ECOLOGICAL INFRASTRUCTURE IMPORTANCE FOR DROUGHT MITIGATION IN RURAL SOUTH AFRICAN CATCHMENTS: THE CACADU CATCHMENT CASE EXAMPLE

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Declaration

I, Sinetemba Xoxo, certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University. The thesis was part of the K5/2928 project, and other versions of the results have been submitted to the Water Research Commission for project reporting. Moreover, since the project was part of a large collaboration, the approach used herein was also applied to a different setting (Tsitsa catchment) by another masters student. Therefore, overlaps in methodology may exist between this thesis and that of Miss Mahlaba.

B. Sinetemba Xoxo

Abstract

Water scarcity is recognised as one of the significant challenges facing many countries, including South Africa. The threat of water scarcity is exacerbated by the coupled impacts of climate and anthropogenic drivers. Ongoing droughts and continued land cover change and degradation influence the ability of catchments to partition rainwater runoff, thereby affecting streamflow returns. However, quantifying land degradation accurately remains a challenge. This thesis used the theoretical lens of investing in ecological infrastructure to improve the drought mitigation function in rural catchments. This theoretical framework allows for a social-ecological systems approach to understand and facilitate science-based strategies for promoting ecosystem recovery. Specifically, this study aimed to explore the role and benefit of ecological infrastructure for improving drought mitigation, and consequently, water security for rural communities. Thus, this study sought to assess the consequences of human actions to catchment health status using the 15th Sustainable Development Goal indicator for the proportion of degraded land over the total land area as a surrogate. Secondly, hydrological modelling was used to describe how different land covers influence catchment hydrology, which related to how ecological infrastructure enables drought risk-reduction for mitigation regulation. Finally, this study developed a spatial prioritisation plan for restoration to improve drought mitigation for four focal ecological infrastructure (EI) categories (i.e. wetlands, riparian margins, abandoned agricultural fields and grasslands). The focal EI categories were selected for their importance in delivering water-related ecosystem services when sustainably managed.

Chapter 1 sets the scene (i.e. provides the study background) and Chapter 2 provides a review of the literature. In Chapter 3, the recently released global GIS toolbox (TRENDS.EARTH) was used for tracking land change and for assessing the SDG 15.3.1 degradation indicator of i.e. Cacadu catchment over 15 years at a 300 m resolution. The results showed a declining trend in biomass productivity within the Cacadu catchment led to moderate degradation, with 16.79% of the total landscape degraded, which was determined by the pugin using the one-out, all-out rule. The incidence of degradation was detected in middle reaches of the catchment (i.e. S10F-J), while some improvement was detected in upper reaches (S10A-C) and lower reaches (S10J). In Chapter 4, a GIS-based Analytic Hierarchical Process (AHP) based on community stakeholder priorities, open-access spatial datasets and expert opinions, was used to identify EI focal areas that are best suitable for restoration to increase the drought mitigation capacity of the Cacadu catchment. The collected datasets provided three broad criteria (ecosystem health, water provision and social benefit) for establishing the AHP model using 12 spatial attributes. Prioritisation results show that up to 89% of the Cacadu catchment is suitable for restoration to improve drought mitigation. Catchments S10B-D, and S10F, S10G

and S10J were highly prioritised while S10A, S10E and S10H received low priority, due to improving environmental conditions and low hydrological potential. Areas that were prioritised with consideration for local livelihoods overlap the areas for drought mitigation and form a network of villages from the middle to lower catchment reaches. Prioritised restoration areas with a consideration of societal benefit made up 0.56% of wetlands, 4.27% of riparian margins, 92.06% of abandoned croplands, and 51.86% of grasslands. Chapter 5 reports on use of the Pitman groundwater model to help understand the influence of land modification on catchment hydrology, and highlight the role of restoration interventions. The Cacadu catchment is ungauged, therefore the neighbouring Indwe catchment was used for parameter transfer through a spatial regionalisation technique. Results suggest that degradation increases surface runoff and aggravates recharge reduction, thereby reducing streamflow during low flow periods. In areas where there is natural land cover recovery, the Pitman Model simulated similar dry season streamflow to the natural land cover.

Combining the outcomes from the three assessments allowed the study to highlight the role and benefits of ecological infrastructure in terms of drought mitigation. Study findings were interpreted to make recommendations for the role and benefit of ecological infrastructure for drought mitigation at a landscape scale and tertiary catchment level, within the context of available management options. The results support the notion that multiple science data sources can promote investments in ecological infrastructure. However, better spatial and temporal resolution datasets at a national level are still needed to improve the accuracy of studies such as the one outlined in this thesis. The study recommends adopting better ecosystem protection approaches and collaborative governance at multiple levels to reduce the vulnerability of rural communities to drought impacts.

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List of abbreviations

AHP	Analytic Hierarchical Process
CBD	Convention on Biological Diversity
CCI-LC	Climate Change Initiative-Land Cover
CHIRPS	Climate Hazards Group Infrared Precipitation with Stations
CSIR	Council for Scientific and Industrial Research
DEA	Department of Environmental Affairs
DEM	Digital Elevation Model
DWAF	Department of Water Affairs & Forestry
DWS	Department of Water and Sanitation
EI	Ecological Infrastructure
ESA	European Space Agency
FAO	Food and Agriculture Organisation of the United Nations
GIDMAPS	Global Integrated Drought Monitoring and Prediction System
GTI	GEOTERRAIMAGE
GWP	Global Water Partnership
IAP	Invasive Alien Plants
IPCC	Intergovernmental Panel on Climate Change
GEF	Global Environment Facility
MAR	Mean Annual Runoff
MEA	Millennium Ecosystem Assessment
NBA	National Biodiversity Assessment
NDP	National Development Plan
NDVI	Normalised Difference Vegetation Index
NTCSA	National Terrestrial Carbon Sinks Assessment
OOAO	One-out, All-out
PDSI	Palmer Drought Severity Index
PGIS	Participatory Geographic Information System

RESTREND	Residual Trend Analysis
RUE	Rain Use Efficiency
RUESC	Rhodes University Ethics Standards Committee
SDG	Sustainable Development Goal
SLM	Sustainable Land Management
SPI	Standardised Precipitation Index
SANBI	South African National Biodiversity Institute
SANLC	South African National Land Cover
SAPIA	Southern African Plant Invaders Atlas
StatsSA	Statistics South Africa
SPATSIM	Spatial and Time Series Information Modelling
SRTM	Shuttle Radar Topography Mission
TRMM	Tropical Rainfall Measuring Mission
UN	United Nations
UNCCD	United Nations Convention to Combat Desertification
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNSTATS	United Nations Statistical Division
WMO	World Meteorological Organisation
WR	Water Resources South Africa
WUEe	Ecosystem Water-use Efficiency

1. CHAPTER 1: GENERAL INTRODUCTION

1.1. Background

Droughts and water scarcity are recognised as major challenges in arid to semi-arid countries, including South Africa, threatening over 2 billion people globally (IPCC, 2014; Mekonnen & Hoekstra, 2016). Water scarcity could be attributed to low average rainfall that leads to shortages in freshwater availability, which is intensified by land degradation because of its negative effect on water flow regulation; moreover it could be due to poor water governance that could lead to inadequate water supply (UN Water, 2015; Mekonnen & Hoekstra, 2016; Veldkamp et al., 2017). UN Water (2015) noted with concern the added pressure of unsustainable agriculture and overexploitation of groundwater reserves as a threat to future freshwater supplies. Burek et al. (2016) revealed that close to 70% of the global water budget is allocated for irrigation, while 20% goes to industry and 10% to domestic needs, emphasising the concern raised by UN Water (2015). Global water budgets for industry and domestic water needs are projected to increase by 10-24% by 2050 (Burek et al., 2016). Kummu et al. (2016) found that the quadruple increase in regional population between 1900 and early 2000s accompanied a 16-fold increase in the southern African population that is vulnerable to water scarcity (i.e. an increase from 0.24 billion people in 1900s to 3.8 billion people by early 2000s).

Regarding water security, droughts are the most notorious threats for water security in arid and semi-arid contexts (Clarke et al., 2012; Masih et al., 2014), but they are one of many environmental issues that threaten the wellbeing of global populations. For instance, climate change impacts and the land cover changes are pushing global biodiversity beyond its stable limits, which is anticipated to negatively impact human well-being (Foley et al., 2005; Rockström et al., 2009). The projections became a reality as the loss of plant biodiversity, and forest fragmentation in the Asian region has been associated with the insurgence of the COVID-19 pandemic (Platto et al., 2020). Thus, there is need for intervention on challenges related with freshwater shortages, including in developed countries, as discussed by Hanasaki et al. (2012) and targeted by the sixth Sustainable Development Goal (SDG) (United Nations, 2015). The United Nations SDGs are a set of global targets for guiding environmental protection, reducing inequality, and stimulate economic growth (United Nations, 2015). Within the SDGs, land management is related to SDG 15, and water security is related to SDG 6 (United Nations, 2015). In South Africa, the two SDGs are closely linked with the 2030 agenda under the National Development Plan (NDP) – a South African framework to address poverty and inequality (National Planning Commission, 2010; United Nations, 2015; Cumming et al., 2017).

1.2. Motivation for research

As natural resources get scarcer, future production and supply of ecosystem services are increasingly threatened; therefore a critical research need is to develop a better understanding of terrestrial resources (SANBI, 2014; Stavi & Lal, 2015). One major threat to terrestrial resources is land degradation, which is defined by the Food and Agriculture Organisation (FAO) of the United Nations as the reduced ability of ecosystems to perform their ecological functions (FAO, 2011a). The dominant drivers of land degradation (particularly in grasslands and savannahs) include climate variation (Bai et al., 2007; Liniger et al., 2019) and soil erosion (Rowntree et al., 2004; von Maltitz et al., 2019). Anthropogenic related land degradation includes invasive alien plants (IAP) (van Wilgen et al., 2008; Yapi et al., 2018) and woody encroachment (Ludwig et al., 2016; Luvuno et al., 2018). Declining land productivity performance (De Klerk et al., 2016; Graw et al., 2017; Smithwick, 2019), loss of soil organic carbon and fertility (Mills & Fey, 2003; Vågen et al., 2005, 2016), poor grazing management (Hein et al., 2011; Kotzé et al., 2013; Palmer & Bennett, 2013) and cropland abandonment (Benayas et al., 2007; Blair et al., 2018; Scorer et al., 2019) are also linked to human-driven land degradation. While unprecedented land continues to deteriorate natural resources, incomplete understanding of the state of natural resources affect the societal capacity to respond to the pressures mentioned above (Foley et al., 2005; Gibbs & Salmon, 2015; Easdale et al., 2018, 2019). These impacts are often pronounced for rural and farming communities who primarily depend on land for their socio-economic wellbeing (Shackleton et al., 2007; Sigwela et al., 2017; Murata et al., 2019; Pantshwa & Buschke, 2019).

Droughts are expected to intensify within the 21st century, implying that ecosystems are also expected to deteriorate further (Dai, 2011; Masih et al., 2014; Graw et al., 2017; Lehner et al., 2017). In response to the drought risk concerns, the United Nations subsidiaries promote the option of using integrated water resource management strategies as drought mitigation strategy and a sustainable ecosystem-based option that can lead to improved provision of other ecosystem services (UNCCD, 2016; WMO and GWP, 2016). By reducing the risks faced by ecosystems, the supply of ecosystem services can be sustained (De Groot et al., 2010; Dias et al., 2016; Le Maître et al., 2016) and social-ecological resiliency can be built (Folke, 2006; Chapin et al., 2010; UNCCD, 2016). Moreover, there is an interesting discussion about the extent to which ecosystems can be rehabilitated and the benefits thereof (Stavi & Lal, 2015; Cohen-Shacham et al., 2016; De Klerk et al., 2016), which has led to the ratification of the UN Decade for Ecosystem Restoration (United Nations, 2019). Ecosystem management approaches such as ecosystem-based adaptation (CBD, 2009; Munang et al., 2013) and nature-based solutions (Cohen-Shacham et al., 2016) have been mainly promoted as cost-effective approaches to deliver the benefits of reducing ecosystem degradation risk. The concept of ecosystem-based adaptation was conceptualised with a particular focus in mitigating and coping with climate change impacts (Munang et al., 2013), whereas nature-based solutions has focused on alleviating an array of social-ecological challenges that are not limited to climate change (Cohen-Shacham et al., 2016).

There has been an increased interest in ecosystem restoration to reduce the risk faced by terrestrial ecosystems due to climate and anthropogenic influences (FAO, 2011b; Keith et al., 2013; Stavi & Lal, 2015; Suding et al., 2015). Attempts to restore natural ecosystems is recommended for increasing the capacity of such systems to deal with external shocks, and therefore improve the integrity of such ecosystems to provide benefit (Chapin et al., 2010; Biggs et al., 2012; Folke et al., 2016; Gann et al., 2019). In practice, however, management of social-ecological systems is often surrounded with management challenges, and the desired outcomes are not often achieved (Bullock et al., 2011). For instance, in the late 1990s, South Africa embarked on a journey to recover native biodiversity, protect water resources and prevent further socio-economic losses from invasive alien plants (IAP) through the Working for Ecosystems programmes (van Wilgen et al., 2008). The Working for Water programme invested over R86 million on managing the spread of *Acacia sp.* within the Grassland biome, yet at present the distribution of *Acacia* sp*.* is still on the rise (van Wilgen et al., 2012b). The issue of poor ecosystem recovery post-intervention has also been raised in a workshop, where participants suggested the prioritisation of IAP clearing is necessary to effectively use available funds and increase programme effectiveness (van Wilgen et al., 2012a). Wetland restoration on the other hand has been hailed for its ability to promote integrity of wetland ecosystems in the long-term despite the high restoration costs (Rebelo et al., 2015; Kotze et al., 2019).

Since catchments are complex-social-ecological systems (Preiser et al., 2018), their complex-adaptive nature often makes it challenging for land managers to implement decisions that will keep catchments within desired conditions (Parrott & Meyer, 2012; van Wilgen et al., 2012a). This complexity and uncertainty challenge for land managers can be reduced through collaborations with researchers (Parrott & Meyer, 2012; Angelstam et al., 2017; Mahajan et al., 2019). Hence, this dissertation focuses on degraded semi-arid rural/farming landscapes to provide a decision-support system for prioritising restoration to improve the water retention, storage and release in drought periods, which is directly linked to water security (SDG 6.6, SDG 6b) and land management (SDG 15.3).

1.3. Conceptual framework: The ecological infrastructure investment approach

Motivated by the need for guiding actions and supporting sustainable development, different countries have adopted country-specific models, such as the concept of ecological infrastructure (EI) (Benedict & McMahon, 2006; European Commission, 2013; Kubiszewski et al., 2017; Silva & Wheeler, 2017). The EI concept was first derived for urban planning towards sustainable cities in 1984 (UNESCO, 1984) and was re-defined by various authors (European Commission, 2013; SANBI, 2014; Li et al., 2017; Silva & Wheeler, 2017). The South African National Biodiversity Institute (2014, p.3) defines EI as social-ecological systems that exist naturally, are in natural or semi-natural conditions, and are functional in that they can deliver ecosystem services as a way of supporting the country's economy, reduce inequalities and alleviate poverty by supporting the national and global development agenda. Elsewhere, Li et al. (2017: S13) define EI as "an organic integration of blue (water-based), green (vegetated), and grey (non-living) landscapes, combined with exits (outflows, treatment, or recycling) and arteries (corridors), at an ecosystem scale." Whereas, Silva & Wheeler (2017: p33) defined EI as connected natural, semi-natural and restored areas that are designed and managed at various spatial scales (from local to global), and contains all major types of ecosystems (terrestrial, freshwater and marine). Silva & Wheeler (2017: 33) go further and state that EI "aims to conserve biodiversity, mitigate emissions of greenhouse gases, enable societal adaptation to climate change, and deliver a wide range of other ecosystem services." The three definitions link with natural capital as a provider of ecosystems and EI as a suitable alternative for traditional built infrastructure (Li et al., 2014; SANBI, 2014; Silva & Wheeler, 2017). Although the different definitions of EI have their foundation in the notion that EI is a natural equivalent of traditional infrastructure (i.e. grey/built infrastructure), the definitions vary. For example, SANBI (2014) broadly defines EI for capturing the social, ecological and economic benefits of natural capital. In China, however, Li et al.'s (2017) framework focuses on EI for urban sustainability and captures different forms of physical capital, including human-made green infrastructure. Silva and Wheeler (2017) proposed a globally unifying EI definition, and they incorporated benefits from natural, altered and restored ecosystems under EI.

From the above definitions, land cover types can be deemed as EI if they are naturally occurring, self-sustaining and capable of providing a compound of benefits (Cohen-Shacham et al., 2016; Doko et al., 2016; Shackleton et al., 2017; Silva & Wheeler, 2017). Due to variations in different EI definitions because of context, this study adopts the SANBI (2014) definition for EI, which refers to EI as a self-sustaining ecosystem that can provide benefits to people. The definition is chosen for its applicability and relevance to the South African context, for water and climate risk reduction in particular (SANBI, 2014). Examples of EI in the context of the SANBI framework include mountain catchments, riparian areas, wetlands, grasslands, and coastal dunes (SANBI, 2014). Ecological infrastructure examples include but are not limited to, healthy mountain catchments, streams, wetlands, estuaries, and dunes (SANBI, 2014). The framework of investing in EI was established to foster and advocate understanding for maintaining and restoring environmental systems in South Africa - henceforth referred to as the SANBI framework (SANBI, 2014).

The SANBI framework provides the basis for improving human well-being by investing in EI through actions or monetary investments (Figure 1.1). Actions that count as an investment in EI comprise of restoring degraded lands, protecting landscapes from further degradation or new degradation, and avoiding degradation (Figure 1.1). These actions reflect those prescribed in the Land Degradation Neutrality (LDN) framework (Orr et al., 2017), thereby making the SANBI framework nationally and globally relevant (Cumming et al., 2017). The EI definition and approach in the South African context impact socio-economic profitability and risk reduction through promoting a reliable flow of ecosystem service for all communities across the catchments (SANBI, 2014). For example, with water being the most critical resource for social, economic, and ecological sustainability (FAO, 2011b), the SANBI framework promotes the ability of catchments to provide strong water regulation ecosystem service (SANBI, 2014), which translates to drought mitigation as discussed later. Sustained water yields enable the drought mitigation service through improved soil water retention, regulated base flow, and reduced sediment load in rivers (Figure 1.1). Therefore, conceptualising EI in terms of land cover and natural capital is essential because it impacts on catchment hydrology.

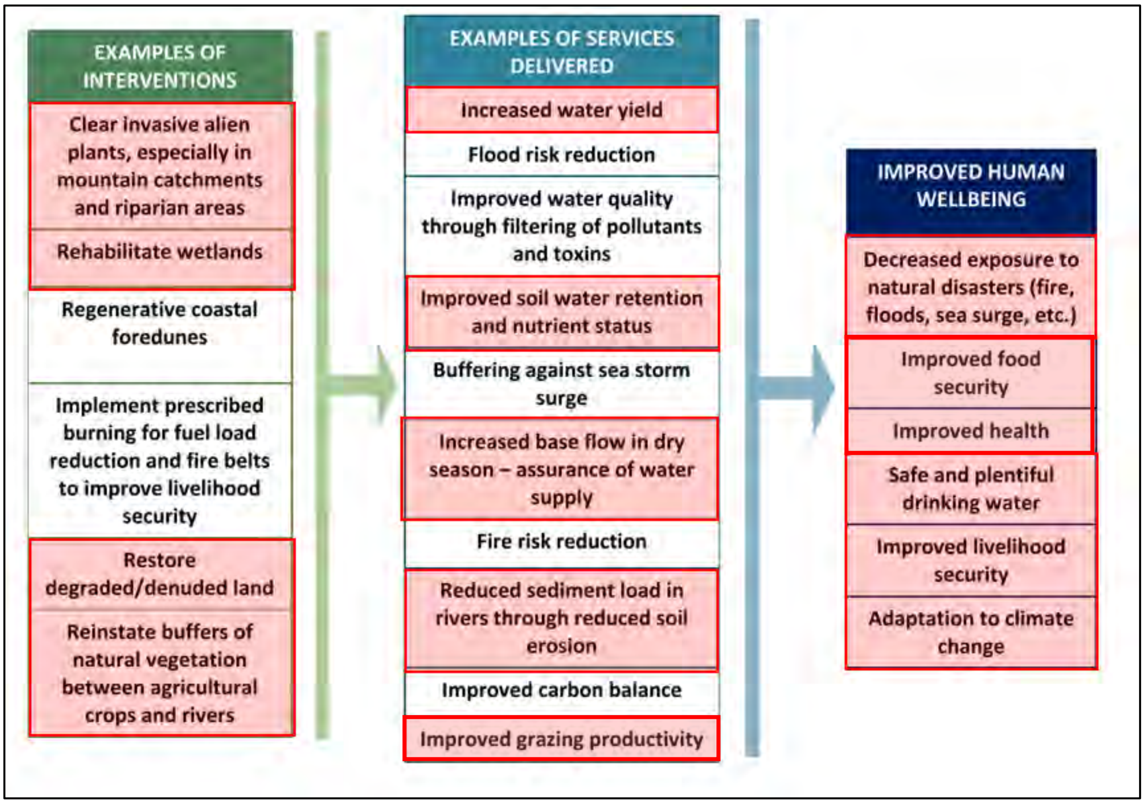

Figure 1.1: Tangible benefits obtained by society through EI investments (SANBI, 2014). The items in red boxes are related to the drought mitigation service.

Within the South African context, the SANBI framework has proved useful in several studies (Cumming et al., 2017; Mander et al., 2017) to provide an evidence base for securing investments in natural capital, and benefits of EI for disaster risk reduction and biodiversity protection, amongst others. The utility of the SANBI framework is based on the fact that the framework embodies the restoration of degraded ecosystems and maintaining ecosystem structures and functions (SANBI, 2014). Several approaches for investing in EI have been proposed under the framework including linking EI to land-use planning, restoring grassland and savannah ecosystems, rehabilitating wetlands, maintaining or restoring natural vegetation buffers in riparian margins, and improving landscape and grazing management. The focus on ecosystem management by the SANBI framework enhances EI investments as an application of the ecosystem-based adaptation and nature-based solution concepts (CBD, 2009; Cohen-Shacham et al., 2016). Therefore, the SANBI framework will be used as an appropriate theoretical path relevant to South Africa for this study.

1.4. Study aim and research questions

This study highlights the importance of healthy EI for the flow regulation function (i.e. rainwater retention, storage and subsequent discharge) of catchments during dry periods and in times of droughts, especially for rural communities in the grassland region. Healthy catchments can sustain the water provision service of ecosystems (Sigwela et al., 2017) through phenomena such as groundwater renewal (van Tol & Lorentz, 2018), and increased water storage and hydrological returns (Gong et al., 2013; Rebelo et al., 2015; Maseyk et al., 2017; Miguez et al., 2019; Pantshwa & Buschke, 2019). Flow regulation is the capacity of catchments to retain and store water from precipitation, reduce direct runoff and flood peaks, and release the water more slowly to sustain streamflow into or through the dry season (De Groot et al., 2002: 396; Brauman et al., 2007). Soil water storage and groundwater recharge are the most crucial catchment properties for flow regulation since they regulate flow events during dry periods through baseflow (Hughes, 2004; van Tol & Lorentz, 2018).

While hydrological response monitoring and protection have gained research attention in the past years, more work is still needed to unlock investments for practical application of literature recommendations such as large-scale rehabilitation targeted at protecting freshwater sources (Le Maître et al., 2016; Mander et al., 2017; Thorslund et al., 2017). Despite this recognition, there has been little to no investment in the protection of healthy ecological infrastructure as nature-based solutions for water security (Turpie et al., 2008; Schäffler & Swilling, 2013; Dias et al., 2016; Cumming et al., 2017; Shackleton et al., 2017; Adamowicz et al., 2019).

Based on the need for more data-based evidence for investing in EI to improve water security for communities, the main aim of the study was to understand better the role and benefits of optimising ecological infrastructure restoration to enhance community water security in times of drought mitigation and during dry seasons. The main aim is achieved by answering four research questions in an identified catchment:

(i) How do droughts and anthropogenic influences threaten water security for communities, and how can catchments be strengthened to improve streamflow regulation? (Chapter 2)

(ii) What is the current condition of the focal catchments? (Chapter 3)

(iii) Which focal ecological infrastructure areas can be targeted for restoration that can help improve the drought mitigation capacity of catchments, and how do these areas compare to core areas for rural household livelihoods? (Chapter 4)How different land cover changes impact water flow regulation, with a particular focus on dry periods? (Chapter 5)

1.5. Study area description

The Cacadu catchment (S10 tertiary catchment) was identified as an ideal case site to address the research questions (Figure 1.2). The Cacadu catchment is situated in a mild to extreme drought exposure area since 1992 (Malherbe et al., 2016). Moreover, the study area is a headwater catchment in rural settings that are vulnerable to degradation (Shackleton &

Gambiza, 2008). Parts of the focal catchment already have pre-existing working relationships with the host institution of the researcher. Rhodes University is currently involved in work in Machubeni communal area (S10F quaternary catchment) through the GEF5 Sustainable Land Management project, Eastern Cape (The GEF, 2013). The Cacadu catchment falls within the Grassland biome, with patches of Savannah and Forest Biomes (Figure 1.2). The mosaic vegetation for the focal catchment indicates eight main vegetation species in Cacadu catchment (Rutherford et al., 2012).

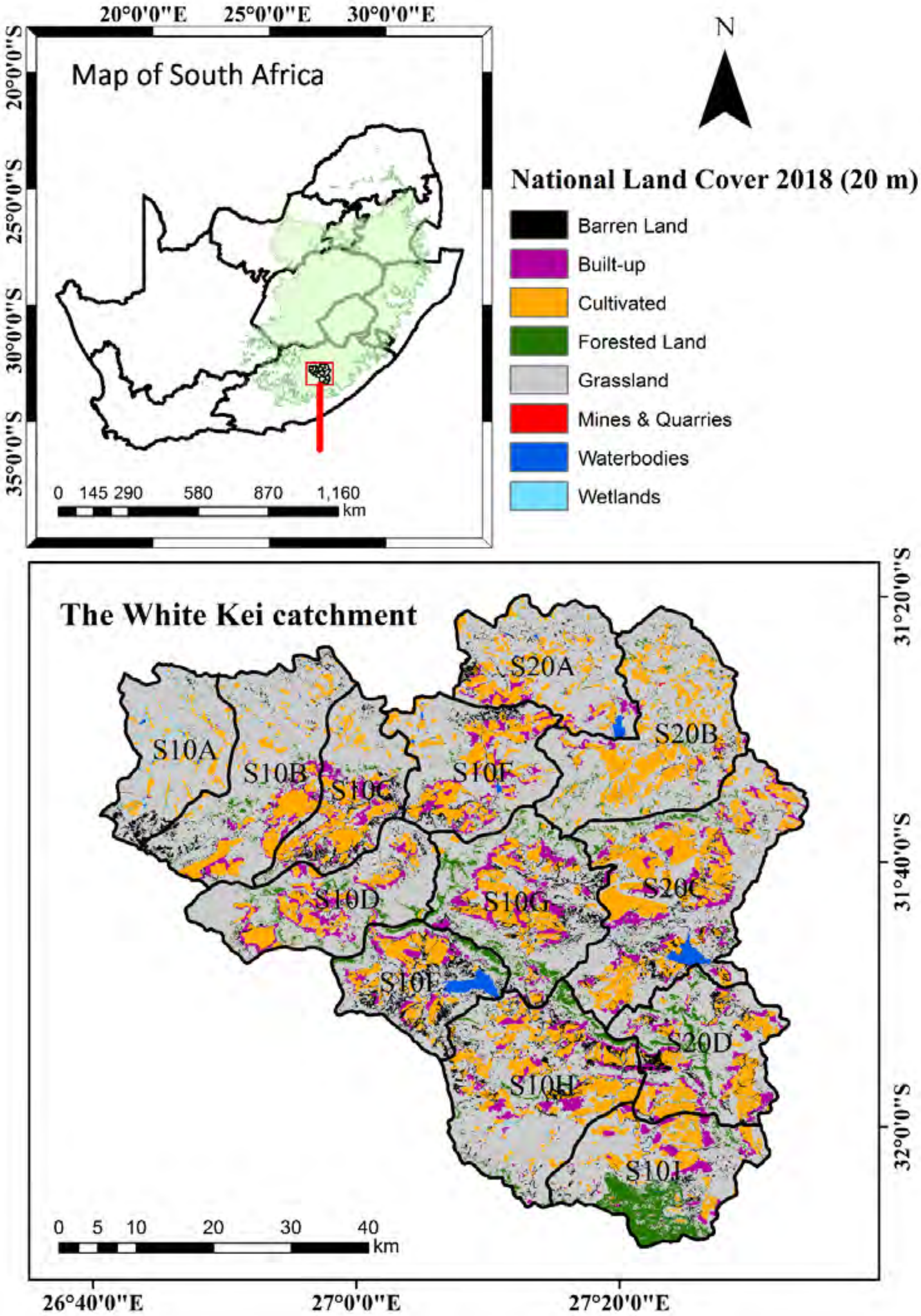

Figure 1.2: The location of the focal catchment within South Africa and context map of the White Kei catchment. The focal catchment has been underlain with the grassland biome in green (SANBI, 2012). The land cover map was extracted from GTI (2018) dataset.

1.5.1. Cacadu catchment (S10 catchment)

1.5.1.1. The biophysical context

The Cacadu catchment (27° 00-30' E; 31-32° 16' S) has an approximate land mass of 292 472.8 ha and is located within the Chris Hani District Municipality (falling within Enoch Mgijima, Emalahleni and Intsika Yethu Local Municipalities) in the Eastern Cape, South Africa (Figure 1.2). The catchment is underlain by five rock types: the Karoo Dolerite suite, the Tarkastad sub-group, Molteno Formation, the Elliot Formation, and some unconsolidated sediments (Smith et al., 1993; CGS, 2018). The catchment falls within the S1A Quaternary catchment rainfall zone with an annual rainfall range between 200 to 700 mm/year, and a dry season during April to September, while mean annual runoff ranges between 20 and 50 mm/year (Bailey & Pitman, 2015; Funk et al., 2015). The Cacadu catchment drains about 14% of the upper Great Kei River system (whose area is 20 489 km^2), through the Cacadu, Wit-Kei and Grootvleispruit major tributaries (DWS, 2006). Most of the catchment is covered by grasslands and cultivated land cover types (Figure 1.2). Like most South African catchments, IAPs are prevalent in the Great Kei catchment (Weni et al., 2010). The dominant soil make-up of the Cacadu catchment is dominated by Leptosols and Luvisols (FAO, 2015). The soils in the catchment are mostly moderate to deep sandy loam (Bailey & Pitman, 2016: 7), with high soil erodibility factors (SE range = 0.5 to 0.7) (Schulze et al., 2007). Parts of the catchment present a unique ecological degradation case characterised by erosion (sheet and gully) and *Euryops floribundus* encroachment (Shackleton and Gambiza, 2008). Natural phenomena and land management practices are linked to erosion in the area (Shackleton & Gambiza, 2008). Evidence of poor land management is validated by the naturalisation of the *E. floribundus* plant. *Euryops floribundus* is a multi-shrub pioneer species that is widely dispersed in the Eastern Cape, and it can grow to 2.5 m. Rural communities use it as a medicinal plant or a source of fuelwood (Shackleton & Gambiza, 2008).

1.5.1.2. The socio-economic environment

The Local Municipalities that host the Cacadu catchment have low population size and density (Table 1.3). There are slightly more female inhabitants than males in Cacadu catchment, and most of the population is within the range of 15-64 years of age (Table 1.3). Livelihoods in the area include subsistence crop farming, livestock farming, and natural resources extraction (e.g. brick making soil, coal, fuelwood, fruits, medicinal plants) (Chris Hani District Municipality, 2017). Some of the typical crops include maize, beans, pumpkin and oats for animal feed, while livestock includes cattle, sheep, goats, poultry and donkey (Chris Hani District Municipality, 2017). The dominant tenure system in the Cacadu catchment is common property, with some private property in the commercial agricultural regions (Chris Hani District

Municipality, 2017). The GEF5 demonstration villages based in the study area (5 out of 14 villages), were selected by the local leaders and the elected leadership based on several criteria. The criteria included the proximity of the villages to each other, their varied location within the catchment, the different land-use types and activities they represent, and previous involvement with Rhodes University projects (Sisitka & Ntshudu, 2017). The participating villages are Platkop, Gxojeni, Qhoboshane, Bomplaas, and Helushe (Sisitka & Ntshudu, 2017).

Table 1.1: Summary of demographic information for Cacadu catchment local Municipalities (StatsSA, 2011).

Quaternary catchments	S10A	S10B-G	S10H-J
Local municipalities	Enoch Mgijima	Emalahleni	Intsika Yethu
Population size	21 971	119 460	145 372
Females	51.4%	52.6%	52.7%
Males	48.6%	47.4%	47.3%
Age groups (%)			
15-64	62.6%	55%	53.9%
65+	6.6%	9.9%	10%
Population density (individuals per sq.km)	6	35	54
Economic activity			
Agricultural households	1 899	16 335	23 639
Total unemployed	2 639	8 070	9 363
Youth unemployment	47.6%	55.3%	56.4%
Household details			
Average household size	3.4	3.7	3.5
Formal dwellings (%)	97.3%	56.1%	32.9%
Settlement type			
Farm/ Tribal	14.8%	81.7%	90.4%
Urban	85.2%	19.3%	9.6%
Water source			
Provided by municipality	81.5%	56.9%	41.7%
Borehole	11.1%	9.6%	8%
Spring/River	1.5%	16.3%	32.2%
Other	6%	17.2	18.1%

1.5.1.3. The Fifth Global Environmental Facility Project: Sustainable Land Management Project in Machubeni

The GEF5 funded SLM project in Machubeni is one of three working sites for the project (i.e. Eastern Cape, Limpopo, and the Karoo), whose work covers five years ending in 2021 (The GEF, 2013). The project's fundamental goal is to aid capacity-building for communities residing in degraded communities to equip them with the technical capacity to combat land degradation and restore ecological infrastructure (Xoxo & Mantel, 2019). The project cites SLM interventions, collaborations with the local and national government, alongside higher learning institutions as its enabling factors.

2. CHAPTER 2: LITERATURE REVIEW

2.1. Introduction

Ecosystem services thinking is useful for forging actions to cope with climate change and anthropogenic related impacts (Howells et al., 2013; Cohen-Shacham et al., 2016; Bridgewater, 2018), and a holistic approach to water resource management (Pollard et al., 2014; Angelstam et al., 2017). The ecosystem services framework provides a systematic narrative to inform principles and practices that can be applied to catchment management in order to maximise socio-economic development while considering the integrity of the biosphere within the integrated water resources management community (DWAF, 2013; Clifford-Holmes et al., 2016; Ghosh, 2017). Such actions emphasise the necessity for a paradigm shift in how we view and manage freshwater resources (Rockström et al., 2014). For this paradigm shift, stakeholder involvement should be prioritised to uncover and inform behaviour and roles towards water resources conservation (Bek et al., 2017; Liniger et al., 2019).

This review focuses on understanding land cover change impacts on catchment hydrology and the links to drought risk reduction for water security. First, the review contextualises the drought phenomenon in the southern African context. Then, a discussion about the relationship between unsustainably managed landscapes and implications for catchment hydrology is provided. A web of science search for the relationship between EI and catchment hydrology indicates that the topic has gained increasing attention from researchers, especially post-2006. The increased research output on the topic above could be due to the threat posed by the increasing water scarcity, increasing global socio-economic developments, and climate extremes that add to the water scarcity challenge (Veldkamp, 2009; Carpenter et al., 2011).

2.2. The Drought phenomenon in the African continent

Droughts are not foreign to Southern Africa and are strongly associated with El Niño Southern Oscillation conditions that reoccur every 2 to 7 years (Meque & Abiodun, 2015; Vogels et al., 2017). In the 21st century, the African continent experienced 134 drought events that contributed to $27 billion in economic losses due to natural disasters (UNDRR, 2019). During the El Niño Southern Oscillation events in the south-east Africa, which includes South Africa, the region experienced a delay in the rainfall season (typically October to April), making South Africa vulnerable to drought (Ropelewski et al., 1987; Richard et al., 2000). Drought occurrences are part of the expected climate cycle and have different causes - relating to climatic or catchment characteristics, depending on the area affected, but the process of drought formation is similar (Mishra & Singh, 2010; Burke, 2011; Dai, 2011, 2013).

Ahmadalipour & Moradkhani (2018) combined 28 datasets that encapsulated economic, energy and infrastructure, health, land-use, and water resources factors from 46 African countries to provide a pentad-year based drought vulnerability index over 55 years. Figure 2.1 illustrates the spatial distribution of normalised mean drought vulnerability of African countries over 55 years. The drought vulnerability index (Figure 2.1) suggests that the Sahel and east African region had the most exposure to drought across the assessment period, while the drought vulnerability in the south-most countries slightly declined over time. The results presented in Figure 2.1 partly support earlier literature findings where Dai (2013) studied potential drought risk under green-house gas-induced warming using the Coupled Model Inter-comparison Project phase 3 and Palmer Drought Severity Index (PDSI) from 1980–1999 to 2080–2099. Dai (2013) found that severe droughts are likely to be over most of the Global South and Southern Europe. Other studies (Dai, 2011; Lewis et al., 2011; van Lanen et al., 2013; Hao et al., 2014; Stoelzle et al., 2014) have indicated a year-to-year increase in precipitation variability and intensification of drought events and this trend is likely to progress to the future. One shortcoming of the drought vulnerability index findings reported by Ahmadalipour & Moradkhani (2018) lies in the country-wide normalisation, which underestimates droughts' localised impacts in the smaller spatial scales. Consequently, the index provides anomalous findings for countries like South Africa compared to other authors (Vogel & Drummond, 1993; Malherbe et al., 2016) who have reported localised severe droughts. Nonetheless, the findings outlined in Figure 2.1 outline a comprehensive long-term view of the major natural disaster (droughts) in the African continent.

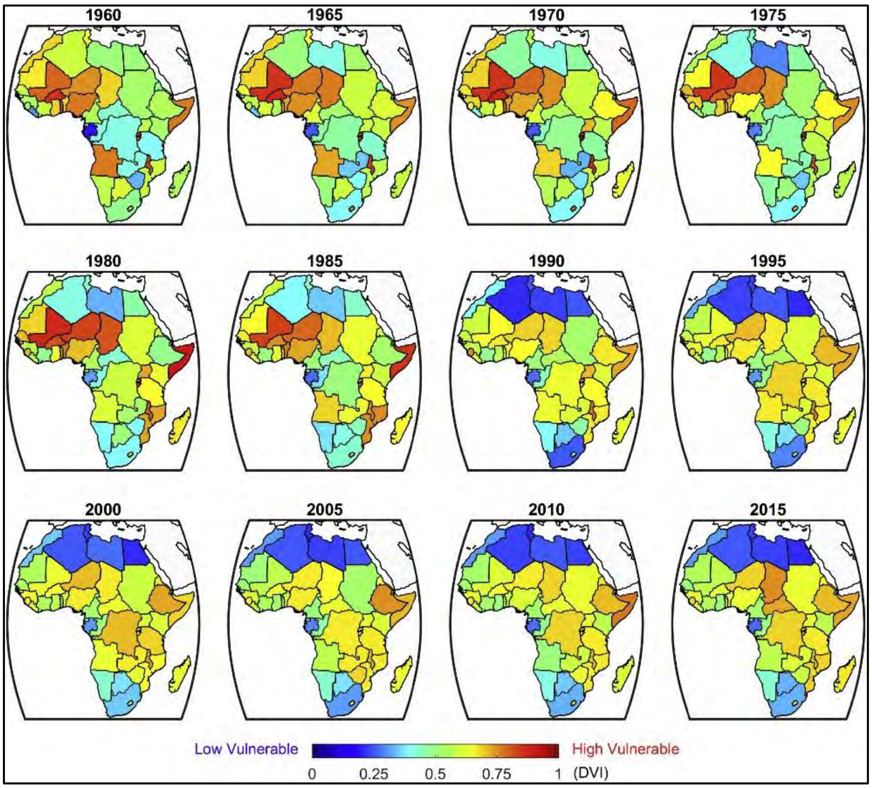

Figure 2.1: Normalised mean drought vulnerability index (DVI) for African countries measured in pentad years between 1960 and 2015 (Ahmandalipour & Moradkhani, 2018).

Besides the deficit of rainfall and reduced soil moisture, the shortage of water in the hydrological system (i.e. hydrological drought) that can be traced as streamflow reduction, low surface water and groundwater volume (van Lanen et al., 2013; Stoelzle et al., 2014; van Loon, 2015). Unlike meteorological and soil moisture drought, hydrological droughts take longer to offset and affect all sectors of the social-ecological system, especially the aquifer dependent (Colvin et al., 2007; van Loon, 2015), and can trigger land degradation (Masih et al., 2014). Knowledge synthesis on the drought topic (Dai, 2013; Masih et al., 2014; van Loon, 2015; Peña-Gallardo et al., 2019) from different regions globally have noted the need to manage droughts especially hydrological droughts due to their associated socio-economic impacts, especially water and food security.

2.2.1. Impact of droughts on water and food security

Droughts and anthropogenic modifications of ecosystems are the main drivers of water scarcity, and have been shown to impact both ecological and community water needs negatively (Clarke et al., 2012; Gandure et al., 2013; Sterling et al., 2013; Rockström et al., 2014). Droughts are linked to reductions in water and food security which threatens the livelihoods of both commercial and subsistence farmers, and communities (Clarke et al., 2012; Gandure et al., 2013; UNDRR, 2019) due to increased desertification, aridity and reduced land productivity (Dai, 2013; Koerner & Collins, 2014). Droughts have also been associated with increased evapotranspiration, causing vegetation mortality and wildfires globally (Allen et al., 1998; Anderegg et al., 2013; McDowell et al., 2013; He et al., 2014, 2017) and lowered water tables (Fang & Pomeroy, 2008). On the other hand, land-use change is linked with shifts in ecosystem structure and function (Vogels et al., 2017; Gibson et al., 2018). Hydrological changes at a landscape scale also relate to land-use change (discussed later) (Le Maître et al., 2014; Rockström et al., 2014; Cuthbert & Tyler, 2017).

The drought phenomenon poses a considerable risk for food security and water-related ecosystem services such as streamflow regulation, water quality regulation and ecosystem health support (Masih et al., 2014; Hughes et al., 2018a, 2018b). Agricultural-related droughts in southern Africa threatened up to 88% of agricultural production between 2000 – 2016 (Winkler et al., 2017). Although agricultural drought patterns were observed across southern Africa, there were spatial variations in East Africa (Winkler et al., 2017). For instance, Kenya and Somalia were affected by droughts in the early 21st century, while the most pronounced droughts in central Ethiopia were observed between 2001 and 2015 (Winkler et al., 2017). The vulnerability of the Southern African region to droughts has also been documented for earlier years (1967-2016) and is projected for the late 21st century if the ongoing climate threat is not addressed soon (Masih et al., 2014; Lehner et al., 2017). Furthermore, unmitigated droughts may threaten future water allocations since groundwater becomes the primary source of water in the absence of adequate surface water storage (e.g. during times of drought) (Masih et al., 2014; UNDRR, 2019). In the South African urban context, droughts could compromise water security of over half the towns within the next ten years (Piesse, 2016).

Implications of severe drought-related risks can be extracted from recent droughts in areas like Cape Town (Botai et al., 2019), the Karoo and other parts of the Eastern Cape. Observations of recent droughts indicate that over 2% of severe drought periods coincide with La Niña periods, even though drought is known to be caused by El Niño Southern Oscillation conditions (Malherbe et al., 2016). This dynamic makes it difficult to project and prepare for these disasters adequately, but projections point to increased drought intensity in South Africa until 2050 (Vogel & Drummond, 1993; Malherbe et al., 2016). The Limpopo, Olifants, Vaal, Mzimvubu and the Pongola-Mntamvuma catchments have been exposed to repeated drought periods (Malherbe et al., 2016; Xulu et al., 2018; Gyamfi et al., 2019).

However, water-related disasters like floods and droughts threaten the livelihoods and sustainability of water resources in the region (USAID Southern Africa, 2013; Malherbe et al., 2016). Dutra et al. (2013) assessed meteorological drought in the catchment based on the Standardised Precipitation Index (SPI) to derive dynamic forecasting system in four regions including the Limpopo River catchment for 30 years ending in 2010. The seasonal forecasting results revealed a reliable monitoring product for the Limpopo catchment that crosses over the wet season due to El Niño Southern Oscillation effects (Dutra et al., 2013). Similar findings were reported by Botai et al. (2019), who used SPI and Streamflow Standardised Index to monitor drought prorogation. Although drought monitoring based on a single climatology index such as the SPI may be limited in detecting droughts due to their complex nature (Hao & Aghakouchak, 2014), the SPI index has been used mainly to monitor meteorological droughts and as an early warning system for droughts in the United States (Mckee et al., 1993; Shukla et al., 2011). Therefore, studies such as Dutra et al. (2013) and Botai et al. (2019) highlight that droughts in South Africa are mostly characterised by a combination of catchment and climate properties. There is a need to understand the hydrological processes that control catchment hydrology to understand how catchments can alleviate drought impacts.

While acknowledging the consequences of climate extremes in this human-driven Epoch – the Anthropocene (Crutzen, 2002), Rockström et al. (2014) appeal that we will need a holistic landscape approach that builds the resilience of both land and water ecosystems, and improved drought monitoring and prediction can aid drought mitigation by increasing disaster readiness (Sahani et al., 2019). However, accurately predicting droughts is a challenging exercise since droughts are a consequence of climate and geographical processes (Hao & Aghakouchak, 2014; Hao et al., 2014). For instance, the Lincon Declaration on Drought Indices recommended the SPI for drought monitoring and prediction (Hayes et al., 2011). Nevertheless, the SPI can only forecast precipitation deficit, which may not be suitable for the timely detection of other drought types, unless the indicator is standardised to reflect soil moisture anomalies (Hao & Aghakouchak, 2014). The Integrated Drought Management Programme recommends using additional indices for monitoring and forecasting to address the challenge of tracking droughts (WMO and GWP, 2016). However, the shortcoming of the second option in developing country contexts is the lack of long-term records needed to compute the forecasting indices (Hughes et al., 2010; Hughes, 2013). However, the availability of open-access earth observation datasets that are high resolution may soon sufficiently address the data scarcity limitation (Robinson et al., 2017; Li et al., 2019), although the challenge regarding the lack of long-term records will remain a challenge. The remaining option for reducing drought risk in data-deficient contexts is optimising holistic water management approaches, through identifying priority areas for drought risk intervention (WMO and GWP, 2016). The use of integrated water management approaches fits well with Rockström et al.'s (2014) plea.

2.3. Modification of catchment hydrology through land cover change

Land cover change is one of two factors that directly affect hydrologic processes at catchment scales. These changes in hydrological processes can be experienced as frequent/severe flood events (Rebelo et al., 2015) or reduced baseflow and quickflow (Le Maître et al., 2014). Since ecosystems change in response to climate change and human modifications (Folke, 2006; Levin et al., 2013), the impacts are often cumulative and hard to distinguish (Li et al., 2009). This challenge is also relevant for hydrological assessments (Graham et al., 2011; Leketa & Abiye, 2019).

The anthropogenic risk to the flow regulation function of catchments includes loss of groundwater recharge sources (Jewitt et al., 2004; Le Maître et al., 2014, 2015; Rebelo et al., 2015; Gyamfi et al., 2016), land conversions and IAPs which lead to unsustainable water withdrawals (Le Maître et al., 1996, 2000, 2014, 2019; Rebelo et al., 2015, 2018; Mander et al., 2017), and increased surface runoff due to compaction and poor landscape designs (Zhao et al., 2014). Several case studies in South Africa (Turpie et al., 2008; Koerner & Collins, 2014; Le Maître et al., 2014) have discussed the impacts of fire regimes and noted soil structure disturbance, soil erosion, biodiversity loss, and increased surface runoff. The concerns of wildfires on ecosystem integrity have relevance in forested ecosystems, but often have to balance the counter-arguments for the importance of managed fires for maintaining species richness in semi-arid grasslands and savannahs (Bennett et al., 2012; Cleland et al., 2013; Koerner & Collins, 2014). The dispute is due to the important role played by fires in maintaining the ecosystem structure of savannah and grassland ecosystems, which in the absence of fire could become forested ecosystems (Koerner & Collins, 2014; Veldman, 2016; Skowno et al., 2017). The risk of grasslands or savannah to forest regime shifts is supported by a decline of over 20% in wildfires globally in the recent period (Allen et al., 2016; Andela et al., 2017; Luvuno et al., 2018), which could result in changes in catchment hydrological properties such as interception and evapotranspiration, even if the encroaching woody plants are indigenous (Le Maître et al., 1999, 2014). Others (Blair et al., 2018; Gibson et al., 2018) have focused on land-use and land cover change, and resulting alteration in productivity and vulnerability of native biodiversity.

As demonstrated above, individual and cumulative human impacts on landscapes can negatively impact on regulating ecosystem ability as they change ecosystem structure, integrity and processes (Rockström et al., 2009; Parrott & Meyer, 2012; Folke et al., 2016). Changes in ecosystem characteristics often expose the social-ecological system to external shocks and can trigger a feedback mechanism that exposes communities to climate risk in the long term (Chapin et al., 2010; Parrott & Meyer, 2012; Clifford-Holmes et al., 2016; Sun et al., 2017; Santos et al., 2018; Pantshwa & Buschke, 2019). While resource managers often know how to respond to shocks and exposure to risk (De Vriend & Iedema, 1995; European Commission, 2013; Ni, 2013; De Klerk et al., 2016), the required response strategy may cost additional human and financial resources which could have been allocated elsewhere if the shock was avoided initially. The same goes for drought mitigation.

Climate change and land cover change and their hydrological impacts have been investigated during the past two decades for the southern African region (Troy et al., 2007; Kabanda & Palamuleni, 2013; McIntyre et al., 2014; Haregeweyn et al., 2015; Rebelo et al., 2015; Gyamfi et al., 2016; Gumindoga et al., 2018). Impacts of land cover change on water yields in catchments are often investigated using time series analysis (coupled GIS and statistical methods), and hydrological models (Le Maître et al., 2000, 2014; Li et al., 2009; Kabanda & Palamuleni, 2013; Gyamfi et al., 2016; Dinka & Klik, 2019). However, all modelling approaches are faced with various shortcomings including data scarcity in the African continent leading to greater uncertainty (De Groot et al., 2010; Hughes et al., 2010; Münch et al., 2017). Regardless of the method used, and inherent uncertainties, the above-cited literature and other research are crucial for framing the understanding of anthropogenic impacts on the ecological integrity and functioning of catchment ecosystems.

2.3.1. Links between modified landscape units and water flow regulation

One key idea related to mitigating hydrological extremes (i.e. droughts and floods) is targeting catchment water storage through appropriate management (van Loon & Laaha, 2015). The current understanding of catchment hydrology is centred around the premise that catchment characteristics control the partitioning of rainfall into the surface, subsurface and deep groundwater runoff (Brauman et al., 2007; Le Maître et al., 2014), leading to a reduction in drought duration and intensity (van Lanen et al., 1997; van Loon & Laaha, 2015). While the potential of catchments to regulate droughts is widely accepted (Smakhtin, 2001; Bullock et al., 2003; Ticehurst et al., 2007), the dominant catchment characteristics that influence drought (i.e. climate, geology, soils, topography, land-use, and vegetation) are not clearly understood (van Lanen et al., 2013). The poor understanding of droughts has led to the need to broaden the knowledge of the physical catchment properties and how they influence the capacity of catchments to partition rainfall to regulate streamflow (van Lanen et al., 2013).

The essential characteristic of hydrological drought development and management is catchment storage capacity at the catchment scale since the physical catchment properties are responsible for water flow regulation (van Loon & Laaha, 2015). Generally, a combination of the following factors: climate, area geology, topography, soils, drainage network, landscape units, and vegetation regulate catchment storage (van Loon & Laaha, 2015). Since catchment characteristics evolve, storage behaviour can be challenging to predict primarily because drought lag in fast responding vs slow responding catchments varies with catchment characteristics (van Loon, 2015).

Presumably, in light of the weakened water flow regulation in degraded landscapes (Figure 2.2; Le Maître et al., 2014), slow responding catchments ought to experience a more extended drought propagation periods (Apurv et al., 2017) compared to fast responding catchments (van Loon, 2015). However, trends for the effects of human-induced degradation (e.g. EI modification) on catchment response and drought regulation are not uniform in reality. For instance, a meta-analysis of 244 case studies found that degradation drivers that reduce indigenous vegetation coverage (i.e. woody encroachment into grassland areas) had no net impact on soil infiltration (Eldridge et al., 2011). The infiltration stability in woody encroached grasslands suggests a consistent drought regime under the woody encroachment degradation driver. A field-based study in the Afromontane grassland area reports double to triple infiltrability of light and densely IAP invaded areas compared to natural grassland areas (Yapi et al., 2018), which supports the potentially positive role of introduced woody plants into the grassland system. Furthermore, if revegetation of eroded or exposed soil surfaces encourages better flow regulation (Mander et al., 2017; Hughes et al., 2018a) woody introductions can enhance the infiltration rates in barren environments, implying that drought recovery in such environments can be improved. However, this is not the case due to the risk posed by the proliferation of woody plants to groundwater resources in grassland and savannah regions (Scott et al., 2006; Le Maître et al., 2015, 2020). As demonstrated by Figure 2.2, land cover change influences the balance between infiltration and surface runoff, influencing water storage and release. Hydrological droughts lead to subsurface water reductions which invasive and encroaching plants also have access to; therefore degradation of catchments through the proliferation of woody plants intensifies droughts through reducing the soil storage capacity as shown in the prairie grasslands of Oklahoma, United State (Acharya et al., 2017). Consequently, catchment response time to recharge following a drought event influences drought propagation periods, e.g. drought extends beyond the meteorological drought season due to changes in groundwater volume, as observed in Cape Town, South Africa (Stoelzle et al., 2014; Apurv et al., 2017; Botai et al., 2019). Therefore, catchment management is essential to mitigate the impacts of drought, such as soil moisture and hydrological droughts (Tallaksen & van Lanen, 2004; Stoelzle et al., 2014).

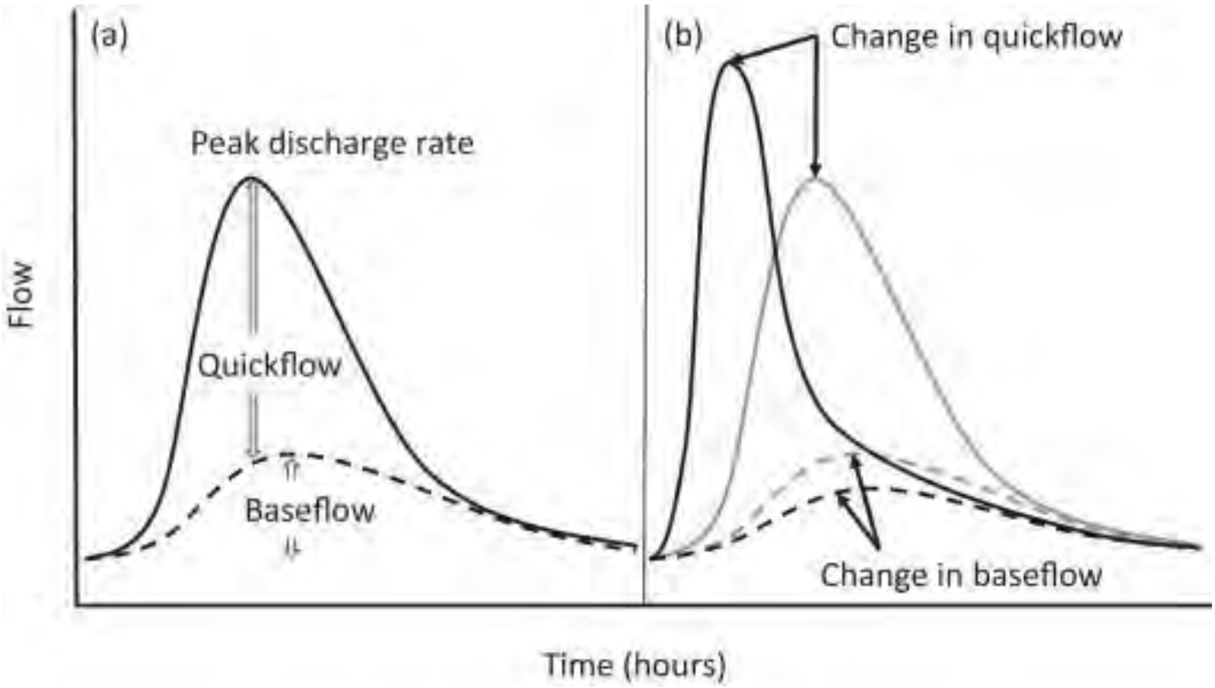

Figure 2.2: Schematic hydrographs representing rainfall-runoff changes between intact and disturbed catchments after a rainfall event (Le Maitre et al. (2014).

In addition to the impact of woody proliferation on biodiversity loss and catchment hydrology (Mander et al., 2017; Hughes et al., 2018b; Le Maître et al., 2020), changes in climate extremes and management styles have been associated with changes in the integrity and functionality of terrestrial ecosystems and vice-versa (Nepstad et al., 2004; Allen et al., 2010; Rowntree, 2013; He et al., 2014; Evans et al., 2015). While grazing and wildfires can facilitate ecosystem resilience (Koerner & Collins, 2014; Andela et al., 2017), overgrazing and wildfires may worsen ecosystem recovery (He et al., 2017). Intense Grazing and wildfires have a potential to trigger tipping points characterised with reduced land cover and increased surface runoff especially in prolonged dry periods (Figure 2.2; Le Maître et al., 2014; He et al., 2017), pointing the need to also pay attention to these two regimes.

Actions directed at maintaining natural traits of the catchment can help restore natural catchment behaviours, which in return may reduce the time lag for drought recovery to mitigate the impacts of droughts (Stoelzle et al., 2014). Severe droughts can result in water supply deficits either in atmospheric, surface or groundwater (UNCCD, 2016). Combined with temperature extremes, droughts can cause soil moisture anomalies that can be observed from the relationship between temperature and precipitation (IPCC, 2014), resulting in wildfires, biodiversity losses and increased desertification as illustrated in Figure 2.3 (He et al., 2014).

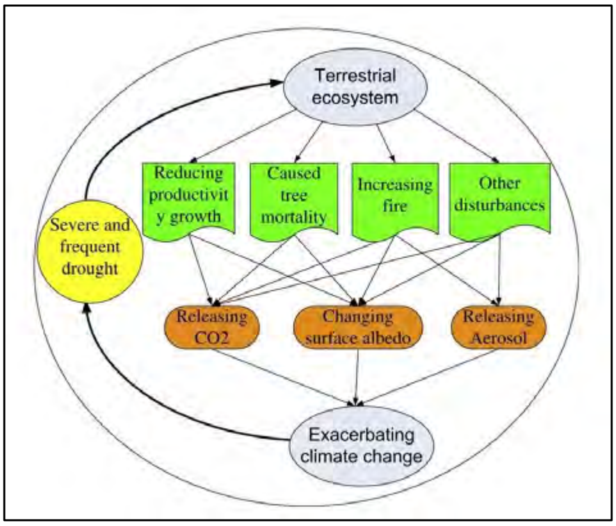

Figure 2.3: The interaction between drought-atmosphere-biosphere systems (He et al. 2014).

2.3.2. The importance of sustainable catchment management for drought mitigation and water security

Based on the current South African landscape ecological conditions (Skowno et al., 2019), the focus of EI investments towards aquatic and water-related ecosystems can be justified by two perspectives. Due to human-induced impacts on catchments such as land cover change, and climate extremes such as droughts, the quality and quantity of freshwater resources is gradually deteriorating (van Deventer et al., 2018). In South Africa, the threat to water quantity is mostly presented by the threatened inland aquatic ecosystems, grasslands and natural woodlands (Skowno et al., 2019). South Africa's long-term policy recognises the centrality of water to achieve sustainable development despite the country being ranked among low-rainfall and water-scarce countries (National Planning Commission, 2010). In their appeal for better management of freshwater resources, Harrison et al. (2016) urged for the protection of high water yielding areas for water security importance. Nel et al. (2013) used hydrological modelling techniques to identify Strategic Water Source Areas for South Africa based on mean annual runoff at a quaternary catchment scale. The Strategic Water Source Areas database has 22 Strategic Water Source Areas of which only 13% fall within formerly protected areas (Nel et al., 2013, 2017), which threatens the capacity of these areas to supply downstream areas (often urban areas) (Nel et al., 2013, 2017). The Strategic Water Source Areas contribute half of the nation's water supply and support nearly two-thirds of the country's economic activities despite being unprotected (Nel et al., 2017). Recognising the important role of Strategic Water Source Areas to the countries development and social wellbeing has informed the decision to include water resource protection to sustainably manage water resources in the second National Water Resource Strategy (DWAF, 2013: 5).

South Africa is one of the most important signatories to the Convention for Biological Diversity (CBD) (United Nations, 1992), since it hosts up to 6% of global biodiversity, and has long been committed to playing a part in native biodiversity conservation (Turpie et al., 2008, 2017; Forsyth et al., 2012). The biodiversity conservation strategy is informed by the National Biodiversity Management: Biodiversity Act (Act 10 of 2004) which provides a guide for achieving the national biodiversity targets. In 2018, South Africa conducted the second National Biodiversity Assessment (NBA) led by SANBI (Skowno et al., 2019). The NBA is a biodiversity-based assessment used to report on the country's biodiversity and management status to inform decision making by various sectors (Skowno et al., 2019). One key indicator provided in the NBA is the levels of protection, which is based on the ecosystem biodiversity targets (van Niekerk & Turpie, 2011; Skowno et al., 2019: 7.2). In brief, Figure 2.4 provides an outline of the EPL nationwide, showing that 26% of ecosystems are well protected, while 25% are not protected at all, and the rest (49%) are under-protected. Majority of the fully protected landscapes are the legally designated Protected Areas such as the Kruger National Park region, the Albany Thicket biome, the Cape Floristic region and parts of the Drakensburg (Figure 2.4). However, the dataset is limited by the absence of considering how effective the ecosystem's management is, thereby leading to potential discrepancies in the indicator (Skowno et al., 2019). Nevertheless, the indicator is crucial as it provides a valuable tool that can be used to inform decisions for biodiversity protection and integrated catchment management.

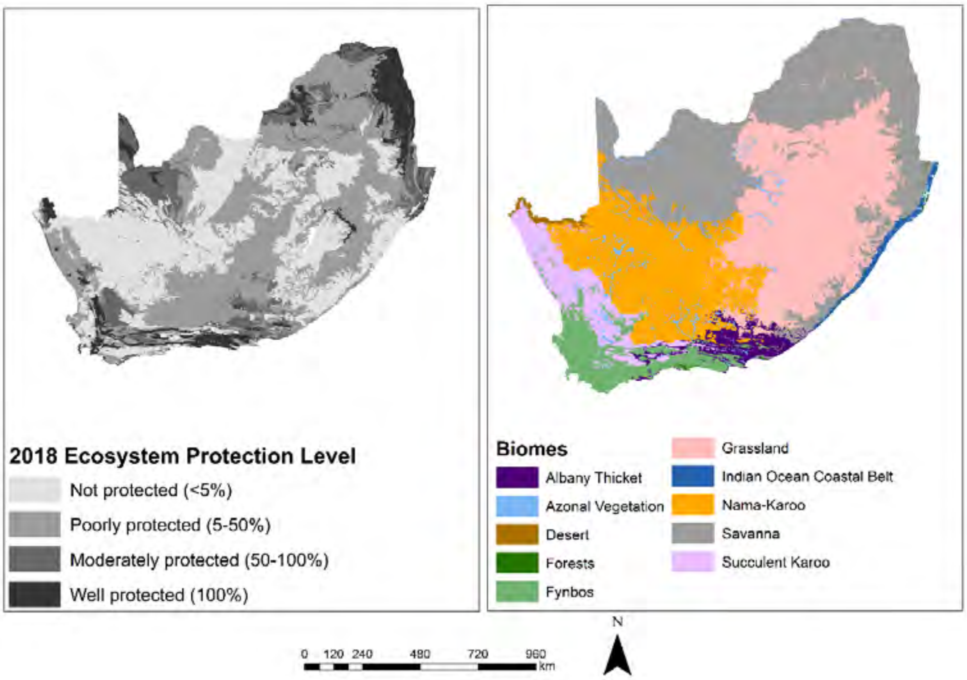

Figure 2.4: Map showing 4 levels of ecosystem protection in South Africa (Skowno et al. 2019) side-by-side with corresponding biomes (Mucina & Rutherford, 2006).

Since South Africa has a long legacy of IAPs with impacts that are mostly attributed to water security and biodiversity loss, the country committed itself to restore ecosystems under the "Working for" programmes (van Wilgen et al., 2012b). Under the Working for Water program, priority invasive species have been targeted and removed from the country's landscapes, especially the Strategic Water Source Areas (van Wilgen et al., 2012b). The IAP intervention strategies are closely linked to ecosystem vulnerability and protection levels (Skowno et al., 2019), considering that IAPs offset indigenous biodiversity, threaten socio-economic wellbeing and water security (Turpie et al., 2008; van Wilgen et al., 2008). Thus, EI literature's focus on IAP intervention strategies can be viewed as a strategic step to advance investments in EI projects for meeting national and global targets (van Wilgen et al., 2012b; Cumming et al., 2017; DEA, 2018). However, one challenge for ecosystem protection relates to defining achievable intervention targets, given the wide range of properties that get eroded by ecosystem degradation (Parrott & Meyer, 2012).

Examples of four possible EI investment pathways from disturbed catchments are presented in the modified illustration (Figure 2.5). First, Figure 2.5 emphasises the growing uncertainty about the impacts of climate and land cover change to baseline conditions in an ecosystem. Plot B shows a moderate or well-protected managed ecosystem that is within the stable state. An excellent example of such a system would be a protected land where biodiversity protection is a priority, and the indirect benefit is increased integrity of the landscape to facilitate water-related ecosystem services (Le Maître et al., 2014; Shumba et al., 2020). Plot C shows a novel ecosystem – ecosystems that have been heavily impacted by anthropogenic actions and have exceeded their resilience limits and cannot be restored (Morse et al., 2014); sometimes even in light of EI investments (management intervention) most of the ecosystem properties cannot be recovered (Figure 2.5). The gully erosion and IAPs in South African landscapes are examples of such systems (van Wilgen et al., 2012b; van der Waal & Rowntree, 2018). Restoration literature emphasises the consideration of novelty when dealing with novel ecosystems (Perring et al., 2013). Doing so would reduce investment costs and attain high recovery of systems as plot C (Kimball et al., 2015; Gann et al., 2019). In plot D, a restored ecosystem has a reduced future ecosystem uncertainty due to climate and land cover change (Figure 2.5). That is because the adopted management interventions yield the highest level of ecosystem recovery, and the ecosystem structure and functions are restored (Figure 2.5). Compared with possible landscape states B and D shown in Figure 2.5, the removal of IAPs is crucial because it represents actions that promote the resilience of lands, and communities that inhabit those lands, through biodiversity protection, provision of multiple ecosystem services, climate change adaptation and reducing emissions. Although IAP invasion is one driver of land degradation in South Africa, the above discussion emphasises the articulation of EI in terms of land-use (identified using land cover change) which is essential when interrogating water-related ecosystem services in catchments (SANBI, 2014; von Maltitz et al., 2019).

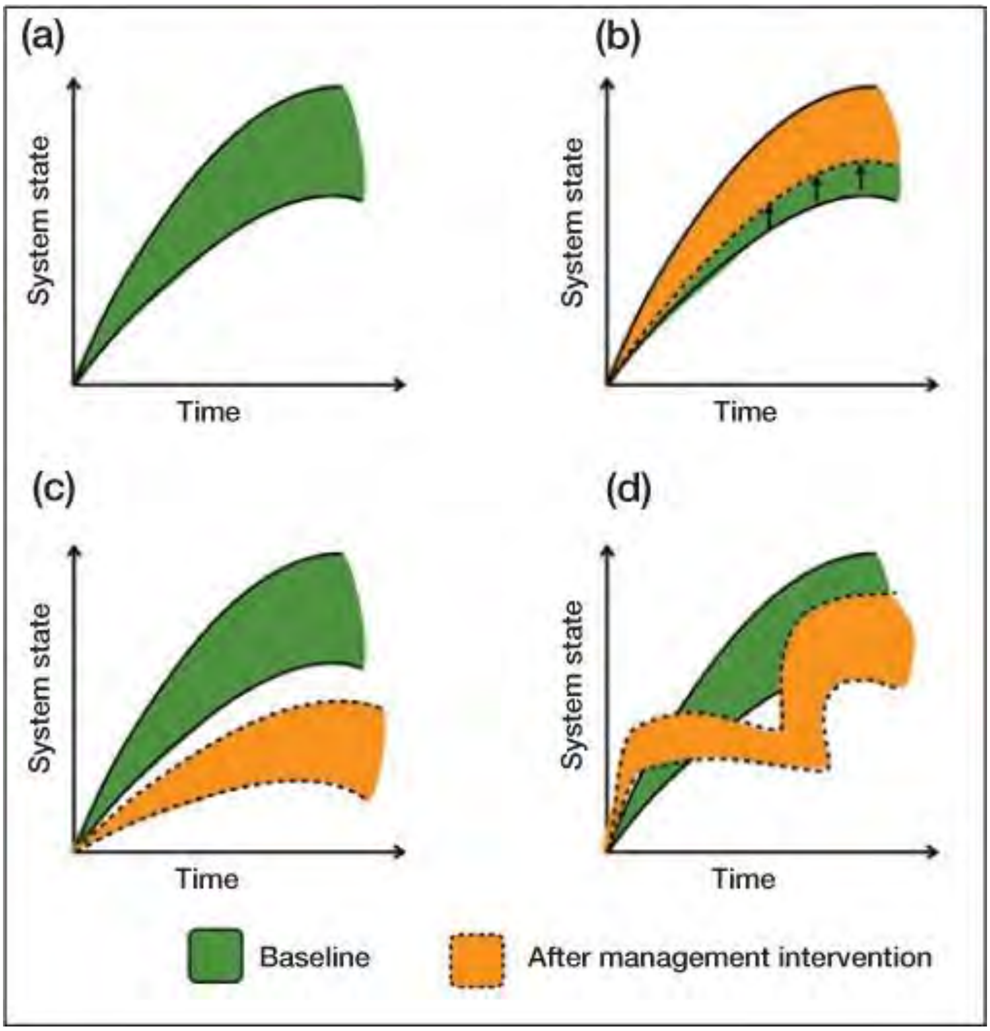

Figure 2.5: Ranges of possible landscape states for social-ecological systems (Parrot and Meyer, 2012: Figure 5). Parrot and Meyer (2015) visualised the factors that describe ecosystem state as a single axis for ease of analysis. In terms of water flow regulation, (**a**) depicts an increasing uncertainty about the impacts of climate and land cover change impacts on flow regulation over time. (**b**) Depicts a sustainably managed system (e.g. protected catchment from further degradation) in which alterations in water balance components could be reduced. (**c**) Shows a system that passes a tipping-point (e.g. transformation of grasslands into forested land). (**d**) Shows a transformed system that undergoes intensive inteventions that would establish a new dynamic state (e.g. restoration of wetlands).

2.3.2.1. The role of restoration interventions in catchment management and drought mitigation

Offsetting land degradation and reversing the loss of ecosystem services is amongst global priorities (United Nations, 2015), and the recent global commitment to make this target a reality under the Decade for Ecosystem Restoration is a promising pathway (United Nations, 2019). Although the restoration process (reclaiming lost ecosystems or establishing new ones in degraded areas) is a complex and challenging process (Gann et al., 2019), achieving restoration has a potential to increase social-ecological resiliency and improving human well-being (De Wit et al., 2012; Stafford et al., 2017). To reduce the practical challenges surrounding the restoration process, an international collaboration of restoration professionals developed eight guiding principles for the restoration of all ecosystem types (Gann et al., 2019). The international standards for restoration stress the importance of stakeholder inclusion, recognising traditional knowledge sources, outline the process of setting restoration baselines, and present a wheel for social progress to supplement the ecological wheel (Gann et al., 2019). The social benefits wheel is intended for tracking the social benefits provided by restoration activities, and it is directly related to the first principle (stakeholder engagement) that was overlooked in earlier standards (Gann et al., 2019: S8). Following a recognition that ecosystem restoration can benefit from practical experience, which is not limited to formal scientific knowledge, the recent restoration standards emphasises the inclusion of various knowledge sources – principle 2 (Brancalion & van Melis, 2017; Gann et al., 2019: S10). As highlighted under novel ecosystems (Morse et al., 2014) and regime shifts (Folke et al., 2004), ecosystem may move into an irreversible state; therefore, restoration activities need to identify achievable reference ecosystem conditions that facilitate ecosystem recovery process – principles 3 & 4 (Gann et al., 2019: S11–S14). Having identified the preferred reference ecosystem, restoration projects should target the highest recovery possible based on a clear set of objectives and targets, which can be visualised using the 5-star recovery scale and measurable social-ecological indicators (Gann et al., 2019). The seventh principle emphasise the understanding that restoration activities and benefits are scale-dependant, and appropriate operating scales should be identified (Gann et al., 2019: S18). The standards also define restoration in affiliation with allied activities via the restoration continuum (Gann et al., 2019). The restorative continuum highlights that restorative activities could facilitate partial or full recovery of ecosystems (Gann et al., 2019: S22). The restorative continuum is a gradient of six steps that range from mitigating social impacts to a full recovery of native ecosystems (Gann et al., 2019: Figure 5). While all levels of the restorative continuum are appropriate and necessary in different locations, ecosystem restoration is recognised as any activity that promotes native recover, while affiliated activates could be target at offsetting social impacts, improving ecosystem management and repairing ecosystem function (Gann et al., 2019). Therefore, drawing from numerous studies within South Africa (Rebelo et al., 2015, 2018; Mander et al., 2017; Hughes et al., 2018b; van der Waal & Rowntree, 2018), some of the key social-ecological benefits for a sustainably managed catchment are related to improved supply of water-related ecosystem services and increased land productivity and food security.

- Rebelo et al. (2018) note that both flora and fauna play an imperative role in soil porosity, and soil character is important for rainwater infiltration and runoff. Under a severe land degradation scenario, soils cannot adequately perform this ecological function, and instead, overland runoff increases (Rebelo et al., 2018).

- Based on a study in the Tsitsa catchment, van der Waal & Rowntree (2018) note that sustainably managed vegetation maintains high infiltration rates, which is an important determining factor for runoff and groundwater recharge and reducing sediment load. Impacts of high runoff include flooding at valley-bottoms and areas that are not well-drained (Rebelo et al., 2015; van der Waal & Rowntree, 2018). However, the benefits include vegetation recovery and crop production in dry years (van der Waal & Rowntree, 2018).
- Rehabilitating overgrazed and degraded grasslands could increase dry season baseflow by up to 520 m^3/ha per year (Mander et al., 2017). While clearing IAP can increase year-round streamflow, nutrient retention (Rebelo et al., 2015; Mander et al., 2017). A similar trend was also detected in the uMngeni catchment, where intact grasslands had a higher capacity to retain rainfall and contribute to streamflow than degraded landscape units (Hughes et al., 2018b).

Given that only a small portion of Strategic Water Source Areas are formally protected, targeting the “Poorly” to “Moderately Protected ecosystems” (49%) (Figure 2.4) for EI investment could significantly improve water yields (Harrison et al., 2016). However, successful EI investments in targeted areas such as protected ecosystems (Figure 2.4) will require continuous decision-making that is underpinned by adaptive management and shared-responsibility by the state and citizens (SANBI, 2014; Freeman et al., 2015). The above indications point out that rainfall partitioning tends to be improved in intact catchments, which can be compromised by land cover change (Le Maître et al., 2014).

2.4. Summary points

Efforts for alleviating drought impacts and water security require understanding the complex physical catchment processes that regulate streamflow, in addition to suitable monitoring and forecasting tools, and robust frameworks that can be adopted into practice. The review highlights that natural catchments contribute to drought mitigation through the flow regulation ecosystem service that partitions rainfall, thereby potentially contributing to streamflow in dry periods. However, this service is negatively affected by changes in climate and land cover. The review supports literature highlighting a lack of understanding of the flow regulation service, due to the complex nature of underlying land cover dynamics, which require further interrogation to inform decision making. Without an adequate understanding of catchment processes, and in light of economic development in the Global South, fulfilling sustainability agenda may remain challenging. Fortunately, national frameworks such as the SANBI framework for Investing in EI can be optimised for water and livelihoods security. The SANBI framework may have utility in building resilient ecosystems and societies that depend on those ecosystems, which is needed to cope with drought risks. The review highlights that the sustainable supply of surface water resources depends on the availability of relatively intact catchments, which largely owe their protection to landowners, civil society, and the state.

3. CHAPTER 3: ASSESSMENT OF DEGRADATION STATUS FOR THE CACADU CATCHMENT

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Conceptualisation, BSX, WM, SKM, ADV; Methodology, BSX, WM, SKM, ADV, DLM; Investigation: BSX, BM; Formal Analysis, BSX, BM; Resources, SKM, JT; Writing - Original draft preparation, BSX; Writing – Review and Editing, SKM, ADV, JT, DLM; Visualisation, BSX; Supervision: SKM, JT, ADV; Project administration: SKM, JT; Funding acquisition, SKM, JT, BM, BSX.

3.1. Introduction

Land degradation assessment approaches are vital to guide and support land degradation interventions (Easdale et al., 2019; Gonzalez-Roglich et al., 2019; Liniger et al., 2019), which supports the use of land degradation (SDG 15.3.1) indicator. A key challenge with past degradation assessments relates to the reproducibility of degradation observations. Besides methodological limitations of earlier degradation assessments, flawed interpretation of biophysical processes has been identified as another roadblock to land degradation assessments (Hein et al., 2011; Easdale et al., 2019). In the Sahel, numerous studies (Prince et al., 1998; Fensholt & Rasmussen, 2011; Fensholt et al., 2013; Dardel et al., 2014; Kaptué et al., 2015) monitored vegetation mostly through satellite-based observations to quantify degradation but reached contradictory conclusions. Some researchers (Dardel et al., 2014) concluded that the Sahel was experiencing ongoing degradation, while others concluded recovery in process (Prince et al., 1998), and still others could not find conclusive degradation (Fensholt & Rasmussen, 2011; Fensholt et al., 2013). Part of the problem was neglecting the resilience of semi-arid vegetation in the region, leading to temporary declines being interpreted as degradation (Hein et al., 2011). To reduce this error, Kaptué et al. (2015) proposed an NDVI anomaly framework that can capture long-term vegetation dynamics. Based on an Argentinian remote sensing study, Easdale et al. (2019) identified five vegetation states using NDVI. They called for the use of trend-cycles instead of monotonic trend-analysis to capture temporal variability when conducting degradation assessments using NDVI as a proxy.

Before the ratification of the Global Development Agenda, there was a long debate about appropriate methods for quantifying land degradation globally (Rowntree et al., 2004; Ibañez et al., 2014). In the South African context, land degradation has been previously assessed at a national scale based on the status of agricultural land with the help of expert opinions by 453 agricultural experts (Hoffman & Todd, 2000) and using satellite-based net primary productivity

(NPP) (Bai et al., 2007). The satellite-based NPP degradation assessment used Rain-Use Efficiency (RUE), and Residual Trend Analysis climate-adjusted land productivity at 1 km spatial resolution as proxies to measure degradation (Bai et al., 2007; Wessels et al., 2007). Bai et al.'s (2007) results were later verified by Bai and Dent (2008) using field visits. The early efforts to quantify degradation by Bai et al. (2008) were criticised for various reasons, including overlooking some land degradation processes which are specific to biome scale, global inconsistency for quantifying and reporting on degradation, and lack of consideration for other biophysical factors (Gibbs & Salmon, 2015). The above criticism applies to Bai et al.'s (2007) national degradation assessment since it followed the same methodology as the global assessment (Bai et al., 2008; Gibbs & Salmon, 2015). Von Maltitz et al. (2019) called for the biome scale as an appropriate monitoring scale for land degradation instead of the predominantly used administrative scale. As demonstrated in the Sahelian degradation case (Fensholt & Rasmussen, 2011; Hein et al., 2011; Dardel et al., 2014), degradation assessments that use administrative boundaries are often misleading because ecological processes are often misinterpreted and unreliable (not reproducible). The recommendation for using the biome setting as opposed to administrative boundaries acknowledged the important role played by earlier degradation assessments which have shaped the understanding of the degradation phenomenon (Hoffman et al., 1999; Bai et al., 2008; Vågen et al., 2016; Graw et al., 2017; Easdale et al., 2019). The methodological approach to assessing land degradation has evolved (Wessels et al., 2007; Bai et al., 2008; FAO, 2011a; Gamoun, 2016), including using better resolution, freely available satellite data, which has provided higher quality monitoring outcomes (Robinson et al., 2017; Sidhu et al., 2018).

The Sustainable Development Goals' (SDG) framework is a United Nation's initiative that acts as a global guiding tool to help the global community realise sustainable development at different operating levels by adopting the 2030 Development Agenda (United Nations, 2015). Within the 2030 Agenda, SDG 15 sets out to protect terrestrial ecosystems and biodiversity (United Nations, 2015). One of the key targets for SDG 15 is target 15.3, which aims to "combat desertification, restore degraded land and soil, and to foster ways to achieve a land degradation-neutral world" (United Nations, 2015). The ratified indicator for target 15.3 is referred to as SDG 15.3.1, which aims "to monitor for the proportion of land that is degraded over the total land area" under the custodianship of the United Nations Convention to Combat Desertification (UNCCD) (United Nations, 2015; Orr et al., 2017). Sustainable Development Goal 15.3.1 has three key sub-indicators for monitoring: land productivity, land cover change, and soil organic carbon (United Nations, 2015).

The first sub-indicator for SDG 15.3.1, land productivity, refers to the biological productive capacity of the land ; it represents the source of all food, fibre and fuel that sustains humans

(Clark et al., 2001). The land productivity sub-indicator is based on the notion that loss of vegetation production in productive lands can result in land degradation, and vice-versa (Munyati & Ratshibvumo, 2011; Bennett et al., 2012; Graw et al., 2017). As a step towards increasing resilience of land and populations dependent on the land, SDG 15.3.1 assesses change in soil organic carbon (third sub-indicator) under the rationale that loss of soil organic carbon is a form of land degradation that contributes to reduced soil quality and fertility, which consequently impacts on biodiversity and food security (Lal et al., 2012; Stavi & Lal, 2015). Soil organic carbon stocks are influenced mainly by land-use and management choices that affect nutrient input and output rates (Solomon et al., 2000; Mills & Fey, 2003). To compute land degradation, the three sub-indicators need to be combined to identify areas that have experienced significant changes (Orr et al., 2017).

This study assessed the terrestrial ecosystem health status of the Cacadu catchment from the land degradation viewpoint by considering the above limitations and recommended methods using remote sensing imagery. To estimate land degradation, this study employed the TRENDS.EARTH plugin hosted by Quantum GIS for degradation monitoring, which calculates SDG 15.3.1 indicator (Conservation International 2018). The plugin was introduced by the Global Environmental Facility (GEF) to extend the availability and use of global data sources to study land degradation at multiple scales using a harmonised methodology (Conservation International 2018). Within land degradation research, the plugin has been utilised in several locations for vegetation productivity measures and land degradation evaluation in South African drylands (Hoffman et al., 2018), for land degradation neutrality assessment at a global scale (Gonzalez-Roglich et al., 2019), and to assess the potential of earth observation datasets to estimate degradation in Namibia (Mariathasan et al., 2019). Establishing a terrestrial health status using the UNCCD approach is important because the one-out, all-out (OOAO) statistical rule (stating that an area is considered degraded if at least one of the indicators is degraded and improving if all indicators area is at least improving) provides a reliable monitoring approach that captures different types of degradation; addressing the limitations raised in earlier degradation assessments outlined before. Secondly, using the plugin for assessing terrestrial ecosystem conditions addresses von Maltitz et al. (2019) suggestion that the biome scale is the appropriate land degradation assessment unit, as opposed to the administrative scale.

3.2. Methods

Conservation International developed the TRENDS.EARTH plugin, in partnership with the Global Environmental Facility to provide a globally consistent and effective way to monitor land degradation, achieve SDGs, and help with national and UNCCD reporting (Conservation

International, 2018a). The novelty of the plugin is two-part. Firstly, the plugin allows for degradation definition based on the geographical context (Orr et al., 2017; Conservation International, 2018b). Secondly, when combining the sub-indicators, the plugin uses the precautionary the OOAO principle (Orr et al., 2017; Conservation International, 2018a). The OOAO principle states that degradation occurs when there is a significant decline in one of the sub-indicators, even if the other two show an improvement (Orr et al., 2017). The ability of the plugin to integrate all SDG 15.3.1 sub-indicators allowed this study to better identify and report on different types of degradation. A detailed discussion of the plugin is available in the appendices (Appendix 3.1). For the current assessment, the plugin was set-up to calculate and combine the SDG 15.3.1 sub-indicators for 2000 and 2015 (henceforth referred to as the 'assessment period') (Conservation International 2018). The assessment period is divided into the initial year (the year 2001) and the target year (2015). Although the assessment period only covered the recommended baseline years for the land degradation neutrality framework (Orr et al., 2017), it was adopted in this study because the plugin could only cover the 15 years at the time of the assessment (Conservation International, 2018a).

The default datasets within the plugin were selected to conduct the degradation assessment because South Africa lacks some of the custom input datasets (e.g. Soil carbon stocks) needed to run the plugin (UNCCD, 2018). To compute the first sub-indicator (land productivity), this study used the 16-day 250 m Moderate Resolution Imaging Spectroradiometer dataset (MOD13Q1) to derive normalised difference vegetation difference index (NDVI) values for use as land productivity proxy. The MOD13Q1 dataset covered the period 1992 to present. Additionally, the rain-use efficiency (RUE) index was conducted on the NDVI to reduce climate bias and detect human-induced impacts (Wessels et al., 2007, 2012). The precipitation dataset used in the RUE adjusted land productivity was obtained from the 5 km Climate Hazards Group Infrared Precipitation with Stations (CHIRPS) (Funk et al., 2015). The CHIRPS dataset is based on a combination of local rainfall station data with remotely sensed infrared cloud cover data from the quasi-global area (50°S to 50°N) (Funk et al., 2015). To compute the second sub-indicator (land cover change), this study used the 300 m resolution ESA CCI-LC dataset (EuroSpace Agency, 2017) which has 36 land cover classes that were derived from the 36 classes defined by the UNCCD shown in (Table A4). South Africa has high-quality custom datasets for land cover (GTI, 2016, 2019a), but these both do not match the assessment period, making inappropriate for use.

3.2.1. Computing SDG 15.3.1 sub-indicators

3.2.1.1. Sub-indicator 1: Land productivity analysis

This study divided the land productivity analysis into two steps to compare biomass productivity (a) under climate influence and (b) with rainfall variation bias eliminated. To detect the climate influenced land productivity, this study first chose to compute land productivity based on NDVI trends (Conservation International, 2018a). The productivity trajectory and productivity performance metrics were computed based on the NDVI observations of the seven UNCCD land cover classes for the assessment period (Conservation International, 2018a). The productivity state was calculated by splitting the assessment period into a baseline period (years 2000 to 2012) and a recent period (2013 to 2015) (Conservation International, 2018a).

The RUE index which makes the climate adjustments based on productivity state and rainfall was adopted to detect human-driven land productivity, eliminating the bias due to rainfall variation, (Wessels et al., 2007; Conservation International, 2018a). In the plugin manual, the WUE_e index is recommended for detecting human-induced effects (UNCCD, 2018). This study used the RUE index because land productivity in arid to semi-arid areas is mostly influenced by precipitation (Huxman et al., 2004; Wessels et al., 2007; Fensholt & Rasmussen, 2011; Fensholt et al., 2013). The long-term mean annual rainfall distribution from the CHIRPS dataset for the Cacadu catchment is presented in Figure 3.1. Between the years 1981 and 2016, the mean annual rainfall range in the Cacadu catchment was between 495 to 717 mm/year with quaternary catchments S10A, B and F receiving a slightly higher rainfall than other areas (Figure 3.1). An analysis for the accuracy of the CHIRPS dataset compared to the WR2012 dataset is discussed later in Chapter 4. Limitations of the CHIRPS dataset were discussed by (Bai et al., 2018) based on observed variations on the Chinese mainland. Bai et al. (2018) discovered that the CHIRPS dataset tends to underestimate rainfall in dry periods due to topographic variation. In east Africa, the statistical integration approach was reported to result in wet season rainfall overestimations, indicating the dataset limitations in poorly gauged areas (Dinku et al., 2018).

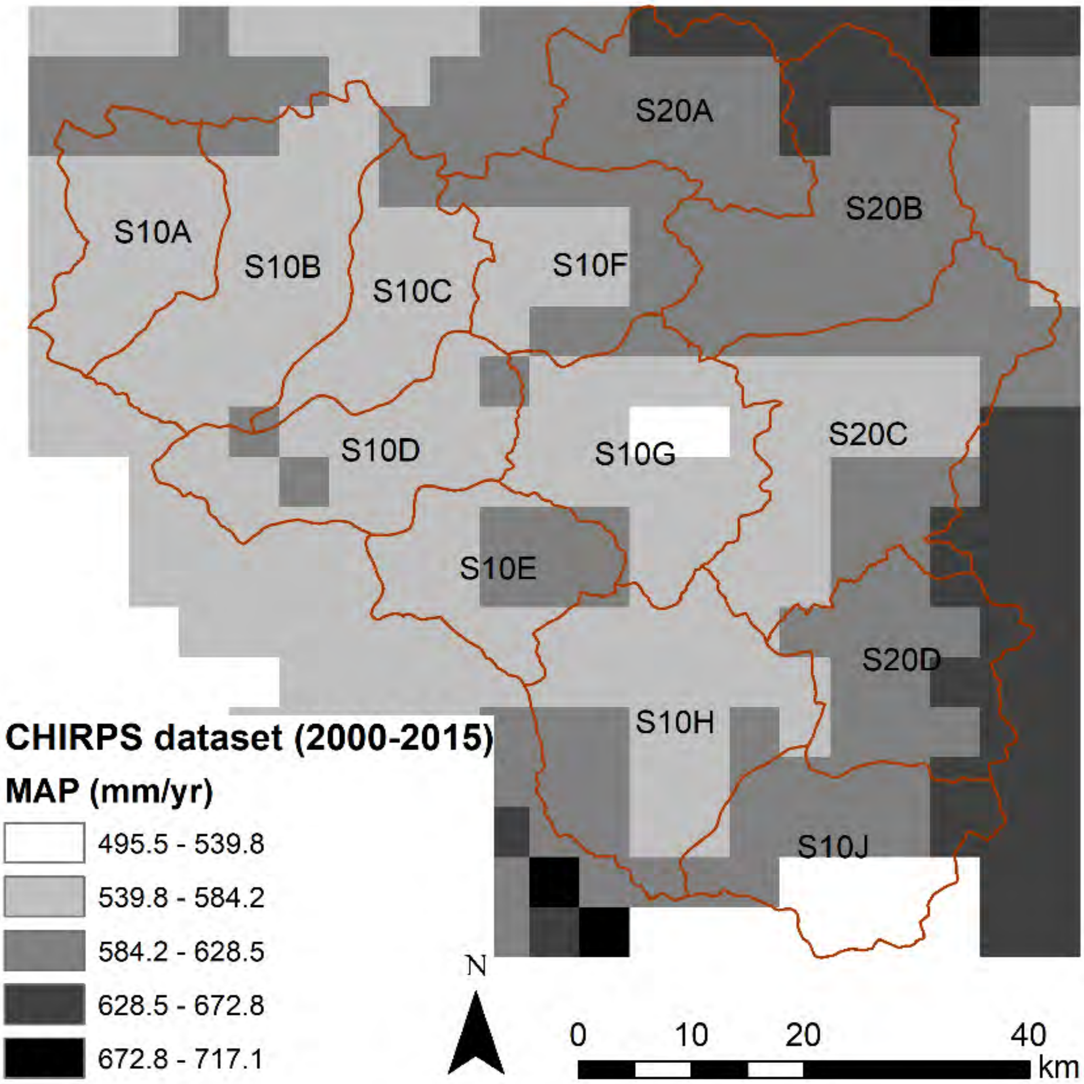


Figure 3.1: CHIRPS mean annual rainfall distribution for the hydrological period covering the years 2000 to 2015 for the Cacadu catchment (Funk et al. 2015).

3.2.1.2. Sub-indicator 2: Land cover change analysis

To analyse land cover degradation, I first defined the land cover transition (Table 3.1) to capture the degradation process of grassland biome. The definition was based on literature understanding of whether a change from one land cover to another is positive or negative (Le Maître et al., 2014; Stafford et al., 2017; Blair et al., 2018; Gibson et al., 2018; Luvuno et al., 2018; Scorer et al., 2019; von Maltitz et al., 2019). The definition has 49 possible transitions through 13 land cover change processes (Table 3.1). For example, grasslands converting to a tree-covered area is considered harmful or negative change since it is generally woody encroachment (Luvuno et al., 2018). Conversely, tree-covered areas transforming to grassland were classified as grassland reestablishment, presuming the potential loss of native woody cover due to external drivers such as wildfires, herbivory, and excessive harvest by humans (Mograbi et al., 2015). Cropland areas that are abandoned are also vulnerable to woody species (e.g. IAPs) and loss of the productive ecosystem, and hence, a change from cropland to trees is also defined as negative change (Blair et al., 2018; Scorer et al., 2019), although there is a possibility for the passive restoration of croplands following abandonment (Benayas et al., 2007). Any artificial structure converting to any of the natural land covers is considered positive, considering naturalisation of IAPs and woody encroachment on artificial lands (Richardson et al., 2000). Introduction of surface water reservoirs (water bodies) from any land cover type except wetlands is perceived as an improvement process as reservoirs are intended to improve water security , although dam structures may negatively affect catchment water balance (Mantel et al., 2010; Ouyang et al., 2011).

Table 3.1: A matrix of changes, which shows the land degradation definition used for the assessment showing 42 possible transitions. Land cover state transitions are highlighted as degradation (red box), stable (amber), or improvement (green). In some cases, a transition may fall under improvement (green box) or stable (amber box), but due to the plugin's aggregation, the same transition may be considered harmful and thus such transitions are labelled in red ink.

2000 \ 2015	Tree-covered	Grassland	Cropland	Wetland	Artificial	Bare land	Water bodies
Tree-covered	Stable	Vegetation establishment	Agricultural expansion	Wetland establishment	Deforestation	Vegetation loss	Inundation
Grassland	Woody encroachment	Stable	Loss of vegetation	Wetland establishment	Urban expansion	Vegetation loss	Inundation
Cropland	Woody encroachment	Withdrawal of agriculture	Stable	Wetland establishment	Urban expansion	Vegetation loss	Inundation
Wetland	Woody encroachment	Wetland drainage	Wetland drainage	Stable	Wetland drainage	Wetland drainage	Inundation
Artificial	Afforestation	Vegetation establishment	Agricultural expansion	Wetland establishment	Stable	Withdrawal of settlements	Inundation
Bare land	Afforestation	Vegetation establishment	Agricultural expansion	Wetland establishment	Urban expansion	Stable	Inundation
Water bodies	Afforestation	Dry-up	Draining	Wetland establishment	Urban expansion	Dry-up	Stable

3.2.1.3. Sub-indicator 3: Soil organic carbon estimation

The plugin's default option estimates soil organic carbon stocks in 2000 from the SOILGRIDS250 dataset for the 0-30 cm soil depth (Hengl et al., 2017). Change in soil organic carbon from 2000 to 2015 is computed by estimating the combined impact of climate (based on climatic zone) and land-use change (based on the land cover change sub-indicator) in each pixel within the focal catchment (Conservation International, 2018b). Finally, the plugin aggregates the changes in soil organic carbon stocks over the assessment period to detect any false results, and any significant changes (soil organic carbon change over 10%) are recorded as improvement or degradation (Conservation International, 2018a). Potential false positives arise for example when tree-cover establishment in grassland areas leads to an increase in soil organic carbon stocks, but in terms of degradation, the process itself is desertification (UNCCD, 2018).

To estimate the soil organic carbon stocks, this study used the plugin's default option to estimate the topsoil soil organic carbon stocks that were derived from the SOILGRIDS250 project (Hengl et al., 2017). The choice of default soil organic carbon stocks dataset was made because the soil organic carbon stocks estimated by the National Terrestrial Carbon Sinks Assessment (NTCSA) was at a coarser spatial resolution (1 km) (DEA, 2015). Additionally, since the study area is a tertiary catchment, the coarse-scale dataset is less suitable.

3.2.2. Computing SDG 15.3.1: The proportion of degraded land

The plugin combines the three sub-indicators following the OOAO rule to provide the SDG 15.3.1 (Orr et al., 2017). The complimentary integration of the three sub-indicators is given in Table 3.2 meaning that for a pixel with any of the sub-indicators determined to be degraded, the pixel is considered as with any of the sub-indicators determined to be degraded, the pixel is considered degraded (Table 3.2). Improvement can be attained when all three sub-indicators improve or 1-2 sub-indicators are stable, and the rest are improved during the assessment period. A stable result for SDG 15.3.1 indicator can only be attained when all three sub-indicators are categorised as stable (Table 3.2).

Table 3.2: Complementary integration of SDG 15.3.1 metrics derived following the one-out all-out rule utilised by TRENDS.EARTH (Conservation International, 2019).

Productivity	**Land cover**	**Soil organic carbon**		**SDG 15.3.1**
Improvement	Improvement	Improvement		Improvement
Improvement	Improvement	Stable		Improvement
Improvement	Improvement	Degradation		Degradation
Improvement	Stable	Improvement		Improvement
Improvement	Stable	Stable		Improvement
Improvement	Stable	Degradation		Degradation
Improvement	Degradation	Improvement		Degradation
Improvement	Degradation	Stable		Degradation
Improvement	Degradation	Degradation		Degradation
Stable	Improvement	Improvement		Improvement
Stable	Improvement	Stable		Improvement
Stable	Improvement	Degradation		Degradation
Stable	Stable	Improvement		Improvement
Stable	Stable	Stable		Stable
Stable	Stable	Degradation		Degradation
Stable	Degradation	Improvement		Degradation
Stable	Degradation	Stable		Degradation
Stable	Degradation	Degradation		Degradation
Degradation	Improvement	Improvement		Degradation
Degradation	Improvement	Stable		Degradation
Degradation	Improvement	Degradation		Degradation
Degradation	Stable	Improvement		Degradation
Degradation	Stable	Stable		Degradation
Degradation	Stable	Degradation		Degradation
Degradation	Degradation	Improvement		Degradation
Degradation	Degradation	Stable		Degradation
Degradation	Degradation	Degradation		Degradation

3.2.2.1. Comparison of degradation results with national datasets

A comparison of the results obtained was made against high-resolution imagery (the 1990-2018 SANLC Change Assessment dataset) that is a locally validated dataset, as recommended by the 'Good Practice' guide for land cover change accuracy assessment, to check whether the estimates provided by the TRENDS.EARTH plugin are consistent with locally derived data (Olofsson et al., 2014; GTI, 2019a). The comparison used here followed the approach that is used to validate remote-sensing products, which is a three-step process (Stehman & Czaplewski, 1998). The three steps followed were the sampling design used, the response design used to obtain reference product for classification, and the estimation procedure (Stehman & Czaplewski, 1998). The comparison exercise utilised in this study focused on describing the overall accuracy of the degradation results estimated by the plugin in the Cacadu catchment (Stehman & Foody, 2019). The sampling points and the comparison products were projected to Africa Albers equal-area to avoid geolocation errors (Olofsson et al., 2014).

3.2.2.1.1. Sampling design for comparison

The comparison was based on 1 000 randomly selected points in each focal catchment using the *Sampling* geoprocessing tool in the ArcGIS platform (Figure 3.2). The option of randomly selecting points was to cover the most geographical area in each focal catchment (Figure 3.2).

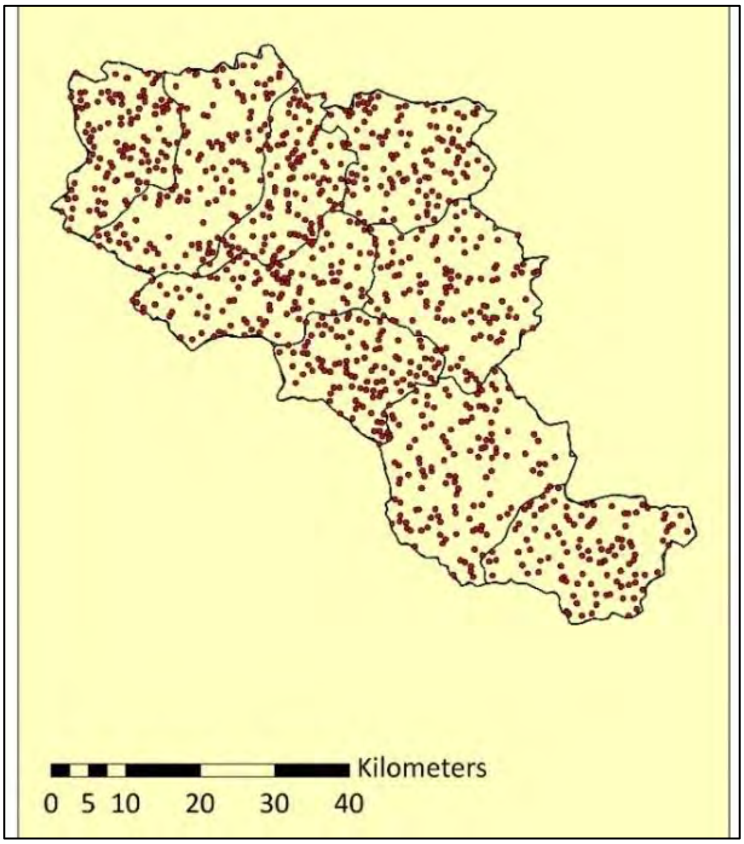


Figure 3.2: Sample distribution of copmarison points (n = 1000) in the Cacadu catchment.

3.2.2.1.2. Response design for comparison

The 1990 to 2018 South African NLC Change Assessment (SANLC Change Assessment; GTI, 2019a) product was used to compare the results of the TRENDS.EARTH evaluation. The SANLC Change Assessment product was chosen for its national acceptance, validation and accuracy (GTI, 2019a; Stehman & Foody, 2019). The SANLC Change Assessment product is a 30 m spatial resolution product created using land cover change estimates based on the 1990 to 2018 NLC datasets (GTI, 2016, 2019b). Since the land degradation datasets are medium resolution (250-300 m), the 30 m SALC Change Assessment product is a suitable response design product as per the Good Practise guidelines for accuracy assessment (Olofsson et al., 2014). The SANLC Change Assessment dataset is only a land cover change product, implying that it is not in line with the SDG 15.3.1 indicator (GTI, 2019a). However, as the only up-to-date and robust national product, it does have a utility in providing an idea of all the three sub-indicators that are combined to compute the SDG 15.3.1 indicator as a note by the dataset producers (GTI, 2019a). Hence the dataset is used as a reference tool to compare the SDG 15.3.1 indicator in this study.

The 1990-2018 SANLC Change Assessment dataset had 20 land cover classes, which were used to assign the corresponding degradation status based on the degradation definition (Table 3.2). The z-values were extracted using the *Extract Values to Points* geoprocessing tool in the ArcGIS platform to extract the values from the two datasets. The attribute tables were exported to Microsoft Excel for conducting the kappa statistics.

3.2.2.1.3. The estimation procedure for comparison

The frequency tables summarised the z-values for the three degradation states in comparison to the 2018 SANLC Change Assessment findings. Using the frequency tables, pivot tables were created to compute Cohen's kappa coefficient (K) (Cohen, 1960). The kappa coefficient (K) provides dataset reproducibility (i.e. reliability) by quantitatively measuring the degree of a joint agreement between two independent observations after a random agreement has been removed (Cohen, 1960). The kappa value is computed by following equation 3.1, which requires the percentage agreement (P_a) and the agreement by chance (P_e) between the products. Percentage agreement is the proportion of the diagonals in the matrix table, and P_e is the sum product of the off-diagonal entries (Cohen, 1960). The kappa value typically ranges between -1 and 1 (i.e. poor to excellent reliability). Using the kappa coefficient, the comparison procedure checked whether the overall accuracy exceeded the random chance agreement (i.e. H_0: $K = 0$).

$$\mathcal{K} = \frac{P_a - P_e}{1 - P_e} \qquad (3.1)$$

3.3. Results

3.3.1. Proportion of land degraded (SDG 15.3.1 indicator)

The spatial patterns of the results for the pixel RUE climate corrected SDG 15.3.1 degradation for the Cacadu catchment is illustrated in Figure 3.3. Most of the Cacadu catchment pixels (73%) represent a non-degraded state (Figure 3.3). Poor conditions were mostly detected in S10F, S10G, S10H and S10J (Figure 3.3). Most of the catchment was stable over the 15 years, while anthropogenic actions led to less than 10% of the landscape health recovery (Figure 3.3). The most recovery was detected in the upper reaches (S10A-C) and the lower reaches S10J (Figure 3.3).

Legend

Cacadu catchment

SDG 15.3.1 indicator

No data

Degradation

Stable

Improvement

	Area (ha)	Percent of total land area
Total land area:	292,472.8	100.00%
Improved	28,678.8	9.81%
Stable:	214,316.8	73.28%
Degraded:	49,101.0	16.79%
No data	376.1	0.13%

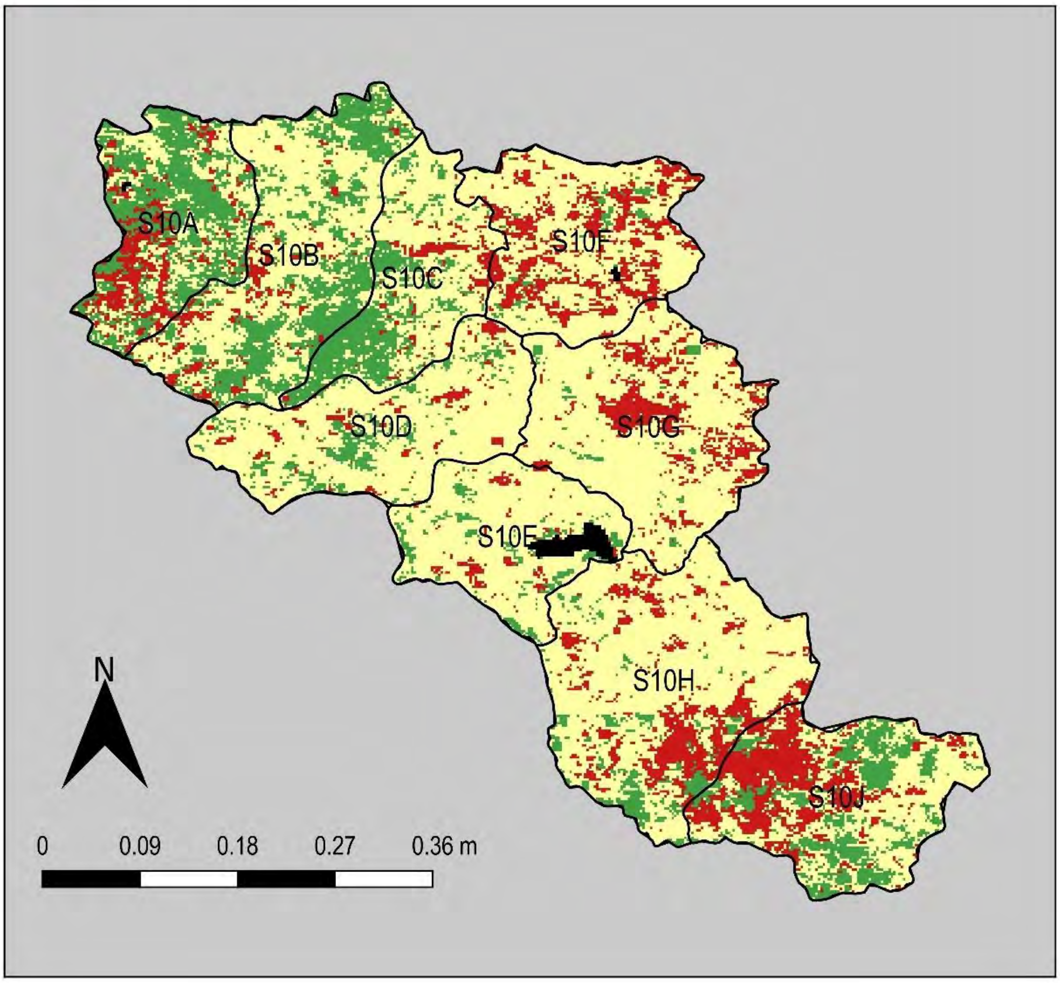


Figure 3.3 Patterns of human-induced land degradation/improvement in the Cacadu catchment at 300 m spatial resolution for the assessment period covering the years 2000 to 2015. The table show summaries of the land degradation indicator.

3.3.1.1. Sub-indicator 1: Land productivity

To estimate the anthropogenic impacts in the Cacadu catchment, the land productivity trends indicator (with potential rainfall bias) and the climate corrected land productivity (RUE-corrected land productivity) were computed and visualised side-by-side for the focal catchment

(**Error! Reference source not found.**). Both land productivity indices reveal that majority (>74%) of Cacadu catchment was stable, while 16% was in early stages of degradation under both land productivity indices (**Error! Reference source not found.**). The impact of human actions that account for the 16% moderate decline is evident in parts of S10A, S10F, S10G, S10H and S10J (**Error! Reference source not found.**B). A slight recovery due to human actions was also observed in S10A, S10E and S10J (**Error! Reference source not found.**B). The land productivity trends without climate correction (**Error! Reference source not found.**) noticeably underestimated improving land productivity status by 4.43% and overestimated declining land productivity status by a negligible 0.34% compared to the RUE adjusted outcome (**Error! Reference source not found.**B).

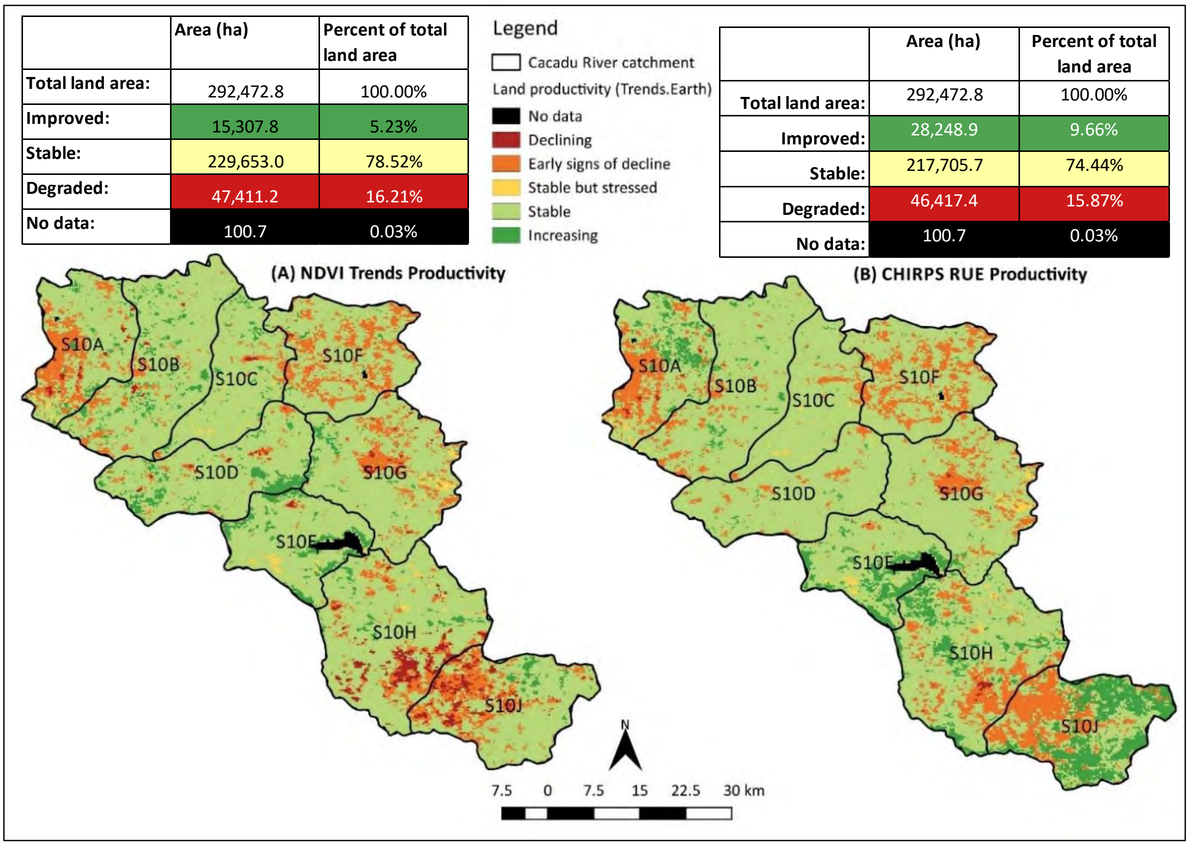


Figure 3.4: Results for land productivity (combined productivity measures) at tertiary catchment level for the years 2000 to 2015 at a coarse scale (250 m) in Cacadu catchment. Pixels are considered degraded if they are classified as “stable but stressed”, “early signs of decline” or “declining”. **A)** The land productivity result has not been corrected for climate influence. **B)** The land productivity result has been corrected for climate impacts using the rain use efficiency index (Wessels et al. 2007).

3.3.1.2. Sub-indicator 2: Land cover change

Figure 3.5 shows gains and losses per land cover class determined by the plugin over the 15-year assessment period, which expresses land cover degradation in Cacadu catchment. Based on the land cover transition analysis, a highly significant proportion of the catchment area was stable across all land cover classes (~99%), with a few pixels indicating negative changes (Figure 3.5).

Legend

Cacadu catchment

Land cover degradation (2000 to 2015)

No data
Degradation
Stable
Improvement

	Area (ha)	Percent of total land area
Total land area:	292,472.8	100.00%
Improved:	874.5	0.30%
Stable:	288,983.6	98.81%
Degraded:	2,614.7	0.89%
No data:	0.0	0.00%

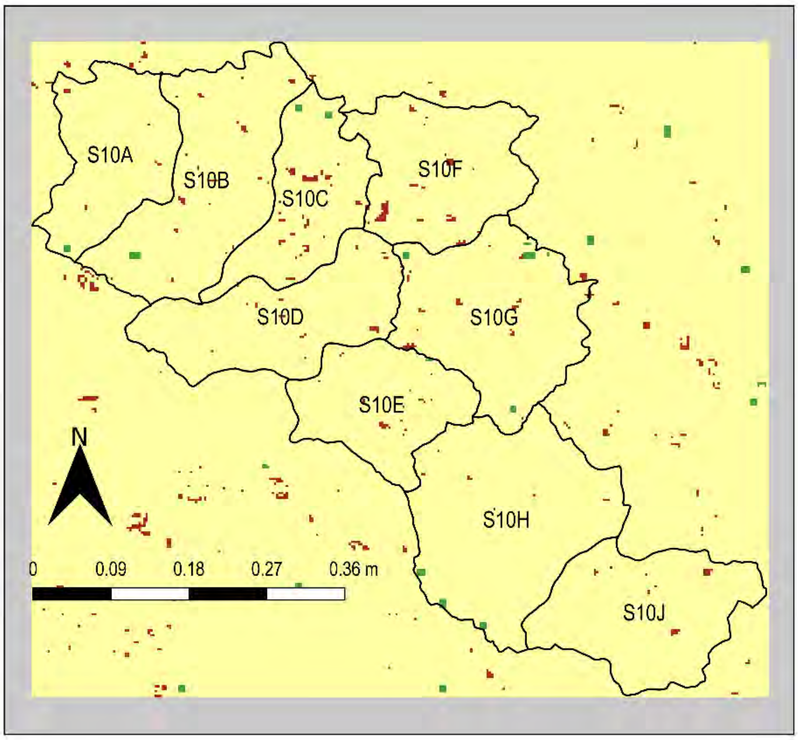


Figure 3.5: Land cover degradation for seven land cover classes in the Cacadu catchment 2000 and 2015 based on Euro-Space Agency satellite imagery. The table shows a summary of land cover degradation.

To gain a better understanding of the land cover change findings in the Cacadu catchment, the land cover transitions between the seven UNCCD land cover classes were organised in a matrix of area changes (ha) per focal catchment (Table 3.3). The majority of the land cover classes remained unchanged over the assessment period (Table 3.3). The tree-covered class mostly changed to the grassland class (Table 3.3), which is defined as an improvement in this study. Most grassland class changes were to croplands and tree-covered areas, both of which are defined as degradation in this study (Table 3.3). Artificial areas remained unchanged but dramatically expanded in areas that were tree-covered or grasslands (Table 3.3). Therefore, changes in grassland and tree-covered areas can explain the degraded and improved pixels in the above maps (Figure 3.5).

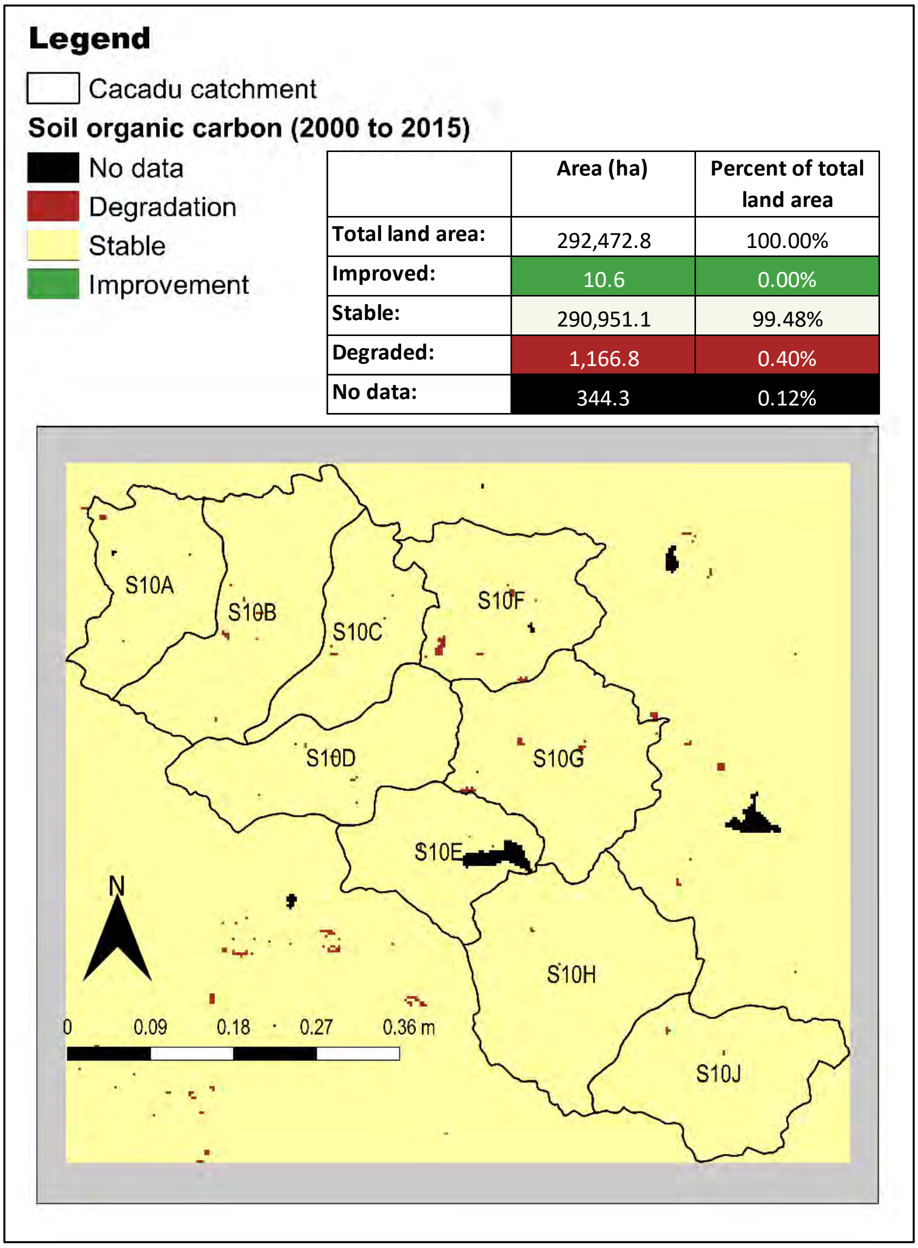
Table 3.3: Matrix showing Cacadu catchment land area by type of land cover transition (ha). Rows represented land cover classes in 2000, columns represent land cover classes in 2015, and fill colour represents the land cover change definition (Table 3.1).

UNCCD legend	Tree-covered	Grassland	Croplands	Wetlands	Artificial	Other lands	Water bodies
Tree-covered	14,174.8	869.2	0.0	0.0	37.1	0.0	0.0
Grassland	965.1	244,770.8	1,548.9	0.0	52.9	10.6	0.0
Croplands	0.0	0.0	29,799.7	0.0	0.0	0.0	0.0
Wetlands	0.0	0.0	0.0	10.6	0.0	0.0	0.0
Artificial areas	0.0	0.0	0.0	0.0	31.7	0.0	0.0
Other lands	0.0	5.3	0.0	0.0	0.0	195.9	0.0
Water bodies	0.0	0.0	0.0	0.0	0.0	0.0	1,551.7

3.3.1.3. Sub-indicator 3: Soil organic carbon

Figure 3.6 compares the average soil organic carbon stocks per land cover for the baseline year (2000) and the target year (2015) for the Cacadu catchment. Like the land cover degradation indicator, over 99% of the catchment was stable (Figure 3.6). However, unlike the land cover degradation results, half as many soil organic carbon pixels were estimated as degraded (Figure 3.6). Nonetheless, the degraded proportions were negligible (Figure 3.6).

The plugin averages the changes in soil organic carbon stocks on annual bases to show fluctuations in soil organic carbon per land cover class. Annual soil organic carbon stocks are presented as cumulative losses across the land cover class to provide additional insight into soil organic carbon stocks in the Cacadu catchment (Table 3.4, Figure 3.6). For the full assessment period, soil organic carbon stocks decreased slightly in the Cacadu catchment with the most loss approximating 15% in artificial surfaces (Table 3.5). Wetlands and tree-covered areas had more initial soil organic carbon stocks in the Cacadu catchment (soil organic carbon = 71.5 t/ha and 61.1 t/ha) in comparison to other land cover classes, while other lands had the least soil organic carbon stocks (soil organic carbon < 35 t/ha) (Table 3.4).



	Area (ha)	Percent of total land area
Total land area:	292,472.8	100.00%
Improved:	10.6	0.00%
Stable:	290,951.1	99.48%
Degraded:	1,166.8	0.40%
No data:	344.3	0.12%

Figure 3.6: Soil organic carbon stocks in the Cacadu catchment for the years 2000-2015.

Table 3.4: Estimates of soil organic carbon stocks (in tonnes per hectare) for each land cover class over 15 years (2000-2015) in the Cacadu catchment. The shaded area shows the initial carbon stocks; the cells below show changes relative to the year-2000.

Time (years)	SOC by LCC (tonnes/ha)					
	Tree-covered	Grassland	Cropland	Wetland	Artificial surfaces	Other lands
2000	61.07	48.03	42.71	71.50	41.00	33.28
2001	0.00	0.00	0.00	0.00	0.00	0.00
2002	0.00	0.00	0.00	0.00	0.00	0.00
2003	0.00	0.00	-0.01	0.00	0.00	0.00
2004	0.00	-0.01	-0.01	0.00	0.00	0.00
2005	0.00	-0.01	-0.01	0.00	0.00	0.00
2006	0.00	-0.01	-0.01	0.00	0.00	0.00
2007	0.00	-0.01	-0.01	0.00	0.00	0.00
2008	0.00	-0.01	-0.02	0.00	0.00	0.00
2009	-0.01	-0.01	-0.02	0.00	0.00	0.00
2010	-0.01	-0.02	-0.02	0.00	0.00	0.00
2011	-0.01	-0.02	-0.02	0.00	0.00	0.00
2012	-0.01	-0.02	-0.02	0.00	0.00	0.00
2013	-0.01	-0.02	-0.03	0.00	0.00	0.00
2014	-0.01	-0.02	-0.03	0.00	0.00	0.00
2015	-0.02	-0.02	-0.03	0.00	0.00	0.00

Comparing soil organic carbon for the initial and target years indicated no change in any land cover types (Table 3.5). Land cover change in the Cacadu catchment led to a significant loss of soil organic carbon stocks (soil organic carbon > 10%) in tree-covered areas (Table 3.5).

Table 3.5: Soil organic carbon change at initial and target years represented as relative frequencies of initial carbon stocks, accounting for all land cover transitions that occurred during the 15 years for each land cover class in the Cacadu catchment.

Land cover conversion		Net SOC change (%)
From	To	
Tree-covered areas	Tree-covered areas	-0.03
	Grassland	0.00
	Croplands	0.00
	Artificial areas	-15.17
Grassland	Tree-covered areas	0.00
	Grassland	-0.03
	Croplands	-4.50
	Artificial areas	-6.74
	Other lands	0.00
Croplands	Tree-covered areas	0.00
	Croplands	-0.08
Wetlands	Tree-covered areas	0.00
	Wetlands	0.00
Artificial areas	Artificial areas	0.00

3.3.2. Satellite data comparison: SDG 15.3.1 vs SANLC Change 1990-2018

After the SDG 15.3.1 indicator was computed from the plugin, a comparison was made between the SDG 15.3.1 indicator and the SANLC Change dataset using a Cohen's kappa statistic. The error matrix presented as Table 3.6 summarises the key data obtained from the SDG 15.3.1 indicator and the 2018 SANLC Change Assessment dataset. Inconsistence spatial scales and land cover changes in years after 2015 may have led to discrepancies between the SDG 15.3.1 indicator and the 1990-2018 SANLC Change product. In both 1990-2018 SANLC Change product and the SDG 15.3.1 indicator product, the largest proportion of comparison points was covered by stable pixels showing (85% and 72% of comparison points as stable) (Table 3.6). Comparing the SDG 15.3.1 indicator and 1990 to 2018 SANL Change product, matched 2.2% of degraded points, 61.6% of stable points and 0.2% of improved points (Table 3.6). The diagonal entries represent the agreement between the two products, showing a higher agreement in non-degraded observations (i.e., stable and improved observations) than the degraded state (Table 3.6). The differences in spatial and temporal resolutions of the two products were highlighted by the slight mismatch of pixels indicators in the two products, which had a 6% to 13% difference range, (Table 3.6).

Cohen's kappa was computed to assess the allocation agreement between two products (the land degradation dataset and the SANLC Change Assessment dataset) in 1 000 sample points in the Cacadu catchment. There was a slight agreement between the two datasets with K = 0.011 for the Cacadu catchment, while the overall agreement (*P_a)* between the datasets was 0.64, which falls within the 0.61 to 0.99 acceptance kappa agreement range.

Table 3.6: Cohen’s comparison error matrix table for the Cacadu catchment based on 1 000 sampling selected points. The shaded cells represent an agreement between the two products.

		2000-2015 SDG 15.3.1 indicator					
		Degraded	**Stable**	**Improved**	**No data**	**Sum**	**Percent**
1990-2018 LCC	**Degraded**	22	77	15	2	116	11.6%
	Stable	150	616	76	11	853	85.3%
	Improved	2	22	2	1	27	2.7%
	No data	0	4	0	0	4	0.4%
	Sum	174	719	93	14	1000 points	
	Percent	17.4%	71.9%	9.3%	1.4%		

The land cover definition (Table 3.1) was used to create Table 3.7, which details the land cover degradation in the Cacadu catchment based on the 1990-2018 SANL Change dataset. The prevalence of neutral land cover between 1990 and 2018 was the main characteristic feature in the catchment, with over 90% persistence in S10A, S10B and S10F (Table 3.7). Like the SDG 15.3.1 indicator (Figure 3.3) and the land cover sub-indicator (Figure 3.5), clearing of tree-covered areas and grassland recovery were the main mechanisms behind nearly 10% of the Cacadu catchment land cover improvement (Table 3.7). The highest land cover degradation fell within the 10-90% margin due to loss of grasslands, and was detected in S10A, S10C, S10E, S10G, S10H and S10J (Table 3.7), which partly coincides with the SDG 15.3.1 indicator (Figure 3.3B). Surprisingly, Table 3.7 shows only 8% of land cover degradation in the S10F catchment which contrasts Figure 3.3.

Table 3.7: Tabular summary of the 1990-2018 South African National Land Cover Change Assessment showing the land cover transitions of the Cacadu catchment (ha) that were derived using the land degradation definition (Table 3.2). The table is organised according to land cover processes: **top =** improvement; **middle** = persistence; **bottom** = degradation. The ink colour for totals depict range of change, **black** = 0-10%; **orange** = 10-90%; **red** = 90-100%.

Catchment	S10A	S10B	S10C	S10D	S10E	S10F	S10G	S10H	S10J
Grassland recovery	22.05	181.89	268.56	177.21	300.33	218.52	254.61	335.61	220.23
Inundation	5.31	7.38	2.61	1.98	33.75	4.41	3.33	46.71	51.66
Recently cleared	86.94	280.53	180.45	488.07	393.48	209.52	813.69	617.76	535.68
Loss of Plantations	9.63	3.69	0.63	1.35	4.41	11.16	22.32	0.54	3.78
Loss of Settlements	0.36	10.53	16.47	13.50	13.86	11.07	26.55	25.02	4.05
Loss of Waterbodies	25.29	11.07	0.99	0.27	128.16	15.12	4.59	5.85	21.87
Loss of Other land	0.54	28.35	23.67	12.24	17.82	8.55	12.78	8.28	4.05
Total improved	***197.91***	***523.44***	***493.38***	***694.65***	***891.81***	***478.35***	***1137.87***	***1039.77***	***841.32***
Stable Croplands	1168.11	4411.35	3647.79	3775.86	3566.25	5971.23	4828.14	6392.61	3208.23
Stable Grasslands	21336.93	29834.28	14111.01	21544.83	12200.58	18989.10	22770.72	30570.12	18541.53
Stable Other land	66.24	130.23	541.35	104.94	841.32	276.84	494.28	909.45	176.04
Stable Plantations	2.97	1.89	0.09	1.80	10.62	11.70	23.49	0.00	8.10
Stable Settlements	30.15	1335.78	1428.84	2661.66	1430.28	1898.10	2892.96	2405.88	1704.78
Stable Tree-covered	2.16	30.42	9.09	119.07	74.25	6.30	233.91	118.62	112.41
Stable Waterbodies	67.32	18.99	5.49	0.09	1152.36	95.76	0.45	3.42	8.82
Stable Wetlands	440.64	56.79	3.33	10.98	0.72	14.31	20.43	62.19	1.53
Total stable	***23114.52***	***35819.73***	***19746.99***	***28219.23***	***19276.38***	***27263.34***	***31264.38***	***40462.29***	***23761.44***
Loss of Grasslands	17,11	25,34	22,04	23,39	36,17	19,50	48,62	50,66	70,36
Loss of Wetlands	5,18	1,69	0,26	0,34	0,12	1,22	0,49	0,73	0,41
New Croplands	0,01	6,26	9,96	1,36	0,52	0,97	1,84	0,69	0,21
New Plantations	0,01	0,00	0,00	0,05	0,00	0,00	0,02	0,00	0,00
Abandoned croplands	2,63	1,98	1,68	2,47	1,84	1,84	2,26	5,82	5,82
Plantations	0,00	0,00	0,00	0,17	0,00	0,02	0,03	0,00	0,00
Afforestation	0,01	0,02	0,02	0,10	0,08	0,01	0,04	0,03	1,20
Total decline	***24,95***	***35,28***	***33,96***	***27,88***	***38,73***	***23,55***	***53,29***	***57,93***	***78,00***

3.4. Discussion

This study sought to assess the environmental status of an upstream rural catchment by combining the SDG 15.3.1 sub-indicators using the recently produced TRENDS.EARTH plugin (Conservation International, 2018a). The plugin combines the three sub-indicators using the one-out, all-out statistical rule prescribed in the land degradation neutrality framework (Orr et al., 2017). Of the three sub-indicators in the Cacadu catchment, the most degradation was detected in the land productivity sub-indicator. Consequently, the SDG 15.3.1 indicator suggests that the Cacadu catchment was moderately degraded [the extent of land area degraded ranges between 10-25% following FAO, (2002) degradation definition] between 2000 and 2015.

Land productivity was the dominant type of degradation affecting the human-induced degradation status of the focal catchment, leading to 491 ha (16.79%) of the degraded area. Since vegetation covers most of the terrestrial ecosystem, abnormalities in vegetation productivity have been used as an indicator of degradation at different scales in different parts of the world (Bai et al., 2008; Fensholt & Rasmussen, 2011; Bennett et al., 2012; Wessels et al., 2012; Graw et al., 2017; Hoffman et al., 2018). The use of land productivity indicator alone for degradation assessment has been criticised for overlooking other forms of degradation (Gibbs & Salmon, 2015). Thus this study took into consideration two other degradation indicators as prescribed by the land degradation neutrality conceptual framework (Orr et al., 2017; Gonzalez-Roglich et al., 2019). Climate effects were removed in this study, and the significant decline detected for land productive capacity could be an indication of ecological response to plant phenological change (the seasonal timing for plant changes) (De Jong et al., 2011).

Soil erosion is also likely to encourage a decline in land productivity (Chappell et al., 2019). South African landscapes, especially in the Eastern Cape, Northern Cape and KwaZulu-Natal, which have a long legacy of soil erosion as a major natural and human-induced geohazard (Le Roux, 2011). Soil erosion can facilitate fertility and sediment transfer in the short term, and it can lead to irreversible soil damage, threaten water resources and compromise grey infrastructure (Lal, 1998). The discussion above explains the prevalence of degradation in the hilly region areas (S10A, S10C, S10E, S10F) of the Cacadu catchment since most of the soils in the catchment are highly susceptible to soil erosion, with soil erodibility values ranging between 0.5 and 0.7 (i.e. highly erodible) (Schulze et al., 2007). In a slightly similar catchment (the Tsitsa catchment), the highly erodible duplex soils have led to widespread development of gully soil erosion (van der Waal & Rowntree, 2018).

Studies focusing on drought reported an increasing trend of droughts in the 21st century in various regions, including South Africa and parts of the Cacadu catchment (Dai, 2013; Malherbe et al., 2016). Persistent drought conditions lead to carbon and nitrogen imbalance in terrestrial ecosystems such as grasslands (Coners et al., 2016). The drought impact as a driver of degradation is because water and productivity-limited grasslands are more vulnerable to aridity (Ponce-Campos et al., 2013) as the phenology of grassland ecosystems primarily depends on water (Hao et al., 2019). Therefore, even though the degradation process in the Cacadu catchment is mostly attributed to human actions, the long-term impacts of climate extremes cannot be overlooked as this would affect the natural grassland recovery process.

The detected degradation in the Cacadu catchment could also be driven by desertification - the replacement of natural grassland cover by forb species, invasive succulents and karraiod shrubs such as the *E.floribundus* plant (Mucina & Rutherford, 2006; Shackleton & Gambiza, 2008). The recent National Biodiversity Assessment also stressed the desertification process (in terms of species changes) in grassland areas through the loss of C_4 grassland species and the establishment of Savanna vegetation (Skowno et al., 2019). The findings for the mesic grassland parts of South Africa in the recent National Biodiversity Assessment, which was conducted at a fine scale (30 m resolution), showed that the dominant vegetation taxa in the Cacadu catchment (i.e. the Tsomo Montane Shrubland) have declined by 4% at a rate of -5.74% y^{-1} between the years 1990 and 2014 (Skowno et al., 2019: 38). The degradation results in this study and the grassland biodiversity loss reported in Skowno et al. (2019) support earlier field-based studies in the S10F quaternary catchment (Blane, 2017; Molepo, 2017; Chappel, 2018). Blane (2017) conducted an ecological survey of the rangelands and found that the invading *E. floribundus* woody species dispersal had doubled from 2006 to 2017 and could have led to a decline in native grass species in the Machubeni communal area. Molepo et al. (2017) assessed the rangelands' condition and functionality using the line point assessment method and the Landscape Functionality Analysis method from 200 transect points. They found a significantly low proportion of palatable grass species and extensive evidence of rangeland degradation due to sheet erosion and overgrazing. A separate postgraduate study (Chappel, 2018) identified degradation in a third of grasslands at a landscape scale using a landscape organisation analysis.

3.4.1. Limitations and conclusions

This study was conducted over a 15-year assessment period; however, the literature suggests that soil organic carbon reaches equilibrium after 20 years (Penman et al., 2003: 2.7). Therefore, the assessment period might have limited the detectability of soil organic carbon stock degradation. Penman et al.'s (2003) estimation that the soil organic carbon stock can reach equilibrium after 20 years was first determined for below-ground tropical forest ecosystems (Malhi et al., 1999), where the soil organic carbon stock residence time recorded was 29 years. However, the 20-year threshold might not apply to cropland areas due to frequent disturbance. Vågen et al. (2005) suggest that for cropland areas, the change in soil organic carbon stock could be observed in as little as three years. However, in a study in KwaZulu-Natal, Sithole et al. (2019) found that the soil organic carbon stocks in cropland areas are more dynamic in the top 0-10 cm than the IPCC recommended 30 cm minimum depth, which could further complicate the plugin’s ability to detect soil organic carbon stock dynamics for croplands during the 15-year assessment period in this study.

This study was conducted at a tertiary catchment scale; therefore, the accuracy may be reduced in other land classes. For instance, the European Space Agency land cover dataset is a globally consistent and quick mechanism to classify land cover change at 300 m resolution (EuroSpace Agency, 2018; UNCCD, 2018). Thus the larger and higher contrast land cover classes (e.g. grasslands, tree-covered areas, croplands, and in some cases bare regions) have a higher probability of detection compared to smaller land cover classes such as wetlands (Ludwig et al., 2016; Liu et al., 2018). Discrepancies due to spatial resolution are evident when comparing the SDG 15.3.1 indicator and the 1990-2018 SANLC Change product, where the S10F catchment was omitted amongst those with considerably degraded pixels (section 3.3.2). Such inconsistencies are one reason the plugin was designed flexibly enough to allow for the use of custom data that is nationally acceptable to modify the land cover definition for context relevance (Conservation International, 2018a). However, this study could not use custom data from South African datasets covering the years 1990 – 2018 because the earliest baseline year for the plugin was 2000 and not 1990. Nevertheless, the SANLC Change Assessment product was important for comparison, as it showed some agreement with the plugins default datasets. Therefore, this study recognises the limited ability of globally derived and coarse-scale resolution datasets to accurately detect tertiary catchment scale changes, reducing the reliability of findings as discussed in a Namibian setting using the TRENDS.EARTH toolbox (Mariathasan et al., 2019). Mariathasan et al. (2019) detected inconsistencies between the reported woody proliferation in Namibia and the SDG 15.3.1 indicator, which is unsurprising because they never modified the land degradation definition to match the Namibian biosphere context.

In closing, based on three land degradation indicators (i.e. land productivity, soil organic carbon and land cover change) in the semi-arid grassland region of South Africa, moderate land degradation of the landscape was determined in the study catchment. Most Cacadu catchment areas are not significantly affected by human actions, and the degraded areas are not catchment-wide, nor do they follow the spatial gradient. Instead, the degradation process in the catchment is localised, probably at sub-catchment levels. Land cover and soil organic carbon stocks largely remained unchanged, while land productivity showed a declining trend, possibly due to natural and human-induced stress. Consequently, land productivity changes influenced the degradation results obtained, suggesting a new degradation process through a moderate reduction in biomass productivity. Therefore, with the direction of change pointing towards land degradation and less towards recovery, the findings from this study emphasise the need to adopt management interventions in rural grassland ecosystems to protect the security of vulnerable rural communities. The approach used in this chapter is generalisable and can be used in other similar settings, but caution has to be paid to the limitations. Beyond the assessment itself, a further step (i.e. ground-truthing) would be a valuable addition.

4. CHAPTER 4: USING MULTI-CRITERIA ANALYSIS TO PRIORITISE RESTORATION AREAS TO INCREASE WATER FLOW REGULATION IN RURAL CATCHMENTS: CASES FROM CACADU CATCHMENT

This chapter is being prepared for submission in Restoration Ecology as: Xoxo, BS, Mahlaba B, Mantel SK, De Vos A, Le Maître DC Tanner J, ,. Using multi-criteria analysis to prioritise restoration areas to increase water flow regulation in rural catchments: Cases from Cacadu and Tsitsa catchments.

Conceptualisation, BSX, WM, SKM, ADV; Methodology, BSX, WM, SKM, ADV,DLM; Investigation: BSX (Cacadu and Upper Crocodile catchments), BM (Tsitsa catchment); Formal Analysis, BSX, BM; Resources, SKM, JT; Writing - Original draft preparation, BSX; Writing – Review and Editing, SKM, ADV, JT, DLM; Visualisation, BSX; Supervision: SKM, JT, ADV; Project administration: SKM, JT; Funding acquisition, SKM, JT, BM, BSX.

4.1. Introduction

Ecosystem degradation threatens over 41% of humans globally and has contributed to over 17% of global water stress (UN, 2020). The clustering of rural communities also exacerbates acute impacts of landscapes prone to degradation, which has exposed nearly 40% of rural population Southern African countries to poverty between 2000 and 2010; this number is expected to increase if not disrupted (Barbier & Hochard, 2016). Therefore, ecosystem restoration is a key strategy to improve landscape resilience and its services (Blignaut et al., 2014; Akhtar-Schuster et al., 2017). Restoration is defined as a procedure of facilitating ecosystem recovery from degradation, modification or destruction to reference site conditions (Aronson & Alexander, 2013a; Orr et al., 2017). The land degradation neutrality framework provides three actions for restoration, (i) reverse degraded land, (ii) reduce ongoing degradation, (iii) and avoid severe degradation (Orr et al., 2017). To promote action, the 15th Aichi Biodiversity Target aimed at restoring 15% of global ecosystems by 2020, but half of the global signatories have missed the target (CBD, 2020). In practice, the United Nations has ratified the period 2021 – 2030 as a time of action towards SDG 15.3 in what has been named The Decade for Restoration (United Nations, 2019).

Although restoration is favoured for environmental improvement and societal benefits, hindrances arise due to vague restoration targets and diverse societal values for natural resources (Tallis et al., 2008; De Groot et al., 2010). The Sloping Land Conversion Program in China sought to reclaim soil erosion by introducing alien woody plants (Tallis et al., 2008). Instead of the intended outcomes, the interventions led to increments in groundwater abstractions and reduced native biodiversity cover, which is a sign of unfavourable conditions due to vague restoration targets that may later impact ecosystem service supply. An emphasis on biodiversity recovery symbolises restoration planning improvements without compromising ecosystem service supply (Gann et al., 2019). In support of the ecosystem service approach, Bullock et al. (2011) and Sapkota et al. (2018) contest that biodiversity targeted restoration alone does not always translate into the enhancement of ecosystem services; instead, restoration should be targeted at how the biodiversity-ecosystem service interplay could react to management interventions.

Finding ways for attaining the highest level of ecosystem recovery given the degradation problem and limited funds available for ecosystem restoration is challenging and thus, a focal area for restoration planning (Allan et al., 2013; Aronson & Alexander, 2013b; Angelstam et al., 2017). To aid decision making, the Land Degradation Neutrality framework encourages local buy-ins for increasing the chances of restoration projects (Reed et al., 2008; Orr et al., 2017). Progress has been made in terms of prioritisation as South Africa recently undertook using the SLM planning to address the land degradation problem at a national level in 2017/18 (DEA, 2018; von Maltitz et al., 2019). Sustainable Land Management involves an integrated approach based on four principles, “targeted policy and institutional support”, “land-user-driven and participatory approaches”, “the integrated use of resources across the landscape and ecosystem-scale”, and “multilevel, multi-stakeholder involvement and partnerships” (Reed et al., 2015; Gann et al., 2019). The ecosystem transformation actions needed to halt the impacts of degradation have been outlined in the national land degradation neutrality target setting (DEA, 2018). In brief, the national target setting for land degradation neutrality focuses on reversing grassland degradation, improving cropland activity, restoring and protecting wetland ecosystems, and improving the flow augmentation function from artificial areas (DEA, 2018). Adopting SLM practises has been recommended in the land degradation neutrality framework to achieve productive and healthy ecosystems by including all spheres of social-ecological systems (Orr et al., 2017). However, one of the challenges with SLM planning and achieving the land degradation neutrality targets lies in the complex governance nature of traditional communities, complicating the collaborative planning and management processes (Fabricius & Cundill, 2010; Israel & Wynberg, 2018; Gann et al., 2019).

As Tengö et al. (2014) stressed, restoration planning can benefit from adopting an approach that can help unpack the social-ecological drivers and feedback to make informed decisions about governing landscapes. This chapter prioritises EI resources for restoration to improve the drought mitigation function of the Cacadu catchment. The relevance of the AHP is its ability to integrate multiple evidence sources, including reaching an acceptable decision under complex conditions in one model (Saaty, 1990, 2000; Ananda & Herath, 2003; Huang et al., 2011). The ability of the AHP process to handle mixed-methods can benefit restoration planners through providing relevant knowledge that can uncover ecosystem trade-offs and synergies that will aid the success of ecosystem protection (Brown & Fagerholm, 2015; Gonzalez-Roglich et al., 2019). Secondly, the AHP can create a collaborative and learning space for making decisions (Belton & Stewart, 2002: 1), which can help establish long-term partnerships that are vital for the sustainability of restoration projects (Angelstam et al., 2017). This study applies the AHP approach at an EI category level based on four focal EI resources (see section 4.1.1). Even though assessments on EI investments have been conducted in other catchments (Forsyth et al., 2011, 2012; Hughes et al., 2018a; Snaddon et al., 2018), those catchments are mostly urban to semi-urban, meaning the recommendations may not be necessarily applicable to rural catchments. The outputs from this study are expected to be beneficial for the SLM planning in the ongoing restoration projects around the focal study areas.

4.1.1. The rationale for selecting the focal EI categories studied

The most important aspect of flow regulation is runoff partitioning into quickflow or subsurface flow to sustain streamflow (Brauman et al., 2007; Le Maître et al., 2014). The capacity of catchments to maintain surface water flow is influenced mainly by geology and land cover since these two catchment properties determine the amount of infiltration that could occur (Le Maître et al., 2014). Land cover is the most dynamic of these two (Foley et al., 2005; van Loon & Laaha, 2015). Therefore, this study focused on naturally occurring ecosystems that can be restored to play a role in flow regulation and consequently mitigate the impacts of droughts on communities (SANBI, 2014). The focal EI resources that were chosen for this study are wetlands, riparian margins, abandoned cultivated fields and grasslands. These four EI categories (Table 4.1) were chosen based on the ecosystem services they provide to communities, a decision that echoes the importance of EI for public ecosystem service provision as discussed by Kubiszewski et al. (2017). Examples of ecosystem services provided by the four focal EI resources and EI investment approaches that could facilitate drought mitigation are outlined in Table 4.1. All these EI categories support an essential ecosystem service of flow regulation when sustainably managed (SANBI, 2014; Le Maître et al., 2016; Hughes et al., 2018b). Investing in the four EI resources also supports many other ecosystem benefits to rural communities (Table 4.1). Provisional benefits for rural communities include water, food, energy, and livestock fodder provision (Table 4.1). The most relevant regulating function of rural landscape are water-related services such as water quality and sediment control and flow regulation (Table 4.1). Aesthetic, cultural and religious values are the most relevant cultural services to rural communities (Table 4.1).

Wetlands were selected because they are ecosystems that provide or retain water resources (in addition to various other ecosystem services such as sediment trapping, water and habitat regulation) but are vulnerable to modification by humans (Rebelo et al., 2015; Pantshwa & Buschke, 2019). The 2018 National Biodiversity Assessment (NBA) highlights that wetland ecosystems in South Africa are the most threatened of all national ecosystems (Skowno et al., 2019). The ecological integrity of wetlands is impacted mainly by anthropogenic impacts such as the conversion of wetlands for cultivation and mining, which reduces the ability of wetlands to perform crucial ecosystem functions such as flow regulation for slow release into streams or subsurface (Foley et al., 2005; Rebelo et al., 2015).

Riparian margins represent catchment drainage systems that connect rivers with the catchment. By nature, riparian zones have diverse vegetation that is crucial for streambank stabilisation, habitat provision, flood and sediment regulation, and water filtration, among other services (Salemi et al., 2012; Acuña et al., 2013). Similar to wetlands, the 2018 NBA results indicate that river ecosystems are highly threatened due to IAP naturalisation and soil erosion, which reduces their functionality (Hoffman & Rohde, 2011; Acuña et al., 2013; Modiba et al., 2017; Ntshidi et al., 2018; van Deventer et al., 2018).

Abandoned croplands and grasslands were chosen because they are a significant component of the land cover and they are vulnerable to invasion by IAPs (Sigwela et al., 2017; Blair et al., 2018; Scorer et al., 2019), woody encroachment (Stafford et al., 2017; Luvuno et al., 2018) and soil erosion (van der Waal & Rowntree, 2018). Cropland areas represent semi-natural land covers that have been converted to increase food production (MEA, 2005). However, climate extremes, poor farming practices and changes in socio-economic factors have led to cropland abandonment (Blair et al., 2018; Shackleton et al., 2019). Cropland abandonment may lead to the reestablishment of natural vegetation (Benayas et al., 2007), or degradation in the form of soil erosion (Bai et al., 2007; van der Waal & Rowntree, 2018), invasion of exotic species and IAP (Scorer et al., 2019). Active croplands contribute to enhanced water infiltration due to tillage and higher evapotranspiration due to increased soil moisture presence (Twomlow & Bruneau, 2000; Gyamfi et al., 2016). Abandoned croplands contribute to reduced infiltration and lower evapotranspiration when naturally vegetated but higher evapotranspiration when covered with non-native biomass (Foley et al., 2005). Therefore, the presence of abandoned croplands in the focal catchments could imply a reduction in the drought mitigation capacity of the catchment (Le Maître et al., 2014).

Grasslands make up the second-largest biome nationwide and are crucial for livestock production, and they facilitate infiltration in catchments (Mucina & Rutherford, 2006; Eldridge et al., 2011). At a biome scale, 22% of grassland ecosystem types are threatened, making grasslands one of the top three biomes with the highest number of threatened ecosystem types (Skowno et al., 2019). The vulnerability of grasslands to degradation can be attributed to soil erosion and infestation by exotic plants (Mills & Fey, 2003; van Wilgen et al., 2008). Degradation of grasslands contributes to siltation of watercourses, increased overland flow, and consequently reduced streamflow during dry seasons and drought periods (Gyamfi et al., 2016; Le Maître et al., 2019; Gao et al., 2020).

Table 4.1: Focal resource definitions and their link with the SANBI framework. The SANBI framework is discussed earlier in Chapter 1 & 2. The italicised ecosystem services examples are the most relevant for rural contexts.

Focal EI resources	**Description**	**Fit with the SANBI framework**	**Ecosystem service examples**		
			Provisioning	**Regulating**	**Cultural**
Wetlands	Areas that contain alluvial vegetation including Shallow-water ecosystems, marshes, ponds and flood plains	Rehabilitating and protecting wetlands	*Livestock fodder; Food production*; *Harvested renewable resources; Water provision*	Carbon storage; Regulation of local climate, pests and pathogens; *Regulation of soil erosion, water quality and water flow*; Habitat control	*Aesthetic values; Cultural and religious value*; Non-use values; Scientific and educational values
Riparian margins	Transitional areas between land and water	Clearing of IAP; Improving landscape management practices			
Abandoned croplands	Areas formerly used for crop production other than homestead gardens	Clearing of IAP; Improving landscape management practices			
Grasslands	Areas within the Grassland biome that could be degraded or proliferated by IAPs, woody encroachment	Clearing of IAP; Improving landscape management practices; rehabilitation of grasslands			

4.2. Methods

4.2.1. Ecological infrastructure extent in focal catchments from spatial databases

Various national datasets were used to identify and compile the spatial database of focal EI resources (DWS, 2006; GTI, 2015, 2019b; CSIR, 2018). Wetlands were obtained from the fifth National Wetland Map, which was part of the 2018 NBA (van Deventer et al., 2020). Wetlands make up about 1% of the total land cover area in the catchment (Figure 4.1). Wetlands are distributed mainly in the upper and lower parts of the Cacadu catchment, with a few located in the middle catchments (Figure 4.1). Most wetlands are found along the floodplains (Figure 4.1). Riparian margins are not presented in the 2014 NLC dataset and the river ecosystem type layer even though the methodology for delineating riparian ecosystems existed (DWAF, 2008).

Riparian margins in this study were derived by applying a 150 m buffer to the national river network dataset hosted on the Department of Water Affairs website (DWS, 2006). This buffer is in line with 100 – 500 m the buffer range used while producing the Atlas for Water Production (Nel et al., 2011). The Cacadu catchment is well-drained, with a river network that runs across the catchment (Figure 4.1).

Abandoned croplands were extracted from the 10 m Spot image-derived dataset for the Eastern Cape Province (DAFF, 2015). Cropland abandonment is prevalent in the Cacadu catchment, with over 6 000 ha abandoned in the Cacadu catchment (Figure 4.1). Abandoned croplands are distributed along the valleys and floodplains (Figure 4.1).

The grasslands EI category was based on the grassland and natural shrubland land cover areas that were extracted from the 30 m 2014 national land cover dataset for its consistency with the SDG 15.3.1 assessment period (GTI, 2015). The shrubland land cover was included as grasslands because grassland vegetation is naturally composed of shrubs (Mucina et al., 2006). Most of the Cacadu catchment is covered by grasslands, with patches of transformed land covers in the valleys (Figure 4.1).

In relation to Chapter 3, there was no evidence of degradation between 2000 and 2015 for most portion of the EI resources (Figure 4.1). Over 20% of all wetlands and abandoned croplands were degraded, while grasslands and riparian margins accounted for a lesser extent of degradation (Figure 4.1). Improvement across all four EI resources was below 10%, emphasising the need for restoration

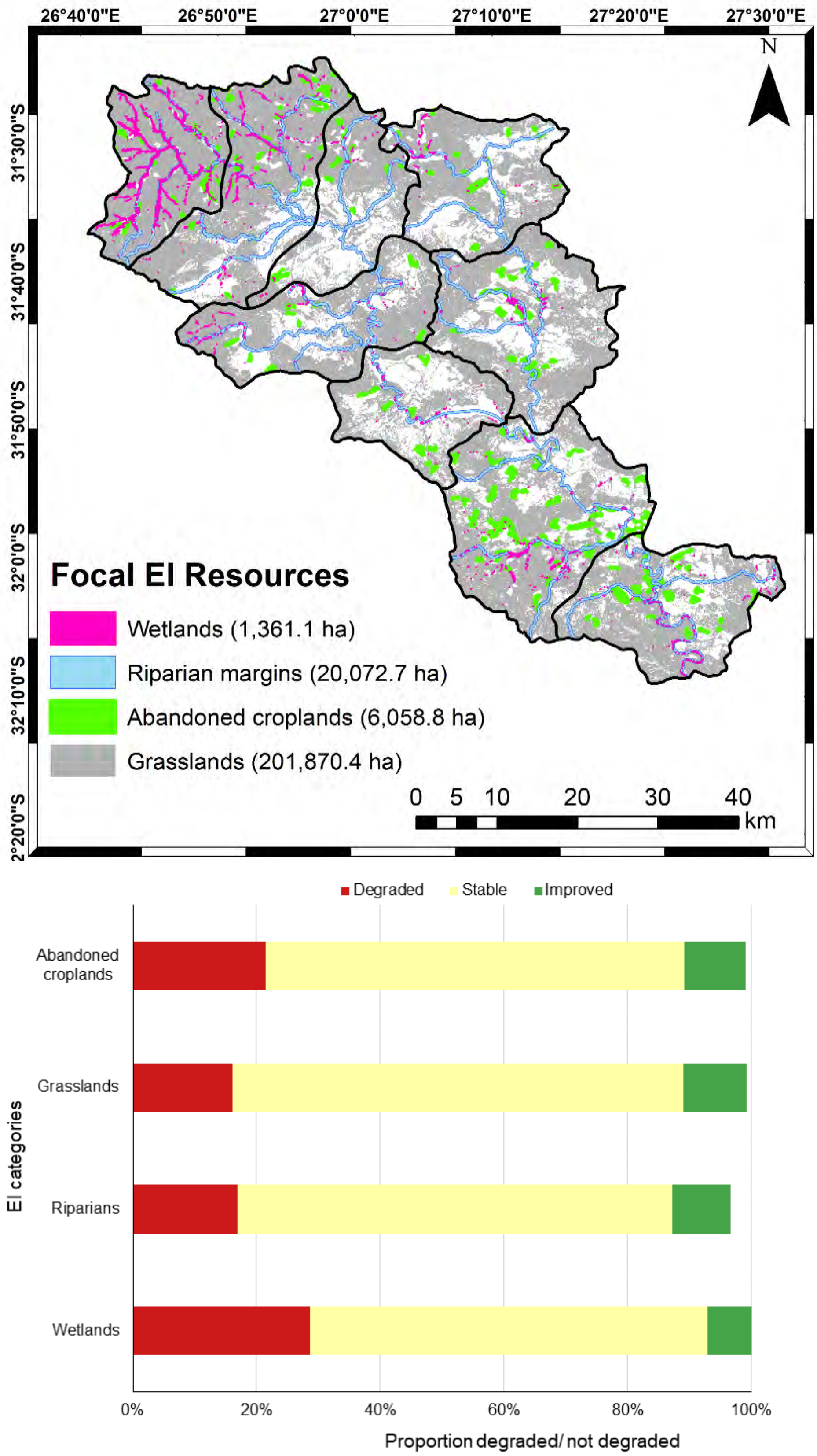


Figure 4.1: Spatial distribution of four key EI categories in the Cacadu catchment (**top**) and link to the SDG 15.3.1 indicator (**bottom**). Hollow (white) background in the map are areas covered by other land cover classes.

4.2.2. Prioritisation of EI for restoration using the AHP method

A GIS-based analytical approach that includes stakeholder preferences integrated with other spatial datasets for identifying priority locations for restoration was used (Forsyth et al., 2012; Favretto et al., 2016). The AHP method is an analytical decision-support framework developed to handle decision problems with complex criteria that can be either qualitative or quantitative, and it also incorporates multiple stakeholders in the process (Saaty, 1990; Belton & Stewart, 2002: 1.1). Like other multi-criteria decision analysis methods, the AHP branch of decision-making theory is instrumental in social-ecological systems decision-making due to its ability to handle mixed-methods and create collaborative decision-making space (Belton & Stewart, 2002). Integrating stakeholder views in the AHP process for decision-support has also been successfully used in landscape planning (Ananda & Herath, 2003; Chow & Sadler, 2010; Komossa et al., 2018). Additionally, the integrated GIS and AHP tools have been used to support groundwater quality assessments (Jhariya et al., 2017), environmental management assessments (Silva et al., 2014), and conflict resolution in marine management assessments (Tuda et al., 2014; Janssen et al., 2015) amongst other uses. Favretto et al. (2016) also used the AHP to identify the economic returns of semi-arid dryland ecosystems in Botswana. Integrating the AHP technique with GIS and stakeholder views in the above-listed fields makes the technique suitable for this study. The AHP incorporates stakeholders through the Delphi technique to combine multiple views and uses a linear matrix to determine the acceptable consistency of individual and group decisions (Saaty, 1990; Ananda & Herath, 2003).

The AHP process is a six-step method, (i) defining the main AHP objective; (ii) determining the main criteria and spatial attributes; (iii) creating the pairwise comparison matrix; (iv) determining relative weights; (v) checking for the consistency of scores in each criterion; and (vi) determining overall ratings for the weight (Saaty, 1990). Figure 4.2 outlines the AHP process followed to prioritise the focal EI resources for restoration to improve drought mitigation. The main criteria (ecosystem health, hydrological functionality and social benefit) were determined by communal stakeholders (described below), and the attributes were derived by the research team based on the criteria. The priority list was derived based on normalised scores using the *Weighted Summation* approach from the ArcGIS Spatial Analyst toolbox (Figure 4.2). The *Weighted Sum* approach is an additive overlay analysis that uses the values of each input raster, multiplied by the specified weight (as continuous variables), then sums all the input raster values to create a final raster.

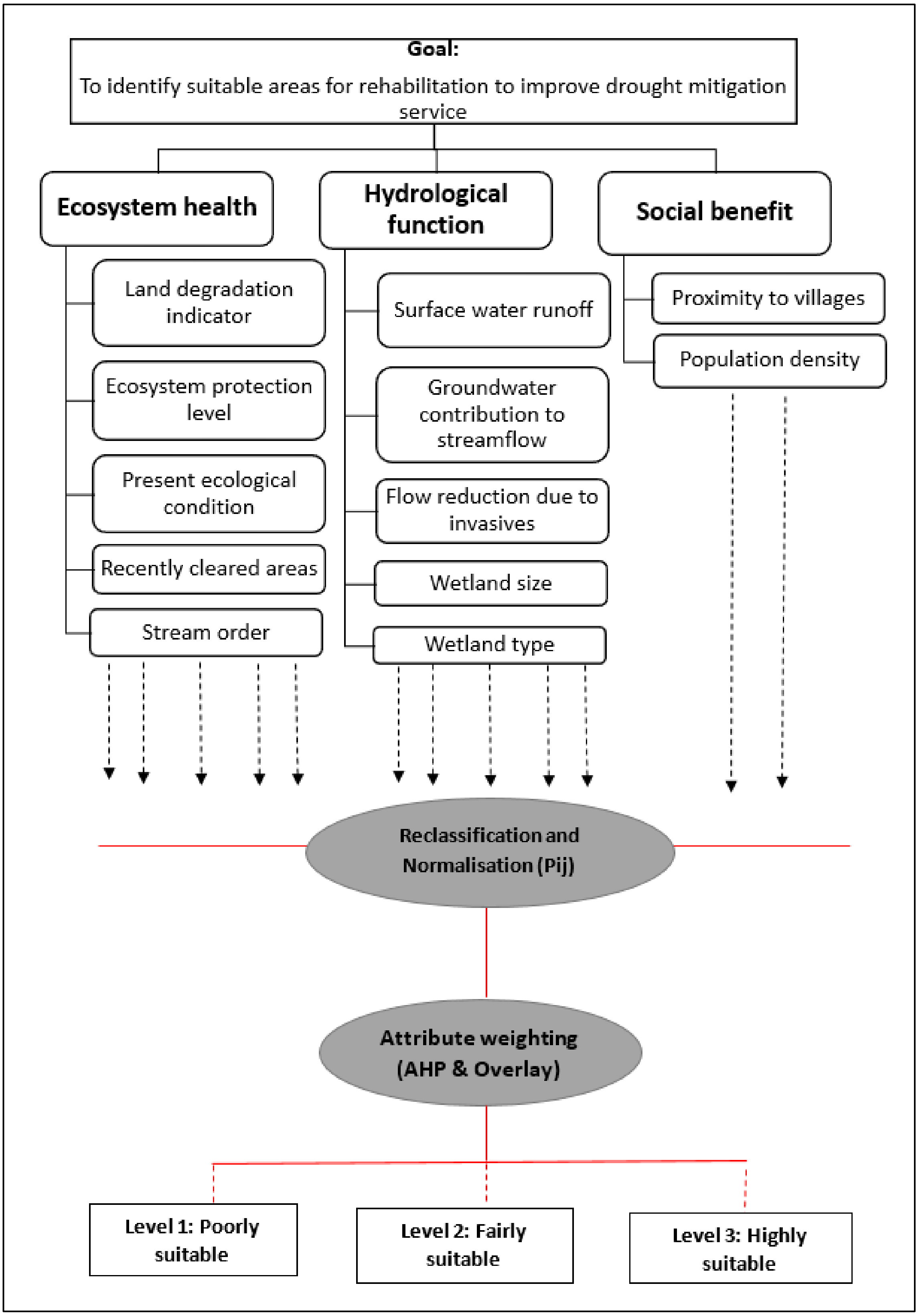


Figure 4.2: Schematic diagram showing the general steps followed for the AHP model process. Provide more details – insufficient and there the reader is going to be lost…explain in general what is going to come ahead and what this shows.

4.2.2.1. AHP Step 1: Definition of the main objective

The central AHP objective was defined by the researcher as follows, “which locations of key ecological infrastructure categories should be prioritised for restoration to improve the catchment’s drought mitigation capacity?” The prioritisation was applied in the Cacadu catchment at the scale of focal EI categories. The different EI categories were separately prioritised because they respond differently to degradation drivers and are managed differently.

4.2.2.2. AHP Step 2: Stakeholder inputs used to derive the AHP criteria

The second step of the AHP technique is designed to split the main objective into smaller segments to reduce the decision-making complexity, which is where discussion is incited among active stakeholders and knowledge exchange occurs (Belton & Stewart, 2002: 1). At this stage, stakeholders must identify the main criteria and indicators that can be compared to reach the main goal (Ananda & Herath, 2003). Advice from community stakeholders within the catchment was used to identify key resources (EI) and the criteria used to identify the relevant spatial attributes and important EI locations. The purpose of this step was to elicit stakeholder values for the four key EI categories that can be used for scoring and weighting the spatial attributes. Stakeholder values were collected through interactive group discussions (digital participatory mapping) in one quaternary catchment following the protocol described in Weyer et al. (2019).

Two one-day participatory geographic information systems (PGIS) group discussions were conducted with stakeholder groups from five villages in the Machubeni communal area (S10F) (Appendix 4.1). Participatory GIS is a participatory mapping approach used in rural settings from developing countries that leads to community empowerment and social capital building by integrating participatory action and learning (Reed et al., 2008, 2010; Levine & Feinholz, 2015). The stakeholder discussions took place in one quaternary catchment (S10F). However, it was assumed that stakeholder views in one area would be similar across the tertiary catchment. The assumption was based on demographic similarities within the catchment (Table 1.1). As described in Chapter 1 (Table 1.1), the Cacadu catchment is divided into 3 Local Municipalities (i.e. Enoch Mgijima, Emalahleni and Intsika Yethu). Amongst other things, areas falling within these municipalities have similar economic activities, a density of agricultural households, a slight dominance of females, a dominance of tribal dwellings and a dependence on natural resources (Table 1.1). The group discussions were convened with stakeholders representing farmers associations from the five GEF5, SLM Project: Eastern Cape -Machubeni in May 2019. One meeting was held with the eastern part of the study area encompassing Gxojeni and Platkop villages with 15 participants. The second was convened in the western part of the study areas encompassing Helushe, Qhoboshane and Boomplaas villages with 25 participants. A detailed workshop report is available (Appendix 4.1). The Farmers Association user groups were invited to participate in the group discussions. The stakeholder discussions were done under the Rhodes University Ethical Standard Committee (Rhodes University, 2014), and the ethics review reference is 2019-0448-489. Before the group discussions, gate-keeper's permissions from the head-men were obtained. Participants were provided information about informed consent, and confidentiality and data ownership protocols were observed as per human ethics standards (Rhodes University, 2014).

The participants' home language - IsiXhosa, was used to conduct group discussions to improve the discussions' efficiency. The discussions lasted for 35-45 minutes. The first step of the group discussions was to ask the participants to verify the previously mapped focal resources for their extent and their locations. For ease of description, the previously mapped community resources under the GEF5: SLM project together with known locations (e.g. schools, churches, and other landmarks) were projected on the wall (Image 4.1). The first group discussion was set in a sheering shed in one of the communities, and the second was set in a community hall Image 4.1. Stakeholder-driven community mapping was used to map resources by following stakeholder instructions on locating the polygons and points features into the Google Earth platform (Image 4.1). Different polygon colours represented different features, for instance, green for croplands and orange for rangelands (Appendix 5.1). Secondly, the participants were as ked to share their knowledge on which criteria they use to identify the most valuable focal resources, and this was used to aid the final prioritisation product.

Image 4.1: Image of the researcher co-facilitating an interactive group discussion and digital mapping in Machubeni.

4.2.2.2.1. Defining the set of attributes for AHP assessment and their restoration importance for drought mitigation

The set of attributes for the AHP assessment were derived based on the overall goal and the advice received from community stakeholders (Table 4.2 – discussed below). The optimal attributes were presented to a group of experts in restoration management and ecological infrastructure in the second Reference Group Meeting for the Water Research Commission Project K5/2928 on 28 May 2020. The purpose of the presentation was to ensure that the attributes reflect the relevant information needed to perform the prioritisations exercise. The attributes contain a range of spatial-temporal, social and ecological based datasets that provide information on the utility of EI locations for flow regulation and benefit to locals within the catchment. The datasets in Table 4.2 were selected based on the understanding that the physical catchment has a role in the flow regulation service (discussed in Chapter 2). All the datasets were selected because of their public availability (Table 4.2). The attributes in Table 4.2 were extracted from the focal EI zones and reclassified to a 30 m resolution.

Table 4.2: Summary of input datasets used to derive the attributes for the prioritisation assessment. The main criteria are used to arrange the attributes.

Main criteria	Attributes (spatial datasets)	Description	Resolution
Ecosystem health status	SDG 15.3.1 indicator (Conservation International, 2018b)	Raster layer showing degradation states for focal EI	250 m
	2018 Ecosystem protection level (SANBI, 2018)	Raster file showing protection level of terrestrial ecosystems excluding aquatic ecosystems	30 m
	Recently cleared areas (EuroSpace Agency, 2017)	Raster layer showing the transformation of tree-covered areas area to other land cover classes covering the period 1990-2018.	300 m
	Stream order (DWS, 2006)	Vector layer with 1-7 stream order levels as line features	1: 500 000
	Present ecological status (CSIR, 2018)	NBA layer for inland aquatic ecological conditions	1: 500 000
Hydrologic functionality	Estimated flow reduction by IAPs (Le Maître et al., 2016)	Raster layer of % annual reduction factor by IAPs	250 m
	Groundwater recharge (Le Maître et al., 2018a)	Raster map showing annual average aquifer recharge	1 km
	Surface water runoff (Le Maître et al., 2018b)	Raster map showing water source areas by mean annual runoff	1 arc minute
	National Wetlands Map (CSIR, 2018)	NBA vector layer of ecological condition, hydro-geomorphic type and protection level. Wetland size is also included as area (ha)	1: 5 000
Social benefit	EI distance from the villages (StatsSA, 2011)	Proximity derived from RSA sub-areas vector layer	1:250 000
	Population density (StatsSA, 2011)	Vector layer showing population density per km2	1: 250 000

4.2.2.2.2. Attributes relating to the health status criterion

I. **Land degradation indicator (SDG 15.3.1)**: The proportion of degraded land is an essential indicator of the location and extent of degradation at the focal catchment (see Chapter 3). The attribute SDG 15.3.1 indicator encapsulates the catchment health status (with climate extremes removed), which has been used as a surrogate for human-induced land degradation (Wessels et al., 2007, 2012). The degradation indicator was derived based on land productivity, soil organic carbon and land cover change as input sub-indicators (Chapter 3; Orr et al., 2017). The indicator is vital to provide information about the potential of land cover to reduce immediate runoff and contribute to streamflow during low-flow periods (Le Maître et al., 2014). The land degradation indicator is useful for weighing vegetation and land cover capacity in catchments to regulate runoff (Le Maître et al., 2014).

II. **Ecosystem protection level (EPL)**: The ecosystem protection dataset considers ecosystem connectivity and monitors the extent of ecosystem protection across landscapes (Skowno et al., 2019: 7.2). The EPL legend categorises native biomass into four categories namely, well-protected (i.e. 100% protection of the ecosystem), moderately protected (i.e. 50-100% protection), poorly protected (i.e. 5-50% protection) and not protected (i.e. <5% protection) (Skowno et al., 2019). Restoration literature (Stavi & Lal, 2015; Mander et al., 2017; Hoffman et al., 2018) indicates that protected ecosystems are less vulnerable to further degradation. Therefore, protected ecosystems allow biodiversity to thrive. However, since the dataset is computed using satellite datasets at the ecosystem level, some land covers, such as farmland areas, are not included (Skowno et al., 2019). Over 65% of terrestrial ecosystems are not protected, and the remaining 35.5% of the ecosystems are under some protection in the Cacadu catchment (Figure 4.3A).

III. **Present ecological status (PES)**: Riverine and wetland conditions are presented using a set of the freshwater ecological status of categories that were designed by the Department of Human Settlement, Water and Sanitation used for describing the ecological condition of rivers, and a similar set of ecological conditions for to wetlands (Macfarlane et al., 2009). The present ecological status assessment considers a range of factors, including physio-chemical conditions, flow, and habitat quality (CSIR, 2018; van Deventer et al., 2018). This attribute was derived from the 2018 NBA inland aquatic database, which presents the river conditions and wetland conditions on a range from A (natural) through to F (severe modification) (CSIR, 2018; van Deventer et al., 2018). Most aquatic ecosystems in the Cacadu catchment are within accepted levels (A-C) (Figure 4.3B). The present ecological status attribute was selected for the AHP model because it reflects the status of river and wetland ecosystems and their vulnerability to ecosystem collapse if neglected (Hoffman & Rohde, 2011; Le Maître et al., 2015; Modiba et al., 2017).

IV. **Stream order**: The choice of including stream order in prioritising riparian areas for rehabilitation is due to the assumption that vegetation type influences streamflow, with non-native plants causing the most streamflow reductions (Le Maître et al., 1996, 2000, 2015). Therefore, management interventions such as the Working for Water programme give a higher priority to headwaters to prevent further reinvasion downstream as per the provision of the Mountain Catchment Areas Act (Act 63 of 1970) (Turpie et al., 2008; Wilson et al., 2013). The Cacadu catchment is well-drained with over 20 first-order tributaries (Figure 4.3C).

V. **Recently cleared areas**: The Cacadu catchment falls within the grassland biome and are often threatened by woody encroachment and IAP naturalisation (van Wilgen et al., 2008; Luvuno et al., 2018). Therefore, the transformation from tree-covered areas (presumed to be IAPs) to other EI categories was viewed as a form of restoration activity in this study (SANBI, 2014; Yapi et al., 2018). The attribute for previously cleared areas was based on the recent change analysis using the 300 m European Space Agency global dataset covering 2000 - 2018 (EuroSpace Agency, 2018; Section 3.3.1.1). The previously cleared areas show tree-covered areas that were transformed into other EI categories (Figure 4.3D). Using the ESA dataset, over 2 500 ha in the Cacadu catchment were cleared between 2000 and 2018 (Figure 4.3D). Recently cleared areas within EI category were given a higher ranking to ensure that the effectiveness of interventions is achieved through ongoing treatments and maintenance (Gann et al., 2019: s8).

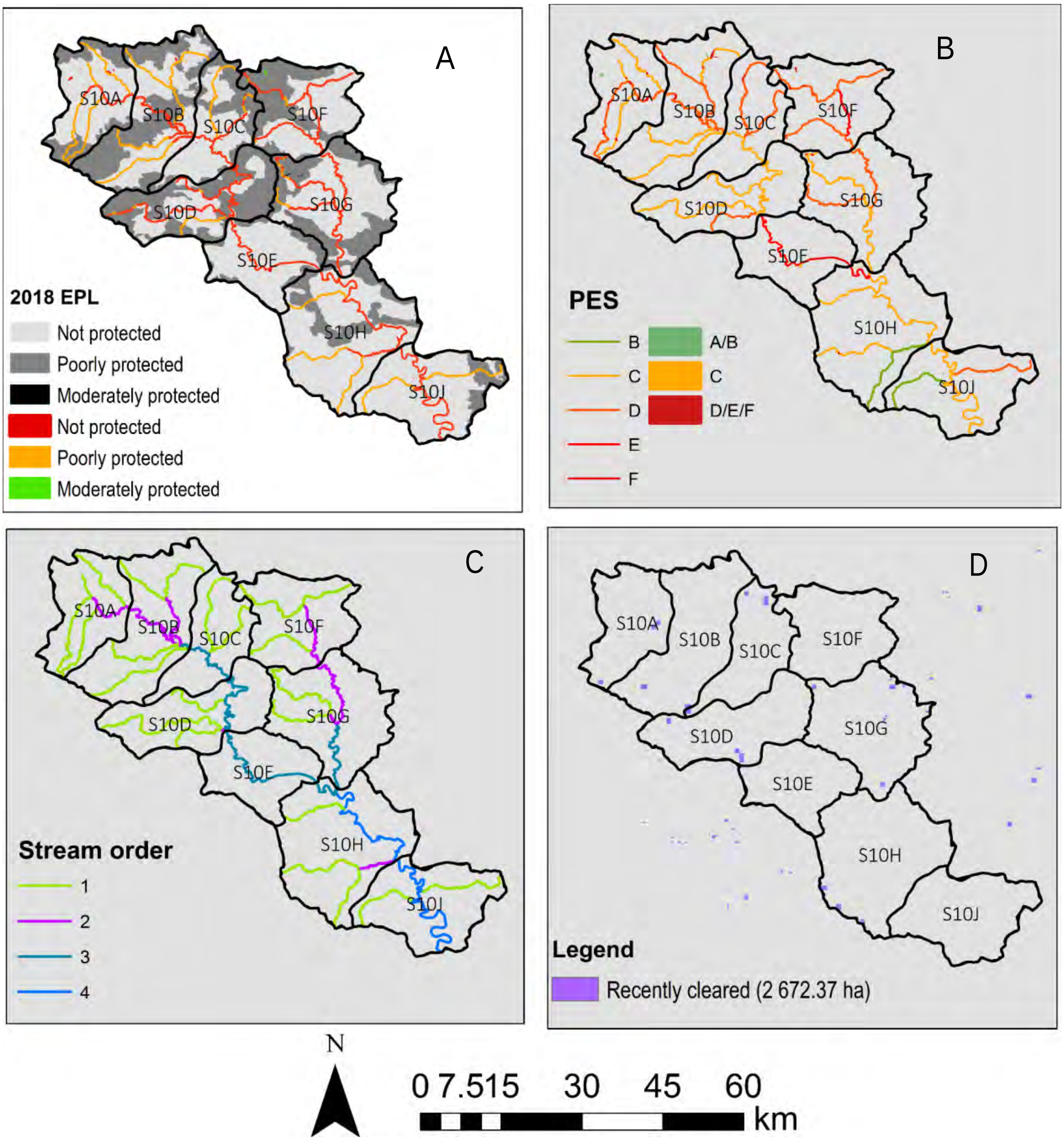


Figure 4.3: Spatial attributes relating to the ecosystem health criterion for the Cacadu catchment.

4.2.2.2.3. Attributes relating to the hydrological function criterion

I. **Groundwater contribution to streamflow**: Precipitation is the primary source of catchment water recharge. However, during the dry season and drought periods, precipitation declines, and the groundwater becomes the primary source of stream recharge through baseflow (Smakhtin, 2001; Stoelzle et al., 2014). Therefore, high groundwater recharge areas are hotspots for reliable water supply and need to be prioritised for rehabilitation. This study used mean annual groundwater recharge as a proxy for the groundwater availability and contribution to streamflow through baseflow (Figure 4.4A). Recharge data were obtained from the second Groundwater National Recharge Assessment hosted in the South African Water Resources database (Bailey & Pitman, 2016).

II. **Surface water runoff (MAR)**: The surface water runoff attribute is an essential indicator for the streamflow regulation function of catchments (Brauman et al., 2007; Le Maître et al., 2014). The attribute also encompasses Strategic Water Source Areas, which are high water production areas with an average annual runoff that exceeds 135 mm (Nel et al., 2013; Le Maître et al., 2018c). Strategic Water Source Areas occupy a small surface area (8%) of South Africa's total area but contribute to over 50% of water supply, making them important freshwater source areas (Nel et al., 2017). Therefore, the restoration of the EI categories within the higher-yielding area needs prioritisation for strong water regulation within the catchment and, consequently, drought mitigation (Nel et al., 2011). The surface water runoff attribute used in this study is at a quaternary catchment resolution (Figure 4.4B). The Cacadu catchment has a low to moderate mean annual runoff range and does not encompass any high yielding areas (Figure 4.4B).

III. **The estimated flow reduction due to IAP (MAR reduction)**: Although the introduction of IAPs has been included in the land cover change definition as degradation, the degradation definition for tree-covered areas leaves uncertainty since it includes all tree-cover classes (e.g. native and planted forests) in one class and is limited to a period of 15 years (UNCCD, 2018). Moreover, IAPs constitute a significant threat to freshwater in South African catchments (Turpie et al., 2008; Le Maître et al., 2015, 2019). Therefore, the inclusion of the estimated flow reduction by Le Maître et al. (2016) was used to represent the degree of impact of IAPs on surface water yields (and thus prioritising high flow reduction areas) each EI category location. The estimated proportion of flow reduction by IAP ranges from 0% to 34% reduction nationwide at 250 m spatial resolution(Le Maître et al., 2016). The Cacadu catchment has a broader flow reduction range with the highest invasion in the upper quaternary catchments (Figure 4.4C).

IV. **Wetland type**: The wetland geomorphic type is an indicator for water movement through a wetland and is estimated using the geomorphic location (Macfarlane et al., 2009: 26). The fifth national wetlands' dataset provides wetland hydro-geomorphic units at a quaternary catchment scale, and they are classified into seven categories (van Deventer et al., 2018, 2020). Although all wetland types play an essential role in flood attenuation (Kotze et al., 2009), channelled seepage and unchanneled valley-bottom wetlands are notably most likely to contribute to streamflow regulation. Most of the wetlands found in the catchment are channelled valley-bottom, seepage and depression types (Figure 4.4D).

V. **Wetland size**: The choice to include wetland size as an attribute for wetland prioritisation is based on the assumption that larger wetlands have a higher potential of contributing to surface water flows (Rebelo et al., 2015; van Deventer et al., 2018; Kotze et al., 2019). The wetland size attribute was derived from the fifth national wetland assessment dataset (CSIR, 2018; van Deventer et al., 2018). Consequently, larger wetlands were given a higher score for their potential role for hydrological flow regulation (Kotze et al., 2019). Less than 60 wetlands areas were detected in the Cacadu catchment, the majority of less than 20 ha in size (Figure 4.4E).

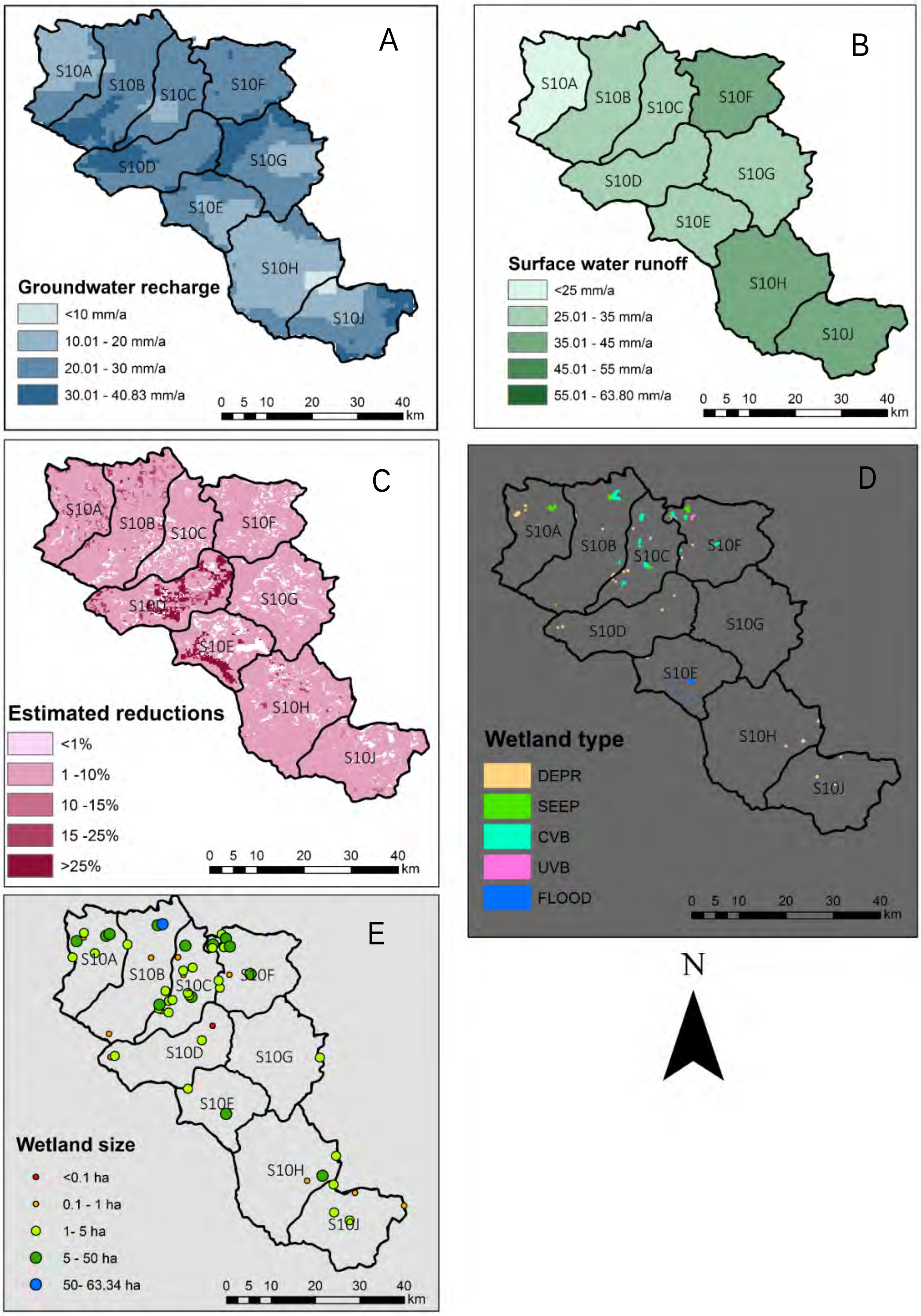


Figure 4.4: Spatial attributes relating to the hydrological function of the Cacadu catchment. **A** shows the groundwater potential for streamflow recharge; **B** surface runoff generation; **C** streamflow reduction due to IAPs; **D** wetland types based on hydrogeomophic location; **E** wetland size.

4.2.2.2.4. Attributes relating to the social benefit criterion

I. **Proximity to homesteads**: The primary goal of restoration is to increase the wellbeing of the social-ecological systems through increasing ecosystem resilience (Biggs et al., 2012; De Groot et al., 2013; SANBI, 2014; Cohen-Shacham et al., 2016; Orr et al., 2017). A proximity to homesteads index links to values derived from community stakeholder discussions (Figure 4.5A; Section 5.3.1). The proximity indicator is essential for access to healthy fodder and freshwater supplies for livestock, crops and other local livelihoods such as brickmaking. The index was obtained by deriving buffer zones (2 km max) around the village/farm area within each focal catchment using the *Multiple Ring Buffer* tool in the ArcGIS platform (Figure 4.5A). The *Multiple Ring Buffer* tool in ArcGIS uses the Euclidian Distance to compute buffer zones.

II. **Population density**: The first principle of ecological restoration highlights that EI investments can lead to social benefits and improve social-ecological resilience (Gann et al., 2019). To demonstrate the distribution of benefits to locals, which is a key indicator for monitoring restoration outcomes (Gann et al., 2019: s2), this study used the number of people that can benefit from the restored ecosystem as a quantitative attribute, expressed as population density (Figure 4.5B). Traditional communities primarily rely on natural resources for their livelihoods; therefore, core areas of ecosystem services are crucial in rural landscapes (Sigwela et al., 2017; Elbakidze et al., 2018). The population density indicator was derived using population per sub-area over the total surface area of the sub-area using the 2011 census dataset (StatsSA, 2011). The attribute expresses population density per sub-place (i.e. township, village or farm). The catchment is sparsely populated, with most of the population occupying less than 5% of the total catchment area (Figure 4.5B).

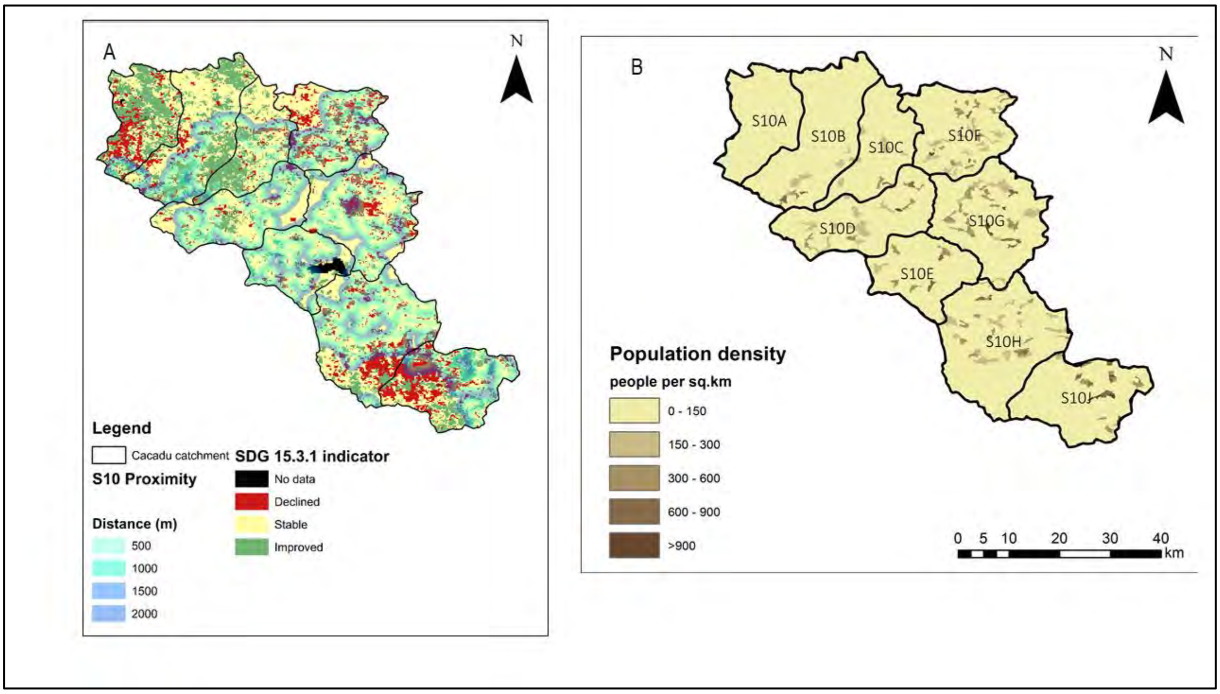


Figure 4.5: Map showing the attributes relating to the social benefit criterion in the Cacadu catchment. **A** shows an overlay of the 2 km buffer zone on the land degradation indicator in the villages of the Cacadu catchment; **B** shows population density at village level in the Cacadu catchment.

4.2.2.3. AHP Step 3: Reclassification

Each EI category was prioritised based on the main criteria provided by local stakeholders [ecosystem health, hydrological function and social benefit (Table 5.3)]. All but the stream order attribute was applicable to the wetland EI category (Table 5.3). All but two attributes (wetland size and wetland type) were applied to the riparian EI category (Table 5.3). Eight were relevant to the grassland EI category from the list of attributes, and those excluded present ecological status, stream order, wetland size, and wetland type (Table 5.3). Lastly, the old croplands EI category was prioritised based on four attributes: land degradation indicator, recently cleared areas, groundwater contribution to streamflow, and surface runoff (Table 5.3).

The attributes use different measurement units and linguistic terms and have varying numerical ranges; therefore, a 5-point indicator range was used to convert the attribute units to a consistent scale (Table 5.3). This indicator range was important to maintain consistency and comparability between the different spatial dataset (Malczewski, 2000). The allocation of scores is detailed in the following sub-section and illustrated in Table 5.3 under attribute scores.

4.2.2.3.1. Reclassification for the ecosystem health criterion

The land degradation indicator provides three states: degraded, stable and improved (UNCCD, 2018). Therefore, to maximise the benefits of restoration interventions for drought mitigation, the degraded state was given a higher rating, followed by the stable state (Table 5.3). The most protected ecosystems were given a higher score under the assumption that protected ecosystems already have an enabling environment for implementing restoration interventions (Table 5.3). The least modified wetlands and riparian margins next to the least modified inland aquatic ecosystem were given the highest score as these require the most protection to prevent new and further degradation. In contrast, the heavily modified ones were given a lower score since they will require high costs for modest benefits, and moderately degraded ones were given a moderate score as ideal candidates for rehabilitation (Table 5.3). Recently cleared areas were given the highest scores since resources have already been invested in these areas, and follow-up on these areas is essential to keep the gains in flow regulation made (Table 5.3). In line with the Mountain Catchment Areas Act (Act 63 of 1970), the lower order streams were given a higher priority as these tributaries act as conduits for invasion in the lower reaches of the catchment (Table 5.3). Therefore, focusing on the lower order streams can prevent the introduction of IAPs (Wilson et al., 2013) and water losses in riparian areas (Le Maître et al., 2016).

4.2.2.3.2. Reclassification for the hydrological functionality criterion

The highest water yielding areas (groundwater potential to contribute streamflow, mean annual runoff [MAR], wetland size and wetland type) were given the highest rating for their streamflow regulation importance (Table 4.3). Valley-bottom and seepage wetland types were given a higher rating for their significant contribution to streamflow regulation, while the low gradient wetland types were given a lower rating (Macfarlane & Atkinson, 2015; Table .43). The highest MAR reduction areas were given the highest scores, which is in line with the need for IAP mitigation for water security (Le Maître et al., 2015, Table 4.3).

4.2.2.3.1. Reclassification for the social benefit functionality criterion

The grasslands, wetlands and riparian margins closest to the villages were given a higher score, which was in line with community stakeholder advice (Table 4.3; Section 4.3.1). Areas with the highest population density were given the highest ranking under the assumption that these areas contribute the most to ecosystem service provision (Table 4.3).

Table 4.3: EI Prioritisation attributes and their relative performance matrix. The table is organised based on the main criteria (ecosystem health, hydrological functionality and social benefit). Shaded cells depict non-applicable items.

Attributes (j)	**Applicable EI category**				**Indicator ranges (i)**				
	Wetlands	Riparian margins	Grassland areas	Abandoned croplands	**1**	**2**	**3**	**4**	**5**
Land degradation indicator	X	X	X	X			Improved	Stable	Degraded
Ecosystem protection level	X	X	X			Not protected	Poorly protected	Moderate protection	Well Protected
Present Ecological Status	X	X			F-Z	E	D	C	A-B
Recently cleared areas (ha)	X	X	X	X					Present
Stream order		X			6-7	5	4	3	1-2
Groundwater contribution to baseflow (mm/a)	X	X	X	X	0-35	35-65	65-100	100-150	>150
Surface water runoff (mm/a)	X	X	X	X	0-25	25-60	60-135	135-220	>220
Estimated flow reduction by IAP (%)	X	X	X		0-5	5-10	10-25	15-25	>25
Wetland size (ha)	X				0.1-1	1-5	5-50	50-200	>200
Wetland type	X				Depression	Flat and Floodplain	Seepage	On-channel valley-bottom	Off-channel valley-bottom
Distance from villages (km)	X	X	X		>4	3-4	2-3	1-2	<1
Population density per km^2	X	X	X		<300	300-600	600-900	900-1200	>1200

4.2.2.4. AHP Step 4: Establishment of attribute weights

The selected attributes all have an impact on selecting suitable restoration sites for drought mitigation. Therefore, in line with AHP rules, attribute weights must be defined for each attribute to express the importance of each criterion on the final result (Belton & Stewart, 2002). Attribute weights were determined based on stakeholder advice for the most beneficial locations of EI resources. Weights were derived using the AHP mathematical method, which derives ratio scales from paired comparisons (Saaty, 1990). An automated questionnaire was sent out to a group of experts consisting of practitioners, researchers, postgraduate researchers and research managers working in the Cacadu catchment to derive the relative weights. The experts were asked to establish a hierarchical structure of the attributes through a Google Forms questionnaire on 27 July 2020 (Appendix 5.2). The questionnaire aimed to rank the various attributes based on their relative importance for the goal of restoring the key EI categories to improve the flow regulation capacity of the catchment. A total of 48 experts were approached to respond within a period of 40 days to the survey, and 22 responses were received.

4.2.2.4.1. Establishment of the comparison matrix using the AHP method

The experts were expected to establish AHP comparison matrices (pairwise comparison matrix) for the ecosystem health and hydrological functionality with the guide of the questionnaire (Saaty, 1990). Each expert completed two five-by-5 matrices of spatial attributes relating to the ecosystem health and hydrological benefit criteria to be considered as an approximation of attribute weights (Appendix 5.2). Experts were provided with attribute pairs to complete each matrix independently (Appendix 5.2). Participants typically reach consensus with the aid of the Delphi approach (Saaty, 2000; Chow & Sadler, 2010; Forsyth et al., 2012), which allows them to deliberate amongst each other in a group setting. However, since the questionnaire was conducted online without the option for experts to reach a consensus, the expert responses were aggregated by taking the medians to accommodate the variability of responses. The Saaty scale is an ordinal data type; therefore, using the median instead of other central tendency indicators is justified. In the AHP mathematical process, the diagonals of the pairwise comparison matrix are given a similar value (i.e. $X_{i1}/X_{j1} = 1$), and the rest of the utility functions are recorded as reciprocals of the attributes. For example, if the utility function of attribute A compared to B = x time more important than B, then B compared to A is 1/x times more important than A.

4.2.2.4.2. Decomposing and normalising the matrix and normalisation

The first step in determining the weights was deriving the normalised attribute values (P_{ij}) using the ratio approach to get a normal distribution (i.e. 0 to 1 range) of all scores (Saaty, 1990). The normalised matrix is derived by dividing each utility function by the sum of entries in the corresponding column to sum i = 1. The final weights are derived by establishing the average performance score of each attribute based on the normalised matrix (Saaty, 1990).

4.2.2.5. AHP Steps 5 & 6: Consistency check and overall weighting

4.2.2.5.1. Consistency check

The pairwise comparison matrix is inherently subjective and biased since the interviewees can only make two comparisons and lose track of earlier responses (Saaty, 1990, 2000). Therefore, the AHP method requires a consistency check of the weights, which uses a mathematical approach to identify the consistency ratio (CR) that should be less than 10% for acceptability; otherwise, the matrix must be recalibrated (Saaty, 1990; Malczewski, 2000). The consistency of weights is measured using CR, which is derived by dividing the consistency index (CI) by random assignment index (RI) (Saaty, 1990). The RI values are already available in the literature, as shown in Table 5.4. This study computed the consistency ratio in Microsoft Excel. The maximum eigenvalue (λ_{max}) was computed using the matrix product (*MMult)* formula in Microsoft Excel for a twos 5*5 matrix sizes. Finally, the weighted scores (X_{ij}) were calculated by multiplying the normalised scores (P_{ij}) by the weight-coefficient (W_i) and presented in a table (Section 4.3.2.2) for ease of analysis.

Table 4.4: AHP random index (After Saaty, 2000).

Matrix size	2	3	4	5	6	7	8	9	10
Average RI	0	0.58	0.90	*1.12*	1.24	1.32	1.41	1.45	1.51

4.2.3. Application of the AHP model

Figure 4.6 outlines the procedure followed to complete the prioritisation activity. The AHP objective (step 1) was predefined by the researchers (Figure 4.6). The attributes were selected based on the literature and advice from community stakeholders regarding the main criteria (Figure 4.6). The datasets used were all converted to the Albers Africa Equal Area projection, and all attributes were rasterised to 30 m cells for consistency. Each EI category was classified separately because of the different input datasets. Scores were allocated by reclassifying each attribute using the *Reclassify* tool within the ArcGIS software (Table .43, Figure 4.6).

In Steps 8 & 9 of the model (Figure 4.6), a numerical overlay operation (i.e. the *Weighted Summation* function) from ArcGIS was used to combine the multiple attributes and criteria to derive priority areas. The overlay process used in this study was conducted four times to prioritise each EI category (Figure 4.6). Weighting was based on the hierarchical ranking of attributes by the expert stakeholders (Figure 4.6). The priority list was determined based on the unique numerical identifiers that are returned by the numerical overlay, and all outcomes were presented as spatial hotspot maps for visualisation. The priority list was determined based on the unique numerical identifiers that are returned by the numerical summation, and all outcomes were presented as spatial hotspot maps for visualisation. The hotspot maps were presented as a 3-point scale to highlight their relative suitability for improving flow regulation and contributing to local livelihoods within the Cacadu catchment. The priority ranges were given the following legend (1-2 = low, 3 = moderate, 4-5 = high).

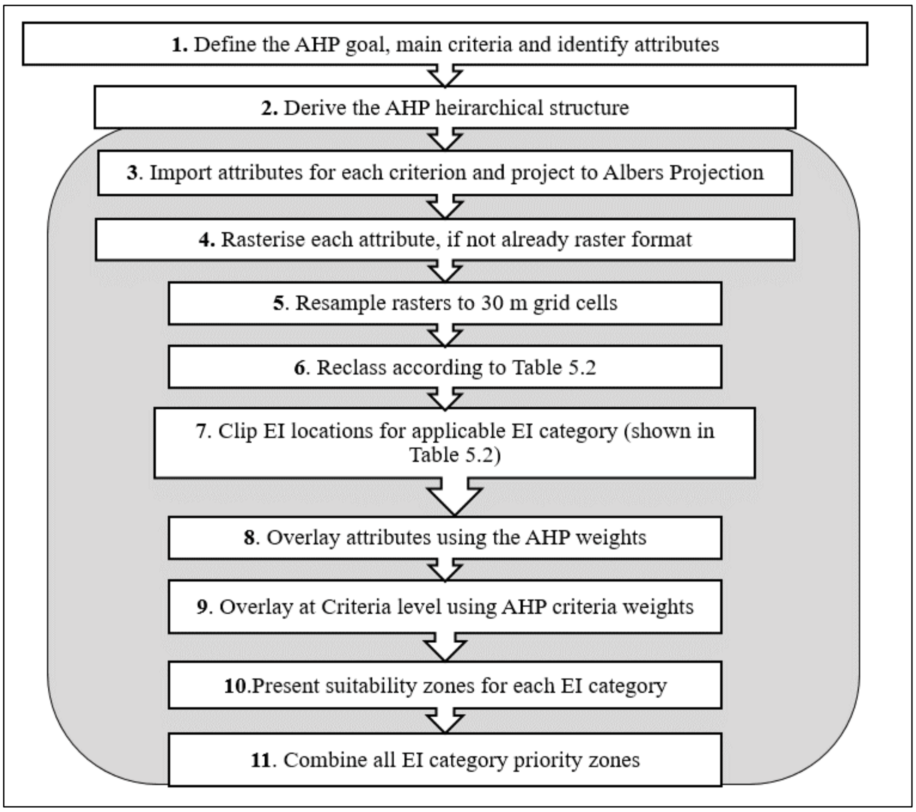


Figure 4.6: AHP model Workflow using ArcGIS.

Although wetlands and riparian margins typically have a higher flow regulation importance, the weighted summation approach used to prioritise the EI resources omitted most of the wetlands and riparian margins (Figure A1, Figure A2). The procedure followed in this study is mostly constrained by the quality of spatial datasets used (wetland coverage and present ecological status), which may lead to discrepancies in identified priority areas. The Cacadu catchment falls into the areas with the least confidence (over 75%) about the coverage, types and ecological conditions (van Deventer et al., 2020: 7), which explains the low detectability of most wetlands. This data limitation was also noted by Snaddon et al. (2018) while prioritising wetland for rehabilitation in the Western Cape Water Supply system; who decided to include the omitted wetlands in the final results based on the contextual understanding of the nature of the wetland in the Waterval River to overcome the limitation. Therefore, following Snaddon et al. (2018), it is sensible to assume that all on-channel and seepage wetlands that are greater than 5 ha in the Cacadu catchment have a high priority for restoration because of their hydrological importance. Including these wetland types would increase the extent of the highly prioritised wetlands from 9.72 ha to 277.78 (20.41% of all wetlands in the catchment).

4.3. Results

4.3.1. Main criteria identification

In this study, each attribute's ratings were derived based on feedback from the stakeholder discussions (Box 4.1). The stakeholder discussions took place in one quaternary Catchment (S10F), where the GEF5 project Machubeni focal villages are located (Appendix 5.1). However, it was assumed that stakeholder views in one area might be similar across the tertiary catchment. In total, 40 stakeholders took part in the group discussions, one-third of which were males. The older stakeholder groups had more in-depth knowledge regarding the EI's longevity and historical state, while the younger generation could better locate the resources. The community stakeholders from Machubeni indicated that the most desirable resources were selected based on the present ecosystem condition (relating to the ecosystem health criterion), water provision (related to the hydrological functionality criterion), accessibility and capacity of the resources to support local livelihoods (relating to the social benefit criterion) (Box 4.1).

Box 4.1: Conclusions about EI categories based on stakeholder opinions in Machubeni communal area.

Rangelands are important for fodder provision. The most important are the healthiest and those that are within walking distance (i.e. less than 2 km away from the homesteads), and have healthy natural vegetation.

Abandoned croplands are predominantly encroached by the woody *Euryops floribundus* plant, or invaded by IAPs. Secondly, since some croplands are still active, the use of cropland areas as surrogate grazing areas is not an acceptable practice.

Wetlands are important for water and fodder provision to livestock. The most important are those that can supply water all-year round.

Riparian zones are all valued for their ability to provide water and fodder. But when heavily eroded, riparian areas are dangerous to livestock, the elderly and children.

To validate the health status importance of rangelands, Figure 4.7 shows the rangelands (2017.4 ha) prioritised by community stakeholder underlain by land productivity status in the Cacadu catchment. The rangelands have been underlain with the productivity sub-indicator to highlight their fodder provision capacity in light of anthropogenic impacts, which can be inferred from (Figure 4.7). Most of the stakeholder prioritised rangelands fall within the stable state, while early signs of decline were detected in some areas (Figure 4.7, Section 3.1.1.1). Some of the areas with early signs of decline are covered with the *E. floribundus* plant (Figure 4.7, Image 4.2A), while the stable areas are within pristine conditions (Figure 4.7; Image 4.2B).

Community stakeholders indicated that lack of government support, climate change and the fast-spreading *E. floribundus* plant further encourage cropland abandonment and delay progress for returning to distant farming (Appendix 4.1). To support the above observation, some of the stakeholders said:

"Hayi, siyakufuna ukulima nto nje asinantsiba kwaye komile"

We want to resume cropping, but we lack resources, and the ongoing drought makes matters even worse

"Kuyenzeka maxa wambi amanzi adlule apha entsimini yakho, kodwa ke ngoba amanzi ahamba ezindongeni ubani uye awabukele edlula lomanzi ngoba kalaku akanayo indlela yokuwenyusa ewawusa ezindongeni awazise ngaphezulu ezitholeni"

There are cases where a stream cuts through or nearby one's cropland, but because the stream channel is deep, we lack the means to bring the water into the crops

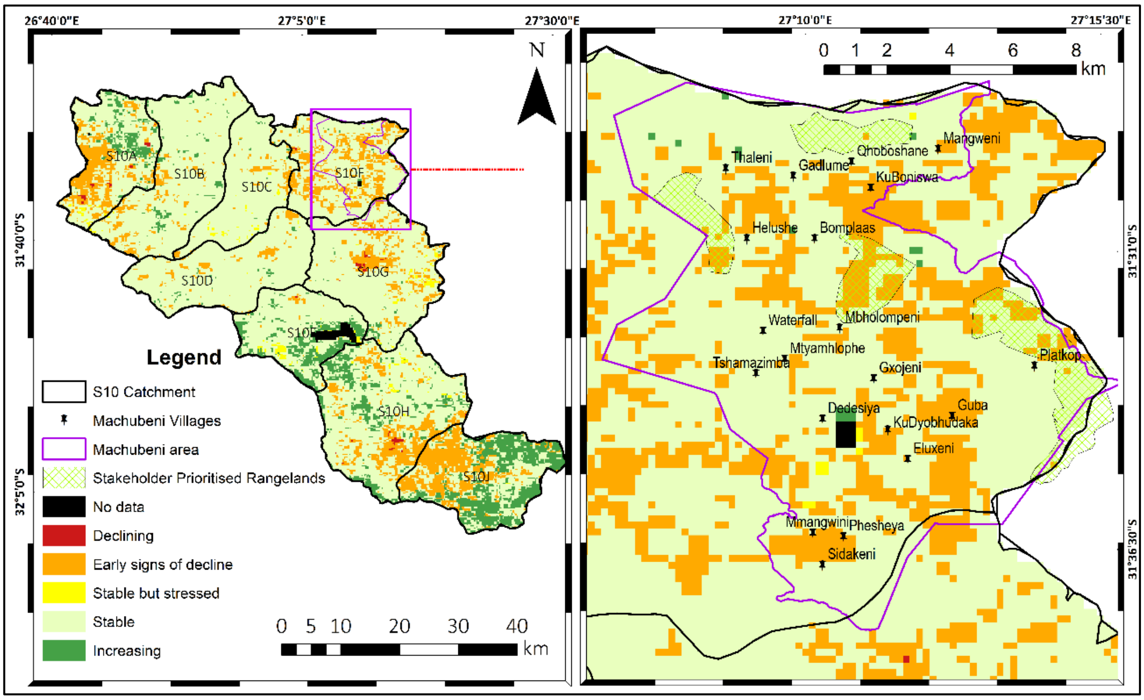


Figure 4.7: Community stakeholder prioritised grazing areas using community mapping and Google Earth within the Machubeni Communal area (S10F) overlain on a 15-year Rain-Use Efficiency index adjusted land productivity results. The RUE index results are outlined in Chapter 3.

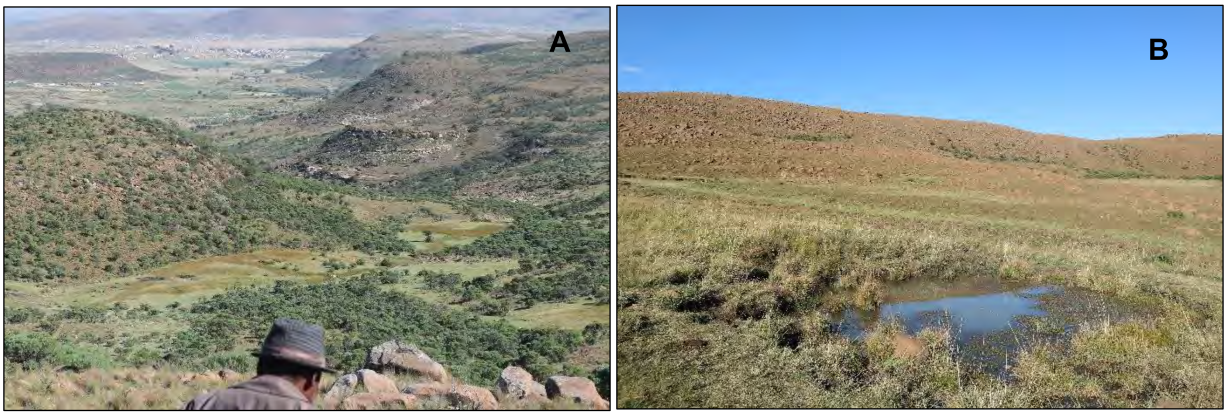


Image 4.2: Priority rangelands in Machubeni that were prioritised by community stakeholders and identified during field walks. **A** depicts and altered grazing shared by Platkop and Bomplaas grazing sites. **B** depicts a pristine grazing site for Platkop and Gxojeni villages.

Community stakeholders identified 33 wetland areas in Machubeni, and 18 wetlands were visited during the field walks (Figure 4.8). However, the sum of wetlands available in the area is still unknown. Most of the wetlands that community stakeholder identified were seeps and depression wetland types (Figure 4.8). Locals noted that wetlands in Machubeni play a crucial role in domestic and livestock water supply, which is why locals have reengineered and piped some of the wetlands.

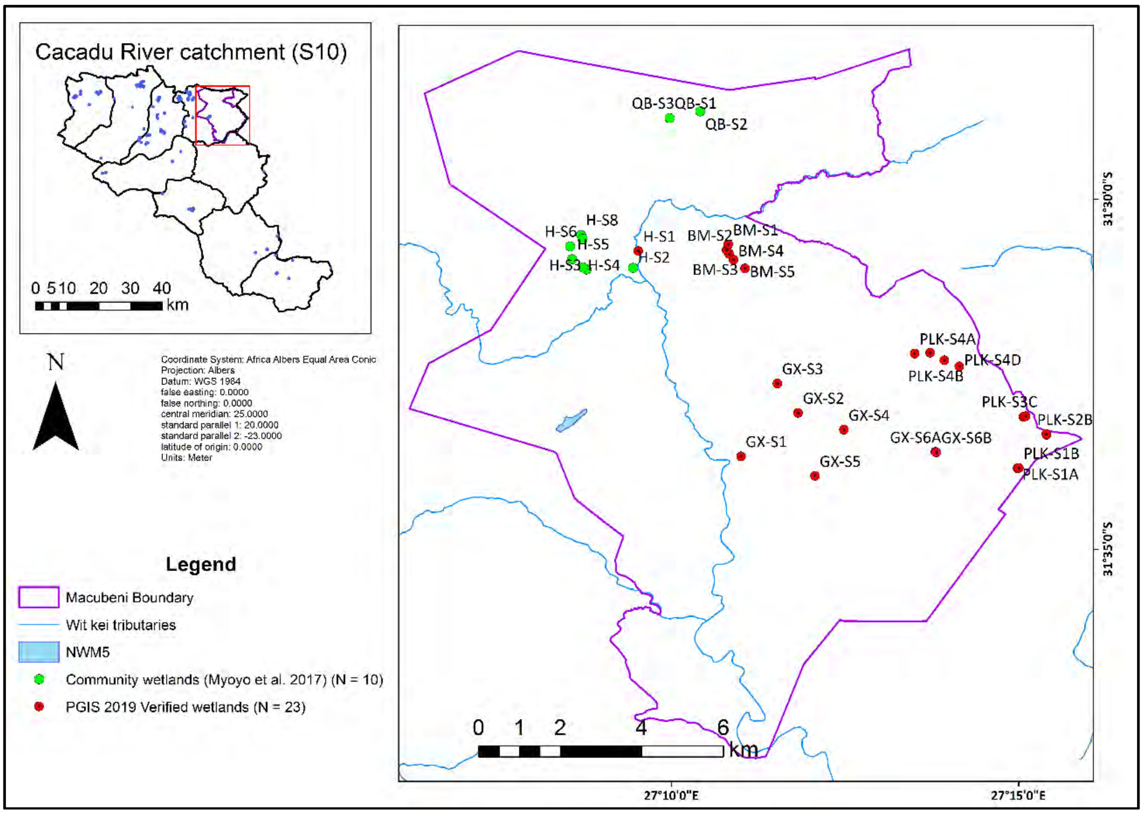


Figure 4.8: Community stakeholder prioritised wetlands using community mapping and field visits approach within the Machubeni Communal area (S10F). The acronyms S = Spring/wetland, BM: =Bomplaas, QB: = Qhoboshane, H = Helushe, GX = Gxojeni, and PLK = Platkop.

4.3.2. Establishment of hierarchical structure

4.3.2.1. Analysis of expert responses

Of the 48 expert participants approached, 46% (22/48) responded. The expert group was represented by three experience levels (extensive, moderate and minimum) in restoration management (Figure 4.9). The majority of the experts (59%) judged themselves to have moderate experience (Figure 4.9). A few respondents (4.6%) denoted NA for their experience in restoration management.

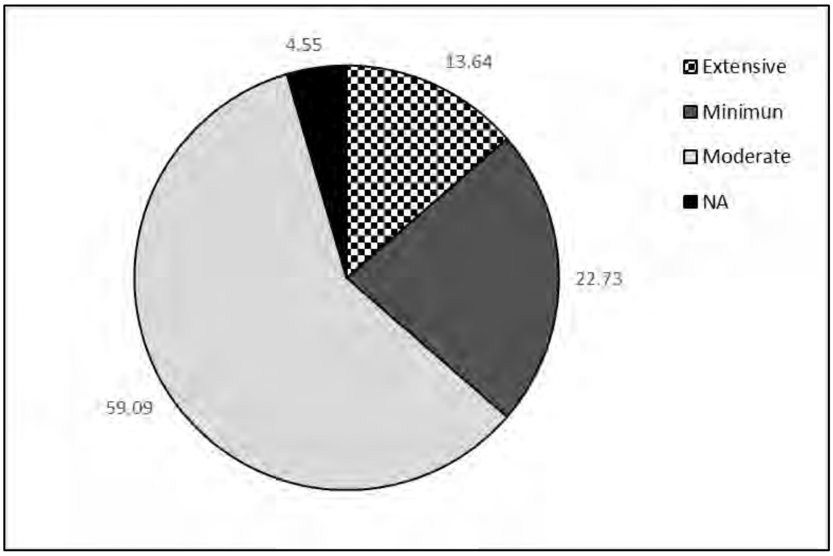


Figure 4.9: Expert's level of restoration experience N=22.

The expert group was asked to rank their level of agreement for the applicability of the selected attributes in the ecosystem health and hydrological function criteria to ensure the appropriateness of the selected spatial attributes. Over 75% of the 22 experts indicated a favourable agreement for all ten attributes, and none of the respondents disagreed.

The expert team was asked to establish the hierarchy of the attributes in each criterion by performing 20 pair-wise evaluations (Figure 4.10). The full range of pair-wise comparisons in the ecosystem health criterion shows some interesting statistics, such that although the confidence intervals overlap, the values of central tendency (medians) have a moderate degree of variability with a range of 1/5 (-5 in the figure) to 5 (Figure 4.10 left)

The small difference in median values indicates that experts had a consensus that there was a ranking difference between comparison-pairs (Figure 4.10). Although most pairs were skewed towards the positive intensity for the hydrologic functionality, the values of central tendency are mostly along with 1 and do not exceed 3 (Figure 4.10). The central tendency range of the hydrologic functionality pairs indicates that the experts had a consensus that most pairs were equally important, and the rest were at most moderately more important (Figure 4.10).

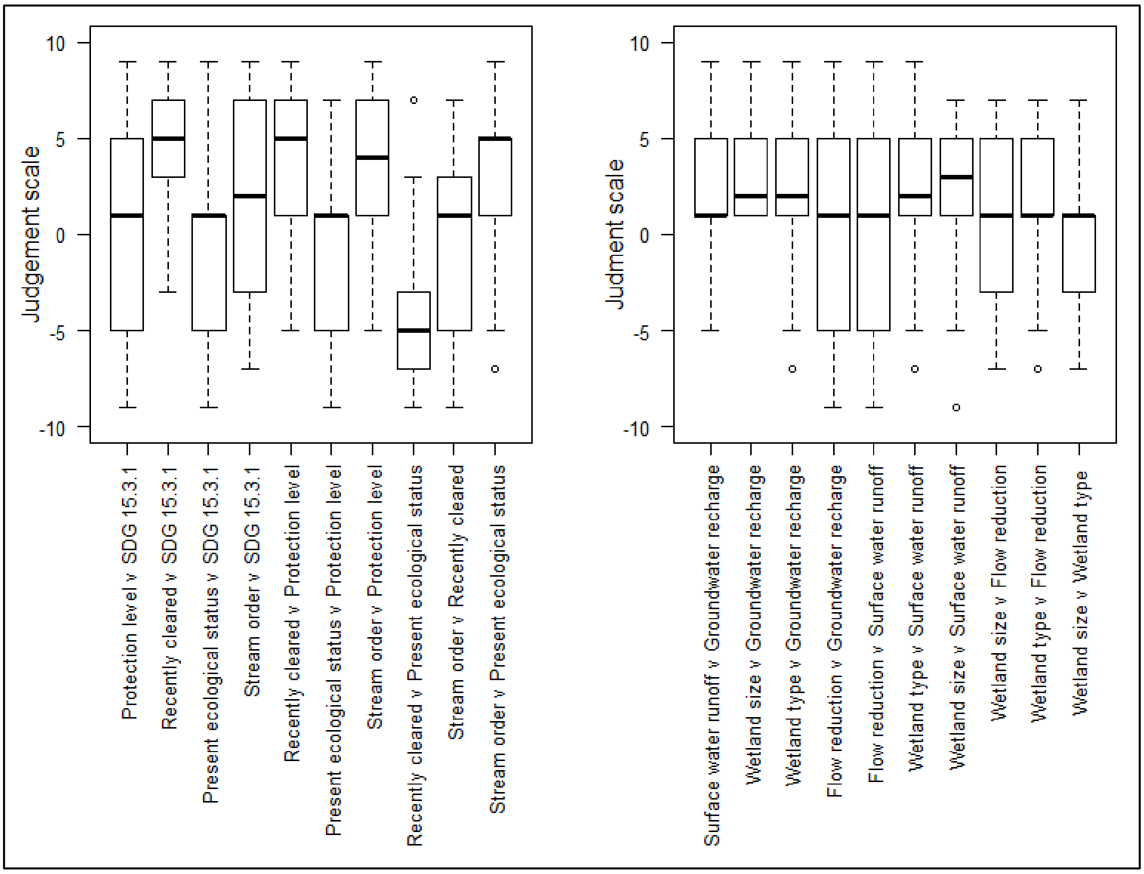


Figure 4.10: Box plots showing pair-wise responses by the 22 experts. **Left**: ecosystem health criterion comparison pairs. **Right**: Hydrologic functionality criterion comparison pairs. The judgement scale represents the Saaty scale. Negative judgement scale values symbolise the reciprocal values of the pair-wise comparisons (e.g. Attribute A < B), such that -3 = 1/3, -5 = 1/5, -7 = 1/9 and -9 = 1/9. The median is represented as the bold line, and the bubbles represent outliers.

4.3.2.2. Pairwise comparison matrices and consistency check

Table 4.5 presents AHP matrices for the ecosystem health and hydrologic functionality criteria that have been aggregated using median values of comparison pairs. The experts filled in the top half of the matrices, and the bottom half was filled in using the reciprocal rule (Table 4.5). The matrices were consistent with CR = 2% and 3%, meaning the matrices were acceptable for AHP usage (Table 4.5).

Table 4.5: Comparison matrices for the ecosystem health and hydrological function criteria attributes. The structure was derived using the AHP approach based on expert responses. The overall consistency for the ecosystem health criterion is 2% and 3% for hydrologic function.

Ecosystem health	**SDG 15.3.1**	**EPL**	**Cleared**	**PES**	**Stream order**
SDG 15.3.1	1	1	5	1	2
EPL	1	1	5	1	4
Cleared	1/5	1/5	1	1/5	1
PES	1	1	5	1	5
Stream order	1/2	1/4	1	1/5	1
Hydrologic functionality	**GW recharge**	**Surface water runoff**	**MAR reduction**	**Wetland size**	**Wetland type**
GW recharge	1	1	1	2	2
Surface water runoff	1	1	1	3	2
MAR reduction	1	1	1	1	1
Wetland size	1/2	1/3	1	1	1
Wetland type	1/2	1/2	1	1	1

Table 4.6 displays a list of attribute weights in each criterion. The main criteria had a similar ranking (33% each) (Table 4.6). The ecosystem health weighting by experts provided the following hierarchy: present ecological status > ecosystem health protection > degradation indicator > stream order > recently cleared areas (Table 4.6). The hydrologic functionality criteria resulted in the following ranking: surface water runoff > groundwater contribution to streamflow > runoff reductions due to IAPs > wetland size = wetland type (Table 4.6). The two attributes under the social benefit criterion were given an equal weighting (50% each) in line with community stakeholder advice (Table 4.6).

Table 4.6: Hierarchical structure for the different attributes and the three main criteria ranked.

Main criteria	**Criteria weights**	**Attributes**	**Attribute weights**
Ecosystem health	0.33	Present ecological status	0.31
		Ecosystem protection	0.29
		Degradation indicator	0.26
		Stream order	0.08
		Recently cleared	0.06
Hydrologic functionality	0.33	Surface water runoff	0.27
		GW contribution to streamflow	0.25
		MAR reduction due to IAPs	0.20
		Wetland size	0.14
		Wetland type	0.14
Social benefit	0.33	Population density	0.50
		Accessibility	0.50

4.3.2.2.1. Normalised scores of main criteria for each EI category

The normalised scores assigned to each EI category based on three main criteria are outlined in Table 4.7. The corresponding map results presented in Appendix 5.3. The normalised scores across the three criteria showed mixed results due to indicator ranges and variability in attribute weights (Table 4.7). The wetland size and wetland type attributes and the social benefit attributes had similar results since they have equal weighting (Table 4.7). The social benefit criterion attributes played the most influential role in the AHP model compared to other attributes, except for the present ecological status attribute (Table 4.7).

Table 4.7: Weight assignment of the twelve attributes arranged similar to Table 5.3. Shaded cells depict null values.

Attributes (j)	Normalised scores (P_{ij})				
	1	2	3	4	5
Land degradation indicator			0.78	1.04	1.3
Ecosystem protection level		1.18	1.77	2.36	2.95
Present Ecological Status	0.31	0.62	0.93	1.24	1.55
Recently cleared areas					0.3
Stream order	0.08	0.16	0.24	0.32	0.4
Groundwater contribution to baseflow	0.25	0.5	0.75	1.00	1.25
Surface water runoff	0.27	0.54	0.81	1.08	1.35
Estimated flow reduction due to IAP	0.20	0.40	0.60	0.80	1.00
Wetland size	0.14	0.28	0.42	0.56	0.7
Wetland type	0.14	0.28	0.42	0.56	0.7
Distance to villages	0.50	1.00	1.50	2.00	2.50
Population density	0.50	1.00	1.50	2.00	2.50

4.3.3. Priority EI areas for restoration to mitigate drought impacts on streamflow

Priority areas for restoration in the Cacadu catchment were summarised in tables of EI extent and proportion of priority EI to the total area of EI (Table 4.8). The results (Table 4.8) are highlighted using two spatial maps. Firstly, the results demonstrate suitable areas that could be restored to improve the flow regulation function of the Cacadu catchment, which was derived using the ecosystem health and hydrological functionality criteria only (Figure 4.11). Secondly, priority areas for restoration to improve drought mitigation while also meeting rural population needs are presented using all three criteria (Figure 4.12). The results of these figures are discussed later.

The prioritisation model omitted 99.29% of the wetland areas and 88.88% of riparian margins because of missing PES data in the catchment at the normalisation stage (Figure A3, Figure A4), showing some limitations with the prioritisation model (Table 4.8). The most detectable wetlands (9.72 ha) that would have no significant limitations for meeting the AHP objective (Table 4.8). A substantial amount of grasslands (80.15%) and abandoned croplands (92.06%) had some suitability to improve the flow regulation function of the Cacadu catchment (Table 4.8).

Table 4.8: Summary of priority areas for restoration to improve flow regulation mitigation at Cacadu catchment. The table expresses the areal (ha) and percentile extent of AHP identified priority EI areas.

EI category	Priority level	Extent (ha)	% of the prioritised area	% of full EI in the catchment
Wetlands	Low	0.00	0.00	0.00
	High	277.8	100.00	20.41
	Total	277.8	100.00	20.41
Riparian zones	Low	1057.3	47.38	5.27
	High	1174.4	52.62	5.85
	Total	2231.7	100.00	11.12
Abandoned croplands	Low	293.4	5.26	4.84
	High	5 284.6	94.74	87.22
	Total	5 578.0	100.00	92.06
Grasslands	Low	63 903.3	39.50	31.66
	High	97 894.6	60.50	48.49
	Total	16 1798	100.00	80.15

The prioritised areas to improve the flow regulation function of the Cacadu catchment mostly mimicked the hydrological functionality criterion (Figure 4.11, Figure A4). The prioritised wetlands in the Cacadu catchment were located in the following quaternary catchments: S10A-C, S10E-F, S10E and S10H, and a visual satellite-image overlay the prioritised wetlands revealed that some were adjacent if not inside the privately-owned property (Figure 4.11). None of the 33 wetlands that community stakeholders prioritised in the Machubeni was identified as suitable by the AHP model, demonstrating the incompleteness of the national wetland dataset (Figure 4.7, Figure 4.11). A decision was made to manually add the omitted wetlands (seepage and channelled-valley bottom wetlands with an extent > 5ha) (Figure 4.11). The highly prioritised riparian margins were second to fourth order riparian zones in the middle quaternary catchments and the lowest quaternary catchment (Figure 4.11). Abandoned croplands throughout the catchment were highly prioritised except for the abandoned croplands in the S10A quaternary catchment (Figure 4.11). The most suitable grassland areas to facilitate flow regulation coincided with the most suitable riparian margins in the Cacadu catchment (Figure 4.11).

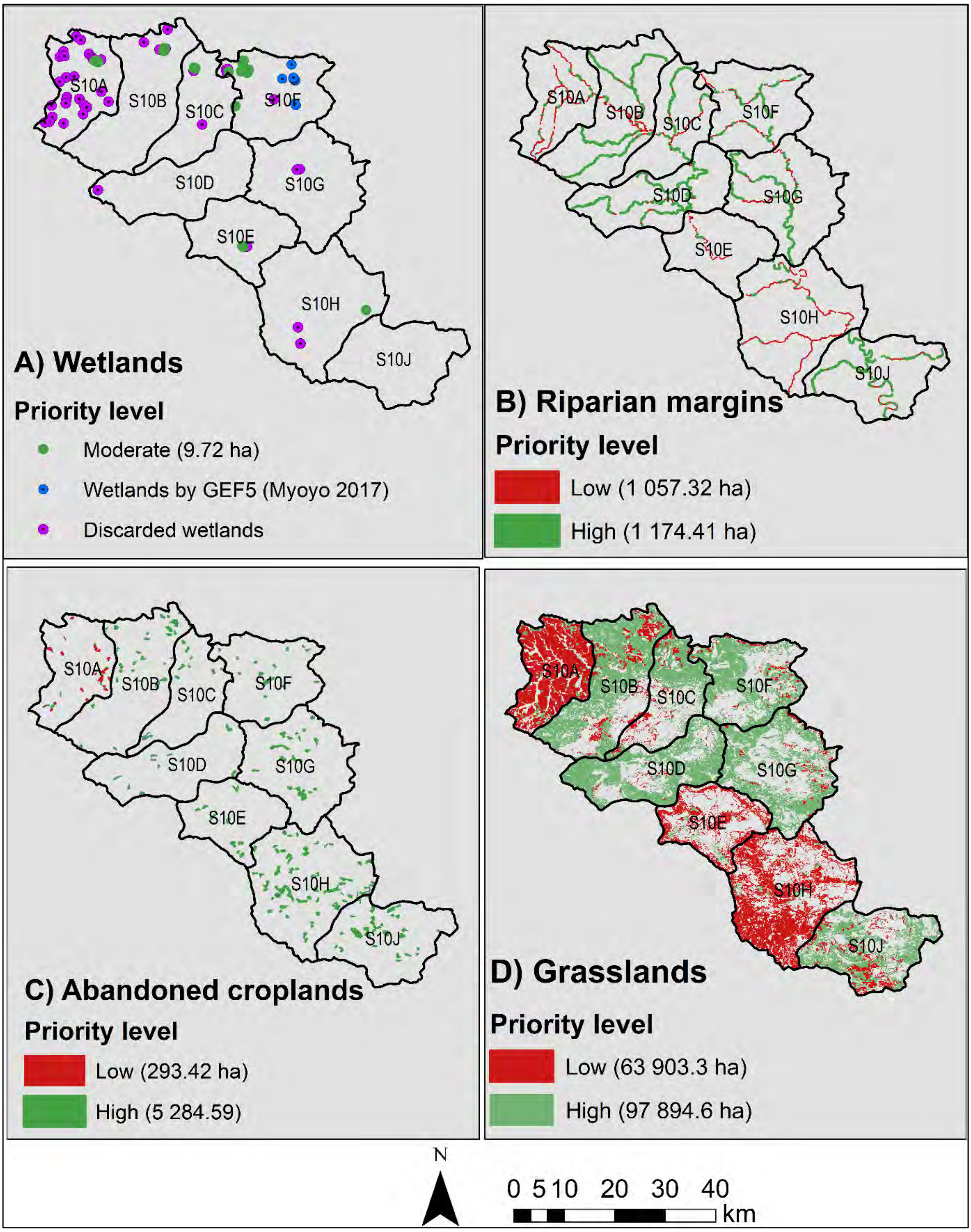


Figure 4.11: AHP model results for priority areas to improve the flow regulation function of the Cacadu catchment.

4.3.3.1. Priority areas for restoration to improve drought mitigation and contribute to local livelihoods

When the social benefit aspect was considered, there was a decline in highly suitable areas for restoration in the Cacadu catchment (Figure 4.12). The prioritised EI areas in the Cacadu catchment formed a network in the middle to lower catchment when considering the local livelihoods, which correlates with the villages in the catchment, except for wetlands (Figure 4.12). Based on community stakeholder advice, all croplands were vital for providing livelihoods. It was assumed that the same drought mitigation suitable areas were equally suitable for improving local livelihoods. Most of the rangelands that the community stakeholders prioritised in the S10F quaternary catchment fell within the poorly to highly suitable restoration levels (Figure 4.12). Most of the highly prioritised grassland areas had proximity to river lines areas, which is an important combination for livestock owners (Box 4.1; Figure 4.12). A significant improvement of other local livelihoods would be obtained from restoring 419.3 and 841.4 ha of riparian margins, and most of these host second and third-order tributaries (Figure 4.12).

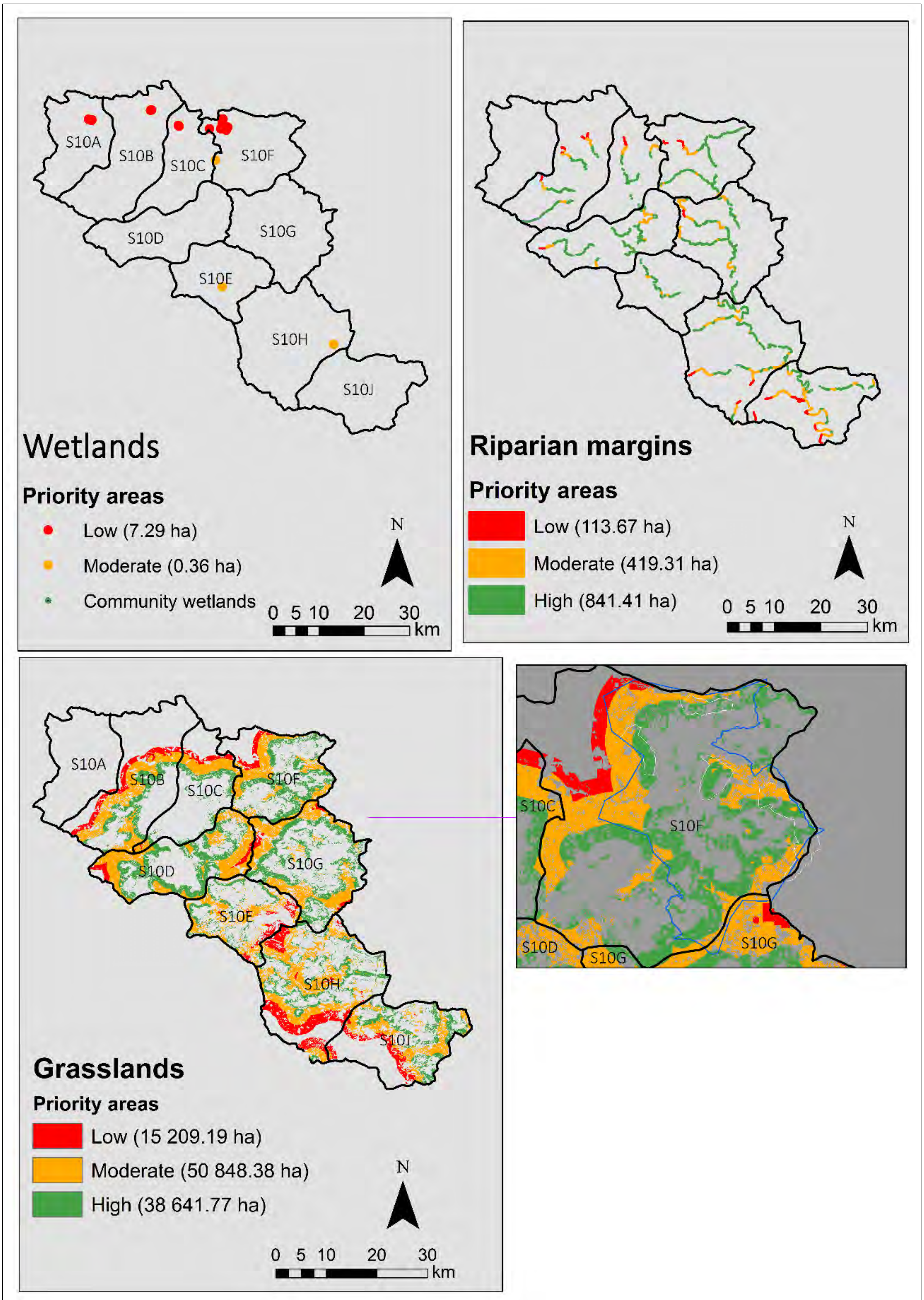


Figure 4.12: The results of priority areas for restoration including priority areas for rural livelihoods in the Cacadu catchment. The annoted map shows priority restoration areas for drought mitigation and areas of potential livelihood benefits in the Machubeni communal areas. The white polygons are stakeholder prioritised rangelands presented in Figure 5.7.

4.4. Discussion

This chapter reports on an approach used (stakeholder informed GIS-AHP approach) to prioritise EI resources for restoration to improve the flow regulation function of catchments. This study used the Cacadu catchment as a case site, and four focal EI resources (wetlands, riparian margins, abandoned croplands and grasslands) were identified as essential resources for their hydrological importance. The utility of stakeholder inclusive multi-criteria decision support has been demonstrated in the IAP prioritisation case study in the Western Cape (Forsyth et al., 2012). Values from diverse actors were combined with spatial datasets to prioritise over 300 quaternary catchments for clearing. Another benefit of stakeholder inclusion in decision planning is the southern Botswana ecosystem service valuation case study, where mixed-methods were used to understand shared-values for ecosystem services (Favretto et al., 2016). Understanding social characteristics in the present study helped determine where EI investments could be targeted, similar to the prioritisation for IAP clearing (Forsyth et al., 2012) and the ecosystem service valuation (Favretto et al., 2016) case studies. The process followed in this study to produce the suitable areas EI for restoration was centred on the numerical overlay in a GIS platform to integrate the attributes and criteria. The numerical overlay requires consistent indicator ranges, for which this study used a 1-5 indicator range. Despite the uncertainty about the omitted wetlands, the above finding suggests that even more wetlands in the catchments will contribute significantly to the flow regulation function of the catchment. Even though the portion of riparian margins detected by the AHP model was nearly insignificant, over 1 100 ha could have some contribution towards meeting the AHP goal (Table 4.8). A substantial portion of all abandoned croplands (87%) would significantly contribute to flow regulation in the Cacadu catchment, while 5% were poorly suitable for meeting the AHP goal (Table 4.8). Restoration of nearly half of all grasslands (97 894.62 ha) in the Cacadu catchment could meet the AHP goal, while another considerable amount (32%) would have some limitations (Table 4.8).

Some countries, including South Africa, have a budget set aside for mitigating ecosystem degradation, but the success of most projects is still low (van Wilgen et al., 2012b; Sapkota et al., 2018). The lack of follow-up restoration, low ecosystem protection (Skowno et al., 2019), vague goals (e.g. the employment target) (van Wilgen & Wannenburgh, 2016), and inappropriate selection of restoration sites (Kraaij et al., 2017) are some of the reasons for poor ecosystem recovery from projects such as the Working for Water Programme; this consequently leads to low repair of ecosystem functions such as flow regulation. A common feature of restoration projects is defining an appropriate operating scale (Pollard et al., 2011; Sayer et al., 2013; The GEF, 2013; Freeman et al., 2015; Liniger et al., 2019), which in the context of this study is the village to tertiary catchment scale. This study used the AHP method (Saaty, 1990) as an evidence-based decision support system to synthesise and analyse existing knowledge for identifying priority restoration areas to improve the flow regulation function of rural catchments. The AHP method used here combined community stakeholder priorities, recent spatial datasets, and expert knowledge of the flow regulation function of catchments to generate suitable areas for restoration. Community stakeholders provided three main criteria, namely ecosystem health, hydrologic functionality and social benefit, which were used to guide the AHP process. Twelve spatial datasets that correspond to the main criteria were identified and used as inputs to conduct the AHP model. In line with the AHP process requirements, the attributes were ranked by a group of 22 experts that have different experience levels in restoration management within the South African context.

4.4.1. Evaluation for prioritised EI areas for restoration to improve drought mitigation

The results suggest that considerable improvements in water-related ecosystem services in semi-arid rural catchments can be attained from restoring less than 50% of grasslands, nearly all abandoned croplands, just over 10% of riparian margins and a few wetlands. The AHP model results also signal that restoration in the Cacadu catchment can be focused in six areas (S10B, S10C, S10D, S10F, S10G, and S10J), which host all four focal EI categories, have a high-water flow regulation potential, but are generally in poor ecosystem conditions. These catchments, except S10J, form a network in the central part of the catchment, which demonstrates the value of considering several quaternary catchments simultaneously to maximise the restoration outcome for flow regulation. The resilience principles (Folke, 2006) and other restoration literature (Rohde et al., 2006; van der Waal & Rowntree, 2018) emphasise the need to consider ecological connectivity when identifying priority areas for restoration. At the grassland biome scale in South Africa, Egoh et al. (2011) estimated that 40% of water supply could be obtained from investing in less than 15% of the area, while 60% could be attained from 56% of the grassland biome. Like this study, Egoh et al.'s (2011) ecosystem service hotspots (catchments) formed bundles, which are important for governance interventions. Considering the social-ecological system and the main goal for the AHP in this study, the clustering of priority areas in the Cacadu catchment is important for facilitating ecosystem recovery and hence improved ecosystem services, including flow regulation.

The AHP model findings suggest that grasslands and old fields might have relatively high importance for water flow regulation in the catchment. In two South African catchments, Mander et al. (2017) estimated substantial stream gains from removing IAPs and revegetation of hillslopes. The findings in the three catchments could be applicable in the Cacadu catchment since the catchment is composed of an undulating topography (Figure 5.2), with hills and abandoned croplands covered by the *E. floribundus* plant (Shackleton & Gambiza, 2008). Woody encroachment is estimated to consume 40% less water than IAPs (Stafford et al., 2017) but woody encroachment leads to soil crusting, as reported by Kakembo (2009), which causes further dry season streamflow reductions (Section 4.2.3.4). On the other hand, Le Maître et al. (2016) estimated streamflow reductions at a national level, and a closer look at their results revealed over 25% reduction in streamflow due to IAPs in the Cacadu catchment. The catchment also has a prominence of erosion due to high stocking rates similar to the Karoo region (Rowntree, 2013). Therefore, encouraging native vegetation in rangelands and maintaining basal cover in abandoned croplands could result in positive flow regulation gains in the Cacadu catchment.

Several other studies within the South African context focused on prioritisation for water security and examples are from the Mpumalanga and Western Cape regions (Forsyth et al., 2011, 2012). However, the studies (Forsyth et al., 2011, 2012) were conducted in distinctly different biome contexts (i.e. the savannah, nama and succulent karoo and the fynbos regions), whereas the present study was explicitly in the grassland biome. Secondly, Forsyth et al. (2012) specifically targeted one degradation driver – IAPs due to their known impact on water resources (Le Maître et al., 1996, 2015). On the other hand, this study incorporated various degradation drivers, including IAPs, to provide a complete view of the restoration need in catchments as recommended elsewhere (Allan et al., 2013). The Mpumalanga Province hosts some of the most important ecological Reserve sites through the Crocodile river (Roux & Selepe, 2013), suggesting the importance of water flow to regulate water quality especially in the dry season (ICMA, 2010). The expert only group in Mpumalanga (Forsyth et al., 2011) provided the following hierarchy: Provision of water (0.404) > Biodiversity displaceability (0.288) > Land capability (0.129) > Protected areas (0.09) > Presence of priority IAPs (0.055) > Negative impact on cultural ecosystem services (0.033). The hierarchy provided by the multi-stakeholder group in the Western Cape province (Forsyth et al., 2012) was as follows: Management capacity of IAPs (0.424) > Potential for control measures to yield positive returns (0.223) > Degree of biodiversity threat by IAPs (0.173) > Conservation (0.104) > Job creation (0.038). Both provinces host Strategic Water Source areas that are important for water supply) (Nel et al., 2011; Le Maître et al., 2018c), and are threatened by IAPs (Le Maître et al., 2016). The Western Cape (The Cape Floristic region) in is also one of the global terrestrial biodiversity priority hotspots (Myers et al., 2000), meaning restoration will focus on protected areas first. The differing weight hierarchies in the two sites demonstrate the contextual-relevance of restoration priorities (Forsyth et al., 2011, 2012). Therefore, the weight hierarchy provided in this study (Section 5.3.2.2) might be applicable to all rural areas within the grassland system.

4.4.2. Evaluation for stakeholder engagement outcomes and livelihood priority areas

The experts allocated the highest rank to present ecological conditions, which could be linked to the influence of hydrological function on water-related services such as water flow regulation, sediment control, and determining the acceptability of water quality levels for domestic and ecological needs. Studies using hydrological modelling have made a compelling case for the importance of ecosystem protection to improve hydrological returns (Le Maître et al., 2014, 2016; Mander et al., 2017; Nel et al., 2017; Hughes et al., 2018a), which could explain the importance placed by experts on the ecosystem protection attribute. Lastly, the experts indicated that annual runoff and potential for streamflow recharge by groundwater have higher importance for determining suitable areas for restoration to improve drought mitigation. The importance allocated to the two attributes (surface water runoff and potential of streamflow recharge through baseflow) could be due to the acknowledgement of the important role played by high water yielding areas in a water-scarce country like South Africa, even though the Cacadu catchment falls outside the SWSA (Nel et al., 2013, 2017).

As discussed by Ananda & Herath (2003) in a case study for forest planning, this study affirms that the AHP protocol effectively fit in with other participatory decision-support tools. Using the AHP protocol, the study prioritised areas that could be restored to improve drought mitigation and highlight other ecosystem service preferences systematically and transparently that can be adopted in other complex decision-support contexts. The outcomes from community stakeholder advice revealed that the most important EI areas are within 2 km of the villages and should benefit the most local population. The emphasis on nearby resources for ecosystem service provision areas in the Cacadu catchment could have been facilitated by safety concerns for the female-dominated households and the ageing rural population who may no longer have the physical strength for tracking long distances to extract resources (Shackleton et al., 2014b; Chris Hani District Municipality, 2017; Sisitka & Ntshudu, 2017). Identification of ecosystem service trade-offs for restoration decision support is an important contribution to the restoration agenda because one of the intended outcomes of restoration is improving the social welfare of locals (Gann et al., 2019) and contribute to international development targets (Cumming et al., 2017). Incorporating multiple stakeholders and various datasets in the decision-making process implies that the outcomes from this study are justifiable, logical and reproducible, which is the fundamental strength of the AHP approach (Belton & Stewart, 2002; Rohde et al., 2006).

The results on priority areas for local livelihoods support the emphasis on the importance of restoration in human-nature relationships (Chan et al., 2016; Gann et al., 2019). This study uncovered up to 51.86% of EI areas (which include 38 641.77 ha of grasslands, 841.41 ha of riparian areas and 0.36 ha of wetlands) that are highly suitable for local livelihoods. As a result of the 2 km buffer limit, EI investment into grasslands and riparian margins could maximise local livelihoods in the catchment, which is in line with the priority areas identified by community stakeholders (i.e. a total area of 2017.14 ha in rangelands, healthy riparian areas and 33 wetland areas in GEF5 Machubeni villages). Abandoned croplands are an essential EI category for food production, and 5284.59 ha of abandoned croplands were prioritised for restoration to improve food security. The importance of restoration in societal benefits has been outlined in a recent review based on 37 local studies (Crookes & Blignaut, 2019). Amongst others, the review highlights that the lack of investment into conservation agriculture is worth $541/ha/year; the cost of degraded communal rangelands has an average annual value of $138/ha; neglecting erosion control is worth an annual average of $52/ha; and unmanaged bush encroachment is worth around $12.3/ha/year (Crookes & Blignaut, 2019).

4.4.3. Chapter limitations and conclusions

Regarding the weight coefficients, this study assumes that the weights would be consistent across the quaternary catchments. However, considering the contexts, the actual weights for prioritising catchments may vary. Varying weights would lead to a different prioritisation; for example, if the downstream benefits (provided by the water from the large dams in the catchment) were considered, the current 1:1 weighting for the social benefit would have to be changed to reflect the number of people benefiting from the dams. The Integrated Development Plan (Emalahleni Local Municipality, 2020) documents a pipeline infrastructure update for bulk water supply to 4158 beneficiaries 25 km away from the Xonxa Dam (S10E), consistent with municipal obligations under the National Water Act 36 of 1998. Based on this water supply case, the population density indicator would have to outweigh the proximity indicator. Changing weight coefficients to meet the prioritisation objective can be seen in two case studies (Macfarlane & Atkinson, 2015; Snaddon et al., 2018) that used followed a similar approach. Macfarlane & Atkinson (2015) allocated a 15% weight to streamflow regulation at a national level while prioritising wetlands for water security. Meanwhile, Snaddon et al. (2018) followed the same approach in the Western Cape but used higher weights to the streamflow regulation factor (40% weighting) since the attribute has more influence on water supply in the upper catchment areas. The influence of weighting on the prioritisation model is evident in the prioritised wetlands as wetland size and wetland types were allocated 14% influences, which led to moderately sized wetlands being the most detectible by the model.

Still, on the weighting step, the use of expert-derived weights was unavoidable due to Covid-19 shortcomings[1]; however, the approach reduces the independence of local stakeholder values to present their contextual-knowledge without outside influence (Rodela et al., 2012; Tengö et al., 2014). This limitation implies that stakeholder weights may differ from those given by experts since experts are influenced by their research background and experience (Abrahams et al., 2019). Local stakeholders are influenced by personal needs, cultural practices, social learning (Reed et al., 2010). Nevertheless, the diverse perspectives from combined focus group discussions and expert-knowledge could benefit the conclusions by providing further insights since restoration draws on various knowledge-basis and experts are also involved in the restoration process (Gann et al., 2019).

To conclude, a GIS-based suitability assessment, coupled with stakeholder engagement, is recommended as a crucial tool for identifying priority areas for rehabilitation for the goal of water security, especially in rural areas. According to this study, priority areas for restoration to improve drought mitigation co-occur with local livelihood areas. Despite the uncertainty about the potential of priority areas to facilitate ecosystem recovery, the findings of this study indicate that investing in restoration for drought mitigation could yield positive returns against future hydrological droughts. By including community stakeholders and experts in the prioritisation exercise, this study was able to detect EI areas that could be managed to encourage catchment resilience against drought impacts and areas that could improve rural livelihoods. The most important areas for drought mitigation in the Cacadu catchment are S10B, S10C, S10D, S10F, S10G and S10J, and mostly host all four focal EI categories. The previous chapter investigated the hydrological regime of the catchment in response to the possible degradation drivers, and the next chapter synthesises the findings and provides implications for ecosystem management with a close look at the six catchments mentioned above. The approach adopted in this study could be improved by the presence of better input information on the groundwater contribution to streamflow, estimated flow reductions due to woody proliferation, and a complete ecosystem indicator. Secondly, more studies similar to this that includes the economic cost-benefit of investing in EI need to be conducted to support science-based decision-making.

[1] Following the outbreak of the SARS-COVID-19 pandemic, RSA similar to other regions of the world went into a complete lockdown on 26/03/2020 to help manage the spread of the virus. In line with the Disaster Management Act regulations, gatherings and non-essential movement was limited at the time; making it impossible to do any field work.

5. CHAPTER 5: ESTIMATES OF LAND COVER MODIFICATION ON **RURAL CATCHMENT'S HYDROLOGY IN SOUTH AFRICAN** GRASSLANDS

This chapter is being prepared for submission to the African Journal of Aquatic Sciences: as: Mahlaba B, Xoxo BS, Tanner J, Mantel SK, Mahlaba B, Le Maître DC. Assessment of hydrological impacts due to potential land degradation drivers and implications for drought mitigation: A case study from the Tsitsa and Cacadu catchments.

Conceptualisation, WM, BSX, JT, SKM; Methodology, WM, BSX, SKM, JT, DLM; Investigation: BSX (Cacadu catchment), BM (Tsitsa catchment); Formal Analysis, BM, BSX, JT Resources, SKM, JT; Writing - Original draft preparation, BM; Writing – Review and Editing, BSX, JT, SKM, DLM; Visualisation, BSX, BM; Supervision: SKM, JT; Project administration: SKM, JT; Funding acquisition, SKM, JT, BM, BSX.

5.1. Introduction

Degradation of catchments due to human activities is a major threat to freshwater availability, particularly in the semi-arid regions, as it alters the catchment-specific equilibrium between rainwater that infiltrates into the soil and overland flow. Different catchment processes interact to govern flow regulation throughout the catchment hydrologic cycle (Brauman et al., 2007; van Loon & Laaha, 2015). Land cover is a significant variable that impacts the flow regulation capacity of a catchment and in many catchments is dynamic owing to changing land-use demands (van Loon & Laaha, 2015). The presence of IAPs in South African landscapes, for example, is associated with an estimated 3% reduction in mean annual runoff (an annual loss of close to 1 500 Mil m^{-3}) for the country as a whole (Le Maître et al., 2016). Although not widely reported, the expansion of indigenous woody species (bush encroachment) may have similar consequences to IAP invasions, since trees that encroach on grassland or savanna dominated areas alter surface runoff and result in increased losses through evapotranspiration (Sholto-Douglas et al., 2017; Le Maître et al., 2020). Overgrazing of natural landscapes leads to bare ground exposure that causes an increase in surface runoff due to soil damage, thereby reducing the capacity of catchments to store water for slow release during low-flow periods (Hughes et al., 2018b). Destruction of wetlands reduces the functionality of wetland ecosystems to regulate water, capture sediment, and filter pollutants (Rebelo et al., 2015, 2018; Pantshwa & Buschke, 2019).

Terrestrial landscapes have a system of feedback loops that interact to determine the catchment's hydrological regime (McGuire et al., 2005; Le Maître et al., 2014). Although the term EI is seldom mentioned, based on the literature above, it is evident that water fluxes respond differently to different land covers (EI modification) depending on the location and the climate. Since land cover is one of the fastest-changing catchment variables (van Loon & Laaha, 2015), where changes influence the dominant feedbacks that maintain a particular water flow regime, the streamflow regulation ecosystem service may also be compromised (Chapter 2). Therefore, it is critical to understand specific catchment degradation drivers and link those to consequences on the hydrological regime to help inform the management of catchments. However, the complex African and South African landscape and data scarcity reduce the ability to reliably quantify the EI impact on flow regulation (Hughes et al., 2015).

In light of data scarcity related to flow regulation processes, hydrological models can be used to understand better the factors that lead to changes in the hydrological system. Hydrological models provide simulations for storage and water flow across a defined area (Hughes, 2013; Haregeweyn et al., 2015; Kusangaya et al., 2017). In the context of modified EI effect on catchment hydrology, the two commonly used analysis approaches are regression indicators (McIntyre et al., 2014) and the modelling of land-use scenarios (Warburton et al., 2012; Rebelo et al., 2015; Gyamfi et al., 2016; Gumindoga et al., 2018).

Having quantified the degradation extent in the Cacadu catchment (Chapter 3), this study assesses the hydrological sensitivity of the catchment to identified degradation drivers through hydrological modelling. Due to the importance of EI as a source of multiple ecosystem services (SANBI, 2014; Table 5.1), comparisons between intact and modified catchment hydrology could act as a valuable data-informed decision-support system for investing in EI (Mander et al., 2017; Hughes et al., 2018b; Rebelo et al., 2018). Le Maitre et al. (2014) conceptualised the flow regulation ecosystem service for intact and modified catchments in the fynbos region to better explain the benefits of a sustainably managed catchment. They (Le Maître et al., 2014) assert that sustainably managed or pristine catchments generally retain more water than altered catchments resulting in higher groundwater recharge and river baseflow, and less surface runoff. This study builds on the conceptual model of Le Maitre et al. (2014) to describe streamflow dynamics in natural vs modified rural catchments found in the grassland biome to make a case for how investing in EI through action can play a role in the flow regulation service of rural catchments and ultimately contribute to drought mitigation.

5.2. Materials and Methods

5.2.1. Model selection and description

The Pitman Model (Pitman GWv3 Model) was selected to represent the runoff regime in natural vs modified catchment areas (Hughes, 2004; Hughes & Forsyth, 2006; Kapangaziwiri & Hughes, 2008). The Pitman GWv3 Pitman Model is a conceptual monthly time-step hydrological model that is applied at a quaternary catchment scale (50 to 1 000 km^2) to simulate natural flows and is hosted by the SPATSIM software (Spatial and Time Series Information Modelling) (Hughes & Forsyth, 2006). Although the model has been used widely within southern Africa (Hughes, 2013), including in data-scarce regions (Ndzabandzaba & Hughes, 2017), the monthly temporal scale of the model remains the largest source of uncertainty that may lead to underestimations; unlike the ACR model which operates at a daily scale (Shackleton et al., 2014a; Kusangaya et al., 2017). A detailed description of the Pitman rainfall-runoff model is available in Hughes (2004, 2013). The Pitman Model selection in this study is motivated by the extensive usage in the southern African region (Hughes et al., 2015) and the expertise on using the model within the Institute for Water Research at Rhodes University.

The model structure is shown in Figure 5.1, and the simulated water balance components include surface runoff, soil moisture/unsaturated zone flow and groundwater (Hughes, 2004; Kapangaziwiri & Hughes, 2008; Figure 5.1). Parameters corresponding to surface and groundwater functions are described in the appendices (Table A3). The model water-use parameters are outlined in Table A4. The hydrological impact of lakes and on-channel wetlands are estimated using the wetland sub-model parameters (Table A4). The use of parameters in the Pitman GWv3 model used here is beneficial for physical relevance (Hughes & Forsyth, 2006), making it suitable for implementing the model by Le Maître et al. (2014), although the model was originally developed for a physical model.

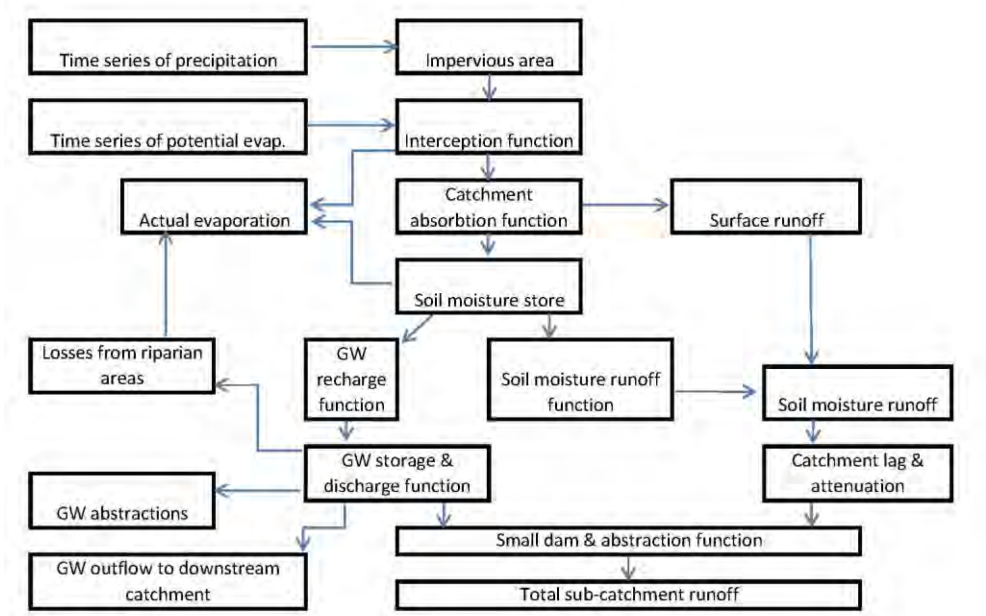


Figure 5.1: Graphical representation of the Pitman model structure.

5.2.1.1. Constructing a water balance

This analysis used a basic water balance computed as the difference in inflows and outflows to the catchment, and the change in storage. In terms of groundwater, outflows from the groundwater storage (aquifer) are a combination of groundwater contribution to streamflow as baseflow, downstream groundwater movement, abstracted groundwater, and evapotranspiration losses (directly from groundwater in riparian areas or wetlands). Inflows to the groundwater storage include groundwater recharge, transmission losses – infiltration into the surrounding aquifer through intermittent streams (de Vries & Simmers, 2002), and upstream groundwater inflow. The groundwater-surface water interactions in response to changing land cover were evaluated using the mass water balance at a quaternary catchment scale. Modelling at the quaternary catchment scale results in inevitable lumping of some of the land uses and catchment characteristics; however, the dominant land uses and types were represented as far as possible.

5.2.2. Differentiating natural from modified land

The first requirement to assess the hydrological impacts of the major degradation drivers in the catchment is reference conditions (i.e. baseline) against which degradation drivers (land modification) can be compared (Warburton et al., 2012; McIntyre et al., 2014; Mander et al., 2017). Hydrological processes are a function of climate and physical catchment properties such as soil type and depth, and therefore a land cover is important in determining how water moves through a catchment. This influence can be viewed at several scales, from detailed water movement assessments through the numerous layers of the unsaturated zone through larger-scale assessments of land cover effects on infiltration vs surface runoff. This study used baseline land-use information from the 30 m resolution 1990 to 2018 NLC Change dataset (GTI, 2019a) and the 20 m resolution 2018 NLC dataset (GTI, 2019b) to simulate the comparison surfaces (scenarios) within quaternary catchments. The 1990 to 2018 NLC Change Assessment dataset was chosen for its consistent land cover interpretation, making it similar to the 2018 NLC dataset (GTI, 2019b), improved from the original 1990 NLC dataset (GTI, 2016).

Using land cover change over 28 years (GTI, 2019a, 2019b), this study consolidated the 72 land cover classes in the datasets into two main land cover types: natural and modified (Table 5.1). The natural land cover classes included grasslands, and native tree-covered areas. Wetlands were represented as riparian areas. Possible hydrological impacts of anthropogenic-induced land modifications (termed as land cover degradation in Chapter 3) were accounted for by combining afforestation, cultivated land and artificial surfaces (Table 5.1). Dams were also represented in the model, although only as storage, and without significant water use due to the lack of water use data.

Table 5.1: Conversion of 1990 to 2018 and 2018 National Land Cover and UNCCD Land Cover categories into flow regulation scenario classes for the rainfall-runoff comparison.

National Land Cover categories	UNCCD categories	New legend
Indigenous Forest Thicket/ dense bush Natural Wooded Land	Tree-covered areas	Natural land/Afforested (if tree-cover was a result of conversion from other land types)
Planted forest		Afforested (Modified land)
Shrubland Grasslands	Grassland	Natural land
Wetlands	Wetlands	Wetlands (Riparian areas)
Barren Land	Other lands	Natural land
Eroded Lands Mines		Modified land
Permanent Orchards Permanent Vines Commercial Annual Irrigated Commercial Annual Non-Pivot Cultivated Subsistence	Cropland	Modified land
Built-up Residential All Built-up Smallholdings Built-up Commercial Built-up Industrial	Artificial surfaces	Modified land
Waterbodies	Waterbodies	Waterbodies (Dams)

Over the 28 years from 1990 to 2018, natural land cover (which includes grasslands, shrublands, barren land, persistent woody cover from 1990 to 2018, and wetlands in both land cover datasets) was generally high (over 70%) across quaternary catchments (Table 5.2).

Across the catchment, natural land cover (including wetlands) declined (decline range between 1.17 and 21.41%) in all but the S10A and S10C quaternary catchments, where there was a recovery of 8.85 and 3.94%, respectively. Modified areas (croplands, abandoned croplands, artificial surfaces and eroded lands) occupied the second-largest land cover class with a percentile range of 17.35 to 32.67% in 1990 (Table 5.2). By 2018, the modified category increased slightly with an incremental range of change 1.27 to 21.45% across areas, except for S10A (declined by 9.33%), and S10C (decline by 3.03%), and S10J (declined by 1.20%) (Table 5.2). Since modified lands combined a range of uses, further analysis was done to detect the dominant modified land class. Woody encroached subsistence croplands were found as a dominant modified land class across the White Kei. Afforestation in the catchment is very low, with a variable change ranging between -3.25 to 0.44% between 1990 and 2018 (Table 5.2).

Table 5.2: Summary of percentage of land cover in the White Kei sub-basins in 1990 and 2018 arranged using the new legend in Table 5.1 and afforested riparian areas added. The figures in brackets show the difference between 1990 and 2018. Differences are represented using the **green** ink for expansion and the **red** ink for loss.

Catchment	Natural land		Wetlands		Waterbodies		Modified		Afforestation		Afforested riparian	
	1990	**2018**	**1990**	**2018**	**1990**	**2018**	**1990**	**2018**	**1990**	**2018**	**1990**	**2018**
S10A	81.08	88.45 (7.37)	0.60	2.96 (2.36)	0.08	0.47 (0.39)	17.34	8.01 (9.33)	0.90	0.12 (0.78)	0.09	1.17 (1.08)
S10B	81.18	79.69 (1.49)	0.60	0.36 (0.24)	0.08	0.23 (0.15)	18.12	19.69 (1.57)	0.02	0.04 (0.02)	0.03	27.33 (27.3)
S10C	65.86	70.14 (4.28)	0.39	0.05 (0.34)	0.52	0.15 (0.37)	32.67	29.64 (3.03)	0.55	0.02 (0.53)	0	1.48 (1.48)
S10D	77.42	75.47 (1.95)	0.15	0.11 (0.04)	0.00	0.77 (0.77)	22.34	23.61 (1.27)	0.1	0.04 (0.06)	0.12	3.95 (3.83)
S10E	68.14	61.78 (6.36)	0.06	0.02 0.04)	5.34	5.6 (0.26)	26.4	32.52 (6.12)	0.07	0.08 (0.01)	0	0.69 (0.69)
S10F	69.49	67.09 (2.40)	0.46	0.08 (0.38)	0.37	0.97 (0.60)	28.79	31.74 (10.29)	0.89	0.12 (0.77)	0.18	4.23 (4.05)
S10G	75.97	69.43 (6.54)	0.20	0.1 (0.10)	0.01	1.48 (1.47)	23.47	28.85 (5.38)	0.35	0.15 (0.20)	0.48	5.47 (4.99)
S10H	76.97	68.98 (7.99)	0.31	0.25 (0.06)	0.02	0.78 (0.76)	22.7	29.99 (7.29)	0.00	0.01 (0.01)	1.09	1.4 (0.31)
S10J	72.64	76.23 (3.59)	0.15	0.04 (0.11)	1.77	1.16 (0.61)	23.73	22.53 (1.20)	1.71	0.03 (1.68)	0	14.17 (14.17)
S20A	77.31	74.07 (3.24)	0.33	0.06 (0.27)	1.46	1.65 (0.14)	19.92	23.88 (3.96)	0.98	0.34 (0.64)	0.08	0.22 (0.14)
S20B	77.32	75.6 (1.72)	0.46	0.03 0.43)	0.36	0.44 (0.08)	21.61	23.24 (1.63)	0.24	0.68 (0.44)	0.02	2.88 (2.86)
S20C	73.00	61.18 (11.82)	0.15	0.03 (0.12)	1.92	2.55 (0.63)	24.89	36.22 (11.33)	0.04	0.03 (0.01)	0.04	1.15 (1.11)
S20D	66.36	66.02 (0.34)	0.13	0.03 (0.10)	0.02	0.87 (0.85)	30.20	33.04 (3.14)	3.29	0.04 (3.25)	0.14	5.19 (5.05)

5.2.3. Physical catchment properties

The 13 selected quaternary catchments of the White Kei vary in size from 23638 to 55211ha, with the Cacadu quaternary catchments covering 236.38 to 472.99 km^2 and where the Indwe quaternary catchments (S20) covers 29810 to 55211 ha (Table 5.3). Lower rainfall is measured in the Cacadu catchment (S10), with a maximum average of 621.10 mm/year lower than the minimum annual average of 623.49 mm for the Indwe catchment (Table 5.3). The higher annual rainfall in the Indwe catchment is despite the slightly lower elevation (mean catchment elevation ranges between 1059.9 to 1497.3 m.a.s.l) compared to the Cacadu catchment, which had a much wider elevation range of 889 to 1602.2 m.a.s.l.

Table 5.3: Physical catchment properties for the 13 quaternary catchments of the White Kei (9 in the Cacadu catchment and 4 in Indwe catchment).

Catchment	Size (ha)	Rainfall (mm/a)	Potential Evapotranspiration (mm/a)
S10A	25,765	527.62	1650
S10B	39,868	579.28	1650
S10C	23,638	545.62	1650
S20D	31,680	593.67	1650
S20E	24,043	588.53	1600
S10F	30,100	584.09	1650
S10G	37,732	621.30	1600
S10H	47,299	571.49	1600
S10J	32,408	565.50	1550
S20A	29,810	626.97	1600
S20B	44,684	623.49	1600
S20C	55,211	634.24	1600
S20D	30,962	682.02	1550

The White Kei catchment is predominantly underlain by mudrocks (i.e. the Tarkastad Subgroup, the Molteno, Clarens and Elliot Formations), with intrusions of Karoo basalts (Smith et al., 1993; CGS, 2018). Although the Karoo aquifers have variable yields, the rocks that underly the White Kei catchment are known for low permeability, leading to low groundwater storage and subsequently low recharge from the surface when compared to other Karoo Supergroup deposits (Woodford & Chevallier, 2002). The elevation profiles for the catchment, expressed as terrain components in Figure 5.2, adapted from Schulze et al. (2007), indicate that an undulating topography characterises most of the White Kei catchment with a gentle relief at the margins and steep segments towards the interior. Bailey & Pitman (2015) soil classification for the White Kei (Figure 5.2) shows that the catchment mostly consists of moderate to deep clay soils in the north-eastern parts and moderate to deep sandy-loam soils in the rest of the catchment. Soils in the catchment have low to moderate drainage and moderate to high susceptibility to erosion (Figure 5.2).

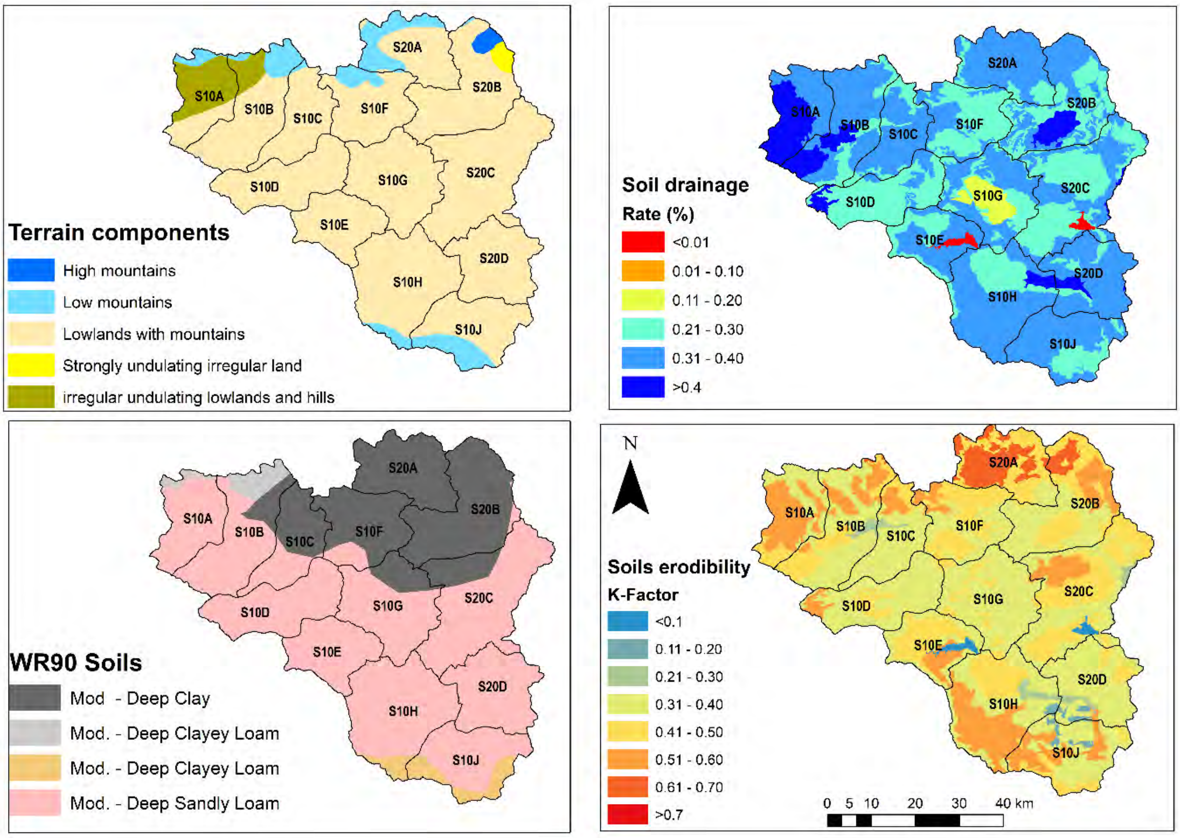


Figure 5.2: Terrain and soil in the White Kei catchment extracted from Schulze (2007) and Bailey & Pitman (2015).

5.2.3.1. Rainfall data

The minimum required data for the Pitman hydrological model are rainfall and potential evapotranspiration (Hughes, 2005). This study used historical time-series rainfall data from the 2012 Water Resources (WR 2012) dataset, which covered the period from 1920 to 2009 (WR, 2012). Since the WR 2012 data ended in 2009, and the assessment runs until 2018, remotely sensed data alternatives were considered to complete the dataset. Remotely sensed data options in hydrological modelling exercises have been used by hydrologists for the last two decades ago following the recognition that satellite data observations have utility in hydrological modelling in data-deficient regions (Sivapalan et al., 2003; Hrachowitz et al., 2013).

Figure 5.3 shows frequency distribution curves for the WR2012 monthly rainfall data and two remote sensed datasets in the White Kei catchment at different times between 1920 and 2019. The WR 2012 dataset spanned over 1920 to 2010, while the CHIRPS and TRMM datasets covered the years 1981 to 2019 (Figure 5.3). The WR 2012 dataset and the CHIRPS datasets corresponded well in the upper (S10A-C) and lower (S10G-S10J) quaternary catchments of the Cacadu catchment (Figure 5.3). The same trend was detected across the quaternary catchments in the Indwe catchment (Figure 5.3). In the middle quaternary catchments of

Cacadu (S10D-S10F), the WR 2012 dataset had slightly lower rainfall estimates than its counterparts, except at low and high rainfall periods (Figure 5.3). The TRMM dataset often exceeded its analytical counterparts (Figure 5.3). Since the TRMM dataset slightly over-estimated rainfall in many of the quaternary catchments, the CHIRPS monthly data were used to supplement the WR2012 rainfall data (Funk et al., 2015). The data was converted into a time series format for input into the SPATSIM software.

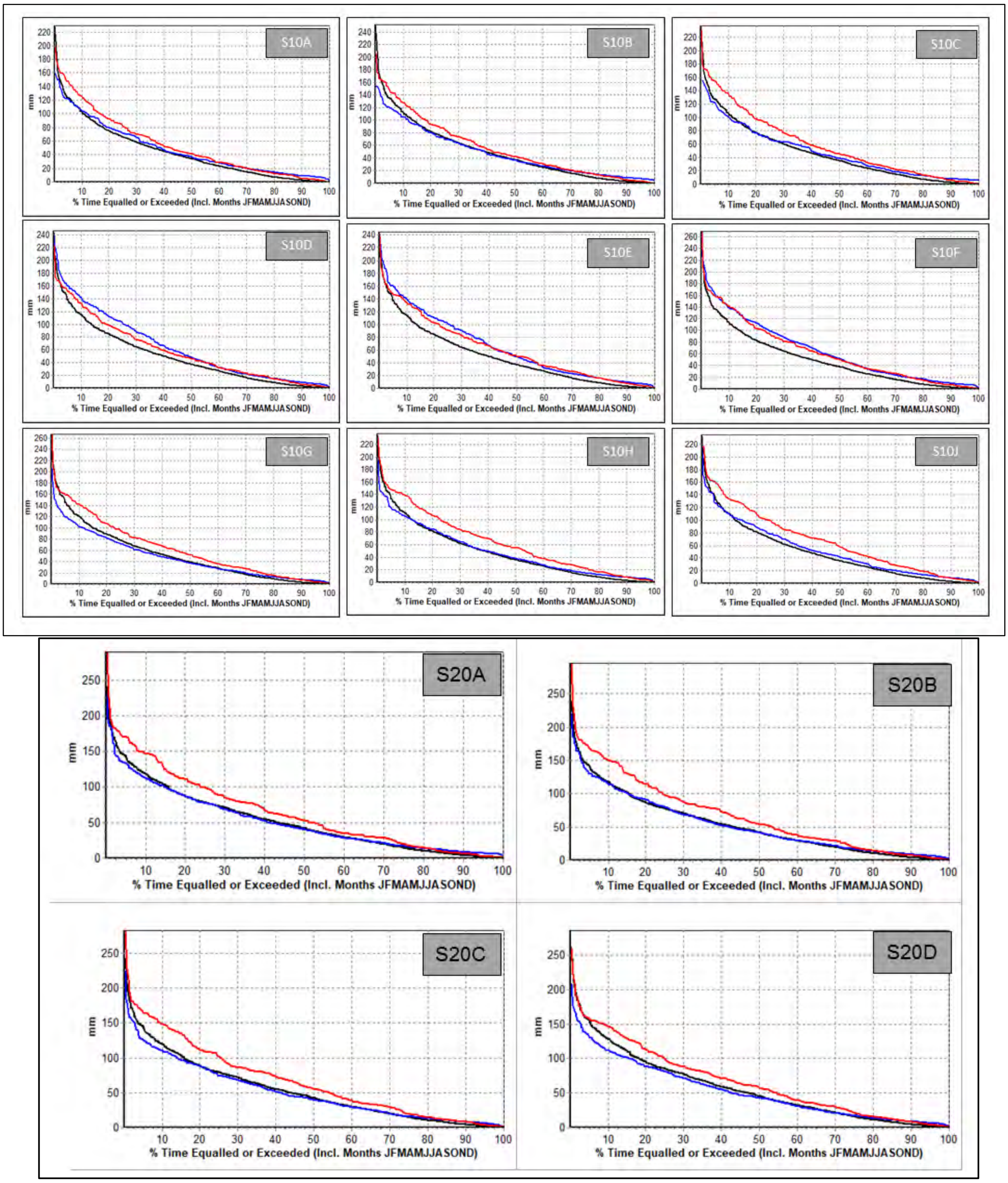


Figure 5.3: Rainfall duration curves showing a monthly distribution analysis for three datasets covering covering the period for WR 2012, and for CHIRPS and TRMM for the quaternary catchments in the White Kei catchment. The lines are coded by dataset, **black**: WR 2012 (1920 to 2010); **blue**: CHIRPS (1999 to 2019), and **red**: TRMM (1997 to 2019). The y-axis display exceedence frequency based on months January to December.

5.2.3.2. Stream gauge input data

The Department of Human Settlements, Water and Sanitation was approached to obtain gauged runoff (daily flow) data for White Kei stream gauges (Table 5.4). At most, the operational stream gauges cover 49 years (Table 5.4). However, only one of the stations in the Cacadu catchment is active (i.e. S1H004); hence, data from the adjacent tertiary catchment (Indwe catchment) was used to improve the model's ability to predict catchment behaviour (Kapangaziwiri et al., 2012). The Indwe catchment has four active stations (S2H005, S2H006, S2H007 and S2H008) (Table 5.4).

Table 5.4: Summary of surface water discharge information for the White Kei catchment.

Catchment	Station ID	Location	Start	End	Gauge status
S10E	S1H003	-31.8156; 27.0542	1986	-	Closed & No data
S10H	S1H001*	-31.8502; 27.2196	1984	-	Closed & No data
	S1H004	-31.8474; 27.1852	2003	2019	Active
S10J	S1H002	-32.0189; 21.3603	1980	-	Closed & No data captured
S20B	S2R002*	-31.5131; 27.3347	1970	1990	Limited data
	S2H006	-31.5231; 27.3319	1970	2019	Active
	S2H007*	-31.5158; 27.3303	1971	2019	Active
	S2H003	-31.7914; 27.4525	1970	-	Closed & No data
	S2H008*	-31.5142; 27.3303	1985	2019	Active
S20C	S2R001*	-31.7958; 27.4311	1968	-	Under investigation
	S2H001	-31.7744; 27.4214	1947	1965	Closed
	S2H005*	-31.7981; 27.4311	1968	2019	Active
S20D	S2H002	-31.8581; 27.4178	1968	-	Closed & No data
	S2H004	-31.8581; 27.4178	1968	-	Closed & No data found

* denotes gauges that are located below or inside a large dam

Figure 5.4 shows flow duration curves for mean daily runoff over different coverage periods for five active stream gauges at the S10H, S20B and S20C outlets obtained from the ministry of Human Settlements, Water and Sanitation (DWS, 2008). The White Kei streamflow is highly variable, with periodic zero flows indicating typical semi-arid conditions in the Eastern Cape Province (Figure 5.4). Based on S1H004 observations below, mean daily river discharge ranged between 1 and 55 m^3/s (Figure 5.4).

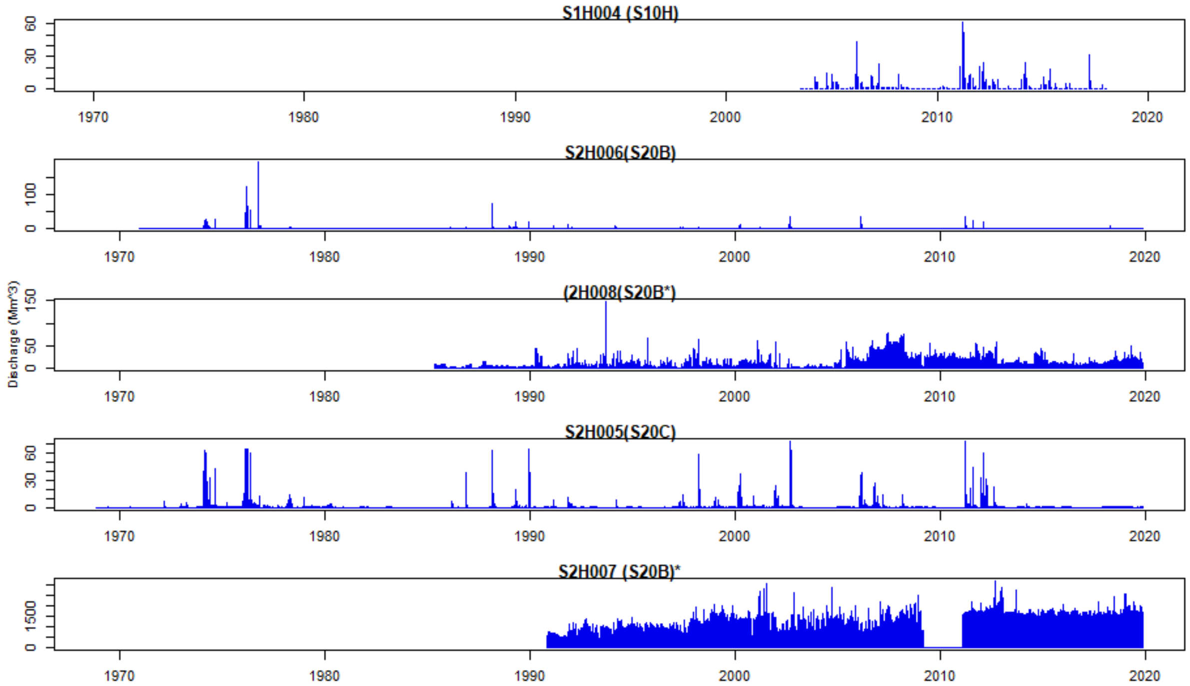


Figure 5.4: Streamflow hydrographs showing observed daily discharge data until 2019 in five active gauges (gauges: S1H004, S2H005, S2H006, S2H007, and S2H008. Gauges S2H007 and S2H008 are dam outlets.

5.2.3.3. Hydrological effects of land modification in rural grassland catchments

This study builds on Le Maître et al. (2014) 's work that was conducted in the fynbos biome region to highlight the flow regulation impacts of land modification. Based on the land cover change between 1990 and 2018 presented in Table 5.2, the most significant EI modification drivers in the Cacadu catchment are cropland areas and afforestation. A conceptual representation of how the two EI modification processes affect water balance is presented below (Figure 5.5). Figure 5.5 is adopted from a review on the impact of urbanisation on groundwater recharge (Schirmer et al., 2013). Since this study is not focused on recharge, the diagram was revised to reflect the impact of land modification on flow regulation as outlined in ecohydrology literature. Four anthropogenic induced land modifications, defined as afforestation, cropland expansion, expansion of settlements, and eroded surfaces, were detected in the catchment (Figure 5.5). All the land cover modifications lead to increments in evapotranspiration and quickflow and reductions in groundwater recharge and baseflow. The land modifications have a variable influence on infiltration, interflow and groundwater usage.

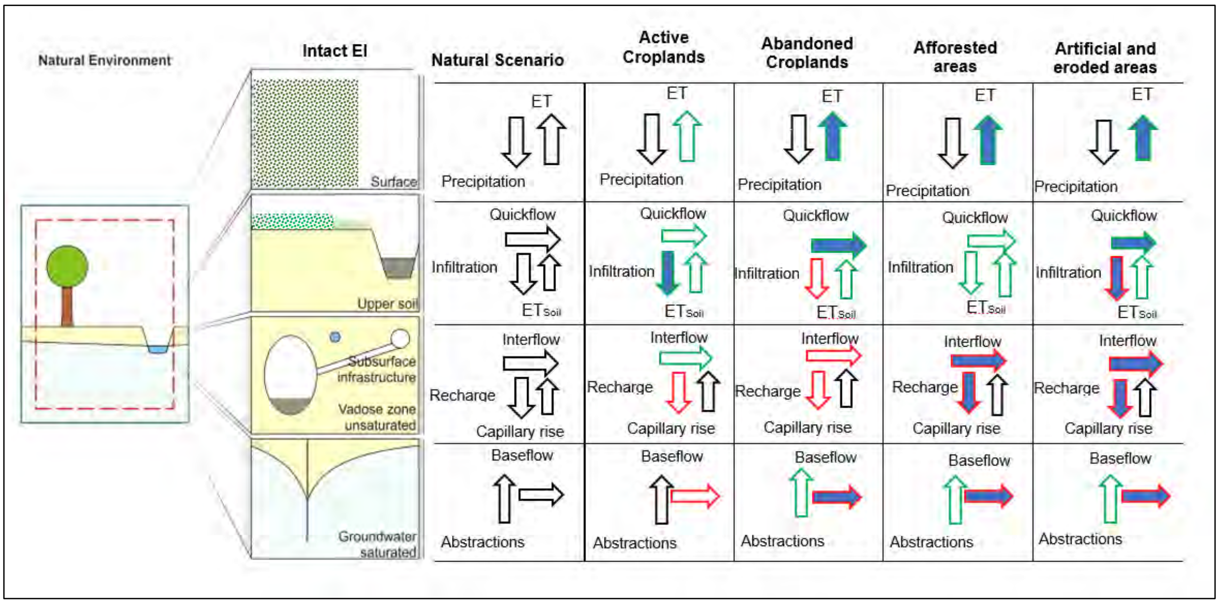


Figure 5.5: Simplified impact of major rural land modification processes in catchment hydrology (adopted from Schirmer et al., 2013). The changes in water balance components are denoted by arrow colours (**black** = natural; **green** = increase in volume; **red** = decrease in volume; **blue fill** = severe modification). Quickflow is a combination of surface and rapid sub-surface runoff.

VI. Hydrology of cropland areas

Cropland agriculture affects the water balance by facilitating rainwater loss through evapotranspiration and quickflow but contribute to catchment hydrology by increasing infiltration and interflow (Figure 5.5). Croplands transform vegetation cover (Foley et al., 2005; Mucina et al., 2006), lead to wetland losses (Rebelo et al., 2015), and fragment landscape connectivity (Figure 5.5), leading to a weakened flow regulation. As discussed in Chapter 2, the hydrological process in cultivated systems is complex. On the one hand, long-term exposure of bare ground between ploughing seasons reduces rainfall interception and encourages quickflow as found in uMngeni catchment, South Africa, which impacts streamflow regulation in dry seasons (Hughes et al., 2018b). On the other hand, tillage activities increase surface friction leading to better infiltrability and groundwater recharge as reported in a semi-arid area in the USA (Scanlon et al., 2008), in a global review by Kim & Jackson (2012) and the Sahel region (Ibrahim et al., 2014). In areas dominated by hillslopes and subsistence farming like the White Kei (Figure 5.2, Table 5.2), tillage is expected to increase infiltration and, therefore, subsurface flow (and maybe groundwater recharge) as demonstrated on different slope types by van Tol et al. (2013). Most croplands are rainwater dependent, but in other areas, irrigation does occur using small farm dams (Cai & Rosegrant, 2002). Therefore cultivation is not expected to influence groundwater.

However, depending on climate conditions, inadequate management of cropland systems leads to environmental degradation (Springmann et al., 2018; Shackleton et al., 2019), including woody proliferation (Blair et al., 2018; Scorer et al., 2019) that has access to subsurface and groundwater sources leading to higher evapotranspiration losses in abandoned cropland areas (Huxman et al., 2005; Scott et al., 2006; Le Maître et al., 2016). Therefore, a combination of bare soil and woody evapotranspiration losses in abandoned cropland areas is expected to negatively impact the low flow hydrology of rural catchments (Figure 5.5).

VII. **Hydrology of afforested areas**

Afforestation in grassland areas leads to negative impacts on catchment hydrology, however, the intensity of the impacts may not be easily quantified doe to the variable hydrological influence of different woody species. Afforested areas in this context included planted forests, IAPs and woody encroached areas (Table 5.2). Le Maître et al. (2015) review that the nature of woody species controls groundwater usage and evapotranspiration rates. At a national level, Le Maître et al. (2016) estimated the impact of IAPs on streamflow, showing that IAPs in the Great Kei catchment (encompassing the White Kei) led to a long-term streamflow reduction of 4.5%. Others (Schulze et al., 2004; Dye & Versfeld, 2007; Tuswa et al., 2019) specifically emphasise the threat posed by commercial afforestation to the South Africa's delicate water resources. In a Namibian and South African study by Stafford et al. (2017), woody encroachment was estimated to consume 40% less water than IAPs. Both studies (Le Maître et al., 2016; Stafford et al., 2017) support the knowledge that afforestation negatively impacts catchment hydrology.

Quickflow is naturally high in steep environments, but the removal of natural vegetation cover intensifies quickflow losses. The dominance of proliferating drought-resistant woody species due to climate change and long-term mismanagement of the catchment as discussed in earlier studies (Shackleton & Gambiza, 2008; Rowntree, 2013), and sparse IAPs (Kotzé et al., 2010) should therefore offset quickflow losses in the catchment. However, this is not often the case since the establishment (up to 40%) of unpalatable grasses in rangelands as reported in Eastern Cape communal lands (Kakembo, 2009; Masibonge et al., 2019) may encourage veld fires in an attempt to restore favourable grass species, which facilitates further quickflow (Le Maître et al., 2014). Soil crusting is also characteristic of woody encroached areas (Kakembo, 2009), further increasing quickflow.

Reduction in native vegetation cover that is important for regulating soil quality and infiltration are consequences that are linked to afforestation as shown in wetland, riparian and terrestrial systems of South African catchments (Le Maître et al., 2015; Rebelo et al., 2015; Ntshidi et al., 2018). Native, agroforest, and planted forest systems facilitate infiltration due to increased bioturbation (Ogden et al., 2013; Yapi et al., 2018). However, possible soil crusting in woody encroached hillslopes (Kakembo, 2009) may offset the high soil infiltration rates due to increased root activity. Soil moisture recharge through infiltration is expected to increase subsurface runoff, except when the woody species are losing the water through transpiration (Figure 5.5). The deep root structure of afforested surfaces (specific to IAPs) gives them access to groundwater sources year-round in many locations, leading to reductions in water tables (Le Maître et al., 2015, 2016), as discussed before for abandoned cropland areas.

VIII. **Hydrology of artificial and eroded surfaces**

One of the main changes in terrestrial systems is establishing artificial surfaces for economic activities (Scanlon et al., 2008). Artificial surfaces are dominated by impervious areas such as constructed surface and sewers (Table 5.1). Like eroded surfaces, artificial areas facilitate quickflow, largely eliminate infiltration and groundwater recharge, consequently reduce subsurface runoff (Sterling et al., 2013; Hughes et al., 2018; Figure 4.5). The socio-economic activities in artificial areas may lead to groundwater withdrawals that reduce water tables and consequently diminish the baseflow contribution to streamflow recharge (Schirmer et al., 2013). However, in areas with municipal water infrastructure, pipe leakages can contribute to subsurface runoff (Lerner, 2002).

5.2.4. Model setup

A total of 13 quaternary catchments were used to study the water balance response to the 1990 and 2018 land modification scenarios compared to natural land cover. The model was run for 97 years (spanning from October 1920 to September 2018), with the stationary land cover datasets included for each of the full-time periods. Of the five active flow gauges in the study area, two (S2H007 and S2H008) are located at the outlet of dams, and they are both highly impacted as a result (Table 5.4, Figure 5.4). The least impacted gauges located on dam outlets (S1H004 and S2h005) were used, although the data was uncertain. The model was, therefore setup using three gauges (S1H004, S2H005 and S2H006).

Despite the impacted observed data, the gauged data from S10H, S20B and S20C were used to calibrate the model. The model was set up as far as possible to represent realistic catchment physical characteristics (e.g. catchment size, slope, soils and land use) and processes that are expected to occur in the various environments. Due to the lack of observed data, streamflow calibration was largely based on the sensible parameterisation of the model. The focus was on setting the model up to represent the processes we would expect to see in these environments. The model was set up with S20B split into two sub-catchments to use the one flow gauge located on a tributary of S20B (S2H006).

Since afforestation forms a small proportion of the catchment, it was integrated into the modified land class (Table 5.2). Wetlands and lakes in the Pitman Model are assessed using a sub-model (Hughes et al., 2013). The Pitman wetland sub-model can only represent valley-bottom (on-channel) wetlands, but the catchment includes on-channel, seepage and depression wetland types. Therefore, wetland types were incorporated into natural land cover types as riparian areas. Wetlands were a minor component of land cover, and therefore not representing them explicitly in the model was not expected to have a significant impact on the hydrology. All dams were integrated and parameterised into the main model.

5.2.4.1. Pitman Model application

Table 5.1 outlines the model parameters for both the natural and land disturbance scenarios. The model can only differentiate two land cover types (besides riparian areas, irrigated areas, wetlands and dams). Natural land cover was represented as the primary vegetation type, with riparian areas (which included wetlands) also included in the model setup. Disturbed land included afforested areas, croplands, and artificial surfaces (Table 5.1) represented as a single land type. Hydrological impacts of modified land were estimated using the second vegetation type, which is typically applied for forestry but was applied for all modified land as discussed earlier (Section 5.2.2). In addition, afforestation in riparian areas and dams were included in the setup. Only relevant land use parameters were modified between scenarios to reflect the changing land use. This was to avoid 'forcing' the model to reflect anticipated outcomes.

The disturbed land parameters are given in Table 5.5 and include the ratio between natural to disturbed land, dam storage and the riparian area representing the catchment area where groundwater can be lost to evapotranspiration. The potential evapotranspiration ratio parameter for disturbed land is typically set at 1.4 (factor) to represent increased evapotranspiration potential of the modified vegetation type. The impact of disturbed land on groundwater regimes due to high evapotranspiration rates and changes in the runoff and infiltration ratio resulted in less recharge (without modifying the recharge parameters) (Table 5.5). Direct evapotranspiration from groundwater, which the Riparian Strip Factor - showing the proportion of the catchment in which groundwater losses occur was set at 0.3-0.4 for the natural land cover but adjusted up to 2 for disturbed land (Table 5.5) to account for riparian invasions. Infiltration parameters ZMIN and ZMAX were adjusted to reflect reducing infiltration with the changing land cover. All dam surfaces were lumped in the model (Table 5.5).

Table 5.5: Parameters used to represent the hydrological impact of land disturbance using the Pitman Model for the White Kei catchment. The following fixed parameters were used: POW = 2.5, GPOW = 3.5; SL = 0, SLG = 0. The bold parameters are those that were changed during the simulation.

Parameters	**Optimised ranges**		
	Natural	**1990**	**2018**
Interception rate (Natural vegetation)	1.5	1.5	1.5
Interception rate (Modified vegetation)	4	4	4
Proportion of afforested area (AFOR)	0	10.8-33.2	8.1-32.6
Veg2/Veg1 PET ratio (FF)	1.4	1.4	1.4
Max. absorption rate (ZMAX)	500 - 520	280 - 500	240 - 400
Max. storage capacity (ST)	70 - 180	70 - 180	70 - 180
Runoff rate at ST (FT)	3-10	3-10	3-10
Max. recharge rate GW	0.25	0.25	0.25
Groundwater abstraction (GWA)	0	0	0
Maximum dam storage (MI)	0	7 - 25685	706 - 28114
% Catchment area above dams (DAREA)	10-100	10-100	10-100
Area-volume relationship (A)	0.8	0.7 - 0.95	0.7 – 0.95
Area-volume relationship (B)	0	0.7 - 0.95	0.7-0.95
Riparian strip factor (RSF)	0.3-0.4	1-2.5	1.5-10
Annual abstraction (ABS)	0	0	0

5.2.5. Model evaluation

The realistic representation of the catchment hydrology was achieved by focusing on calibrating the parameters to represent the conceptual model (Beven, 2006; Liu & Gupta, 2007). However, the sparse data availability and impacted gauges in the catchment made it difficult to validate the majority of the hydrological modelling outputs, thereby increasing predictive uncertainty (Liu & Gupta, 2007). Therefore, only the change difference statistic was used to check the model's ability to simulate the hydrological impacts of the land cover changes. The change difference statistic is used to compare the correlation of two time-series. The percentage error of the total volume (%Diff) objective function was used to quantify the percentage deviation in the total volume and maximum discharge of the simulated time-series from the observed. A difference in the time-series results in deviation from a perfect correspondence (denoted as zero), signifying a poor simulation. Poor simulations could be due to model setup errors. Mathematically, the %Diff function can be expressed using the expression below, where Obs and Sim refer to the volume of observed and simulated time-series. The log difference statistic indicates represents the percentage change in annual discharge.

$$\%Flow = 100 * \frac{(Obs - Sim)}{Obs}$$

5.3. Results

5.3.1. Model performance

Figure 5.6 shows the observed mean monthly streamflow data plotted with the streamflow simulations under the three scenarios for S10H, S20B tributary and S20C. The comparison results are shown for the years 1970 to 2019 as the observed streamflow data is only available from 1970 (Table 4.4, Figure 5.6). Although the model statistics were poor, the simulations for the 1990 and 2018 scenarios were more comparable to the observed flows (Figure 5.6, Table 5.6). All simulations show multiple periods of no flow, highlighting prolonged drought periods and the presence of seasonal streams in the catchment (Figure 5.6). The mismatch in observed vs simulated low flow patterns for S10H and S20C can be attributed to impacted observed flows resulting from dam releases and unknown water use patterns. The simulation of the small tributary of S20B also compared poorly to the observed flow which could be due to a number of uncertainties including unknown water use, or gauging problems.

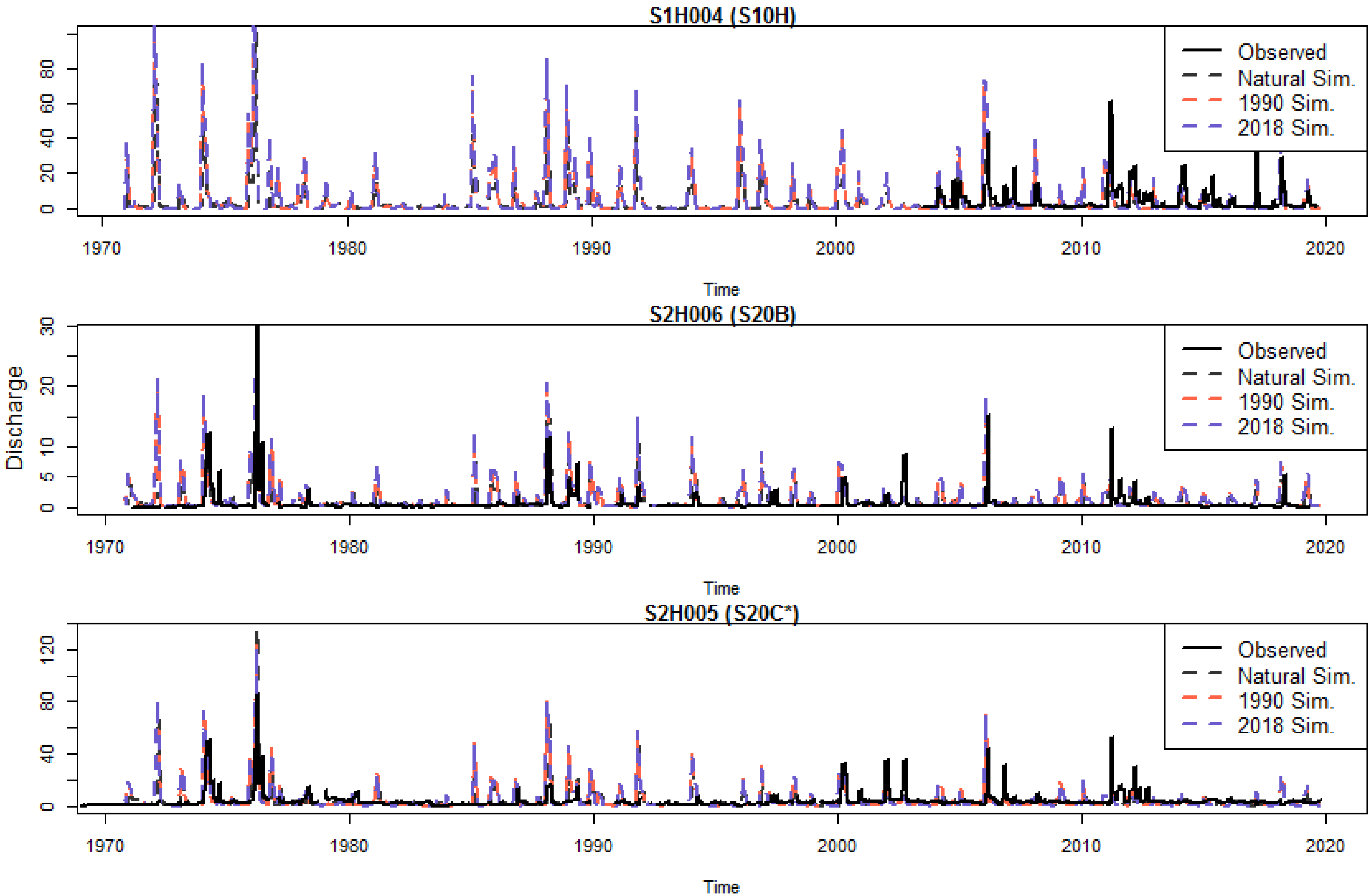


Figure 5.6: Comparison of observed and simulated flows under the 1990 and 2018 scenarios based on three gauges (S1H004, S2H005 and S2H006). The hydrographs graphs show the 1970 to 2018 period for better visualisation.

Table 5.6: Model evaluation statistics to test the acceptability of model structure based on three-stream gauge records, namely: S1H004 (S10H), S2H005 (S20C), and S2H006 (S20B).

	Natural		1990		2018	
Gauge	**% Diff (Nat)**	**% Diff (Log)**	**% Diff (Nat)**	**% Diff (Log)**	**% Diff (Nat)**	**% Diff (Log)**
S1H004	-46.29	-381.73	-14.46	-21.87	-1.49	-14.81
S2H006	67.84	-58.40	108.00	-57.73	125.36	-58.62
S2H005	6.60	-54.52	22.78	-74.98	15.45	-126.21

5.3.2. Simulated streamflow

Simulated streamflow regimes for the Natural, 1990 and 2018 Scenarios covering the years 1920 to 2019 are shown using flow duration curves by quaternary catchment in Figure 5.7. The general impact of land disturbance in the White Kei catchment was higher quickflow (surface runoff and fast released subsurface flow) and a reduction in the magnitude of dry season low flows compared to the Natural Scenario (Figure 5.7). Periods of no-flow covering at least 5% of the time for the natural land cover in the White Kei catchment (S10A-D, S10F-G, and S20A) indicate seasonal streams in the catchment, but land disturbance expands the duration of these by a range of 10.94 to 21.21% of the time (Figure 5.7). The areas dominated by intermittent streams are typically symbolised by a discharge threshold below 0.05 cumecs, whereas those dominated by perennial streams (S10E, S10H, S10J, and S20B-D) have a low flow discharge threshold ranging between 0.5 to 1.04 cumecs (Figure 5.7). Land disturbance in most parts of the White Kei catchment with a dominance of intermittent streams further reduces the catchments' ability to delay rainwater for release in the dry season, shown by the steep falling gradient (Figure 5.7).

5.3.2.1. Hydrological effect of land disturbance to individual water balance components

The percentage change difference (shown as colours) in the long-term mean annual volumes of water balance variables (i.e. evapotranspiration, surface runoff, interflow and baseflow) for the 1990 and 2018 disturbance scenarios compared to the natural land cover scenario are presented in Table 5.7. The corresponding long-term mean volumes for the water balance components are available in the appendices (Table A3).

Evapotranspiration rates in the catchment between 1990 and 2018 were variable, consistent with the modification in the catchment (Table 5.2, Table 5.7). The model simulated over 40% increment of surface runoff volume under both land disturbance scenarios than the Natural Scenario (Table 5.7). An exception for increased surface runoff due to land disturbance was S20D, where 8.56% less disturbance under the 1990 land disturbance led to lower surface runoff than the Natural Scenario (Table 5.7).

The model simulated a catchment-wide reduction in recharge due to land disturbance, except for S20 under the 1990 scenario where the model estimated an increase in recharge (Table 5.7). The long-term mean annual baseflow volume for the Natural Scenario in the catchment ranged between -0.34 to -0.02 mm, suggesting a prevalence of streams in the catchment that lose water to nearby aquifers (transmission losses) (Table A3). Reduction in recharge is also reflected by reduction in baseflow volume, except in five areas (S10A, S10F, S10J, S20A and S20D) where natural land recovered, or there was deforestation (Table 5.2, Table 5.7).

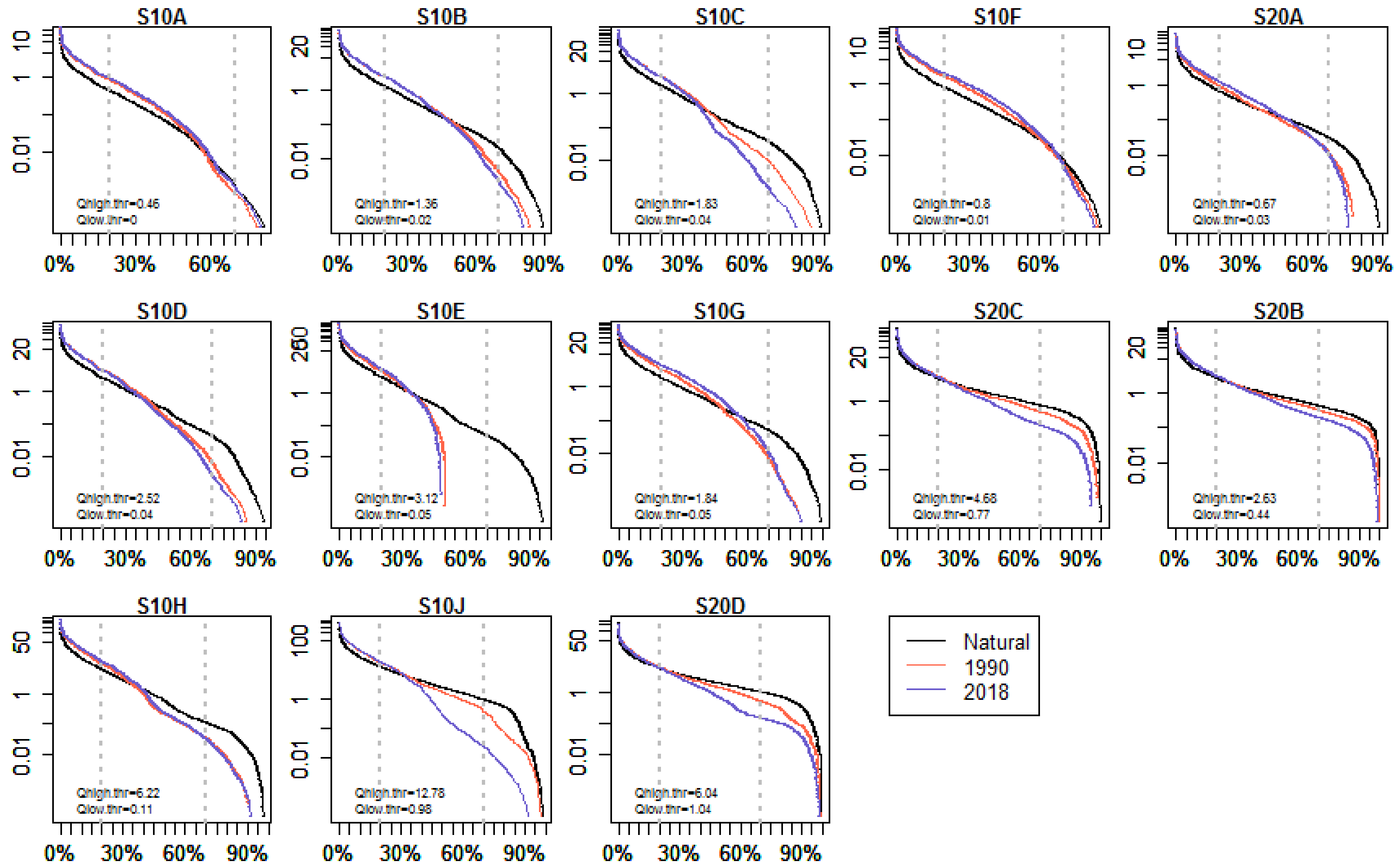


Figure 5.7: Estimated long-term simulated streamflow expressed as monthly distribution curve under the three scenarios of land modification in each quaternary catchment of the White Kei catchment. The plots have been arranged based on landscape from high to low reach. The vertical axis show discharge (in cumecs). The horizontal axis show % exceedence time.

Table 5.7: Long-term annual average rainfall percentage for the main water balance components in each quaternary catchment. The ink depicts a considerable percentage change difference in volume between the land disturbance and Natural Scenarios, **red** = < 5% reduction, and **green** = > 5% increment. The table has been arranged to match the different catchment localities (i.e. upper, middle and lower reaches).

Catchment	Scenario	Evapotranspiration	Surface runoff	Interflow	Baseflow	Recharge
S10A	Natural	27.42	1.59	0.25	-0.28	0.24
	1990	40.87	2.98	0.20	-0.28	0.19
	2018	25.96	3.20	0.21	-0.29	0.21
S10B	Natural	29.02	2.14	0.29	-0.21	0.43
	1990	43.66	3.92	0.22	-0.28	0.33
	2018	27.88	3.87	0.22	-0.30	0.33
S10C	Natural	28.01	1.77	0.26	-0.07	0.36
	1990	42.96	2.90	0.18	-0.12	0.25
	2018	27.69	2.99	0.19	-0.22	0.26
S10F	Natural	29.40	2.24	0.30	-0.27	0.45
	1990	45.96	4.33	0.24	-0.28	0.36
	2018	27.77	5.00	0.19	-0.31	0.29
S20A	Natural	30.89	1.72	0.49	-0.33	0.52
	1990	48.58	2.73	0.38	-0.28	0.40
	2018	30.12	3.29	0.35	-0.33	0.38
S10D	Natural	29.79	2.25	0.30	-0.10	0.52
	1990	44.54	4.02	0.22	-0.23	0.38
	2018	28.78	3.96	0.21	-0.26	0.38
S10E	Natural	29.39	2.18	0.31	-0.03	0.36
	1990	43.83	5.29	0.19	-0.11	0.23
	2018	25.00	8.46	0.16	-0.24	0.18
S10G	Natural	30.21	2.57	0.33	-0.17	0.47
	1990	42.85	4.50	0.24	-0.29	0.34
	2018	28.16	5.76	0.21	-0.34	0.30
S20B	Natural	32.54	1.81	1.25	-0.08	0.63
	1990	47.67	2.65	0.97	-0.13	0.50
	2018	32.12	2.61	1.11	-0.10	0.57
S20C	Natural	32.82	1.91	1.28	-0.02	0.67
	1990	48.35	2.68	0.96	-0.09	0.51
	2018	32.77	2.87	0.85	-0.30	0.45
S10H	Natural	32.18	2.11	0.60	-0.09	0.25
	1990	42.85	3.93	0.43	-0.26	0.18
	2018	29.95	4.10	0.39	-0.26	0.17
S10J	Natural	31.82	2.09	0.62	-0.17	0.41
	1990	42.45	4.00	0.46	0.27	0.31
	2018	29.29	4.25	0.43	-0.17	0.29
S20D	Natural	56.22	4.07	2.45	-0.60	1.35
	1990	91.43	3.18	5.91	-0.16	1.74
	2018	56.40	6.37	1.67	-0.60	0.92

5.4. Discussion

Previous reports in two catchments from the Eastern Cape (Shackleton & Gambiza, 2008; Rowntree, 2013; Van Tol et al., 2016; van der Waal & Rowntree, 2018) affirm the finding of intensified surface runoff throughout the White Kei catchment due to woody expansion (afforestation and woody encroachment), long-term exposure of bare soil in cropland areas, increasing barren lands [which fall under natural land cover (GTI, 2019b)], and the ongoing erosion in rangelands and cropland areas. This study concurs with previous studies and demonstrates that land disturbance in this environment reduces the catchment's capacity to delay rainfall from quickly reaching streams during the wet season. The study catchment's steep slopes drain quicker due to the gravity factor and can be prone to quickflow dominance and lead to soil erosion. Hughes et al. (2018a) affirmed that native vegetation stabilises the soils and increases soil water retention, which could be applicable in environments such as the White Kei. However, in a catchment with widespread IAPs and bare soil exposure (uMngeni catchment, KwaZulu-Natal), this stabilisation does not occur. The simulated surface runoff dynamics due to land disturbance support earlier findings in the Olifants catchment (Gyamfi et al., 2016), where a significant reduction in rangelands and increase in croplands led to nearly 50% more surface runoff. Findings by Gyamfi et al. (2016) are consistent with other literature in South Africa (Rebelo et al., 2015; Mander et al., 2017; Hughes et al., 2018b) and elsewhere in the world (Kabanda & Palamuleni, 2013; Rebelo et al., 2015; Gyamfi et al., 2016; Mander et al., 2017; Hughes et al., 2018b). Despite the differences in methodology and study contexts, the studies collectively agree that land cover alterations such as those in the White Kei combined with climate change impacts intensify surface runoff and result in less resilient catchments regarding drought.

As discussed in Chapter 2, land cover change is the most dynamic catchment structural component (Jencso & McGlynn, 2011; van Lanen et al., 2013), and the Pitman Model findings support this assumption in the White Kei. The land disturbance has resulted in reduced soil water storage, which affected interflow contributions to the rivers, particularly in steeper areas. Hence, streamflow reduction in the catchment could be observed as early as the intermediate or moderate flow period. Since vegetated steep terrains dominate the White Kei catchment, interflow is assumed to be a dominant process because the vegetated slopes encourage infiltration and the steep slopes result in a lateral component to unsaturated zone water movement (Hughes et al., 2018b). The modified land scenarios suggest that the presence of croplands and woody vegetation expansion in the Cacadu catchment reduced infiltration rates and unsaturated zone water movement (both laterally as interflow and vertically as groundwater recharge.

Both afforested areas (Ogden et al., 2013; Yapi et al., 2018) and active cropland areas (Scanlon et al., 2008; Kim & Jackson, 2012; Ibrahim et al., 2014) facilitate better infiltration than other modified land classes due to root zone activity and improved soil porosity due to tillage activities(Scanlon et al., 2008; Kim & Jackson, 2012; Ibrahim et al., 2014). The model simulated reduced infiltration, but this was due to both the presence of abandoned croplands within the modified land and increased evapotranspiration. As expected, the model estimated that water usage by woody encroachment and IAPs decreases available water for subsurface flow and groundwater recharge through water loss by interception, and transpiration. This in turn impacted low flow discharge to regulate dry season runoff (Huxman et al., 2005; Le Maître et al., 2015, 2020).

Uncertainty persists for the influence of cropland areas on catchment hydrology. For instance, a field-based comparison of different tillage processes with croplands under non-tillage in Bergville, KwaZulu-Natal (catchment V13D) showed about five times lesser infiltration rates in the upper soil horizon of non-tillage croplands (possibly similar to abandoned croplands) compared to active croplands (Sithole et al., 2019). Active croplands have also been identified as influential ecosystems for groundwater recharge in the semi-arid Sahel region Ibrahim et al. (2014). The findings in the Sahel (Ibrahim et al., 2014) support outcomes from field observations based in the United States by Scanlon et al. (2008) and global literature synthesis by Kim & Jackson, (2012), which contend that cropland areas contributed substantial amounts of groundwater recharge compared to other land cover types, including grasslands, due to tillage activities that increase surface friction and soil infiltrability. However, Scanlon et al. (2008), Kim & Jackson (2012) and Ibrahim et al. (2014) did not consider the role of bare ground exposure in croplands, which facilitates quickflow and consequently sediment removal (Van Tol et al., 2016) that could trigger a positive feedback mechanism that reduces infiltration in the long-term. The studies also neglected the vulnerability of cropland areas to IAPs and woody encroachment (Scorer et al., 2019). As discussed in a review on the impact of IAPs by Le Maître et al. (2015), woody plants affect the water balance differently based on plant physiology and traits. Woody encroachment in semi-arid grassland areas increases quickflow (Kakembo, 2009; Manjoro et al., 2012) and exploit soil water (Huxman et al., 2005; Scott et al., 2006). However, questions remain about the exact hydrological impact of encroaching woody species in the rural semi-arid grasslands and abandoned croplands.

The simulated streamflow under natural conditions showed sustained dry season streamflow, which reduces drought risk exposure, unlike land modification that leads to increases in the duration of zero flow periods, which is expected to increase drought risk to local communities. Streamflow simulation results, together with the limited observed flow data, show that zero flows are natural in the White Kei catchment, but the dams and reduced infiltration capacity in the catchment extend these dry periods, as can be seen in S10E with approximately 5% of waterbodies. Increasing dam storage and changing land use in the 1990 and 2018 Scenarios substantially impacts the streamflow regime, extending zero flows to 45% of the time. An impact of small dams in a catchment dominated by seasonal streams (S20A) is shown by extended periods of zero flows from 7% (Natural Scenario) to 21% (2018 Scenario). . The impact of dam infrastructure has also been noted in China, where it led to anthropogenic impacted seasonal rivers because of increased no-flow events compared to natural conditions (Xu, 2004). Similarly, a high density of farm dams in South African catchments has a net negative impact on streamflow magnitudes within the H10 catchments in South Africa (Hughes & Mantel, 2010).

5.4.1. Limitations and conclusions

This study could identify four major limitations. Firstly, the Cacadu catchment is largely ungauged, making it difficult to accurately conduct model-based hydrological assessments (Sivapalan et al., 2003; Makungo et al., 2010; Hughes, 2013). To overcome this shortcoming, this study focused on the appropriate parameterisation of the model which aimed to represent the catchment processes according to the conceptual model and land use information. Incomplete flow data meant the simulated flows could not be compared with observed flows, which introduces uncertainty regarding the reliability of the outcomes (Liu & Gupta, 2007; Hughes et al., 2010). Therefore, this study could only make inferences about the nature of land use impact on runoff in the Cacadu catchment.

The second source of uncertainty was estimating channel losses. In field-based studies, channel losses are often estimated using water table levels or stable isotopes (Villeneuve et al., 2015; Costigan et al., 2016). Unless such data is available, hydrological models will remain uncertain regarding channel losses, and in the White Kei catchment, such data is not available. The challenge in estimating channel losses in hydrological modelling is common (Tanner & Hughes, 2015). The uncertain volume of channel losses in this study was estimated based on expert knowledge about the physical setting and the occurrence of zero-flow periods in the catchment. Therefore, the volume of channel losses in the catchment remains uncertain.

The third limitation relates to the lack of available data on water use from both surface water and groundwater, which may have led to underestimations in the results. Lack of real-time data meant the model could not be setup to represent all hydrological processes, especially the water-use components, as shown in Table 5.5 by the groundwater abstraction rates and the annual abstraction rates that were set at zero in the model setup. In reality, pumped boreholes do exist in the catchment, and therefore setting the ABS parameter at zero restrained the model from simulating abstractions, which could not have been avoided. Despite the misrepresentation of some water balance components due to real-time data limitations, the simulated results allowed for a sensible understanding of the effects of land modification on surface and groundwater in the catchment.

The fourth limitation relates to the assumption about the catchment structure that informed the lumping of land covers. To estimate the impact of land alteration in catchment hydrology, the natural scenario was used as a reference condition while the 1990 and 2018 land cover changes were used as impact Scenarios. Since individual land cover changes did not have a significant extent in the catchment, they were lumped – which also meets the requirements of the Pitman GWv3 model used here. Therefore, given the water balance impacts of the individual changes (Figure 5.5) and the detail required by the hydrological framework followed (Le Maître et al., 2014), the results of this study have a limited accuracy. Despite the shortage of information on the model behaviour due to different land covers, the results are still important as they provide estimated catchment behaviour in degraded vs intact catchments – which was the main goal of the assessment.

In closing, this study used an ungauged rural catchment in the grassland biome to describe the impact of rural land use change on catchment hydrology. Over the past 30 years, significant land cover changes in the White Kei sub-basins are characterised by a significant decline in natural grasslands, increasing modified areas (consisting of active croplands, abandoned croplands, artificial surfaces, eroded surfaces and waterbodies), and a variable afforestation trend. Of these changes, croplands and tree-cover expansion accounted for the most significant changes. The Pitman Model results suggest that land modification in the catchment leads to a cumulative impact in the water balance components. First, land modification leads to evapotranspiration losses and increased surface runoff. These changes in surface water dynamics limit the amount of surface water available to replenish soil and groundwater storage, resulting in lowered groundwater tables and, consequently, a reduced low flow regime. Dam infrastructure leads to impacted patterns of flow in rivers. To improve water security in the White Kei catchment and other similar contexts, there is a need to prioritise ecosystem protection and restoration. If neglected, the coupled impacts of climate extremes and unregulated land use will further threaten the communities in the catchment that depend on surface water sources for water supply and possibly downstream users, especially with the reality of drought risks.

6. Chapter 6: CONCLUSION AND WAY FORWARD

6.1. Introduction

Developing an understanding of the role and benefit of ecological infrastructure for the flow regulation function of the catchment is important for building the resilience of rural communities against climate extremes like droughts. The impacts of climate extremes and ecosystem degradation are most evident in poor and vulnerable communities, mainly in rural areas (van der Waal & Rowntree, 2018). The SANBI framework provides a guideline for promoting financial and physical actions for ecosystems that provide socio-economic benefits to people (SANBI, 2014). The SANBI Framework fits with the United Nation's Nature-based solutions approach for water that recommends ecosystem-based approaches to improve water security and mitigate natural disaster risk exposure Actions promoted by the SANBI framework emphasise the importance of natural capital for poverty alleviation and protection from environmental shocks (Cumming et al., 2017; Mander et al., 2017). Recently, the role of ecological infrastructure for economic and social-ecological wellbeing is receiving much attention from different social sectors. While researchers, laypeople and practitioners are involved in the EI discussion, unlocking potential investments remains a challenge, and where investments have existed, the rate of return has been low (van Wilgen et al., 2012b; Cumming et al., 2017; Mander et al., 2017; Adamowicz et al., 2019).

This research used the SANBI framework for EI investments to provide a decision-support system for actions that facilitate flow regulation and, consequently, drought mitigation (SANBI, 2014). In semi-arid landscapes, increasing climate extremes coupled with anthropogenic pressures acting as environmental drivers are expected to facilitate water and land resources' vulnerability further, thereby threatening the ecosystem services they provide (UN Water, 2015). The SANBI framework's narrative is in many ways similar to the ecosystem-based adaptation framework in that it promotes soft adaptation for disaster risk reduction or mitigation to build resilience against threats to ecosystems (CBD, 2009; Cohen-Shacham et al., 2016; Scarano, 2017). However, the SANBI framework distinguishes itself in that, although it adopts most of the ecosystem-based adaptation principles, it moves beyond climate change and invokes the cost-benefit thinking, which is necessary to motivate for changes in policies and secure private sector investments (CBD, 2009: 41; SANBI, 2014). This thinking line is essential for the South African landscape, considering the climate and ecosystem degradation threats to the water-stressed country (DWAF, 2013).

This study sought to better understand the role and benefit of optimising ecological infrastructure to enhance community water security during drought periods and dry seasons. To achieve this aim, the study assessed the degradation status of a rural catchment in the

South African grassland biome. The study additionally provided an example of how anthropogenic modifications affect the runoff regime at a quaternary catchment scale using the study catchment. Finally, EI related to flow regulation in the catchment was prioritised for restoration to improve drought mitigation. This section summarises the three research chapters (Chapters 3-5), and highlights several insights regarding the key research questions outlined in Chapter 1.

6.2. Summary of key findings

6.2.1. What is the current land condition of the Cacadu catchment?

Chapter 3 used a globally developed GIS plugin that computes the SDG 15.3.1 indicator by combining the SDG 15.3.1 sub-indicators using the OOAO statistical rule. Findings suggest that land cover and soil organic carbon were stable over 15 years, and land productivity decline is a potential concern. For the pixels that demonstrate degradation, land productivity loss was the main driver of the degradation process suggesting that the recent degradation process in the Cacadu catchment is a slow process. The most degraded pixels were detected in areas that lost natural vegetation cover (S10A, S10F, S10G, S10H and S10J). The slight recovery in land condition was found in areas revegetated with grasslands, which is consistent with the grassland productivity dependence on rainfall. Comparing the SDG 15.3.1 indicator results and the 1990 to 2018 SANLC Change product revealed a mismatch between degraded and stable pixels, highlighting degradation/improvement before 2000 and post-2015, and the need for higher resolution datasets in the baseline year (2000) to improve the accuracy of the assessment.

The various degradation drivers typically lead to poor flow regulation that could expose rural communities to resource insecurity (Le Maître et al., 1999, 2014). The impact of land degradation drivers such as loss of productive land (Bai et al., 2008; Fensholt et al., 2013), woody proliferation (Shackleton & Gambiza, 2008; Luvuno et al., 2018), soil erosion (van der Waal & Rowntree, 2018), and land conversion to plantations and croplands (Foley et al., 2005; Skowno et al., 2019), often leads to removal of native vegetation and alteration of soil structure, and that leads to the destruction of groundwater recharge properties, and consequently poor freshwater supply by rivers and wetlands (Le Maître et al., 1999, 2014; Brauman et al., 2007). The consequence of degradation and poor flow regulation in rural catchments may further threaten the wellbeing of communities that inhabit these areas, and the spill-over effects may be experienced by downstream communities (Rebelo et al., 2015; Nel et al., 2017; Hughes et al., 2018b).

The availability of appropriate input datasets, knowledge of ecosystem dynamics in case sites, and accurate interpretation of findings remain the main prerequisites for using the plugin, as with any remote sensing tools. Chapter 3 adequately meets the degradation assessment objective by using publicly available data and a simplified protocol to track land degradation. The study supports using the global land tracking tool (TRENDS.EARTH plugin) to monitor and report land degradation. The support is founded on the plugin's ability to be adopted in different contexts, including grassland and rural areas, and be applied consistently to provide a decision-support system towards achieving the land degradation neutrality target by 2030 (Hoffman & Todd, 2000; Orr et al., 2017; Conservation International, 2018a). In light of the need to reduce the degradation threat, the absence of a coherent degradation monitoring method and the broad definition of land degradation has led to studies to focus on a singular indicator for land degradation, which has been criticised for underestimations and overlooking other sources of degradation (Hoffman & Todd, 2000; Gibbs & Salmon, 2015). Although TRENDS.EARTH plugin is not an error free option (Wessels et al., 2012; Prince, 2019); it has some utility in detecting degradation and can offer insight into separating human-induced from climate-driven degradation (Wessels et al., 2007; Prince, 2019). Secondly, by defining land cover degradation for the grassland biome, the study targeted the grassland-specific degradation process, unlike earlier assessments that simplify the degradation assessments to administrative boundaries (Gibbs & Salmon, 2015; von Maltitz et al., 2019). Lastly, the Trends.Earth plugin can only quantify degradation, but more work is needed to include stakeholder voices and other knowledge systems - social elements that are required for restoration planning as outlined in the International Restoration Principles (Gann et al., 2019).

6.2.2. Which focal ecological infrastructure areas can be targeted for restoration to help improve the drought mitigation capacity of catchments, and how do these areas compare to core areas for rural household livelihoods?

Building resilience against challenges such as water security for rural landscapes and communities requires the adoption of holistic natural resource management strategies that include resource protection, prevention of further degradation and reversal of past degradation as provided by the land degradation neutrality framework (Orr et al., 2017) and presented in the SANBI framework (SANBI, 2014). Chapter 5 uses a GIS-based AHP method to detect priority EI areas that could be restored to improve the focal catchment's drought mitigation function. The AHP process was guided by community stakeholder values, 12 spatial datasets, and expert opinions. The focal EI categories were wetlands, riparian margins, abandoned croplands, and grasslands chosen for their hydrological importance when sustainably managed. Multi-stakeholder involvement in the prioritisation step was essential to increase social acceptability of any planned restoration activities since restoration is partly to improve socio-economic benefits (Reed et al., 2008; Sapkota et al., 2018), and is underlined as a key driving pillar for restoration (Gann et al., 2019).

The prioritisation results indicate that most of the EI categories had a high priority level for restoration to augment flow regulation, except for S10A, S10E and S10H, which seemingly divides the catchment into important and least important EI zones for flow regulation. Chapter 5 shows that the flow regulation function of the Cacadu catchment could be improved by restoring 87.22% of abandoned croplands, 49.49% of grasslands, 20.41% of wetlands, and 5.85% of riparian margins (Table 5.8). The water flow regulation of croplands is demonstrated in a global review by Kim & Jackson (2012) and a study on groundwater recharge in a semi-arid area in the United States (Scanlon et al., 2008). The two studies highlight that tillage activities increase the infiltration capacity of cropland areas and encourage recharge due to improved soil porosity. In a rainfed agricultural region in west Africa, Ibrahim et al. (2014) discovered lower groundwater than active croplands, which may apply to abandoned croplands. Therefore, restoration of abandoned croplands and rangelands (grasslands) through revegetation and removal of woody encroachers is expected to repair the flow regulation capacity of rural catchments in the grassland biome. Since the flow regulation capacity of wetlands depends on wetland type and wetland size, the prioritised wetland areas could play a role in delaying and storing rainwater for release in low flow periods, as discussed in a synthesis by Bullock et al. (2003) and modelled by Rebelo et al. (2015) for Kromme wetlands. The connected network formed by the highly prioritised riparian margins, grasslands and abandoned cropland areas under both prioritisations substantiate the importance of networks for building resiliency (Biggs et al., 2012; van der Waal & Rowntree, 2018), which has also been reported in a study by Egoh et al. (2011) on prioritisation for ecosystem service management at a national scale for the grassland biome, and in the Tsitsa catchment for sediment control (van der Waal & Rowntree, 2018).

This study supports the adoption of SLM strategies, multiple data sources, multiple stakeholder groups and remote sensing for restoration planning. Integrating multiple data sources and various stakeholder groups has practical implications for restoration because the success of restoration depends on reliable evidence-base, local buy-ins and sound governance structure. In the end, the prioritisation study recommends the adoption of affordable tools such as the GIS-based AHP model for setting clear goals and identifying restoration hotspots since field-based options may be too costly and are time-consuming. The strength of the prioritisation approach adopted in this study is integrating multiple actors openly and transparently, selecting a measurable restoration target, and identifying potential win-win benefits from future restoration activities. The highlighted strengths of the prioritisation technique in Chapter 5 are important. They have been noted as a shortcoming for existing EI investments, as exemplified by the Working for Water project (van Wilgen & Wannenburgh, 2016; Kraaij et al., 2017). However, similar to the degradation assessment, the study recognises the shortcomings of poor-quality spatial datasets. Moreover, stakeholder participation and willingness to form a collaborative partnership among the different governing bodies (i.e. homeowners, traditional authorities, local and provincial government) in rural catchments remains a challenge for successful natural resource management.

6.1.1. How natural land cover facilitates water flow regulation?

Chapter 4 describes how land modification (land cover change) might affect the flow regulation function of the rural catchments in the grassland biome using the White Kei catchment as a case study. Hydrological processes in intact EI areas are characterised by a sustained capacity to regulate streamflow in terms of the flow regulation regime (i.e. duration, timing and magnitude) (Brauman et al., 2007; Le Maître et al., 2014). A monthly time-step semi-distributed conceptual model (the Pitman groundwater model) was used to fulfil this objective at a quaternary catchment resolution (Section 4.2). The model integrates hydro-climatic data and uses parameters that were calibrated based on expert knowledge of the land cover and flow regulation system of the focal catchment. Runoff data scarcity compelled adopting the regionalisation approach to achieve the runoff assessment. The model was setup to reflect natural (grasslands, off-channel wetlands, indigenous woody cover, and barren lands) and modified land (artificial surfaces, croplands, afforested areas, eroded surfaces, and waterbodies) (Section 4.24). A Pitman sub-model was used to simulate channel wetlands. The White Kei catchment is dominated by natural surfaces, but land modification does exist.

The reliability of Chapter 4 results is limited by poor quality hydrological data that made it impossible to validate the simulations. Improved data would substantially reduce the input data uncertainties. However, the findings of this study are still acceptable as the approach used in this study is based on known techniques for hydrological modelling in ungauged regions (Beven, 2006; Seibert & Beven, 2009; Kapangaziwiri et al., 2012; Swain & Patra, 2017).

Although the model outputs are uncertain due to limited discharge data, the estimates by the Pitman Model helped understand the hydrological behaviour of the catchment following land modification in 1990 and 2018. In most quaternary catchments, the model simulated that land modification in the White Kei is expected to increase quickflow while negatively impacting recharge and dry season streamflow, except for areas where natural land cover has recovered. The simulated hydrological behaviour of the White Kei supports findings in uMngeni catchment (grassland and savannah region) by Hughes et al. (2018a), where the loss of natural vegetation in the hillslope areas reduced the capacity of the catchment to capture and store rainwater for a dry season release. Gyamfi et al. (2016) reported similar findings in the Olifants catchment, where surface runoff increased due to land modification. The Pitman Model simulations in a rural grassland context (shown by a dry season streamflow improvement following natural land cover recovery in S10A, S10F, S10J, S20A and S20D) are affirmed by results of wetland modification in the Kromme catchment (fynbos region) by Rebelo et al. (2015), where over 80% of on-channel wetlands were lost due to anthropogenic activities, leading to streamflow losses of over 1 $M.m^3$. Chapter 4 also shows that the White Kei streams are spatially heterogeneous as the catchment naturally has seasonal streams in some areas. However, the land cover degradation could intensify the presence of seasonal streams in the catchment. Seasonal streams are more apparent in dam infrastructure areas (e.g. S10E and S20A), which emphasises earlier findings of dam impacts on streamflow runoff by Hughes & Mantel (2010). Building on the conceptual framing of ecological infrastructure impacts for water flow regulation by Le Maître et al. (2014), the results in Chapter 4 show that land cover degradation in rural grasslands cold have similar dry season streamflow impacts in other rural grassland contexts.

6.2. Key insights that relate to the mitigation of droughts

To mitigate the impacts of drought (i.e. soil and hydrological drought), catchment management is essential (Tallaksen & van Lanen, 2004). The importance of catchment management to regulate droughts (through water flow regulation) is primarily because drought lag in wet versus dry catchments differs based on catchment rainfall capture and storage characteristics (van Loon & Laaha, 2015). Consequently, catchment response time influences drought propagation periods, e.g. drought can expand beyond the meteorological drought season as observed in Cape Town, South Africa (Botai et al., 2019). However, actions directed at maintaining the physical catchment traits can help restore natural catchment behaviours, which in return may reduce the time lag for drought recovery (Stoelzle et al., 2014).

Reflecting on the research study, it is evident that besides climate impacts, the extent of long-term degradation (Chapter 3), and inadequate ecosystem protection [prevention and control of degradation (Chapter 5)], contribute to poor dry season water yields (Chapter 4) that will lead to reduced water security for communities. As discussed in Chapter 2 and shown in Chapter 4, weak water flow regulation in catchments is characterised as the inability of catchments to maintain dry season flows due to an interaction of climate and human-induced factors that reduce the catchment's capacity to capture and retain rainwater for later release into streams (Brauman et al., 2007; Le Maître et al., 2014). Moreover, the lack of rural access to reliable drought warning systems and the complex nature of droughts necessitate adopting integrated management strategies to alleviate drought impacts.

Restoring native vegetation, reducing woody proliferation, managing riparian areas and protecting wetland resources could reclaim the water flow regulation service of the Cacadu catchment and other similar contexts, and improve future water security. Chapter 3 indicated productivity loss as the primary degradation process in the catchment, but grassland recovery increases land productivity as detected at a provincial level (Eastern Cape Province) by Graw et al. (2017). Water availability is amongst the limiting factors for grassland species recovery (Ponce-Campos et al., 2013), indicating the need to better-manage catchments, which can help offset woody species and soil erosion while contributing to water flow regulation and local livelihoods. (SANBI, 2014; Bridgewater, 2018).

From a development perspective, a restoration commitment from rural areas could significantly contribute to meeting the national and international targets. For instance, a national level restoration target setting using the SLM approach identified the grassland biome and inland aquatic ecosystems amongst South Africa's priorities for restoration (DEA, 2018). In particular, the country committed to restoring 61 900 ha of wetlands, 2 436 170 grasslands, and 6 000 000 ha of soil organic carbon in croplands (DEA, 2018). The suitable restoration areas identified in Chapter 5 could be potential areas for leveraging land degradation neutrality through ecological infrastructure investment (SANBI, 2014; Dias et al., 2016; DEA, 2018; von Maltitz et al., 2019). Understanding that it is impossible to restore an entire landscape, a commitment to restore the highly prioritised areas in the Cacadu catchment for flow regulation (Table 5.8) would contribute the following to the national SLM target: 0.45% wetlands (and riparian management could add a further 1.90% for inland aquatic systems), 4.02% grasslands, and 0.09% soil organic carbon in croplands.

6.2.1. EI investment strategies to mitigate future droughts

The findings indicate that a novel adaptive resource management pathway is needed to protect both the catchment and the livelihoods. Of the three approaches suggested to meet zero-net land degradation (i.e. maintain, rehabilitate and restore), maintaining or protecting the landscape from new or further degradation could be the most effective and affordable option for ecosystem management in the catchment (Stavi & Lal, 2015). However, to offset the socio-economic costs of droughts, financial allocations for rehabilitation and restoration will also be required. While there is some municipal water supply, some rural communities depend on boreholes, rivers and wetlands for their water supply needs (Table 1.1). This dependence on catchment water provision forms the basis for the importance of EI investment commitments that area aimed at maintaining soil moisture and groundwater storage to augment water supply in low flow periods. In the case of food and economic productivity, active croplands and planted forests would also need to be sustainably managed to avoid degradation for these areas to contribute to the drought mitigation service. To meet the anticipated frequency and intensity of future droughts, and reduce other livelihood insecurities, Action 8 of the SANBI Framework advocates for environmental stewardship to build environmental sustainability and resilience against future shocks (SANBI, 2014).

The inclusion of core areas for rural livelihoods in the prioritisation of EI for restoration could help argue for the significance of integrated catchment management at a multi-village scale catchment scale, which can help move from knowledge-to-action. For instance, the overlap in the network of priority EI areas for restoration could help advise locals on how they can better manage their landscapes within the 2 km zones they currently prefer. The possibility of attaining healthier fodder and sustainable water ponds if locals, elected and traditional leadership chose to buy-into the protection of highly prioritised areas that fall on stable and improving areas is one example. However, the success of such a management decision would largely depend on a close collaboration between social players from the household to the provincial level as mandated by Principle 6 of the SANBI framework - EI investments should be cantered by inclusivity and social participation (SANBI, 2014). Insights on how such collaborative governance can be made possible will be clarified by the GEF5 SLM project, which is drawing to an end (The GEF, 2013). The connection of highly prioritised areas for rural livelihoods substantiates the importance of connectivity for social modularity, which is important for managing common-pool natural resources (Biggs et al., 2012; Nemec et al., 2014).

The governance of natural resources in some South African rural areas appears to be symbolised by fragmentation, leading to the absence of resource control (Sisitka & Ntshudu, 2017). In the early 2000s, Machubeni communal area was identified as a site for rural development planning under the Transform project (iKhwezi, 2003). The project aimed to improve land management and mitigate soil erosion in the quaternary catchment (iKhwezi, 2003). Some of the project milestones include establishing social partnerships and land management governance structures, but locals recently indicated their inability to manage the previously prioritised resources (Sisitka & Ntshudu, 2017). With the GEF5 SLM project's introduction, the ceased partnerships between the Transform projects established groups were renewed. New partnerships were merged to support the elected and traditional authorities to collaborate with the municipalities (Sisitka & Ntshudu, 2017). In the Tsitsa catchment, the Tsitsa EI project seeks to achieve zero-net degradation to support local livelihoods (Cockburn et al., 2018). In the Tsitsa catchment, the governance structure is symbolised by tensions between the elected and traditional authorities over land use decision-making leading to poor resource management (van der Waal et al., 2017). The formation of an inclusive governance structure in an urban catchment (the Palmeit catchment) in Kwa-Zulu

Natal could also provide valuable insight into the importance of context-specific governance arrangements for building resilience against social-ecological shocks (Sutherland & Roberts, 2014). Moreover, socio-political history has a role in the acceptability of the project in the catchment. In light of the two case studies described above, the Machubeni case offers a demonstration opportunity for other EI investment initiatives that could be implemented to meet the main restoration goal in the Cacadu catchment and other rural contexts.

6.3. Conclusions and Future research

For terrestrial ecosystems, remote sensing is one promising option, especially in areas where high-resolution data is available (Richardson et al., 2009; Botai et al., 2019). Using methods that depend on satellite-derived datasets, publicly available data, community stakeholder and expert views, this study was able to quantify the ecological health status of a rural catchment, describe the impact of potential land degradation drivers on the catchment hydrology, and prioritised four EI resources for restoration to improve water flow regulation. Fulfilling the three objectives helped understand the role and impotence of healthy EI for delivering the flow regulation ecosystem service in rural catchments. The results highlight that moderate human-induced degradation over the UNCCD baseline period (2000-2015). Secondly, the Pitman Model estimates demonstrate that land modification aggravates the catchment's inability to augment low season water yields. Lastly, the stakeholder-driven GIS-AHP prioritisation technique revealed two priority levels for restoration to improve water flow regulation in the catchment, and areas in most needs of interventions formed a network in the central part of the catchment. Considering the social benefit of restoration, the prioritisation technique also highlighted core areas of livelihoods that could offset food security concerns. In agreement with earlier studies on the topic of EI, this study emphasises the significance of EI for water security and buffer against future droughts for rural communities. In recognising this importance, this study recommends that EI investment actions in rural catchments should facilitate a transformative pathway that meets the frequency of anticipated droughts, as discussed in Chapter 2. Secondly, as promoted by the triple-bottom-line targeted by sustainability, the findings of this study highlight the need for promoting pragmatic solutions that are founded on respect for the social setting of target communities and equitable distribution of the already limited financial resources to meet the restoration targets. Lessons to meet the recommendations can be drawn from the Palmeit catchment (in uMngeni Greater catchment), the Tsitsa and GEF5: SLM projects.

To the best of our knowledge, the integration of remotely-derived produced community stakeholders and hydrological modelling that considers a wide range of degradation drivers and targets water security to inform decision-making for restoration planning is the first attempt, especially in a rural context. Although the outcomes provided in this study contribute a significant evidence-base for the need to invest in EI, there is still a need to ground-truth the results with real-life observations because of the quality of datasets used in this study. This step will help inform restoration planners, land managers, and communities identify restoration opportunities and select the appropriate restoration strategy for the priority areas. The approach and the findings of this study can be applied to other rural parts of the South African grassland system to guide resource protection and enhance ecosystem services such as drought mitigation. The drought mitigation service will probably take time to yield returns; however, through investing in EI in areas like the Cacadu catchment will build a buffer against future climatic risks to water security. Although droughts are expected to be a continuous threat, drought mitigation approaches such as restoring the catchment to facilitate groundwater runoff better can prevent the socio-economic impacts of hydrological droughts.

The results outlined in this study are a good starting point for restoration planning, although the study outcomes are affected by the quality of spatial datasets used that could be attributed to the lack of monitoring in rural contexts (van Deventer et al., 2018). However, the final recommendations need to be considered with caution since the complex governance system in traditional societies may influence the inclusive governance endeavour of the land degradation neutrality and EI Investment Frameworks (Parrott & Meyer, 2012; SANBI, 2014; Orr et al., 2017; Palmer & Munnik, 2017; Pantshwa & Buschke, 2019). By adopting a framework that embraces local stakeholder values, restoration planners can better understand the social-ecological conditions of sites that require restoration and help craft a sustainable restoration pathway (Tengö et al., 2014). More still needs to be done to uncover cross-spatial interactions that can further reduce the uncertainty surrounding the prioritisation of EI for protection.

For catchment management targeted at mitigating drought impacts, research and monitoring need to consider current catchment characteristics and forecast these for how they can affect water security in the future at catchment scale (Gibson et al., 2018). The emergent feedback of investing in restoring the prioritised areas would transform the catchment towards the other ecosystem benefits. In addition to the priority areas for water flow regulation, the prioritisation technique adopted in this study identified core areas for local livelihoods, which could benefit from restoring to improve the flow regulation capacity of catchments. To better accommodate the mosaic nature of South African landscapes, a comprehensive exercise is needed to provide degradation definitions for each of the remaining eight biomes. Secondly, to better assess the role of human-induced degradation, assessments need to go beyond the assumptions relating to main drivers of change, such as the NPP/rainfall relationships. Where possible, inquiries should interrogate other phonological processes that can explain the trends of degradation. Since degradation assessments rely on the baseline and observation periods, it is imperative to set adequate term for the target period to better detect the land degradation state as emphasised in the land degradation neutrality framework (Orr et al., 2017). Moreover, the spatial resolution limitation discussed earlier can be resolved if UNCCD partners with countries like South Africa to develop further a national land cover dataset that is consistent with the 1990 and 2018 NLC datasets. Finally, while remotely sensed data offer valuable and unprecedented methodological opportunities to compute land degradation, future degradation assessments still need to address the implication of the complex-adaptive nature of ecological systems when computing degradation and consider how ecological resiliency influence degradation conclusions especially considering seasonal changes and shifts beyond ecosystem thresholds.

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8. APPENDICES

8.1. Appendix 3.1: Background of the TRENDS.EARTH plugin

8.1.1. How the plugin computes sub-indicator 1: Land productivity

The plugin uses the NDVI products (user-defined) from either the 250 m Moderate Resolution Imaging Spectroradiometer or the 8 km Advanced Very High-Resolution Radiometer for estimating land productivity (MOD13Q1) (Tucker, 1979; Conservation International, 2018a). Land productivity is estimated based on three metrics of net primary productivity, which is the remaining amount of carbon after photosynthesis and autotrophic respiration over a given period (Clark et al., 2001). The three metrics are (a) the trajectory of degradation, (b) land cover performance and (c) productivity state (Conservation International 2018). The trajectory of degradation refers to the rate of change over time (Conservation International, 2018a). The land cover performance is the relative measure of local productivity in comparison to similar vegetation types within the study area (Conservation International, 2018a). Productivity state is the measure of recent productivity changes, which tracks recent changes by splitting the assessment period into a baseline period and the recent period of at least three years (Conservation International 2018; more details under Methods in Section 3.2.1.3). The UNCCD global harmonised methodology has been designed with emphasis is placed on land productivity trend instead of quantifying the extant of change in land productivity biomass (Conservation International, 2018a). Therefore the land productivity dataset provides a logical matrix of five categories (i.e. declining, moderate decline, stressed, stable and increasing) showing long-term trajectory for land productivity (Conservation International, 2018a; UNCCD, 2018).

8.1.1.1. Correcting for climate effects

Annual integrals of productivity are influenced by multiple factors such as anthropogenic drivers, climatic or phenological (seasonal) variations, and rainfall extremes are the most dynamic and can lead to misinterpretation of the overall degradation trends (Ehleringer et al., 1997; Bai et al., 2008). The TRENDS.EARTH plugin was built with an option to monitor degradation status by correcting for climate effects of allowing for the detection of productivity trends that are not driven by variability in climate (Conservation International, 2018a). When the climate effects have been removed, the remaining influences on biomass productivity are those that areas are related to human actions; hence, the climate corrected productivity dynamics can be interpreted as human-induced (Wessels et al., 2004, 2007; Hao et al., 2019). The plugin's built-in feature to adjust the NDVI trajectory values allows for the elimination of seasonal and inter-annual rainfall variability (Conservation International 2018). To achieve the adjustments, the plugin adopts a linear regression model for every NDVI pixel via one of the following three approaches that the user can select based on the context of the area:

Rain Use Efficiency (RUE) – which relies on the relationship between precipitation and productivity for the pixels and is commonly used in areas with arid to semi-arid climate (Wessels et al., 2007, 2012; Fensholt et al., 2013). Use of RUE index is limited by the assumption that NDVI integrals have a strong correlation with biomass production (Nicholson et al., 1998; Wessels et al., 2006), and the NDVI computed in the rainy seasons overestimates biomass production in degraded landscapes (Wessels et al., 2006; Graw et al., 2017). The RUE per pixel trend analysis is used to standardise the year-to-year rainfall variability in net primary productivity and generate a productivity indicator that is free of bias due to rainfall variation (Wessels et al., 2007; Fensholt et al., 2013). This index has been designed for regions where rainfall is a primary driver of productivity (i.e. arid to semi-arid areas), and therefore it does not perform well in high rainfall and sparsely vegetated regions (Wessels et al., 2007; Fensholt et al., 2013).

Residual Trend Analysis (RESTREND) – is an analysis method to assess the relationship between observed NDVI and predicted NDVI using a linear regression model of NDVI and log of annual rainfall or soil moisture to provide NDVI residuals for given precipitation through trend analysis (Wessels et al., 2012). When applied in the Kruger National Park, Wessels et al. (2012) concluded that the RESTREND is an unreliable measure of land degradation assessments after the regression model underperformed in areas with lower degradation intensities, which limits the model's capability for early detection of land degradation. However, RESTREND performs well for detecting extreme and rapid land degradation (Wessels et al. 2012). Usage of RESTREND is centred around the assumption that photosynthetic capacity is a function of rainfall (Wessels et al., 2012). However, this assumption has been challenged since rainwater is not immediately available to plants after a rainfall event which means photosynthetic capacity is more a function of soil moisture instead of actual rainfall, hence there is an option for calculating RESTREND using NDVI-soil moisture in the plugin (Ibrahim et al., 2015).

Ecosystem Water Use Efficiency (WUE_e) – is the ratio of water used by plants through metabolism to the amount of water lost by plants through transpiration (Ponce-Campos et al., 2013). The WUE_e index corrects for climate impacts by accommodating for interception of water by plants (Ponce-Campos et al., 2013). The WUE_e index follows the assumption that productivity and precipitation have a linear relationship, and that during photosynthesis plants use stomatal regulation to trade-off between water loss and carbon gain (Osmond et al., 1982; Ponce-Campos et al., 2013). The hypothesis for WUE trends under the above assumptions is that WUE increases during dry periods and drought, although under severe droughts WUE might decrease (Lu & Zhuang, 2010).

8.1.2. How the plugin computes sub-indicator 2: Land cover change

To calculate land cover change, the plugin uses the land cover imagery provided by the global land cover product of the Climate Change Initiative (ESA CCI-LC) at 300 m resolution by reclassifying the initial year (the year 2001)and the target year (the year 2015) maps with 36 land cover classes into seven land cover classes (UNCCD, 2018). Identifying land cover transition in degradation assessments is necessary to uncover the process of land degradation at study area scale and is based on the IPCC classes shown in Table A1. (Penman et al., 2003; Sims et al., 2019). The land cover classes are tree-covered area, grassland, cropland, wetland, water body, artificial and other lands, and are derived using the aggregation shown in Table A1 (UNCCD, 2018). The plugin then uses a user-defined degradation definition matrix (Table A1).to calculate land cover evolution per land cover class based on pixels.

Table A1: Visual summary showing the possible land cover transitions based on six IPCC classes (Source: Sims et al. 2019). Land cover classes are considered as a degrading process (red), improving process (green) and stable (blue).

Final Class

Original Class

IPCC Class	Forest Land	Grassland	Cropland	Wetlands	Settlements	Other Land
Forest Land	Stable	Vegetation loss	Deforestation	Inundation	Deforestation	Vegetation loss
Grassland	Afforestation	Stable	Agricultural expansion	Inundation	Urban expansion	Vegetation loss
Cropland	Afforestation	Withdrawal of Agriculture	Stable	Inundation	Urban expansion	Vegetation loss
Wetlands	Woody Encroachment	Wetland drainage	Wetland drainage	Stable	Wetland drainage	Wetland drainage
Settlements	Afforestation	Vegetation establishment	Agricultural expansion	Wetland establishment	Stable	Withdrawal of Settlements
Other Land	Afforestation	Vegetation establishment	Agricultural expansion	Wetland establishment	Urban expansion	Stable

Table A2: UNCCD land cover aggregation of 36 land cover classes (Source: UNCCD 2018).

UNCCD Legend	ESA's CCI-LC labels
Tree-covered areas	Tree cover, broadleaved, evergreen, closed to open (>15%)
	Tree cover, broadleaved, deciduous, closed to open (>15%)
	Tree cover, broadleaved, deciduous, closed (>40%)
	Tree cover, broadleaved, deciduous, open (15-40%)
	Tree cover, needle-leaved, evergreen, closed to open (>15%)
	Tree cover, needle-leaved, evergreen, closed (>40%)
	Tree cover, needle-leaved, evergreen, open (15-40%)
	Tree cover, needle-leaved, deciduous, closed to open (>15%)
	Tree cover, needle-leaved, deciduous, closed (> 40%)
	Tree cover, needle-leaved, deciduous, open (15-40%)
	Tree cover, mixed leaf type (broadleaved and needle-leaved)
	Mosaic tree and shrub (>50%) / herbaceous cover (< 50%)
Grassland	Mosaic herbaceous cover (>50%) / tree and shrub (<50%)
	Shrubland
	Shrubland evergreen
	Shrubland deciduous
	Grassland
	Lichen and Mosses
	Sparse trees (<15%)
	Sparse shrub (<15%)
	Sparse herbaceous cover (<15%)
Cropland	Rain-fed cropland
	Herbaceous cover
	Tree or shrub cover
	Cropland, irrigated or post-flooding
	Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%)
	Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (< 50%)
Wetland	Tree cover, aquatic or regularly flooded in fresh or brackish water
	Tree cover, aquatic, regularly flooded in salt or brackish water, Mangroves
	Shrub or herbaceous cover, flooded, fresh/brackish water
Artificial surfaces	Urban areas
Other lands	Bare areas
	Consolidated bare areas
	Unconsolidated bare areas
	Permanent snow and ice
Waterbodies	Waterbodies

8.1.3. How the plugin computes sub-indicator 3: Soil organic carbon stocks

Soil organic carbon stocks are derived from the SoilGrids250m project by the International Soil Reference and Information Centre (ISRIC) in reports to the United Nations Convention to Combat Desertification (UNCCD) (Hengl et al., 2014, 2017). The SoilGrids250m project uses the topsoil (upper 30 cm) to calculate SOC through the integration of multiple datasets and ground-based observations (Hengl et al., 2017). To compute SOC, the plugin considers the land cover transitions in relation to the topsoil for each pixel, based on the global climate data to estimate the average SOC stock for each spatial feature (Conservation International, 2018a).

The plugin then compares SOC for the assessment period (i.e. initial and target years) for each geographic feature (Conservation International, 2018a). Soil organic carbon stocks are averaged over the assessment period to standardise SOC loss reporting at a national and global scale (Conservation International 2018). Therefore, the areas that experience a minimum of 10% magnitude of change are considered to be degraded (10% SOC loss) or improved (10% SOC gain) (Mugagga & Nabaasa, 2016). Eventually, the plugin assesses whether there are potential “false positives and negatives” and justifiable anomalies by considering a 10% SOC variation from initial to target years (Conservation International, 2018a).

8.2. Appendix 4.1: 2019 PGIS Report-Machubeni

RHODES UNIVERSITY GEF5 SUSTAINABLE LAND MANAGEMENT PROJECT, EASTERN CAPE

Macubeni

Deliverable 1. A report indicating methods and outcomes (including a detailed map) of a participatory mapping exercise on priority rehabilitation areas and key resources for the five GEF5 SLM Project villages at Macubeni

Xoxo, Sinetemba & Mantel, Sukhmani

Date: July 2019

Declaration:
Completion of this reports emanates from a masters study entitled:
Assessment of ecological infrastructure for water security using Macubeni and Cro *catchment as case study sites.* The MSc research is being conducted under WRC K5/2928 titled 'The role, benefits and prioritisation of Ecological Infrastructure (EI) i the impacts of droughts in South Africa'.

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Introduction

Over 40% of global ecosystems are considered to be degraded, of which 70% are in developing countries (MEA, 2005; Stavi and Lal, 2015). Anthropogenic and biophysical factors such as climate sensitivity, polluted waterways, increasing rates of biodiversity loss and changes in landscape character, are attributed as causes to this degradation (Rockström *et al.*, 2009). Degraded ecosystems limit ecosystem services that are needed to sustain human well-being and enhance ecosystem resilience (Ellis, 2000; MEA, 2005; Folke, 2006; Morse *et al.*, 2014). The capacity of landscapes to provide ecosystem services can be indirectly linked to landscape composition and structural elements (MEA, 2005; Willemen *et al.*, 2008).

Recognising the importance of ecosystem services when studying and managing landscapes, South African National Biodiversity Institute (SANBI, 2014) focuses on ecological infrastructure via (i) the social-ecological benefits they provide, and (ii) potential investors to protect, maintain or restore the ecosystems. SANBI (2014) defines ecological infrastructure as "naturally occurring ecosystems that provide valuable benefits to people", and has developed a holistic ecological infrastructure framework to adequately capture ecosystem services and ecosystem values for landscape planning and management (SANBI, 2014).

Globally, United Nations' Sustainable Development Goal (SDG) 15, which deals with life on land, sets out to neutralise land degradation by 2030 through (i) protection of lands from any degradation, (ii) prevention of further degradation, and (iii) rehabilitating severely degraded landscape (Suding *et al.*, 2015). The research community has widely recognised that actions to neutralise ecological degradation should be centred around the recognition of human-nature interactions (Folke, 2006; Ostrom, 2009; Biggs *et al.*, 2012). A further value has been placed on restoration interventions which aim to target an increase in the resilience of both the landscape and the communities that depend on the ecosystems (Stavi and Lal, 2015). To support this requirement, the Global Environmental Facility (GEF) mobilised a number of key actors in ecological restoration within South Africa in a project aimed at "strengthening the capacity for

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adoption of knowledge-based sustainable land man
and ecosystem rehabilitation in support for the gre
building capacity of selected government depa
governance and financial incentives" (https:
ecosystems-benefit-through-slm-roductive-degrad
support is very important because for any restorat
for funding to carry out the task (Holl and Howarth

Following lessons learnt from an earlier unsus
(described in Shackleton and Gambiza, 2008), proj
adopted stakeholder engagement as a key tool
approach. For this reason, the participatory map
making tool for the task reported here, which
landscape restoration by placing the users at the c
resources they value and those they prioritise for
note that participatory mapping tools are seldor
planning or land management. Whereas, Reed *et a*
every stage of the decision-making can help secure
of restoration. Hence the GEF5 SLM project
participation.

The present report forms part of the GEF5 SLM p
project that seeks to detect and prioritise key resou
base for investing in ecosystems for water security
Masters study is to make use of stakeholder input
final key resource priorities for rehabilitation
prioritisation outcomes will be made available to th
management plan for Macubeni. The focus of the p

2

presented in Table 1. The target resources have been selected based on the potential for well-managed ecosystems to support flow regulation ecosystem service (Le Maitre *et al.*, 2014).

Table 1: Key focal resources of the project and link with SANBI's Ecological Infrastructure framework.

Focal resource	**Fit with SANBI Framework** (SANBI, 2014)
Grasslands/rangelands	Clearing of IAP; Improving landscape management practices
Abandoned croplands	Clearing of IAP; Improving landscape management practices
Riparian areas	Clearing of IAP; Improving landscape management practices
Wetlands	Rehabilitating wetlands

The report outlines the steps that were taken to conduct the participatory mapping exercises with representatives from the five study villages in Macubeni to verify and prioritise socio-spatial data of the focal resources (using workshops and field surveys with local villagers. In the exercises, PGIS was used both as a low-level method of participation to consult, and a high-level participation to empower people to express their spatial understanding of the area, and to make decisions about ranking local natural resources while sharing understanding of restoration and sustainability needs with all the involved actors (*sensu* Wiedemann and Femers, 1993). The report further presents the limitations associated with the process.

Study area

Macubeni village (27° 01-16′ E; 31° 27-34′ S): The Macubeni area has a landmass of 153.0 km^2 and is located in Ward 13 (21306013) of the Emalahleni Municipality in the Eastern Cape (Figure 1). The Cacadu River quaternary catchment (S10F) drains the hilly and mountainous terrains (altitude ranging 1300-2100 m above sea level) of Macubeni. The underlying rocks in the area are composed of four rock types, namely: the Karoo Dolerite suite, the Tarkastad subgroup, Molteno

formation, and the Elliot Formation (Smith *et al.*, 1
diabase, sandstone, mudstone and shale rocks (Sm

The area presents a unique ecological degradation c
and *Euryops floribundus* encroachment (Shackleto
a multi-shrub species that is widely dispersed in t
which is used by rural communities as a medicinal p
Gambiza, 2008). According to the 2011 Census da
was 5 817 people, living in 1 595 households from 1
in the area include subsistence crop farming, li
resources (e.g. brick making soil, coal, fuelwood, fr
crops include maize, beans, pumpkin and oats for
sheep, goats, poultry and donkey.

In 2017, Emalahleni Municipality listed Macubeni a
which is consistent with the 2011 Census data th
receives water from the service provider (Cen
Macubeni is a relatively dry area, falling within the
between 1950 and 2000 (Figure 2).

The GEF working areas based in the study area (5
leaders and the elected leadership on the basis o
proximity to each other, their varied location withi
and activities they represent, and previous involvem
(Sisitka and Ntshudu, 2017). The participating sub
Boomplaas, and Helushe (Figure 1).

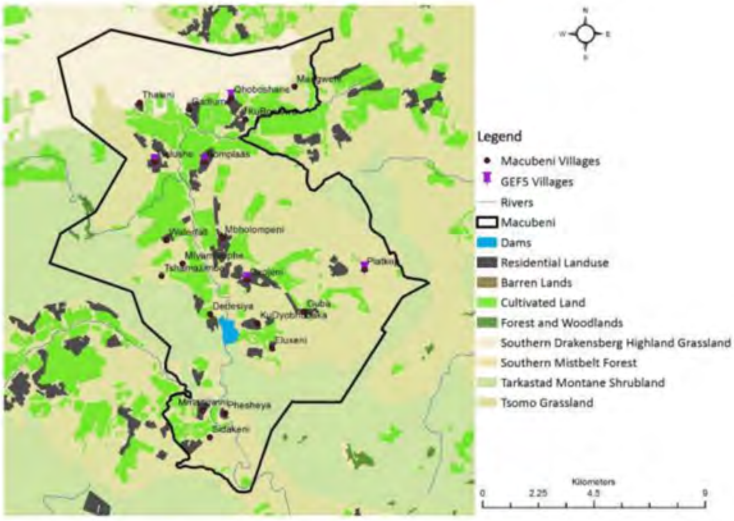

Figure 1: Land use and land cover context map of Macubeni sourced from SANBI (2018) and PlanetGIS (2012).

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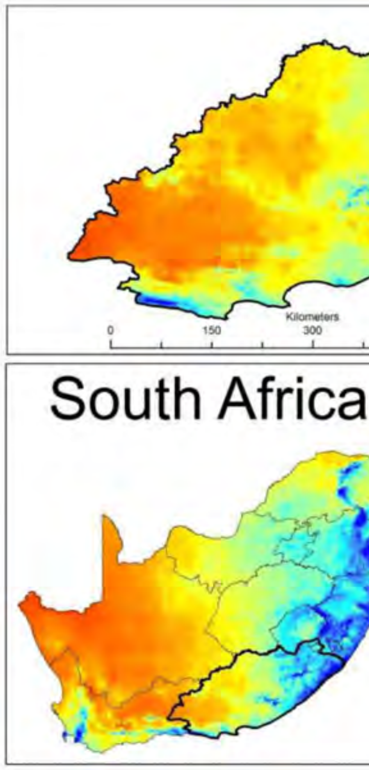

Figure 2: Average annual rainfall data for 1989 - 20

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Methods

A Participatory Geographical Information System (PGIS) exercise (Reed *et al.*, 2008) was conducted in Macubeni to verify and prioritise socio-spatial data of the focal resources. The approach used in the case study area employed the direct-to-digital (D2D) PGIS method (DeRoy, 2016) to verify key resources and help identify priority land covers based on stakeholder views.

Workshop design

Two group meetings were held: group meeting 1 (WS1) for the villages on the eastern side of the case study area involving stakeholders from Platkop and Gxojeni sub-villages with 15 participants, and group meeting 2 (WS2) for the western side villages involving stakeholders from Qhoboshane, Boomplaas and Helushe sub-villages with 25 participants (Appendix 1 – Attendance registers). The participatory mapping group meetings spanned over two days with one half-day group meeting for each side. The group meetings took a form of facilitated discussions with the guidance of a semi-structured interview schedule (outlined in the next subsection) and base maps of project sites with previously mapped key resources. Figure 3 shows a schematic overview of the process followed for the participatory mapping exercises.

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Figure 3: Schematic presentation of the meetir
Workshop one (with participants from Platkop
participants from Helushe, Qhoboshane and Boom

Recruitment of participants

The key user groups that were recruited by mear
Ntshudu, 2017), based on their interests in terms c
invited for the group discussions. The identified p
were the farmers' association and village land co
and over. The headmen and sub-headmen were a
for a separate meeting for a quick briefing since he
participants were expected to have lived in Macut

Participatory mapping process

Prior to the participatory mapping process, the GEI
Then an IsiXhosa speaking facilitator used the
Informed consent) to (i) clearly explain the purp
informed consent, and (iii) explain the expected o

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The community mapped focal EI resources can be accessed from http://iwr.ru.ac.za/iwr/Macubeni/.

provided a breakdown of the workshop schedule. Data were collected by means of semi-structured guidance in an open group session which consisted of 3 questions viz.:

1. Key natural resources

- Using the map (Image 1; Image 2; Image 3; Image 4; Image 6), please verify if the areas shown therein are representative of all the areas you said you value the most.

For croplands and riparian margins, stakeholders were not asked to redraw polygons because these can be extracted from spatial datasets of river shapefiles using ArcGIS tools. Instead, participants simply shared their views regarding these two ecosystems.

2. Priority resource areas

- In which location is the most useful [*key resource name*]?

(*Resources that were included in the exercise: Abandoned croplands* (Image 1; Image 4), *Rangelands* (Image 2; Image 5), *Wetlands* (Image 3; Image 6)*; and Riparian areas*)

- Why are the identified locations ranked as the most important or least important resources?

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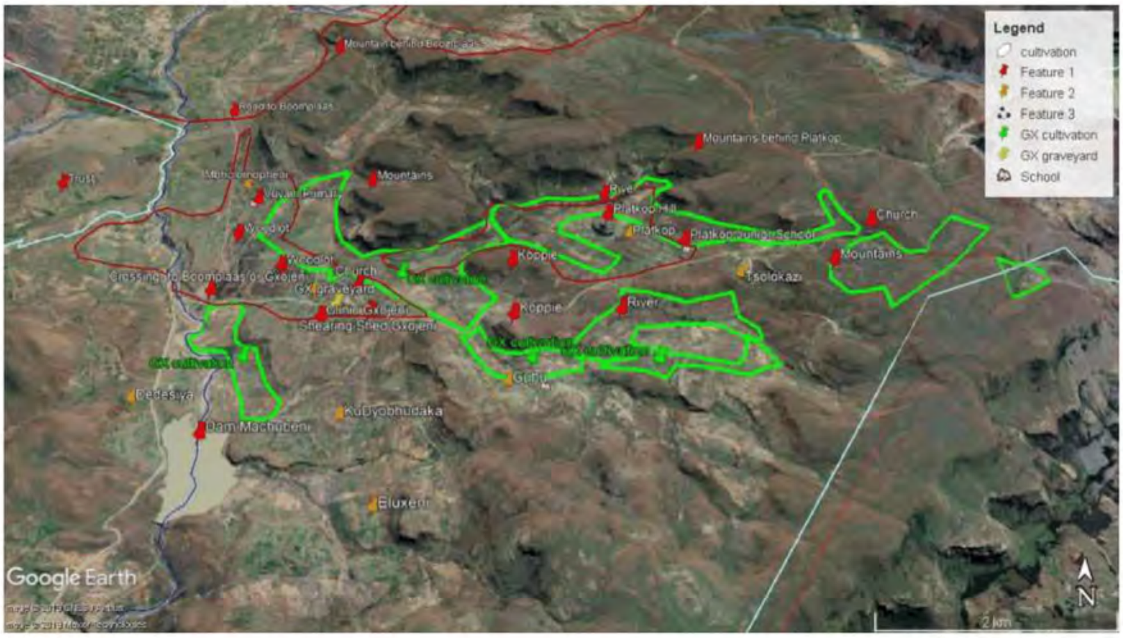

Image 1: Screenshot of the Google Earth base map for WS1 croplands. Red polygon = Village boundaries; green polygon = cultivated areas.

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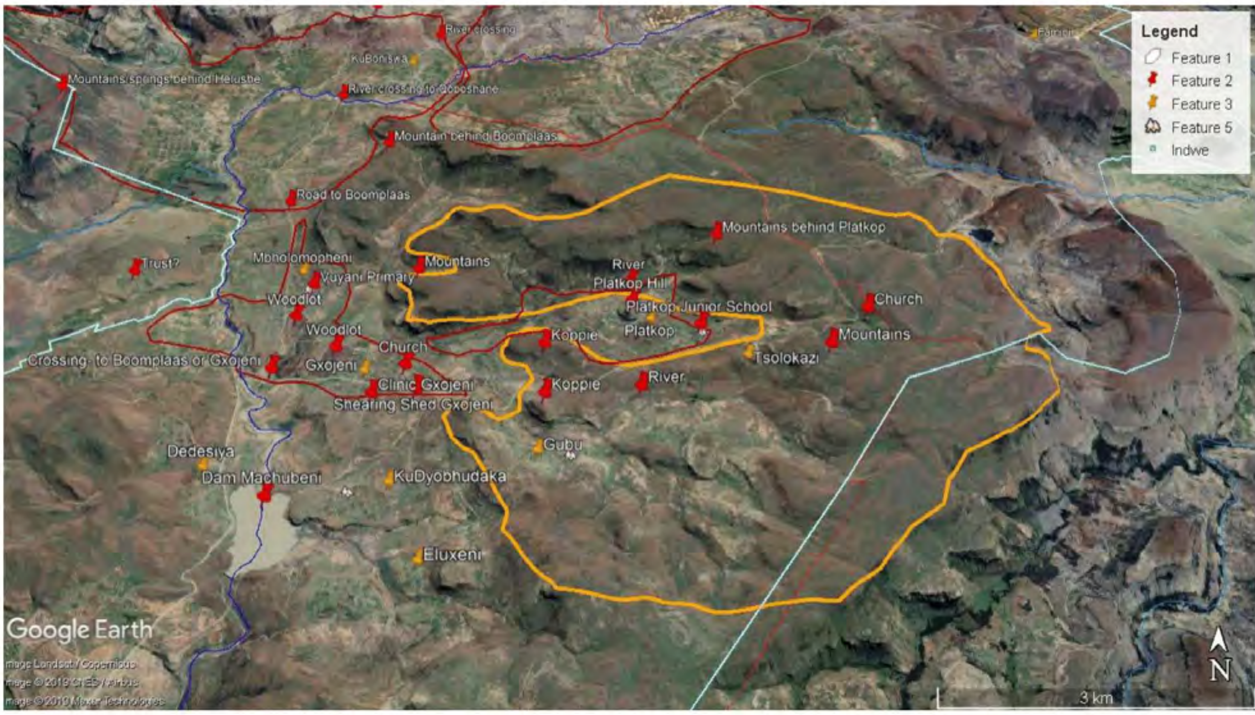

Image 2: Screenshot of the Google Earth base map for WS1 rangelands. Red polygon = Village boundaries; orange polygon = rangel
area.

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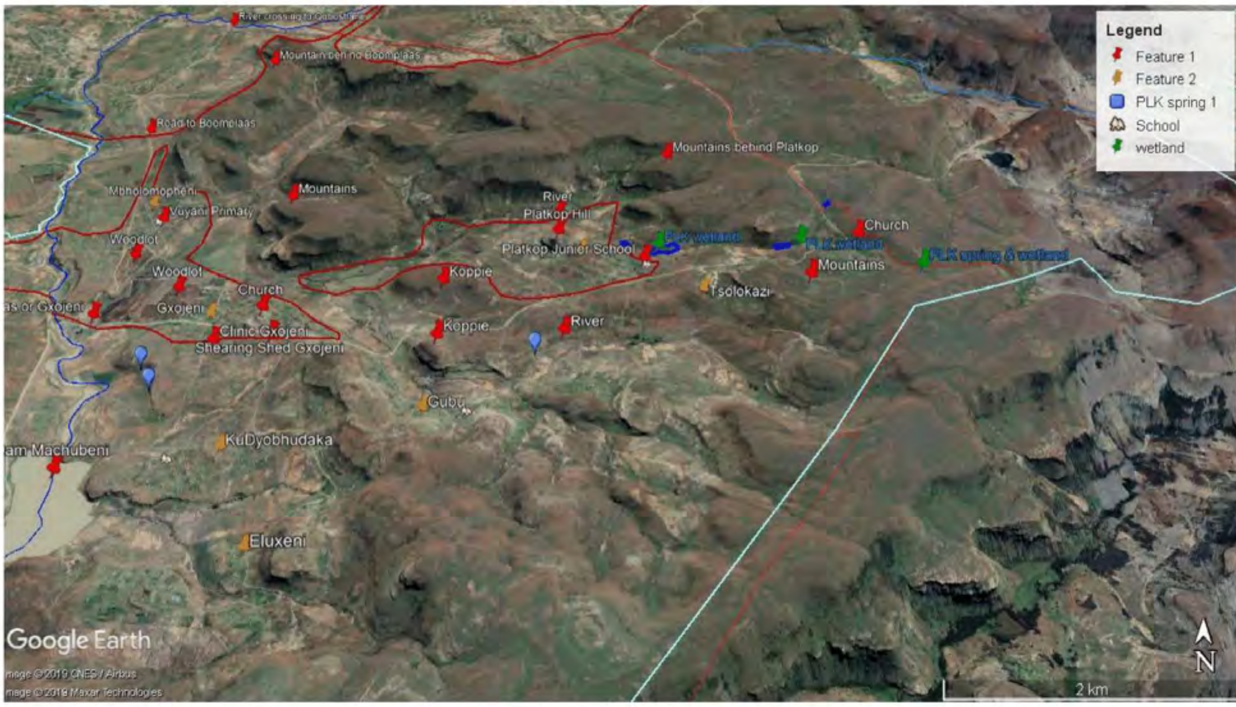

Image 3: Screenshot of the Google Earth base map for WS1 wetlands. Red polygon = Village boundaries.

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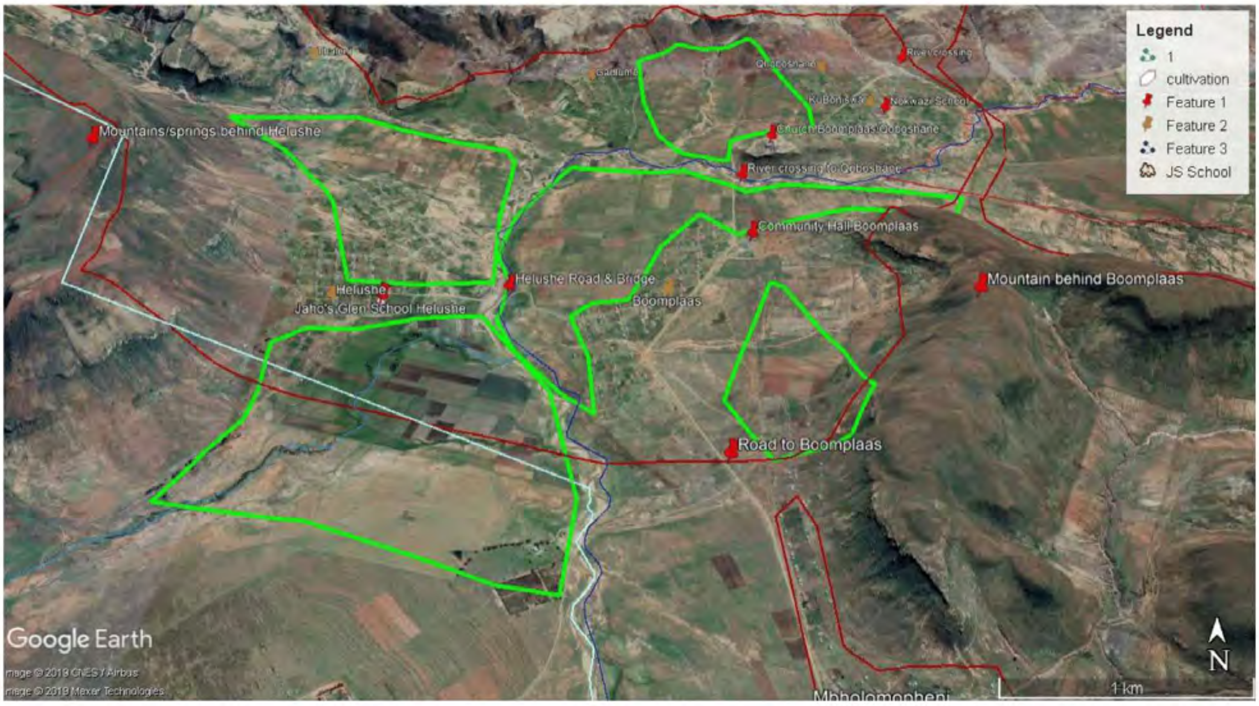

Image 4: Screenshot of the Google Earth base map for WS2 croplands. Red boundary = village boundaries.

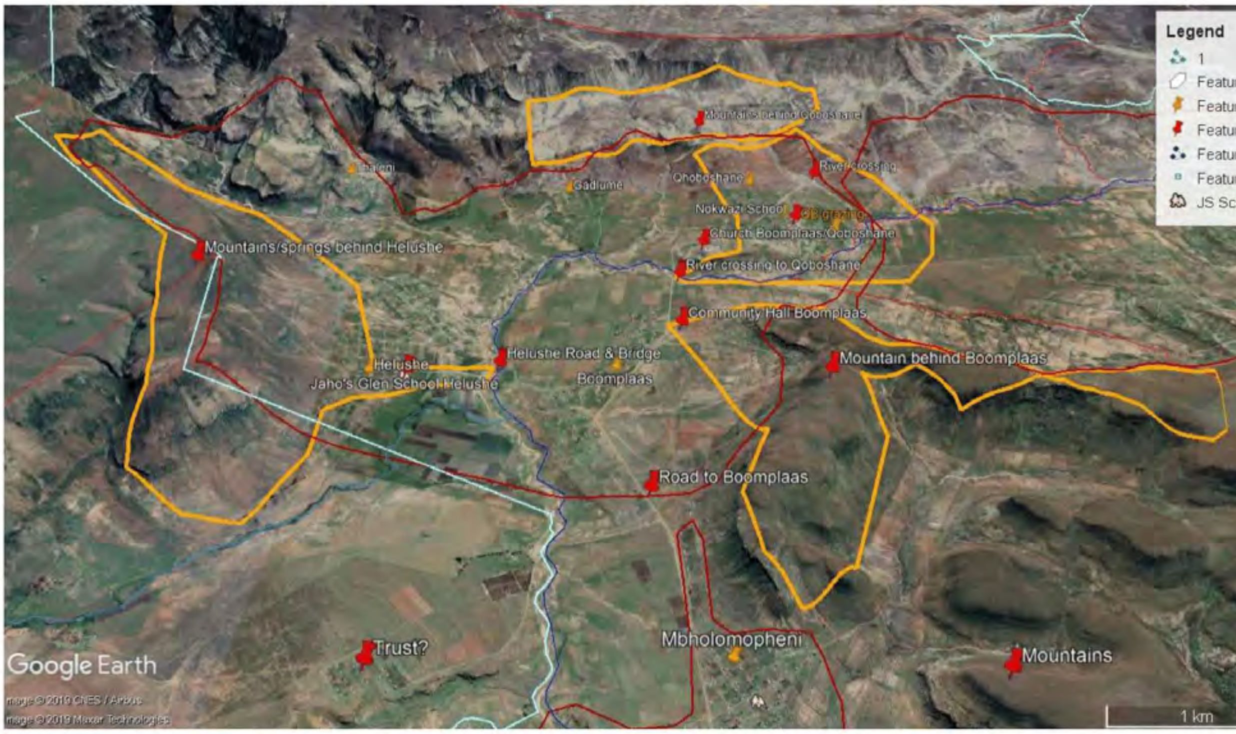

Image 5: Screenshot of the Google Earth base map for WS2 rangeland locations. Red polygon = village boundaries; orange poly
rangeland area.

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Image 6: Screenshot of the Google Earth base map for WS2 wetland locations. Feature 1 = Village landmarks; Feature 2 = GEF5 v
and Feature 3 = Cacadu tributary.

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Verifying and prioritising key resources

Popular place names in the area were used as feature markers (or landmarks) for the participants to locate themselves in the maps, for example, schools and mountain ridges as shown in (Image 1; Image 2; Image 3; Image 4;Image 5; Image 6). In the base maps, villages and subsequent resources were organised by folders in Google Earth for ease of guidance through the workshops (Image 1; Image 2; Image 3; Image 4; Image 5; Image 6). The first step of the workshop was to ask participants to verify areas they deem important for their livelihoods in Macubeni using the identified key resources data from the previous workshop as shown in Image 1; Image 2; Image 3; Image 4; Image 5; Image 6. At this stage, land cover polygons of rangelands and croplands that were collected during a workshop conducted by the GEF land rehabilitation group earlier this year, and wetlands (Myoyo *et al.*, 2017) were used to refer to the land covers for the first question. A Google Earth (GE) imagery of the village outlining the land covers-land (used as a base map) was set up on a projector monitor for visual access to everyone (Image 1; Image 2; Image 3; Image 4; Image 5; Image 6). During the workshop, facilitators recorded the information on notepads.

After verifying the key resources, participants were prompted to share their views on which resources they value the most and what criteria defined how they valued the resource area. The stakeholders were asked to define the relative importance of the resources into two classes only (less important and more important) because the characterisation was not for statistical measurements but to obtain qualitative stakeholder views. As a result, the longest group discussion time was 50 minutes. Image 7 below depicts the setup of the discussion meetings.

Image 7: Participants in WS2 identifying and priori
Stakeholders are shown discussing missing resou
identifying a higher priority location of key resource

Field walks

The researcher conducted field surveys with the h
surroundings of both WS1 and WS2 villages. The p
locate the target resources that had not been ad
wetlands, springs and boreholes) and to confirm
rehabilitation sites, as identified by the stakehold
researcher took photos and GPS coordinates of the
GPS locator, in order to assist with assigning a coars

Data processing

Following the participatory workshops, the researc
(version 7.3.2.5) to digitise and update both the ke
to meet spatial descriptions by stakeholders using
Borders of the resources were drawn using the p
resources (i.e. springs and boreholes) were digitised

The features were allocated according to a classification system that categorises the features according to use and appearance e.g. springs and seeps were allocated as triangles, while rangelands were corrected as red polygons. The final map displays the priority resource locations as points and polygons. Details relating to each resource will be entered into a database and stored for later use in the prioritisation of the key resources as valued by the stakeholders. The digitised areas will then be stored in GE for an interactive map and will later be used for other research needs. The digitised areas will also be imported into ArcGIS and converted into a layer to present the stakeholder priorities as point marks and shaded maps.

Ethics for human subjects

This work adheres by the university research guidelines and was reviewed under the Human Ethics protocol by the Rhodes University Ethics Standards Committee (Rhodes University, 2014). Ethics were granted by the committee as a Participatory GIS workshop for Macubeni-0448, May 2019. Ethics certificate is attached (Appendix 3: Ethics clearance certificate).

Results of the workshop and field su

Stakeholder profile

Participants in the meetings come from a diverse r registers, and have different primary livelihoods a larger communities from which they reside. Of the males), most of the participants were livestock k widely split such that the older participants knew of the resources, while the middle-aged participan the resources. A detailed stakeholder analysis for Stakeholder Report (Sisitka & Ntshudu, 2017).

Verifying and prioritising key resources

The two PGIS focal discussions resulted in stakehol focal resource. The final maps showing locations Figure 6, and a user-friendly interactive map is host noted inaccuracies in the base maps for springs, w (corrected locations shown in Image 8). The follow verified and prioritised key resources.

[7] Digitised google earth map for Macubeni PGIS available at:

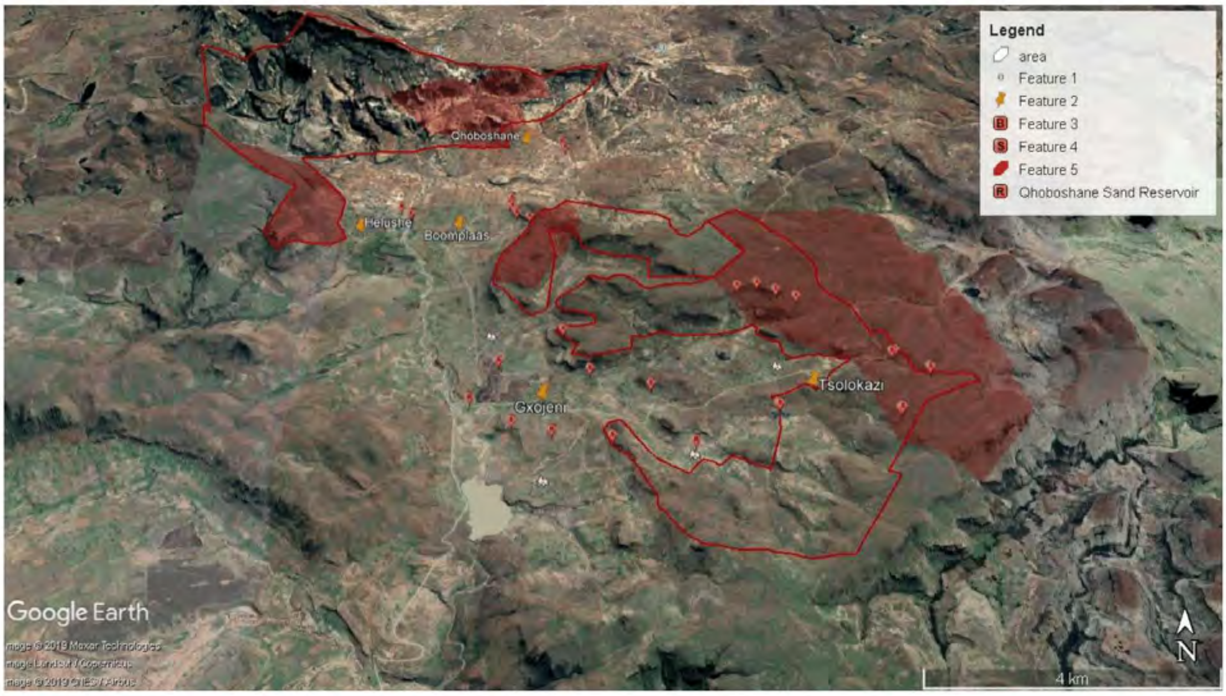

Image 8: Snapshot of the resulting Google Earth map of verified and prioritised rangelands and wetlands for rehabilitation. Feature 2 = GEF 5 Villages; Feature 3 = Boreholes; Feature 4 = Springs; and Feature 5 = identified and prioritised rangelands.

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In summarised Table 2, a criteria-list of proxies for how stakeholders decide on important resources was derived based on reasons for why a certain resource location has a higher priority for rehabilitation. Wetlands and riparian areas were allocated a higher priority based on ecosystem condition (e.g. intact or degraded), and availability of resources (e.g. water and healthy grasses) (Table 2). For wetlands, the ecosystem condition was mostly based on whether a certain wetland or spring that feeds the wetland is still active and remains active even during the dry season, while the availability of resources was based on water and fodder supply especially in dry periods. For the riparian areas, the ecosystem condition for river banks was determined by the availability of constant water flow and grasses, and available resources were determined similar to wetland ecosystems.

Rangelands were allocated a higher ranking based on ecosystem condition (e.g. degraded or not), and availability of resources (e.g. healthy sweet veld grasses) (Table 2). Additionally, rangelands were allocated a higher ranking based on distance from homesteads and surface area of the resource Table 2). The proxies listed in Table 2 will be used with spatial datasets to allocate the final priority weights for the focal resources for rehabilitation in Macubeni.

Table 2: Four criteria list derived from stakeholder views. A tick depicts applicability of criteria to resource priority by stakeholders. The PGIS exercise excluded verification of cropland areas, and the locals decided to prioritise all the croplands. Therefore, the criteria list shows a grey fill for croplands because none of the proxies was used to rank the croplands.

Focal resource	**Criteria**			
	Condition	**Available resources**	**Distance from home**	**Resource size**
Wetlands	✓	✓	Not important	Not important
Rangelands	✓	✓	✓	✓
Croplands				
Riparian areas	✓	✓	Not important	Not important

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Wetlands

Figure 4 shows the location of water source area
identified and verified by the users from both WS1
Boomplaas and Helushe) in relation to underlying
scattered around the landscape of Macubeni, whe
sources are closely located (Image 8; Figure 4). Th
villages still remains unknown, but 18 wetlands w
spring source (Figure 4). As shown in Figure 4, th
within the valley, while in Boomplaas, Platkop a
surfaces of the hillslopes.

It is noteworthy to highlight that the participa
(Marshall *et al.*, 2019; Murata *et al.*, 2019). For ex
that supply provisioning services such as animal
over those whose functions are not of direct bene
the discussions commented:

> *"Ingaba ikhona na indlela yokuba kulun*
> *yam, ngoba ndilumkele ukulimala kw*
> *Ngamanye amazwi ndifuna ukuqonda*
> *kwenzela imfuyo ingalimali?"*

> *Is there a way for the wetland in my cro*
> *anxious that people's livestock will get inj*

During the field walks, it was observed that nume
other wetlands, especially those in Boomplaas (se
Macubeni, these resources altogether, play an imp
in addition to providing forage and acting as drin

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2017). Sada (2000) discusses the importance of wetland water quality (in terms of water temperature and water chemistry) and notes that these ecosystems are important for habitat provision to endemic biota. Non-native ungulate livestock activities, brick making, and irrigation are expected to impact the integrity of wetlands in the area; hence locals in Macubeni have built wells around some springs, particularly the ones with higher priority as a way to protect the water resources against sediment pollution and damage by livestock (Image 9). Most of the springs are in remote areas where there are no roads that allow direct access. Using the classification by Sada and Pohlmann (2004), most springs in Macubeni have been converted by locals into qanats or wells due to water shortages, while others have been semi-converted and they are classified as limnocrenes – *discharging into a pool or pond before flowing into a channel*. A few of these (mostly the furthest from the villages) still remain intact, and these can be classified as rheocrenes – *discharging into a well-defined channel* (Sada and Pohlmann, 2004). See the interactive online map for a detailed view and images of these resources and examples in Image 8.

With respect to prioritised springs and wetlands, participants from WS1 identified three resources per village that are viewed as important based on the criterion in Table 2 and these border the villages (see Figure 4). For the wetlands, participants in WS1 identified three wetlands that are most important to them, while participants in WS2 opted to allocate high priority to all their water resource areas (Figure 4).

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Figure 4: Distribution of identified and prioritised s
to area geology. Geology data source: Council for

Rangelands

Participants presented a mixed understanding of w
land as rangeland while others referred to rang
strictly designated for grazing sites (Reid *et al.*, 200
above statement, Image 10 shows vacant land th
allocated as grazing areas. After some discussion,
sites as the rangelands, hence Figure 5 shows verif
area. These polygons are slightly different from th

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Image 9: Images showing the types of engineered wells and animal drinking ponds in Macubeni

during the verification step of the exercise. Rangeland areas are allocated in the hill-slopes to plateaus that buffer the village area (Figure 5). Participants indicated that livestock from Gxojeni and Platkop generally share the rangelands; the Boomplaas grazing area is also adjacent to the Platkop grazing area (approximate sizes shown in Table 3).

With respect to rangelands, participants in WS1 pointed to a direction that they deem most important for their livestock to feed, which is rangeland to the east of Macubeni (Figure 5), approximately 5 times bigger than other villages (Table 3). The prioritised grazing area from WS1 occupies a larger size compared to WS2 villages because the polygon goes beyond the boundaries

of Macubeni (Figure 5). In WS2, participants ex

based on how far the distance is from residential

Table 3: Approximate surface areas of rangelands
sites in Macubeni. Division lines between rangel
descriptions by stakeholders.

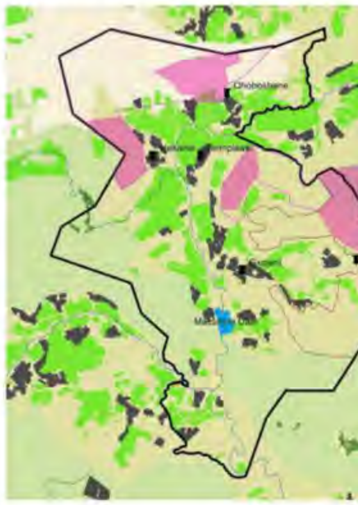

Figure 5: Land use and land cover map (PlanetGIS
sites. Most important grazing sites are shaded pin

Croplands

A uniform perspective regarding croplands emerg

supported the idea of abandoned croplands as indicated that in the past ten years, woody encroachment (*E. floribundus* in this case) has occurred at a faster pace than natural growth of local vegetation and this has been more prominent in abandoned cropland fields. In one workshop, an elder summed this perspective up by saying:

> *"...ukusuka phaya ku-2000, ukuza, uzawqwalasela ukuba amasimi angasalinywayo, kunye namathango agcwele ilapesi. Zindawo ezo ebekufanele zikhula ingca okanye izityalo. Umzekelo, aba banehabile, abanalapesi emasimini tu. Logama abo bagqibela kudala kakhulu ukulima ubona ilapesi. Aliphelelanga amasimini ke elilapesi, lide keengoku lanabela nalapha ezindlini"*

> *Since around the year 2000, resin bush (Euryops sp.) seems to have spread faster on abandoned croplands and ancient homesteads. This dispersal is much higher in comparison to endemic vegetation. Eventually, the resin bush is prominent even around the residential space.*

Image 10: Images showing open areas in Macubeni. Left shows residential reserve areas where livestock graze, right: shows actual rangeland site.

The boundaries of the croplands can be visible o
and shape (Vogels *et al.*, 2017). Since the GEF pro
out in a preceding workshop, verification of th
workshops. In Macubeni, croplands are located i

For prioritisation, stakeholders were unanimous
physical capital and an important livelihood so
resume their cultivation activities.

> *"Hayi, siyakufuna ukulima nto nje asina*

> *We want to resume cropping, but we la*

> *"Kuyenzeka moxa wambi amanzi adlule amanzi ahamba ezindongeni ubani uye a akanayo indlela yokuwenyusa ewawu ezitholeni"*

> *There are cases where a stream cuts t because the stream channel is deep, we the crops*

Riparian margins

While not included in maps contained herein,
Additionally, natural sand reservoirs (Olufayo *et*
important sources of water during dry seasons (
Gxojeni and Qhoboshane (see Image 11), and the
water compared to others, based on stakeholder

> *"Yiyeke imothombo, iyakhawleza ukuts baye bayokomba isanti phaya ezin*

afumanekayo phaya, kodwa ke kuba amanzi akemi unaphakade, nalawo asesantini aye ehlele ngezantsi. Kodwa eladami lesanti, kudala likhona kwaye amanzi okuncenceshela siwafumana phaya"

Besides the springs which often dry up, we also access water for irrigation from sand reservoirs in the river channel. Those supply us with more water than springs, but we often lose that water to flow.

This comment then led to the discussion of whether the river channels with sand reservoirs should get a higher rank, which will be verified with locals in the next feedback meeting. It should be noted that sand reservoirs could be under threat from sand mining which is one key source of livelihood in the area. Some locals also mentioned the possible positive role played by woody encroachment of riparian margins for bank stabilisation.

Image 11: Photos showing sand reservoirs in Macubeni riparian areas. Left image shows a natural sand reservoir in Gxojeni while image in the right shows a natural sand reservoir in Qhoboshane.

Reflections and limitations

In the case of Macubeni, the assessment of fo
incorporating the local voice in any rehabilitation,
by and of value to the local custodians of the area
for obtaining information that will be incorporated

Dealing with place-based resources often requires
this study, the village scale seemed to be appropria
boundaries and locations for the focal resources.
resources were described, where the oldest partic

"Kunomthombo osemaqaqeni, ngumthom
omdala kakhulu, umthombo owawu
nokhokho bokhokho..."

There is a spring below the shrubs, it's a p
has been sipping since the ancient tim
parents...

Such comments were most common when dec
particular, water source area resources such as we

Although the PGIS method was a useful tool to
method had to be slightly altered to take the form
the workshops, we noticed that most participant
themselves using digital maps. This prompted the f
facilitator allowed the participants to use their kn
locations, for example, instead of pointing the r
would use area landmarks to give a direction of
approach proved to be effective as larger groups

discussion began to flow with ease. Therefore, producing precise boundaries for the resources that are presented as polygons was sometimes a challenge, as has been found elsewhere (Levine and Feinholz, 2015). Levine and Feinholz (2015) used a systematic approach to navigate this challenge, through the use of gridded cells, attribute tables and supplementary qualitative text. However, in the present case, the only supplementary text was used as a guide to decide on the rangeland boundaries. Such text was derived from comments such as:

> *"Hayi, imfuyo yethu iyahamba, iyalahleka. Itya apha iyoma edolophini eLady Freyi, nase Indwe ngoba kaloku akho cingo. Apho ingca intl khona, ukuba akukho cingo iyaya"*
>
> *Our livestock covers a large footprint in search of good fodder. At times we collect it in the nearby towns i.e. Indwe and Lady Frere because there is no fencing to act as boundary lines.*

During both workshop discussions, when asked about the resources they considered important and how they decided upon this, participants would quickly negotiate amongst themselves before coming up with a final response. Through this negotiation, participants were able to exchange knowledge and come to agreements about common-pool resources (Ostrom, 2015), an act that can be interpreted as a form of social learning (Reed *et al.*, 2010).

Data usefulness for landscape management

Among other degradation challenges facing the community of Macubeni, the availability of freshwater resources for domestic use and agricultural purposes is a crucial challenge. Some common comments in this regard arose when interrogating the deagrarianisation phenomenon:

> *"Hayi, siyakufuna ukulima nto nje asinantsiba kwaye komile"*
>
> *We want to resume cropping, but we lack means and the soil is dry*

> *"Kuyenzeka maxa wambi amanzi adlule*
> *amanzi ahamba ezindongeni ubani uye a*
> *akanayo indlela yokuwenyusa ewawu*
> *ezitholeni"*
>
> *There are cases where a stream cuts t*
> *because the stream channel is deep, one*
> *into the crops*

The impact of water shortage on subsistence fa
Africa and globally (Mekonnen and Hoekstra, 201
the PGIS exercise as an opportunity to inform th
rehabilitation of water source areas which may b

Spatial information about the location and pr
management actions. The Headmen specificall
demarcations be included in the final maps an
Therefore, once the spatial information has bee
to inform some specific management plans and
team.

Conclusion

Degradation in rangelands, croplands, wetlands,
increased soil erosion and higher run-off rates (P
Van der Waal and Rowntree, 2018). Marshall *et a*
resources can help rehabilitation managers with i
of ecosystems, how they can rehabilitate the
decisions made towards rehabilitating such ecosy
and prioritise key resources was considered ess

project assessment of how ecological infrastructure can facilitate drought mitigation through flow regulation (Le Maitre *et al.*, 2014). Such an assessment is important to provide evidence that can help transform landscape intervention instruments that are available for governments and practitioners (Palmer and Bennett, 2013). Secondly, this PGIS exercise revealed the criterion used by locals to prioritise their key resources. This criterion will be used to allocate weights for the final prioritisation of key resources in the area.

The PGIS exercise reported here enhances the applicability of any emerging recommendations within the Macubeni landscape scale. Moreover, because of their social-ecological functions, in the case of Macubeni, ecological infrastructure resources need to be prioritised and protected simultaneously. For restoration, knowing how focal resources are valued by locals has a potential of increasing restoration success if restoration is linked to social commitment.

Future work will focus on the use of remotely sensed data of these key resources to provide an evidence base of change detection for developing a prioritisation ranking. The existing stakeholder data sets will be used to support the findings reported here and the change analysis results for the prioritisation. Feedback meetings to the stakeholders will take part in early 2020.

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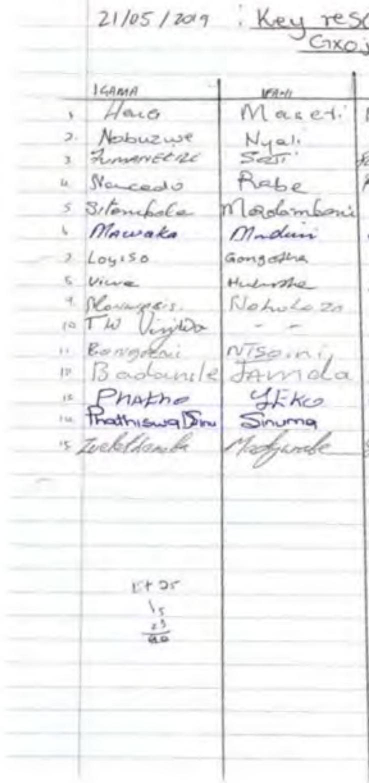
Appendices

Appendix 1 – Attendance registers


21/05/2019 : Key res[illegible]
Gxo[illegible]

	IGAMA	IFANI
1	Hace	Maceti
2	Nobuzwe	Nyali
3	[illegible]	Seti
4	Nocedo	Rabe
5	Sitembele	Madambani
6	Mawaka	Maduni
7	Loyiso	Gongotha
8	Viwe	[illegible]
9	[illegible]	Nohulezo
10	TW Vinjiwe	- -
11	Bongani	Ntsoini
12	Badanile	Jamda
13	Phatho	Yiko
14	Phathiswa Sinu	Sinuma
15	Zukiswa	Madyambe

40

24/05/19

Igama	ifani	ilali	signature	Sx
1. Zolile	Silekwa	Qoboshane	[illegible]	M
2.	M.K nges	nkangala		M
3.	NR Velem	Boomplaas	NRVelem	M
4.	M.D, Silekwa	QoBoshane	nomtusi	F
5.	DM Kondisa	Qoboshane	DMKondisa	M
6.	Silekwa	Qoboshane	N Silekwa	F
7.	NA. MARWANQANA	Qoboshane	[illegible]	M
8.	AM Ndzondza	Qhobosnane	[illegible]	M
9. Nobongile	N.A. Shenxane	Nkangala	N Shen	F
10. Nosizwe	Thaphuko	Boomplaas	N Taphuko	F
11. Nomsingathi	Dywili	Qoboshane	N Dywili	F
12. P.	MKHOSI	Qoboshane	[illegible]	M
13. S.S.	Mayekiso	Qoboshane	[illegible]	M
14. Cordelia	V Adlule	QoBOSHANE	N Vadlule	F
15. Duzana	Nyanga	Qoboshane	F Nyanga	M
16. S Dywili	Dywili	Qoboshane	S Dywili	M
17. NDOYISILE	NYAPHELA	BOOMPLAAS	N Nyaphela	M
18. Y. Madolo	Nelson	Helushe	N Madolo	M
19.				M
20.				M
21.				M
				18 M
				6 F

Appendix 2- Informed consent

Incazelo kunye n

Ndingu Sinetemba Xoxo, umfundi okwizinga elip
neSebe lezeNzululwazi kweZendalo, ndisuka
uphando ngomhlaba yaye ndikhangela ukubal
phando kukuqonda nokuphononga ukuba ind
edibeneyo ingasetyenziswa njani ukuphucula u
impilo zabantu ngamaxesha embalela neentleke
Ngamanzi, kunye noGEF.

Olu phando luzawkwenziwa ngokohlobo lent
abangama-30. Le ndibano izakukhokelwa yimib
ukuba ithathe ngaphaya kweeyure ezimbini. Le
kunye nezo zekhompiyutha. Ukuthatha inxaxhel

Akukho mingcipheko ozakujonganayo nayo
inxaxheba kungokokuzithandela. Umyinge w
ukhululekile unako ukuhamba nanini na. Ak
ndinomdla kumava nolwazi lwakho. Impendulo
kodwa inkcukacha zakho azizukushicilelwa zon
nalowo undongameleyo, kwaye ndizakuzifihla n
uhlala ufihlakele.

Akukho ntlawulo okanye nzuzo ezakufumaneka
inxaxheba kwakho kolu phando kubalulekil
amaqhinga okugcina umhlaba wenu ukwisimo e

Lomsebenzi womganyelwe ngu Gqir. Sukhmani
bonke base Rhodes. Ndizakubhala incwadana e

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wokupapasha iziphumo kwihlabathi jikelele ngendlela yokufundisa nabanye abantu. Apho inkcukacha zakho zivela khona, azoziphumpo azizukupapashwa. Ndizakubuya malunga nekaTshazimpuzi kunyaka ozayo ndiniphathele ingxelo.

Ukuba unemibuzo ungandibuza ngoku, okanye uqhagamshelane nam kule nombolo: 071 284 9784. Unako ukuqhakamshelana nalowo undongameleyo, uGqi. Sukhmani Mantel kulenombolo 046 603 7965.

Apho sithe sanobugwenxa khona, nceda uqhakamshelane no Rebecca ku 046 603 7005.

Imvume yokuthatha inxaxheba

Ngokuqhubeka uhlale kulendibano usixelela ukuba uyavumelana nokubhalwe ngasentla.

Ingxelo yomcwaningi / umqulu womvume wokufunda umntu

Mna obhalwe phantsi, ndifunde ngokuchanekileyo iphepha lolwazi kubachaphazelekayo ngolwimi abalisebenzisayo. Kwaye ngokusemgangathweni imigudu yenziwe ukuqinisekisa ukuba abathathi-nxaxheba abaye baqonda ukuba oku kungentla kuya kwenziwa. Ndiye ndashiya abathathi-nxaxheba ikopi yemvume enolwazi

: [illegible] Tyikitya:Umhla: na: SINETEMBA XOXO

: [illegible] u 2012/5- /ëg- Igama: Zwelethemba Madyuabe

Ingqina: Umhla: 2019/ Cf ,/-A Igama:

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Ecological Infrastructure for Dro
Security – Case Study from Macul

I am Sinetemba Xoxo, a masters degree student
University under the supervision of Dr Sukhma
am Doing research on a recent research area
ecological infrastructure is any naturally funct
ecosystem services as a way of supporting th
ongoing project with Macubeni, under the GEF
management and ecosystem rehabilitation wh

In my research study, I am Doing research on h
to land and water resource management a
livelihoods security in times of disaster using
impacts are affecting a lot of people in the M
locals can help us by sharing with sharing abou
identify areas they deem to be in urgent r
restoration. I want to uncover knowledge abo
protect catchments to maintain or restore th
Therefore, you are being invited to take part i
someone who knows and uses the natural res
will be better equipped to better adapt to clim

Procedures

If you agree to be part of this study, you will be
in a form of group discussion face-to-face wi
hours of your time. The discussion will be gui
community liaison officers. The questions ask

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their importance to your daily life, and are not intended to be sensitive or link directly to personal issues. The workshop will take place in the next two months (sometime between May 1st and May 7th).

For the workshop, a facilitator will ask a prompting question, and everyone will be given a chance to reply. If you do not wish to not answer any of the questions during the workshop, you may do so and we will proceed to the next question. The data will be anonymous i.e. will not be published in a way that reveals identity, but the processed data will be made available to the public. , notes will be taken during the workshop but your names or identity will not be included in any form. The raw notes and the mapped responses will be saved on a separate cloud folder of which access will be directly limited to myself, the GEF project manager and my supervisors, but the processed data will be made available to the general public. At the conclusion of the GEF project (estimated in 3 years' time), the collected data will be deleted, leaving behind only the processed data.

As a participant, you will also have access to the workshop results once they have been fully processed and integrated to some change analysis results. You will be acknowledged as a group in the final maps and any write up that emerges as a results of this workshop. You will not be receiving any incentives for taking part in this study as participation is only voluntary.

Additional information

This form has been approved by the Rhodes University Ethics Standard Committee. If you have any questions, please feel free to contact Sinetemba Xoxo by email at g13x2945@campus.ru.ac.za or by telephone at 046 603 7691.

Alternatively, contact the supervisor, Dr Sukhmani Mantel at 046 603 7695 or s.mantel@ru.ac.za, if you have any questions or concerns about your rights as a research participant.

Consent of subjects

Your proceeding to sit in for this workshop will be

Statement by the researcher/ person rea

I the undersigned, have accurately read out the in in a language understandable to them, and to participants that the participants understand tha

Sign: ______________________ Literate wit

Date: ______________________ *Date*: _______

Appendix 3: Ethics clearance certificate

RHODES UNIVERSITY

Human Ethics subcommittee
Rhodes University Ethical Standards Committee
PO Box 94, Grahamstown, 6140, South Africa

NHREC Registration no. REC-241114-045

27 May 2019

Sinetemba XOXO

Email: g13X2945@campus.ru.ac.za

Dear Sinetemba XOXO

Re: Participatory GIS workshop for Macubeni, Participatory GIS workshop for Macubeni (0448, May, 2019)

Principal Investigator: Dr Sukhmani Mantel

Collaborators: Mr Sinetemba Xoxo,
This letter confirms that the above research proposal has been reviewed and **APPROVED** by the Rhodes University Ethical Standards Committee (RUESC) – Human Ethics (HE) sub-committee.

Approval has been granted for 1 year. An annual progress report will be required in order to renew approval for an additional period. You will receive an email notifying when the annual report is due.

Please ensure that the ethical standards committee is notified should any substantive change(s) be made, for whatever reason, during the research process. This includes changes in investigators. Please also ensure that a brief report is submitted to the ethics committee on completion of the research. The purpose of this report is to indicate whether the research was conducted successfully, if any aspects could not be completed, or if any problems arose that the ethical standards committee should be aware of. If a thesis or dissertation arising from this research is submitted to the library's electronic theses and dissertations (ETD) repository, please notify the committee of the date of submission and/or any reference or cataloguing number allocated. Sincerely

Prof Joanna Dames
Chair: Human Ethics sub-committee, RUESC- HE

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8.3. Appendix 5.2: Survey to experts

This section provides an outline of the automated questionnaire that was sent to experts for establishing the AHP judgement matrices. The questionnaire can also be viewed online at https://forms.gle/99LkMg2EGFuj2L5CA.

1. Section A: Information for Participants

Please refer to the attached document in the email for the information sheet and informed consent.
Please click on Section A to continue to the form.

*Required

1. I have read the attached information / confirm that the above information has been explained to me in a language that I understand, and I am aware of this document's contents. I have asked all questions that I wished to ask, and these have been answered to my satisfaction. I fully understand what is expected of me during the research. I have not been pressurised in any way, and I voluntarily agree to participate in the above-mentioned project. *

   Mark only one oval.

   ⬭ I am giving my consent to participate in the research.

2. How would you rate your experience / knowledge of degradation/restoration in South Africa (select one)

   Mark only one oval.

   ⬭ Extensive
   ⬭ Moderate
   ⬭ Minimun

3. With regards to ecosystem health, how much do you agree that the attributes in the left column have an influence on selecting priority rehabilitation areas to improve the drought mitigation capacity of catchments? *

   Mark only one oval per row.

	Disagree (1)	Moderately agree (3)	Strongly agree (5)	Very strongly agree(7)	Extremely agree(9)
Degradation status	⬭	⬭	⬭	⬭	⬭
Ecosystem protection level	⬭	⬭	⬭	⬭	⬭
Cleared areas	⬭	⬭	⬭	⬭	⬭
Present ecological status	⬭	⬭	⬭	⬭	⬭
Stream order	⬭	⬭	⬭	⬭	⬭

Section B-1: Establishing a hierarchical structure for prioritising key areas for rehabilitation with respect to ecosystem health

Goal: To prioritise EI for rehabilitation for drought mitigation
Catchment health-related attributes:

i. Land degradation indicator: The proportion of land degraded over the total area, calculated over a 15-year period.

ii. Ecosystem protection level: This attribute is being used to reflect the potential of catchment management interventions to take effect to reduce the degradation threat to each ecosystem type due to ecosystem protection and management, which may lead to increased flow regulation capacity of catchments.

iii. Recently cleared areas: The previously cleared areas show tree-covered areas that were transformed into other land cover classes.

iv. Present ecological status: The assessment of the present ecological state considers a range of factors, including physio-chemical conditions, flow, and habitat quality. This indicator is used to reflect the present and future desired aquatic conditions, which is crucial for identifying the ideal rehabilitation sites on the basis of cost-to-benefit.

v. Stream order: Management interventions such as the Working for water programme give a higher priority to headwaters to avoid reinvasion downstream.

4. Using the scale of 1-9 compare the relative importance of the two attributes listed (1 denoting degradation status as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting ecosystem protection as EXTREMELY important), please rank the relative importance of the two attributes. *

   Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Degradation status	⬭	⬭	⬭	⬭	⬭	⬭	⬭	⬭	⬭	Ecosystem protection

5. Compare the two listed attributes using a scale of 1-9 (1 denoting degradation status as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting cleared areas as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Degradation status										Cleared areas

6. Compare the two listed attributes using a scale of 1-9 (1 denoting degradation status as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting present ecological status as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Degradation status										Present ecological status

7. Compare the two listed attributes using a scale of 1-9 (1 denoting degradation status as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting stream order as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Degradation status										Stream order

8. Using the scale of 1-9 (1 denoting ecosystem protection as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting recently cleared areas as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Ecosystem protection										Cleared areas

9. Compare the two listed attributes using a scale of 1-9 (1 denoting ecosystem protection as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting present ecological status as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Ecosystem protection										Present ecological status

10. Compare the two listed attributes using a scale of 1-9 (1 denoting ecosystem protection as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting stream order s as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Ecosystem protection										Stream order

11. Compare the two listed attributes using a scale of 1-9 (1 denoting recently cleared areas as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting present ecological status as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Cleared areas										Present ecological status

12. Compare the two listed attributes using a scale of 1-9 (1 denoting recently cleared areas as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting present stream order as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Cleared areas										Stream order

13. Compare the two listed attributes using a scale of 1-9 (1 denoting recently present ecological status as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting stream order as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Present ecological status										Stream order

Goal: To prioritise EI for rehabilitation for drought mitigation

Attributes relating to hydrological functioning:

i. Estimated flow reduction due to invasive alien plants: Proportion of surface runoff being taken up by alien invasive plants in the quaternary catchment.

ii. Surface water runoff: This attribute is an essential indicator for the partitioning of flows (e.g. overland, interflow etc.), which is essential because land-use change modifies the proportion of surface runoff to infiltration, which affects the responsiveness of a catchment to rainfall.

Section B-2: Establishing a hierarchical structure for prioritising key areas for rehabilitation with respect to hydrological functioning

iii. Groundwater contribution to streamflow: Although precipitation is the primary source of catchment water recharge, during dry periods, groundwater becomes the primary source of water recharge in catchments through baseflow.

iv. Wetland size: For the wetland EI category, the inclusion of wetland size is important because larger wetlands play a more important role for flow regulation than smaller wetlands.

v. Wetland type: Wetland type is an important indicator for the hydrological influence of wetlands to surface water depending on the hydrogeomorphology of wetlands.

14. With regards to the hydrological function of catchments, how much do you agree that the attributes in the left column have an influence on selecting priority rehabilitation areas to improve the drought mitigation capacity of catchments? *

Mark only one oval per row.

	Disagree (1)	Moderately agree (3)	Strongly agree (5)	Very strongly agree (7)	Extremely agree (9)
Groundwater contribution to streamflow					
Surface runoff					
Wetland size					
Flow reduction (due to IAPs)					
Wetland type					

15. Compare the two listed attributes using a scale of 1-9 (1 denoting groundwater contribution to streamflow as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting surface runoff as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Groundwater contribution to streamflow	◯	◯	◯	◯	◯	◯	◯	◯	◯	Surface runoff

16. Compare the two listed attributes using a scale of 1-9 (1 denoting groundwater contribution to streamflow as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland size as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Groundwater contribution to streamflow	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland size

17. Compare the two listed attributes using a scale of 1-9 (1 denoting groundwater contribution to streamflow as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland type as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Groundwater contribution to streamflow	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland type

18. Compare the two listed attributes using a scale of 1-9 (1 denoting groundwater contribution to streamflow as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting flow reduction due to IAPs as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Groundwater contribution to streamflow	◯	◯	◯	◯	◯	◯	◯	◯	◯	Flow reduction (due to IAPs)

19. Compare the two listed attributes using a scale of 1-9 (1 denoting surface water runoff as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting flow reduction due to IAPs as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Surface water runoff	◯	◯	◯	◯	◯	◯	◯	◯	◯	Flow reduction (due to IAPs)

20. Compare the two listed attributes using a scale of 1-9 (1 denoting surface water runoff as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland type as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Surface water runoff	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland type

21. Compare the two listed attributes using a scale of 1-9 (1 denoting surface water runoff as EXTREMELY important, 5 denoting EQUAL importance and 9 wetland size as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Surface water runoff	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland size

22. Compare the two listed attributes using a scale of 1-9 (1 denoting flow reduction due to AIPs as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland size as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Flow reduction due to IAPs	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland size

23. Compare the two listed attributes using a scale of 1-9 (1 denoting flow reduction due to AIPs as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland type as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Flow reduction due to IAPs	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland type

24. Compare the two listed attributes using a scale of 1-9 (1 denoting wetland size as EXTREMELY important, 5 denoting EQUAL importance and 9 denoting wetland type as EXTREMELY important), please rank the relative importance of the two attributes. *

Mark only one oval.

	1	2	3	4	5	6	7	8	9	
Wetland size	◯	◯	◯	◯	◯	◯	◯	◯	◯	Wetland type

8.4. Appendix 5.3: Spatial outcomes of normalised weights

This subsection contains illustrations for normalised scores for the focal EI categories in the Cacadu catchment (Figures A1- A3). The normalised indicator (Pij) of the focal EI areas has a 2 to 4 range (Figure A1). Figure A2 demonstrate the normalised scores corresponding to the hydrological function criterion for the focal EI categories in the focal catchment. The catchment has a low yielding capacity due to low rainfall (Figure A2).

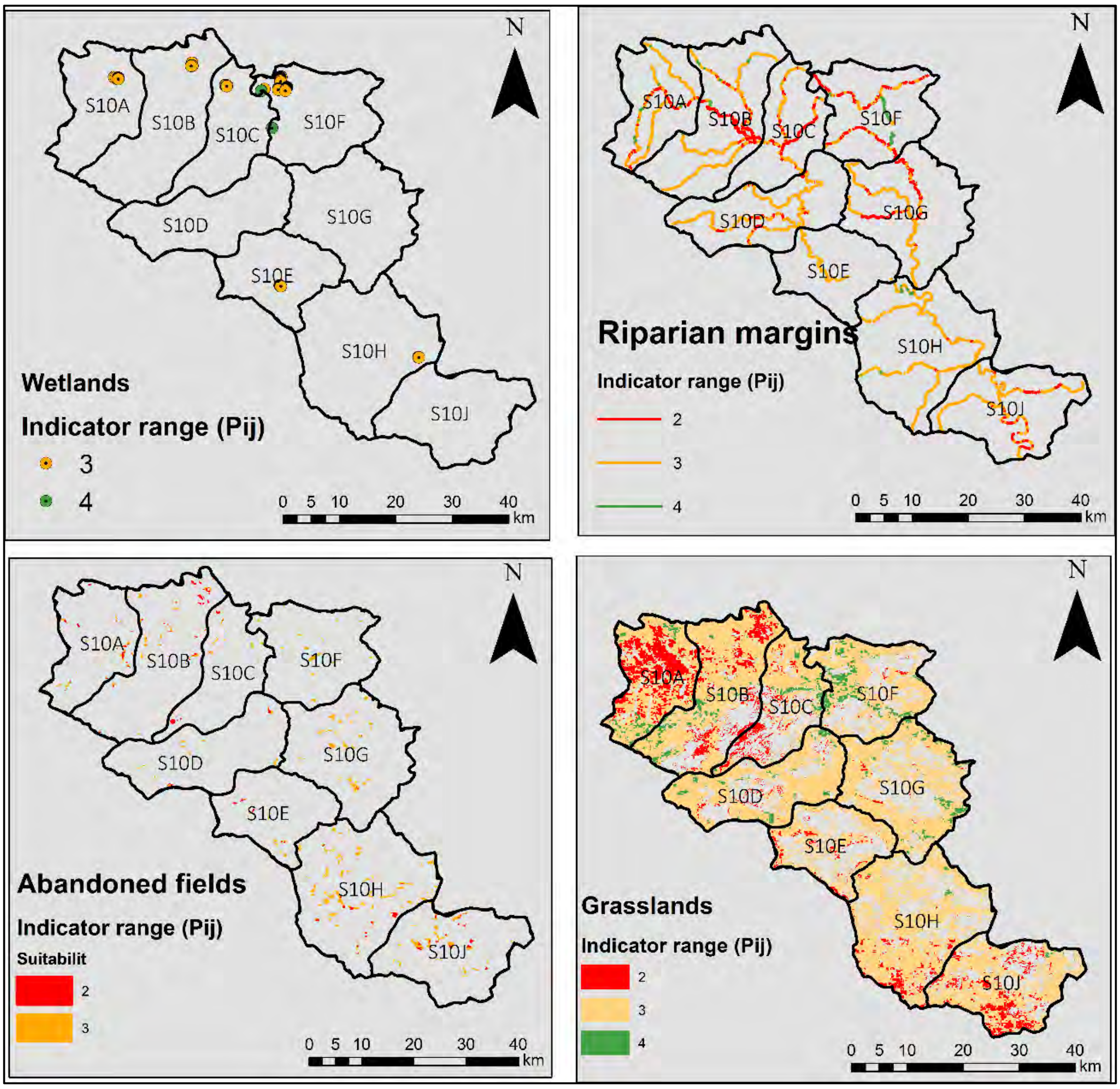

Figure A1: The normalised maps for focal EI with regards to the ecosystem health criterion in the Cacadu catchment.

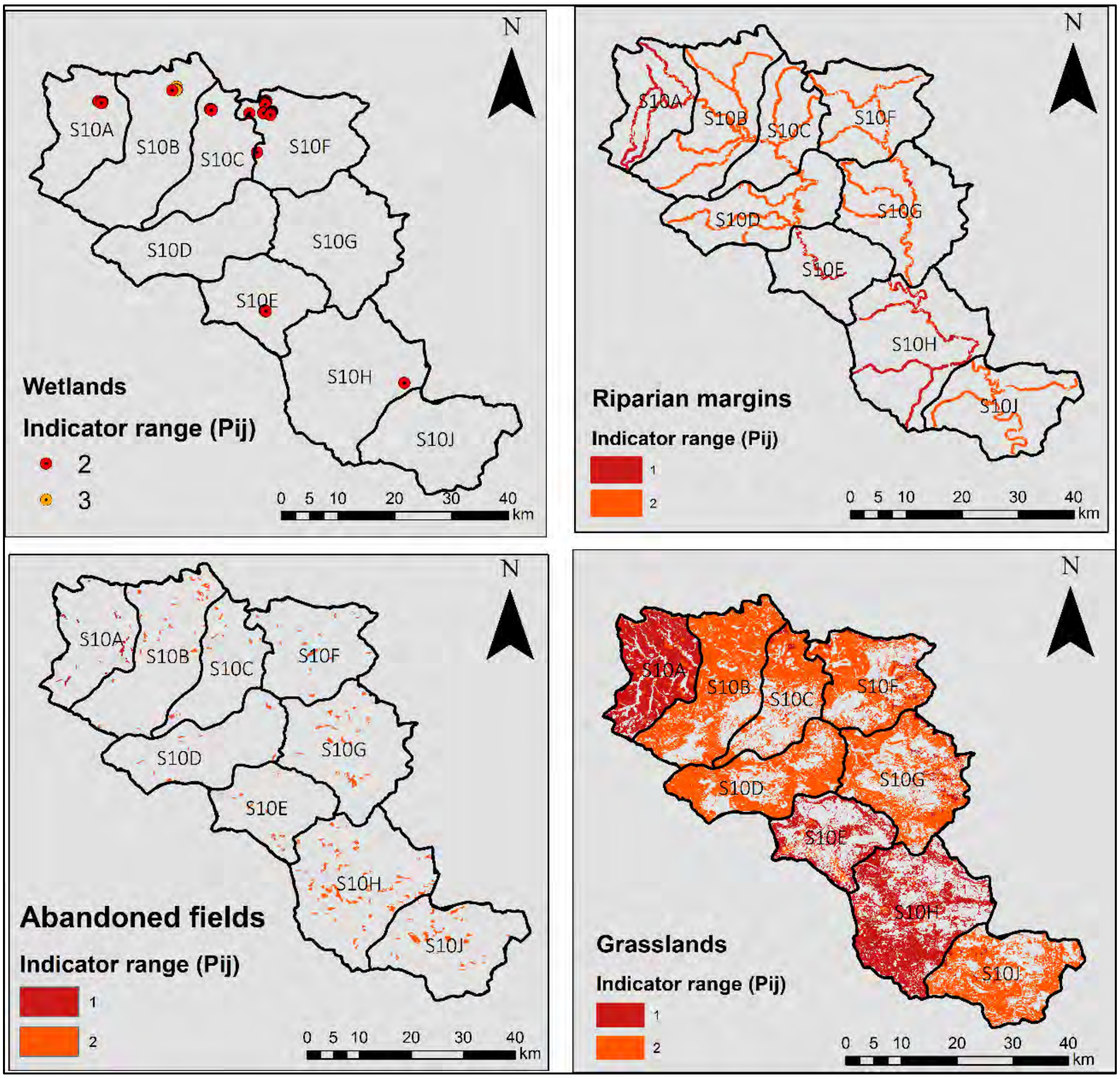

Figure A2: The normalised maps for grasslands with regards to the hydrologic functionality criterion.

8.3. Appendix 4.1

Table A3: Pitman Model parameters and their description.

Parameters	Units	Description
Surface water parameters		
RDF		Rainfall distribution factor. Controls the distribution of total monthly rainfall over four model iterations
AI	fraction	Impervious fraction of sub-basin
PI1s	mm	Summer Interception storage for vegetation type 1
PI1w	mm	Winter Interception storage for vegetation type 1
PI2s	mm	Summer Interception storage for vegetation type 2
PI2w	mm	Winter Interception storage for vegetation type 2
AFOR	%	% area of sub-basin under vegetation type 2
FF		Ratio of potential evaporation rate for Veg2 relative to Veg1
PEVAP	mm	Annual basin potential evaporation
ZMINs	mm $month^{-1}$	Summer minimum basin absorption rate
ZMINw	mm $month^{-1}$	Winter minimum basin absorption rate
ZAVE	mm $month^{-1}$	Mode of distribution of absorption rates
ZMAX	mm $month^{-1}$	Maximum basin absorption rate
ST	mm	Maximum moisture storage capacity
SL	mm	Soil moisture below which there is no recharge
POW		Power of the moisture storage runoff equation
FT	mm $month^{-1}$	Runoff from moisture storage at full capacity
R		Evaporation-moisture storage relationship parameter
TL	months	Lag of surface runoff
Groundwater parameters		
GW	mm $month^{-1}$	Maximum recharge depth at maximum moisture capacity
TLGMax	mm	Maximum channel loss
GPOW		Power of the moisture storage recharge equation
DD	km km^{-2}	Effective drainage density
T	m^2 day^{-1}	Transmissivity
S		Storativity
RG		Regional groundwater drainage slope
Rest RWL	m below surface	Aquifer depth
RSF	% slope width	Riparian Strip Factor

Table A4: Pitman Model water-use and wetland sub-model parameters

Parameters	Units	Description
Surface water use parameters		
AIRR	km^2	Irrigation area
IWR	fraction	Irrigation water return flow fraction
EFFECT	fraction	Effective rainfall fraction
RUSE	Ml/year	Non-irrigation demand from the river
Groundwater use parameters		
GWA (Upper slopes)	Ml $year^{-1}$	Groundwater abstraction far from the channel
GWA (Lower slopes)	Ml $year^{-1}$	Groundwater abstraction near to the channel
Small farm dam parameters		
MDAM	Ml	Small dam storage capacity
DAREA	%	% of sub-basin above dams
A		Parameter in non-linear dam area-volume relationship
B		As above
IRRIG	km^2	Irrigation area from small dams
Large reservoir sub-model parameters		
Reservoir Capacity	MCM	Reservoir storage capacity
Dead Storage	% Capacity	Dead storage of the reservoir
Initial Storage	% Capacity	Reservoir magnitude at the beginning of the simulation period
A in Area(m^2) = A* Volume(m^3)^B		
B in Area(m^2) = A * Volume(m^3)^B		
Reserve level 1-5	% Capacity	5 levels of operating rules used to reduce abstraction of reduced storage.
Annual Abstraction	MCM	Demand from the reservoir
Annual Compensation Flow	MCM	Downstream compensation flow released into the river
Wetland and lake sub-model parameters		
MaxWA	Km2	Maximum wetland area.
RWV	$m^3 * 10^6$	Residual wetland storage volume below which there are no return flows to the river channel.
IWV	$m^3 * 10^6$	Initial wetland storage volume at the start of the simulation.
AVC	m^{-1}	Constant in the WA = AVC * WV^{AVP} relationship, where WA (m^2) and WV (m^3) are the current wetland area (limited to MaxWA) and volume, respectively.
AVP		Power in the WA = AVC * WV^{AVP} relationship.
QCap	$m^3 * 10^6$	Channel capacity below which there is no spill from the channel to the wetland.
QSF		Channel spill factor in SPILL = QSF * (Q – QCAP), where Q is the upstream flow and SPILL is the volume added to wetland storage.
RFC		Return flow constant in the RFF = RFC * $(WV / RWV)^{RFP}$ relationship. RFF is a fraction limited to a maximum of 0.95 and then adjusted when Q is greater than QCap (RFF = RFF * QCap / Q). The return flow volume is calculated from RFLOW = RFF * (WV – RWV).
RFP		Return flow power in the RFF = RFC * $(WV / RWV)^{RFP}$ relationship.
EVAP	mm	Annual evaporation from the wetland (distributed into monthly values using a table of calendar month percentages).
ABS	$m^3 * 10^6$	Annual water abstractions from the wetland (distributed into monthly values using a table of calendar month percentages).

Table A5: Long-term mean monthly volume (mm) for the main streamflow generating parameters in the White Kei catchment.

Catchment	Scenario	Precipitation	Evapotranspiration	Surface runoff	Baseflow	Recharge
S10A	Natural	44.21	27.42	1.59	0.24	0.25
	1990		40.87	2.98	0.19	0.2
	2018		25.96	3.2	0.21	0.21
S10B	Natural	44.10	29.02	2.14	0.43	0.29
	1990		43.66	3.92	0.33	0.22
	2018		27.88	3.87	0.33	0.22
S10C	Natural	45.56	28.01	1.77	0.36	0.26
	1990		42.96	2.9	0.25	0.18
	2018		27.69	2.99	0.26	0.19
S10D	Natural	49.13	29.79	2.25	0.52	0.3
	1990		44.54	4.02	0.38	0.22
	2018		28.78	3.96	0.38	0.21
S10E	Natural	48.71	29.39	2.18	0.36	0.31
	1990		43.83	5.29	0.23	0.19
	2018		25	8.46	0.18	0.16
S10F	Natural	48.74	29.4	2.24	0.45	0.3
	1990		45.96	4.33	0.36	0.24
	2018		27.77	5	0.29	0.19
S10G	Natural	51.01	30.21	2.57	0.47	0.33
	1990		42.85	4.5	0.34	0.24
	2018		28.16	5.76	0.3	0.21
S10H	Natural	47.38	32.18	2.11	0.25	0.6
	1990		42.85	3.93	0.18	0.43
	2018		29.95	4.1	0.17	0.39
S10J	Natural	47.22	31.82	2.09	0.41	0.62
	1990		42.45	4	0.31	0.46
	2018		29.29	4.25	0.29	0.43
S20A	Natural	52.07	30.89	1.72	0.52	0.49
	1990		48.58	2.73	0.4	0.38
	2018		30.12	3.29	0.38	0.35
S20B	Natural	51.77	32.54	1.81	0.63	1.25
	1990		47.67	2.65	0.5	0.97
	2018		32.12	2.61	0.57	1.11
S20C	Natural	52.48	32.82	1.91	0.67	1.28
	1990		48.35	2.68	0.51	0.96
	2018		32.77	2.87	0.45	0.85
S20D	Natural	56.33	31.67	2.29	0.76	1.38
	1990		51.5	1.79	0.98	3.33
	2018		31.77	3.59	0.52	0.94