5.33	4.88	19.23	2.06	2.25	1792.73	4.06	0.76	227.10	34.45	0.45	1.17	906.13	5.26
Co	0.91	0.08	5.09	0.12	0.59	0.11	0.10	0.06	0.05	0.15	0.11	0.07	0.17	0.12	0.13	7.20	0.11	0.22	0.47	0.30	0.06	0.23	7.33	0.13
Ni	6.72	3.67	4.38	3.73	40.19	4.51	3.80	1.36	0.82	3.96	2.07	1.90	4.77	4.39	4.13	13.14	2.44	1.56	8.78	9.52	1.62	5.32	31.22	2.33
Cu	5.98	0.82	0.82	3.74	4.85	2.57	2.04	2.86	0.48	4.37	1.73	1.92	2.22	3.17	2.62	3.17	3.48	1.8	2.85	2.74	2.51	3.95	2.9	1.3
Zn	2.69	4.44	21.33	19.30	60.26	13.88	9.56	19.32	4.79	<0.05	1.07	81.22	5.73	21.37	9.58	89.01	0.82	0.05	11.88	13.42	10.47	12.48	73.74	2.95
As	0.78	0.26	0.59	0.16	0.28	0.30	0.45	0.21	0.22	0.22	0.25	0.23	0.37	0.28	0.38	0.27	0.33	0.19	0.21	0.32	0.44	0.31	0.16	0.43
Se	<0.02	<0.02	<0.02	<0.02	<0.02	2.57	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	2.07	0.02	0.02	0.02	0.02	2.63	0.02	0.02	3.75
Rb	1.02	2.19	10.66	5.80	3.00	1.47	2.42	2.31	1.49	1.41	2.52	6.07	2.19	0.99	3.39	8.01	0.92	2.71	5.73	2.18	0.81	2.16	4.26	2.1
Sr	11.20	217.69	23.39	18.48	818.23	15.05	15.69	6.84	9.41	17.37	26.48	16.69	1735.45	16.33	7.90	322.67	19.12	248.3	17.66	1731.25	17.89	10.24	461.13	11.43

Mo	0.08	0.24	0.10	<0.04	<0.04	0.14	0.14	0.17	0.38	0.04	0.07	<0.04	0.27	0.09	0.09	0.18	0.05	0.25	0.08	0.28	0.05	0.1	0.27	0.71
Cd	0.04	0.03	0.03	<0.01	0.10	<0.01	0.04	<0.01	<0.01	<0.01	<0.01	0.02	0.01	0.02	0.70	0.13	0.01	0.01	0.10	0.03	0.05	0.03	0.13	0.2
Sn	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	<0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Sb	0.02	0.01	0.01	0.01	0.01	0.04	0.01	0.02	0.02	0.01	<0.005	0.02	0.02	0.04	0.01	0.01	0.01	0.01	0.02	0.02	0.04	0.03	0.01	0.06
Te	0.00	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.09	0.00	0.18	0.00	0.17	0.09	0.00	0	0.17	0.00	0.00	0	0.17	0
Cs	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	0.00	0.004	0.00	0.00	0.00	0.004	0.004	0.004
Ba	4.15	13.21	29.51	26.64	<0.08	<0.08	31.07	2.15	10.02	<0.08	9.53	24.37	<0.08	<0.08	<0.08	50.82	0.08	8.29	30.17	<.08	0.08	0.08	5.95	5.59
La	0.25	0.08	0.07	0.06	0.80	0.08	0.28	0.08	0.02	0.02	0.23	0.07	0.03	0.06	0.11	0.28	0.04	0.01	0.07	0.05	0.03	0.1	1.09	0.09
W	<0.010	0.07	0.04	0.02	<0.010	0.06	0.12	0.10	0.29	<0.010	0.03	<0.010	<0.010	0.01	0.05	0.18	0.01	0.06	0.02	0.01	0.02	0.1	0.13	0.33
Pt	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.03
Hg	0.12	0.21	<0.08	<0.08	<0.08	0.41	0.35	0.28	0.64	0.77	<0.08	<0.08	<0.08	<0.08	0.17	0.39	0.50	1.12	0.21	0.08	0.08	0.46	0.4	0.93
Tl	<0.01	<0.01	0.02	0.01	0.01	<0.01	0.01	<0.01	0.01	<0.01	<0.01	0.01	<0.01	<0.01	<0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.05
Pb	0.12	0.02	0.03	0.03	0.05	0.05	0.03	0.04	<0.004	0.02	0.06	0.03	<0.004	0.06	0.04	0.04	0.05	0.01	0.06	0.01	0.03	0.06	0.04	0.06
Bi	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
U	0.01	0.01	0.01	<0.01	0.01	0.01	0.01	0.01	<0.01	0.02	0.01	<0.01	0.01	<0.01	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.03

Appendix D: 2530 Barberton 1:250 00 geological map



Appendix E: 2630 Mbabane 1:250 00 geological map



Appendix F: Winter season SASS5 abundance ratings per macroinvertebrate group (Dicken and Graham, 2002) for each wetland sampled in the X11B catchment. Abundance data: A=2-10, B= 10-100

Family taxa	**Sensitivity**	**1a**	**2a**	**3a**	**4a**	**5a**	**6a**
Oligochaeta	1	A					A
Culicidae	1					1	
Psychodidae	1						
Chironomidae	2			A	A	B	B
Potamonautidae	3						1
Corixidae	3		A	A	A		
Nepidae	3			1			
Notonectidae	3		A			C	B
Physidae	3		A				
Planorbinae	3						
Belostomatidae	3						
Lymnaeidae	3						
Thiaridae	3						
Sphaeriidae	3						
Baetidae 1 sp	4						A
Coenagrionidae	4	A				B	B
Libellulidae	4			A			A
Pleidae	4		B		B		C
Veliidae	5	1					
Dytiscidae	5			B	1	A	
Gyrinidae	5	B					
Hydrophilidae	5	A	A	C	B		
Ceratopogonidae	5	A				A	A
Simuliidae	5	C					
Gerridae	5			1			
Hydropsychidae 1 sp	5						
Corbiculidae	5						
Baetidae 2 sp	6	B		B		A	
Caenidae	6	B					
Gomphidae	6						
Ancylidae	6						
Naucoridae	7	B			1		1
Hydracarina	8			1		B	A
Synlestidae	8	A					
Lestidae	8					B	
Aeshnidae	8	A					

Hyraenidae	8	B				B	
Protoneuridae	8						
Ecnomidae	8						
Elmidae	8						
Leptophlebiidae	9	1					
Platycnemidae	10						
Baetidae >2 sp	12						

Appendix G: Summer season SASS5 abundance ratings per macroinvertebrate group (Dicken and Graham, 2002) for each wetland sampled in the X11B catchment. Abundance data: A=2-10, B= 10-100

Family taxa	Sensitivity	1a	2a	3a	4a	5a	6a	7ma	7aa
Oligochaeta	1			1					
Culicidae	1			A		A			
Psychodidae	1								
Chironomidae	2				1				
Potamonautidae	3	A				1	1		
Belostomatidae	3								
Corixidae	3		A		A		1		
Nepidae	3								
Notonectidae	3	A		A		A			A
Lymnaeidae	3								
Physidae	3		A						1
Planorbinae	3					1	1		
Thiaridae	3								
Sphaeriidae	3								
Baetidae 1 sp	4	A	A	1		A			
Coenagrionidae	4			1		1			
Libellulidae	4			1		1			
Pleidae	4								
Hydropsychidae 1 sp	4								
Gerridae	5			A					
Veliidae	5			1		A			
Dytiscidae	5			A	A	A	A	A	A
Gyrinidae	5	1	1				1		
Hydrophilidae	5					A		A	
Ceratopogonidae	5		1						
Simuliidae	5								
Corbiculidae	5								
Baetidae 2 sp	6								
Caenidae	6								
Gomphidae	6								
Ancylidae	6				1				
Naucoridae	7						1		
Hydracarina	8		1			A			
Synlestidae	8								
Lestidae	8					1			
Protoneuridae	8	A							
Aeshnidae	8			A					1

Ecnomidae	8								
Elmidae	8						1		
Hyraenidae	8					A			
Leptophlebiidae	9								
Platycnemidae	10								
Baetidae >2 sp	12					A			