An assessment of land degradation and alien plants invasion in the Waterberg Biosphere Reserve, Limpopo Province

Lucia Macheba Mokubedi

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University of Cape Town

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Supervisor:

Professor Michael Meadows

Co-Supervisors:

Ms. Zuziwe Jonas

Dr. Sebataolo Rahlao

DECLARATION

I know the meaning of plagiarism, and I declare that all the work in this dissertation is my own.

Lucia Macheba Mokubedi



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ABSTRACT

Land degradation and invasion by alien plants are interconnecting issues that span both natural and socio-economic systems. As a result, addressing the complexity and severity of these pressures requires the use of a multidisciplinary approach. This study aims to assess land degradation and invasive alien plants distribution and their impacts on the ecosystem services within the WBR. An area outside the reserve was designated as a control to compare how degradation processes have evolved over the last few decades. Landsat images captured in 1989, 2000, 2013, and 2017 were used to analyse and map the land use/cover change. The images were classified into the following classes: water bodies, woody vegetation, grasslands, cultivated fields, bare lands, and settlements. Change detection performed between the years showed that the area outside the WBR has been losing grasslands and woody vegetation and gained more bare lands, cultivated fields, and settlements. On the contrary, the area within the WBR lost a significant amount of bare land and gained grasslands, woody vegetation, and a small amount of cultivated fields and settlements. The roadside survey was used to assess the distribution of invasive alien plants along the main roads in the selected areas. A single observer drive-by method was used to record the alien plants present at an interval of 5 km. To understand how the communities are being influenced by land degradation and alien plants invasion, focus group discussions were held to understand how their livelihoods are affected. Four important indicators, namely vegetation loss, land-use intensity, alien plants distribution, and population pressure, were overlaid using ArcMap 10.1 to determine the land degradation severity in both study areas. The results revealed that land degradation was severe outside the WBR, where most human activities occur, regardless of population pressure and alien plant distribution. These issues were intensifying with profound, severe implications for people's livelihoods, as land is the leading resource from which communities, both within and outside the WBR, eke their living. The WBR experienced land recovery following the proclamation of the area as a formal biosphere reserve by UNESCO, which has constrained the harmful impact of human activities. Reversing degradation and invasion by alien plant species requires sustainable land use and planning based on the current land-use practices and landscape attributes. Therefore, integrating local and scientific knowledge is helpful in the planning of sustainable and efficient land management.

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CHAPTER 1. INTRODUCTION

1.1 Introduction

1.1.1 Background to the study

Land degradation, particularly in drylands, is a major environmental issue due to its diverse impacts on environmental quality and food security (Utuk and Daniel, 2015), given that approximately 40% of the earth's surface is covered by drylands and occupied by over two billion people (Gnanavelrajah, 2009). In many drylands, unsustainable use of resources combined with poor socio-economic and challenging climatic conditions have resulted in the deterioration of ecosystems, making it impossible to sustain the increasing population growth (Ravi et al., 2010). When ecosystems lose their functionality and productivity, desertification, the most extreme form of land degradation (Bloem and Dunbar, 2016), may result.

In Southern Africa, land degradation is perceived as a severe environmental problem (Hoffman and Todd, 2000; Scholes and Biggs, 2004). In South Africa, the South African National Report on Land Degradation (NRLD) focused mainly on land degradation in the communal areas, previously known as *Bantustans*, or "homelands" (Hoffman and Todd, 2000; Hoffman and Ashwell, 2001) (Figure 1. 1). In these communal lands, high population densities, excessive harvesting of wood, and overgrazing constitute a significant problem (Hoffman and Todd, 2000) and threaten the livelihoods of the rural communities (Shackleton et al., 2001). These problems may have been fuelled by the involuntary resettlement of people in the previously abandoned areas during the apartheid era (Fox and Rowntree, 2001). Over three million people were forcefully moved from their settlement under the apartheid regime between 1960 and 1985 to these inhospitable areas (Hoffman et al., 1999), possibly resulting in degradation as people sought resources for survival. Hoffman and Todd (2000) and Scholes and Biggs (2004) highlight poverty, unemployment, and lack of governance structures as the primary causes of land degradation. Such problems are globally recognized, and many countries, including South Africa, became signatories to the United Nations Convention to Combat Desertification (UNCCD). This lawfully binding agreement focuses on the association between environmental protection, sustainable development, and land management (UNEP, 1994).

Bai et al. (2008, p.223) define land degradation as the "long-term loss of ecosystem function and productivity caused by disturbances from which land cannot recover unaided." This definition was adopted for this study. Degraded lands are often susceptible to alien plants invasion (Asmelash et al., 2016). Disturbances arising from logging, grazing and trampling, water extraction, and land-use change can trigger the proliferation of alien plant species, many of which exploit the opportunities provided by anthropogenic activities (Richardson et al., 2007). A thorough description of the land, degradation is further given in chapter 2.

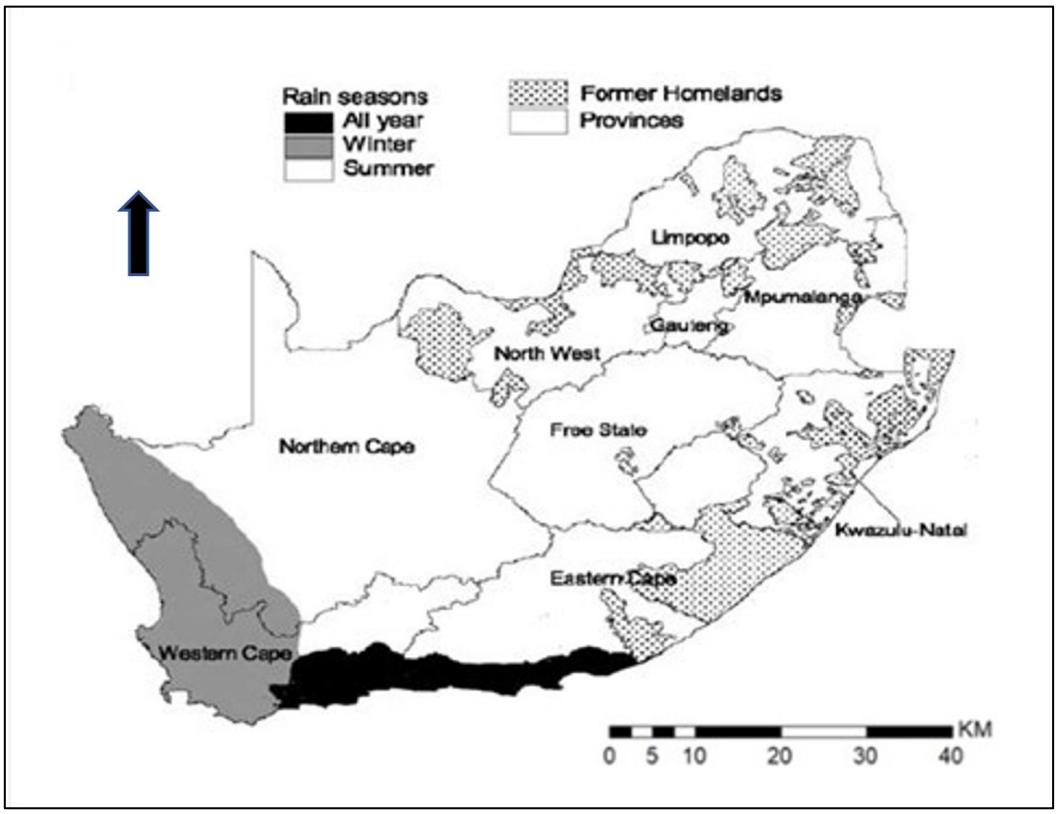


Figure 1.1. The location of former homelands of South Africa (Wessels et al., 2007).

Biological invasions have taken a toll on ecosystems and biodiversity. The intensity of this threat is increasing globally, and more evidence has been gathered based on global trade and patterns (Hulme, 2009). Biological invasion is the process whereby an introduced species dominates in a new environment and transforms the biodiversity and ecosystem attributes of the invaded area. Alien invasive species are perceived as a significant threat alongside habitat destruction and landscape fragmentation (Walker and Steffen, 1997). They threaten ecosystems that deliver services such as water purification, waste decomposition, and

nutrient cycling, soil development, and flood control – all of which are highly beneficial processes to humans (Pejchar and Mooney, 2009). Invasive alien species are also a major issue for the economy globally. They may impose a substantial financial burden through loss in agricultural productivity (van Wilgen et al., 2001) and costs associated with eradication or clearing. In the United States, the environmental damage resulting from invasive alien species has been estimated at more than the U.S $125 billion per year (Mosissa, 2019). It may have a significant impact on agriculture (Paini et al., 2016).

In many parts of South Africa, invasive alien plants pose severe threats to the rich biodiversity and delivery of ecosystem services (Brooks et al., 2004; Le Maitre et al., 2000; van Wilgen et al., 2008; van Wilgen et al., 2001;). Under South African legislation, the invasive alien species are regulated under the National Environmental Management: Biodiversity Act 2004 (Act No. 10 of 2004) (NEMBA) and have been legislated into three different categories. According to the NEMBA regulations, category 1a includes invader plants that need to be removed and destroyed before establishing. Category 1b is invaders that have been established and should be controlled. Category 2 lists invader plants that should be controlled and for which a permit may be required; category 3 is species that do not need to be removed but must not be propagated or sold. Although alien species in South Africa are formally regulated, controlling and managing them remains one of the biggest challenges conservation biologists faces. Therefore, it is important to assess the nature and scale of such impacts in a range of geographical contexts. There is a need to monitor and control the alien and invasive species using techniques and methods. There is a large body of literature highlighting the use of remote sensing and GIS techniques in assessing the occurrence, spread, and impacts of invasive alien species, as well as their potential future range (Bradley and Mustard, 2006; Hestir et al., 2008; Joshi et al., 2004; van den Berg et al., 2013). Other literature highlights successful methods of eradicating and controlling alien and invasive species (Bradley and Mustard, 2006 (Booy et al., 2020; van Wilgen et al., 2018; van Wilgen et al., 2020; van Wilgen et al., 2020).

Ecosystem-based adaptation (EbA) is an internationally recognized approach initiated to assist people in coping with the impacts associated with the decline in ecosystem services, such as those caused by land degradation and invasive alien species. EbA is a sustainable and cost-effective approach to improve communities' adaptive capacity (Roberts et al., 2012). As

described in the Convention on Biological Diversity (CBD), its primary objective is to maintain the quality of ecosystems and improve biodiversity and human livelihoods by ensuring that people adapt to environmental changes (Convention on Biological Diversity, 2009). Such adaptation strategies can help in improving ecosystem resilience (Campbell et al., 2009). Due to the close association between people and ecological systems in addressing climate change impacts (Roberts et al., 2012). Various adaptation policies have been initiated worldwide. In South Africa, adaptation policies include Working for Wetlands, Working for Water, the National Water Act, and the Integrated Development Plan (Macdonald, 2004). These ecosystem adaptation measures benefit the communities and are arguably more sustainable in the long run than more technical solutions (Campbell et al., 2009). For example, controlling or avoiding the introduction of alien invasive plants increases the quantity of water available for human consumption.

In South Africa, best practice case studies on the implementation of EbA measures are beginning to emerge. An example is the Working for Water Programme, which successfully restored water supply in alien-infested catchments (Macdonald, 2004). Hobbs (2004) refers to this initiative as an example of a highly successful integrated land management programme to address biodiversity, water resources and foster socio-economic development. This programme helps in managing or protecting the natural environment while at the same time providing job opportunities. The establishment of specific areas as Biosphere Reserves also aims to serve such purposes, although they are susceptible to pressures such as land degradation and alien plants infestation (Manzo-Delgado et al., 2014; Sahu and Singh, 2008).

1.1.2 The Biosphere Reserve concept

The concept “Biosphere Reserve” was initiated in the 1970s. It formed an essential part of UNESCO’s Man and the Biosphere (MAB) programme (Batisse, 1982), an intergovernmental programme that uses scientific research to foster harmony between humans and their natural environment (Sahler, 2007). Biosphere Reserves, which are integral to MAB, are located in either terrestrial or marine ecosystems or both (UNESCO, 1996). They help address essential questions on balancing biodiversity conservation with social, economic, and cultural values (UNESCO, 1996). One objective of the MAB programme is to ensure that a sustainable balance is achieved between biodiversity conservation and socio-economic development in the

Biosphere Reserves. Thus, Biosphere Reserves are regarded as tools for implementing the MAB programme. Effective Biosphere Reserves draw on the knowledge and experience of scientists (both natural and social), conservationists, development organizations, management authorities, policymakers, and indigenous peoples (UNESCO, 1996). This varied group of actors brings a wealth of expertise, experience, and scientific research to bear on the link between biodiversity conservation and socio-economic development for human benefit. Although the Biosphere Reserve model is widely adopted and regarded as successful, some difficulties are still faced with the implementation (Neumann, 1997; Wells and McShane, 2004; Blom et al., 2010).

Globally, 58 Biosphere Reserves were initially designated (Coetzer et al., 2014). This number increased in 2012 to 610 Biosphere Reserves in 117 countries (Coetzer et al., 2014). Currently, UNESCO records 727 Biosphere Reserves globally in 131 countries, including 22 transboundary sites (https://en.unesco.org/biosphere). Figure 1.2 illustrates the spatial distribution of Biosphere Reserves globally. By 2014, South Africa had six Biosphere Reserves, with three located in the Western Cape, two in Limpopo, and one in Mpumalanga, extending into Limpopo Province (Figure 1.3). In 2015, UNESCO recognized two more Biosphere Reserves, namely Gouritz, which straddles the Eastern and Western Cape, and the Magaliesberg, situated in Gauteng and North-West provinces. Although Biosphere Reserves play a significant role in biodiversity conservation, they continue to be threatened mainly due to illegal human activities. In the northern part of South Africa, illegal activities such as rhino poaching, persecution of wild dogs, cycad theft, and the killing of pangolins for their scales are major problems encountered (land use.waterbergbiosphere.org/za).

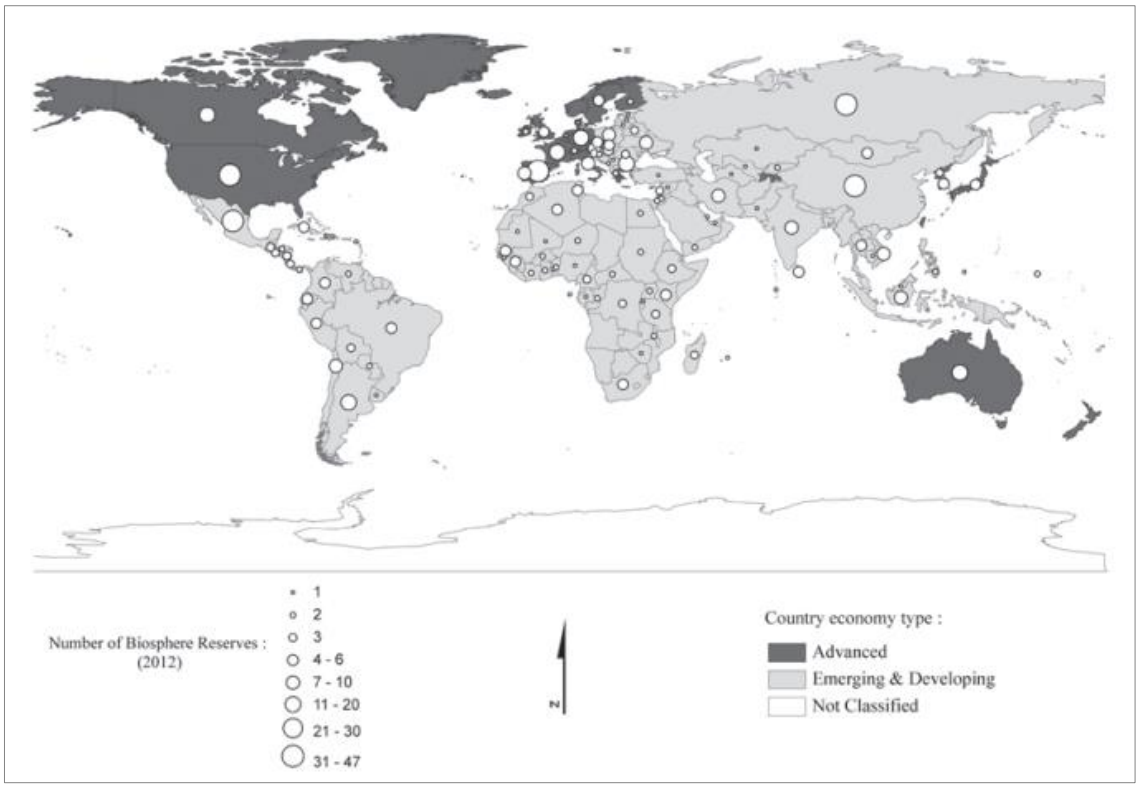


Figure 1.2. 2012 global spatial distribution of Biosphere Reserves by economic type of the countries (Coetzer et al., 2014).

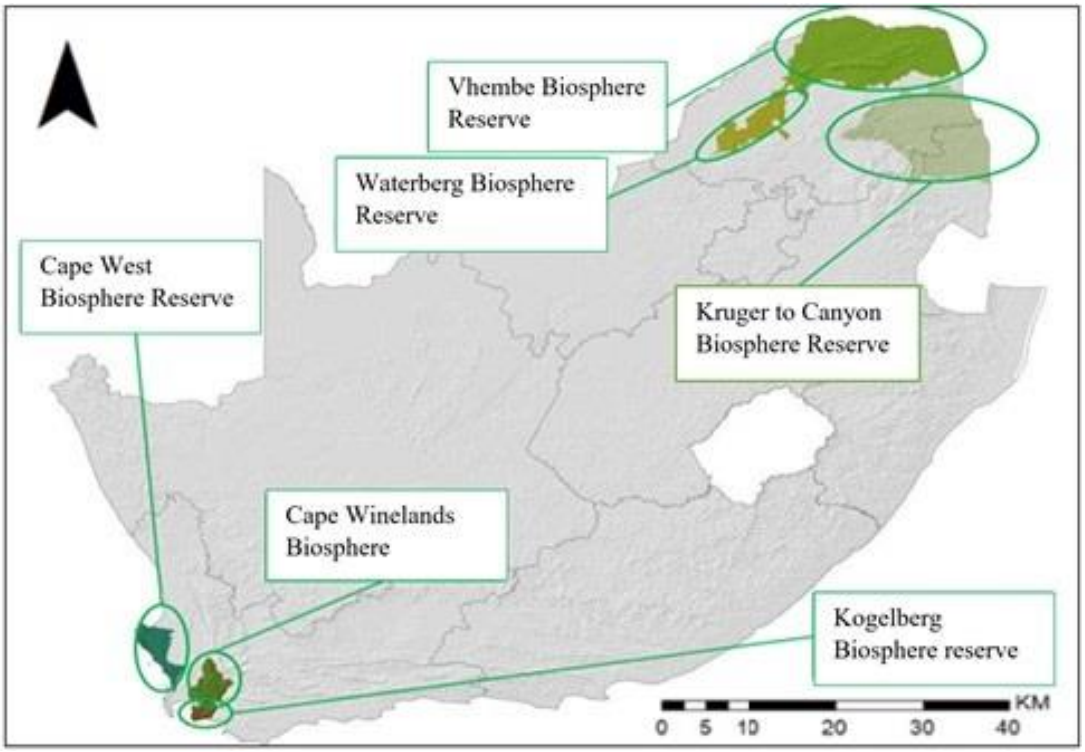


Figure 1.3. The spatial distribution of Biosphere Reserves in South Africa (Map produced by Tendamudzimu Munyai from SANBI).

1.1.3 Rationale

Environmental pressures on Biosphere Reserves are expanding globally, putting the protected areas in jeopardy. Biodiversity and ecosystem services suffer as a result of this. Invasive alien plants contribute to land degradation by producing huge, unproductive monocultures, and alien plants are more prone to colonize damaged areas due to unsustainable land-use practices. Land degradation and alien plants invasion are among the most prominent pressures on ecosystem services (e.g., water resources) that maintain livelihoods in rural areas in particular (Linders et al., 2020). These pressures lower the ability of the affected communities to adapt to the effects of climate change and other challenges. This study focuses on such issues concerning the Waterberg Biosphere Reserve (WBR) in Limpopo Province.

Furthermore, the WBR is within one of the most economically disadvantaged provinces of South Africa. As such, economic growth and poverty alleviation is one of the province's main objectives. Such reserves aim to address essential questions such as balancing biodiversity conservation with social, economic, and cultural values. It is, therefore, important to assess the level of land degradation and biological invasions and their effects on the related communities within the WBR. Understanding the causes, scale, and impacts of land degradation and biological invasions will assist in raising awareness of their effects and foster the development of effective mitigation and adaptation interventions.

Research in several regions globally has been undertaken on the impacts of land degradation and alien plant invasion on the natural environment and human well-being individually (Gisladottir and Stocking, 2005; Kolar and Lodge, 2001; Sala et al., 2000, Richardson et al., 2007). However, literature on the integrated impacts of these two significant pressures considered together is limited. Due to the nature of land degradation and alien plant invasion, there may be cumulative or even multiplier impacts when they occur together. For example, land degradation could result in the loss of soil nutrients, thereby creating conditions favorable for invasion by alien plant species. On the other hand, alien plant species invading an area may reduce biodiversity and make the area less productive and more vulnerable. It is essential to assess how natural environments respond to anthropogenic activities and subsequently affect human well-being.

In the WBR, commercial and subsistence crop cultivation are still conducted and have transformed the natural landscape to a greater or lesser degree (de Klerk, 2003). Commercial cultivation used to be practiced extensively in the region. Still, unreliable rainfall patterns and poor nutrient soil have declined, resulting in more abandoned farming lands (de Klerk, 2003). Nonetheless, commercial irrigated cultivation continues alongside the river courses, with tobacco, cotton, sunflower, sorghum, and maize being the main cultivated crops (de Klerk, 2003). Other activities, such as livestock grazing, have also resulted in the degradation of vegetation cover (de Klerk, 2003).

Furthermore, sand mining and stone digging for commercial purposes place further strains on the area's biodiversity. Figure 1.4 illustrates the impacts that these activities have on the environment within the WBR. Alien plant species invade the northern part of the Waterberg, which further threatens biodiversity and ecosystems (Henderson, 2013). Some of these alien invasive plants were intentionally introduced (Henderson, 2013), such as cultivated and visible along the streams and rivers and within the farms. According to Henderson (2013), more than 100 alien plant species have been naturalized within the WBR, primarily ornamental. The lantana (*Lantana camara*), jacaranda (*Jacaranda mimosifolia*), queen of the night cactus (*Cereus jamacaru*), and syringa (*Melia azedarach*) are the most invasive alien species discovered in this Biosphere Reserve (Henderson, 2013)



Figure 1.4. Areas within the WBR where sand mining and stone digging activities occur.

For the study to document the impacts of land degradation and alien plants invasion, the individual and combined effects of both pressures on ecosystems and livelihoods are to be assessed. Assessing the effects, particularly in rural settings, is crucial for understanding the magnitude of the problem and advising on ways towards preserving the remaining natural resources (i.e., water, land) for future generations. For this research, a comparative study will also be used to assess the difference in land use/ land cover changes that resulted in land degradation and the alien plants' distribution outside the WBR.

1.1.4 Aim and Objectives

This study aims to assess land degradation and invasive alien plants distribution and their impacts on the ecosystem services within the WBR. Specific objectives of this study are to:

- Document the nature, scale, distribution, and temporal trends in land degradation using remote sensing techniques
- Assess and document the distribution (occurrence and abundance) and types of selected alien plant species in the WBR. This focuses on selected main roads and land use/land cover type.
- To investigate the local knowledge on land degradation and alien plants invasion impacts and how the communities' well-being is affected.
- To explore the integration of both scientific knowledge and ecosystem-based adaptation approaches for land management and human well-being.

1.2 Study area description

1.2.1 Geographical location and description

The WBR is one of eight Biosphere Reserves in South Africa recognized by UNESCO. It is located in the Waterberg District of Limpopo province and lies between latitude $27^{0}30'$S and $28^{0}40'$S and longitude $23^{0}10'$E and $24^{0}40'$E (Figure 1.5). The reserve is jointly owned by the Provincial government, the Local and District Municipality, and the local communities in the Waterberg District Municipality. The reserve transverses six local municipalities: Lephalale, Mogalakwena, Modimolle, Thabazimbi, and Mookgopong. The WBR was the first to be established in the northern part of South Africa and included the Waterberg Mountain range,

from which the name originates. The reserve was designated in 2001 by UNESCO, having been formalized in 1996 (Pool-Stanvliet, 2013).

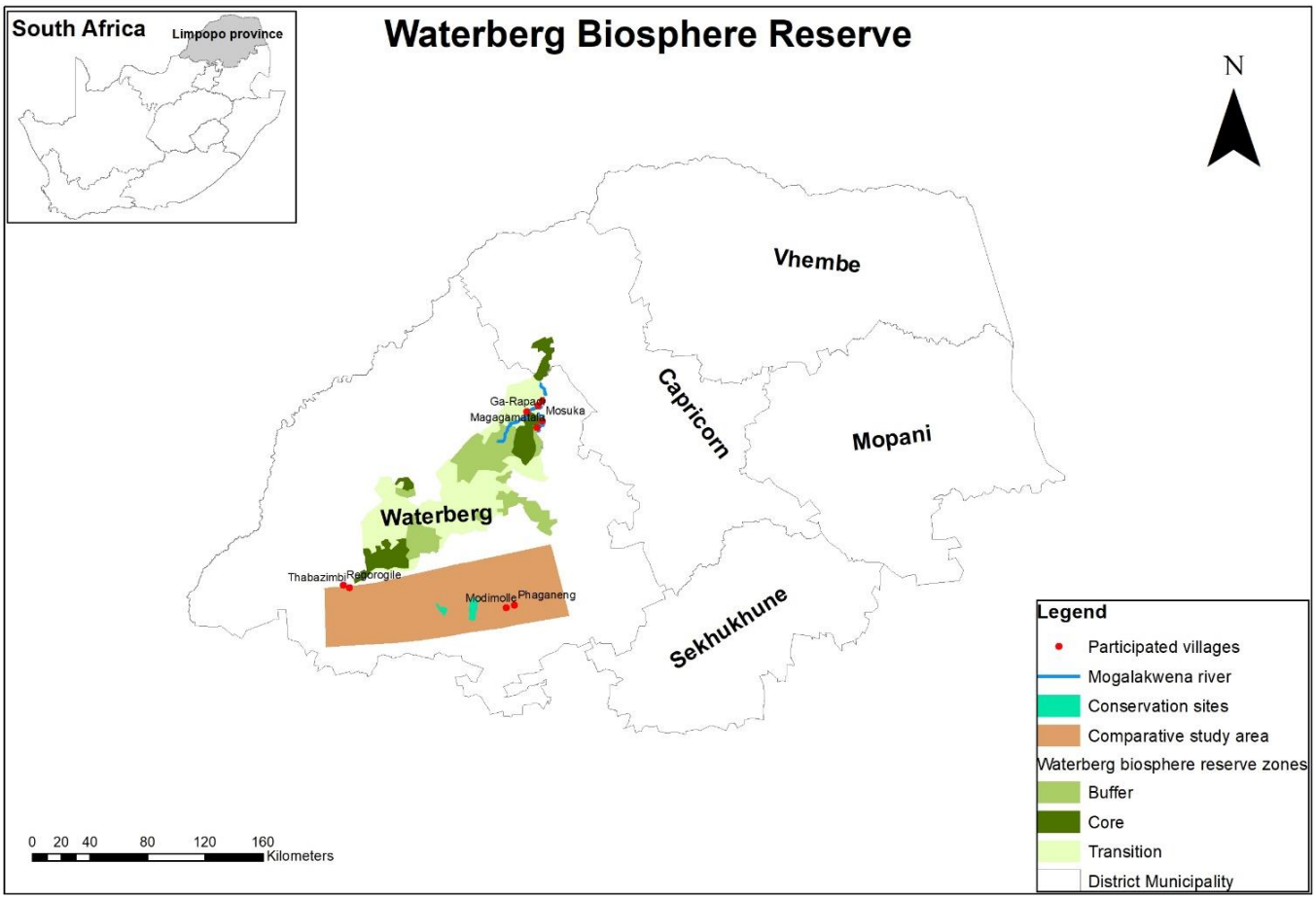


Figure 1.5. The geographical location of study areas.

The reserve is approximately 665 000 hectares in extent, comprising a core area, a buffer zone, and a transition zone (Baber et al., 2003). The core areas within the reserve include the Marakele National Park, Wonderkop, Mokolo Dam, Masebe, and Moepel Nature Reserve. These core areas were classified based on their legal status and their long-term role as protected areas, including the participation of stakeholders (de Klerk, 2003). The protected areas within the Biosphere Reserve are confined to biodiversity and the historical/native landscape of the area, such as bushman paintings in the caves. The buffer zones are areas adjacent to the core zones. However, all buffer areas don't need to be surrounding the core zone. Still, the activities within these zones should be compatible with the conservation objectives of the core areas (UNESCO, 1996). Transition zones, on the other hand, are areas

that contain a diversity of sustainable activities. Figure 1.6 illustrates the zonation of a Biosphere Reserve.

Most activities occurring within the core area of the WBR are environmentally friendly. These include eco-tourism, game farming, controlled hunting, and other conservation projects such as environmental education and research (Baber et al., 2003). Although tourism is seen as an economic pillar of the area, other economic activities such as sand mining and stone diggings are occurring, particularly at the reserve's periphery. There is a need to ensure that ecotourism destinations are well planned and protected within the biosphere.

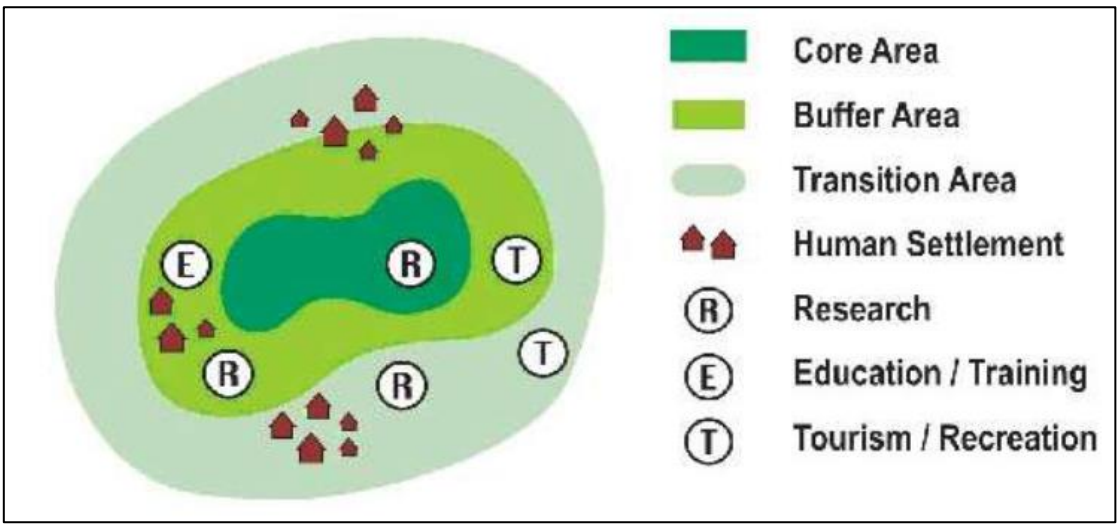


Figure 1.6. Example of activities permitted in the Biosphere Reserves zones. (Structure model obtained from https://unacademy.com/lesson/biodiversity-conservation-part-2/J0S740BI).

The comparative (control) study area is located just south of the WBR. In terms of size, the area covers approximately the same hectares as the WBR. The control area includes Thabazimbi, Modimolle and Mookgopong. Northam borders it in the south and Bela-Bela in the south-west and south-east, respectively (Figure 1.5). The area was chosen because it has the same geographical features (e.g., topography, vegetation) as the WBR but falls outside the designated reserve area. However, two recognized conservation areas do occur within the comparative area, *viz*. Bateleur Nature Reserve and the Phala-Phala Wildlife Reserve. However, in terms of human settlement, the control area is comprised of a more significant number of formal and informal settlements in larger villages and towns. In contrast, the WBR human settlements are dispersed mainly in rural communities. The western part of the

control area is primarily dominated by commercial or industrial mining. The area produces minerals such as Platinum, Iron Ore, and Andalusite (Waterberg District Municipality, 2010).

1.2.2 Demography

The WBR and control area falls within the Waterberg District, occupied by approximately 600 000 with black Africans comprising 90% of the population, 9% whites, and 1% of other ethnic groups (Waterberg District Municipality, 2010). The population within the WBR itself is sparsely populated, although somewhat higher in the eastern part of the reserve (Baber et al., 2013). Outside, the WBR, on the other hand, is densely populated due to a large number of informal settlements and employment opportunities, largely from the commercial mining industry (See Chapter 3 on the population density of both areas). The languages spoken in the area are Sepedi, English, and Afrikaans, with Sepedi being the primary language. The population within and outside the WBR is characterized by high poverty rates and low levels of economic development. Overexploitation of local woody vegetation for energy supply is expected due to the poor living conditions, in general, and lack of access to electricity (de Klerk, 2001). Tourism is an additional source of income. However, people also practice raising cattle, crop production, and increasingly switching to game farming for ecotourism (de Klerk, 2001).

1.2.3 Vegetation

The WBR and the control study area both fall within the savanna biome of southern Africa. The savanna biome is home to various large herbivores, which, together with fire, play a crucial role in promoting ecosystem processes and maintaining plant species diversity (Henning, 2002). Be that as it may, only 36 percent of the savanna biome falls within the formally protected areas in South Africa (Hoveka et al., 2020). This is the case with the WBR, whereby the savanna vegetation is not lawfully protected within the proclaimed nature reserve (de Klerk, 2003). Fifteen different veld types are represented in the study areas, with Sour Bushveld and Mixed Bushveld being the most common savanna vegetation types (de Klerk, 2003; SANBI 2018 vegetation map). The mixed Bushveld vegetation type provides better grazing for various large herbivores within the reserve (de Klerk, 2003). In the northern part, Arid Sweet Bushveld is found in small patches, whereas the southern and central part of the reserve occupies the smaller areas of North-Eastern Mountain Sourveld, the Limpopo's

most threatened veld type (de Klerk, 2003). The Sourish Mixed Bushveld occurs partly in the further south of the reserve. The Waterberg plateau, valleys, and cliffs are characterized by diverse habitat types (Henning, 2002), with the vegetation at the valleys varying from riparian vegetation to Thornveld. Wildlife found in the study areas are rhinoceros, elephant, leopard, lions and buffalo, and other games. Over 400 bird species and a rich diversity of butterflies, insects, and reptiles can also be found in the WBR (Lyon et al., 2017).

1.2.4 Climate

The study areas lie within the summer (December – March) region, with an annual rainfall of 420 mm (Cornelus, 2018). During summer, the maximum temperatures in the area are approximately 32 degrees Celsius, with the average minimum temperature at 18 degrees Celsius. In winter, the average maximum temperatures can reach up to 22 degrees Celsius, with the average minimum temperatures at 4 degrees Celsius, with frost occasionally occurring (Isaacs et al., 2013). The area experiences a marked dry season between May and September.

1.2.5 Geology

The topography of the study areas includes the mountain ranges with an elevation ranging between 800 m and 2 100 m with the steep rocky hills found in the south-western part of the WBR (de Klerk, 2003). The area's geology is characterized by sedimentary rocks, which eroded from the Bushveld Igneous Complex (Kerr et al., 2018). Other features such as numerous streams and small rivers, fountains, rock pools, marshes also define the area's topography (de Klerk, 2003).

1.3 Chapters synopsis

The dissertation is partitioned into six chapters. The **first chapter** gives background to the study, introduces the rationale for the project and the study area. **Chapter 2** presents a literature review of land use/ land cover change, land degradation, alien invasive plants, and ecosystem services and their adaptation strategies. **Chapter 3** explores the land use/ land cover change of the area to determine how the area has transitioned over time and provides a mapping framework that can be used to understand the nature and spatial extent of land degradation. **Chapter 4** presents the results of mapping the abundance and distribution of

selected invasive alien plants in the study areas. **Chapter 5** examines the integrated impacts of land degradation and alien invasive plants on the ecosystem services in the areas. Chapter 5 is followed by the Conclusion, **Chapter 6.**

1.4. References

Asmelash, F., Bekele, T. and Birhane, E., 2016. The potential role of arbuscular mycorrhizal fungi in the restoration of degraded lands. *Frontiers in microbiology*, *7*, p.1095.

Baber, R., De Klerk, A. and Walker, C., 2003. Environmental education and its role in the Waterberg Biosphere Reserve–South Africa. *Prospects*, *33*(3), pp.283-291.

Bai, Z.G., Dent, D.L., Olsson, L. and Schaepman, M.E., 2008. Proxy global assessment of land degradation. *Soil Use and Management*, *24*(3), pp.223-234.

Batisse, M., 1982. The Biosphere Reserve: a tool for environmental conservation and management. *Environmental Conservation*, *9*(02), pp.101-111.

Ben-Shahar, R., 1987. Grasses and habitat relationships on a sour bushveld nature reserve. *Vegetatio*, *72*(1), p.45. cited in Isaacs, L., Somers, M.J. and Dalerum, F., 2013. Effects of prescribed burning and mechanical bush clearing on ungulate space use in an African savannah. *Restoration Ecology*, *21*(2), pp.260-266.

Bloem, J. and Dunbar, M., 2016. Land degradation and desertification. In *Global Soil Biodiversity Atlas* (No. 27236, pp. 130-131), European Commission.

Blom, B., Sunderland, T. and Murdiyarso, D., 2010. Getting REDD to work locally: lessons learned from integrated conservation & development projects. *Environmental Science & Policy* 13, 164–172.

Booy, O., Robertson, P.A., Moore, N., Ward, J., Roy, H.E., Adriaens, T., Shaw, R., Van Valkenburg, J., Wyn, G., Bertolino, S. and Blight, O., 2020. Using structured eradication feasibility assessment to prioritize the management of new and emerging invasive alien species in Europe. *Global Change Biology*, *26*(11), pp.6235-6250.

Bradley, B.A. and Mustard, J.F., 2006. Characterizing the landscape dynamics of an invasive plant and risk of invasion using remote sensing. *Ecological Applications*, *16*(3), pp.1132-1147.

Brooks, M.L., D'antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., DiTomaso, J.M., Hobbs, R.J., Pellant, M. and Pyke, D., 2004. Effects of invasive alien plants on fire regimes. *AIBS Bulletin*, *54*(7), pp.677-688.

Campbell, A., Kapos, V., Chenery, A., Kahn, S., Rashid, M., Scharlemann, J.P.W. and Dickson, B., 2009. *The Linkages between Biodiversity and Climate Change Mitigation*. UNEP World Conservation Monitoring Centre, Cambridge, UK, First meeting of the Second ad hoc Technical Expert Group on Biodiversity and Climate Change, Convention on Biological Diversity.

Coetzer, K.L., Witkowski, E.T. and Erasmus, B.F., 2014. Reviewing Biosphere Reserves globally: effective conservation action or bureaucratic label? *Biological Reviews*, *89*(1), pp.82-104.

Convention on Biological Diversity (CBD)., 2018. *Biodiversity and Climate Change: Ecosystem-Based Approaches to Climate Change Adaptation and Disaster Risk Reduction*. Subsidiary Body on Scientific, Technical and Technological Advice.

Cornelus, L.A.J., 2018. Stakeholder participation in surface water and groundwater quality management in the Waterberg area: Limpopo Province, South Africa (Doctoral dissertation, University of Witwatersrand).

de Klerk, A., 2003. The Waterberg Biosphere Reserve: a Land Use Model for Ecotourism Development (Doctoral dissertation, University of Pretoria Pretoria).

de Klerk, G.J., 2001. Veld Condition Assessment of an Area of the Bakenberg Communal Land in Comparison to the Veld of Masebe Nature Reserve, as a Benchmark. Conducted as part of the Bakenberg Poverty Alleviation Program, Department of Environmental Affairs and Tourism, Pretoria.

Fox, R.C. and Rowntree, K.M., 2001. Redistribution, restitution and reform: prospects for the land in the Eastern Cape Province, South Africa. The GeoJournal Library, 167–186.

Gisladottir, G. and Stocking, M., 2005. Land degradation control and its global environmental benefits. *Land Degradation & Development*, *16*(2), pp.99-112.

Gnanavelrajah, N., 2009. Agriculture, Land degradation, Global warming: Links and Options for Mitigation. *Proceedings of the Jaffna Science Association*, 16th Annual Sessions 10-12 June 2009 Jaffna, Sri Lanka, *16*(02), p.5.

Henderson, L., 2013. SAPIA newsletters are posted at the ARC website: www.arc.agric.za. Volume No. 28.

Henning, B.J., 2002. The relevance of ecosystems to ecotourism in the Waterberg Biosphere Reserve. Doctoral Dissertation, Faculty of Nature and Agricultural Sciences. University of Pretoria.

Hestir, E.L., Khanna, S., Andrew, M.E., Santos, M.J., Viers, J.H., Greenberg, J.A., Rajapakse, S.S. and Ustin, S.L., 2008. Identification of invasive vegetation using hyperspectral remote sensing in the California Delta ecosystem. *Remote Sensing of Environment*, *112*(11), pp.4034-4047.

Hobbs, R.J., 2004. The Working for Water Programme in South Africa: The science behind the success. *Diversity & Distributions* 10, 501-503.

Hoffman, M.T. and Ashwell, A., 2001. Nature Divided: Land Degradation in South Africa. Cape Town: University of Cape Town Press.

Hoffman, M.T. and Todd, S., 2000. A national review of land degradation in South Africa: the influence of biophysical and socio-economic factors. *Journal of Southern African Studies*, *26*(4), pp.743-758.

Hoffman M.T, Todd S, Ntshona Z and Turner S., 1999. Land Degradation in South Africa. Department of Environmental Affairs and Tourism, Pretoria.

Hoveka, L.N., van der Bank, M. and Davies, T.J., 2020. Evaluating the performance of a protected area network in South Africa and its implications for megadiverse countries. *Biological Conservation*, *248*,108577.

Hulme, P.E., 2009. Trade, transport and trouble: Managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, *46*(1), pp.10-18.

Isaacs, L., Somers, M.J. and Dalerum, F., 2013. Effects of prescribed burning and mechanical bush clearing on ungulate space use in an African savannah. *Restoration Ecology*, *21*(2), pp.260-266.

Joshi, C., de Leeuw, J. and van Duren, I.C., 2004. Remote sensing and GIS applications for mapping and spatial modelling of invasive species. In *Proceedings of ISPRS* Vol. 35, p. B7.

Kerr, A., Eglington, B.M. and Milne, G.C., 2018. The geochemistry and geochronology of some Proterozoic granitoid rocks from the Natal structural and metamorphic Province, South-eastern Africa. *Revista Brasileira de Geociências*, *17*(4), pp.543-549.

Kolar, C.S. and Lodge, D.M., 2001. Progress in invasion biology: predicting invaders. *Trends in Ecology & Evolution*, *16*(4), pp.199-204.

Le Maitre, DC, Versfeld, DB and Chapman, RA. 2000. Impact of invading alien plants on surface water resources in South Africa: a preliminary assessment. Water Research Commission, vol. 26(3), pp 397-408.

Linders, T.E., Bekele, K., Schaffner, U., Allan, E., Alamirew, T., Choge, S.K., Eckert, S., Haji, J., Muturi, G., Mbaabu, P.R. and Shiferaw, H., 2020. The impact of invasive species on social-ecological systems: relating supply and use of selected provisioning ecosystem services. *Ecosystem services*, *41*, p.101055.

Lyon, A., Hunter-Jones, P. and Warnaby, G., 2017. Are we any closer to sustainable development? Listening to active stakeholder discourses of tourism development in the Waterberg Biosphere Reserve, South Africa. *Tourism Management*, *61*, pp.234-247.

Macdonald, I.A.W., 2004. Recent research on alien plant invasions and their management in South Africa: a review of the Inaugural Research Symposium of the Working for Water Programme. *South African Journal of Science,* 100, 21-26.

Manzo-Delgado, L., López-García, J. and Alcántara-Ayala, I., 2014. Role of forest conservation in lessening land degradation in a temperate region: The Monarch Butterfly Biosphere Reserve, Mexico. *Journal of Environmental Management*, *138*, pp.55-66.

Mosissa, D., 2019. Invasive Alien Plant Species in the Northwestern & Western Provenance of Ethiopia and Linking Its Phonological Implication to Management and Control. Ethiopian

Biodiversity Institute Assosa Biodiversity Center, Forest and Rangeland Biodiversity Case team.

Neumann, R. P., 1997. Primitive ideas. Protected area buffer zones and the politics of land in Africa. *Development and Change*, 28, 559–582.

Paini, D.R., Sheppard, A.W., Cook, D.C., De Barro, P.J., Worner, S.P. and Thomas, M.B., 2016. Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences*, *113*(27), pp.7575-7579.

Pejchar, L. and Mooney, H.A., 2009. Invasive species, ecosystem services and human well-being. *Trends in Ecology & Evolution*, *24*(9), pp.497-504.

Pool-Stanvliet, R., 2013. A history of the UNESCO Man and the Biosphere Programme in South Africa. *South African Journal of Science*, 109(9/10), 1-6.

Ravi, S., Breshears, D.D., Huxman, T.E. and D'Odorico, P., 2010. Land degradation in drylands: Interactions among hydrologic–aeolian erosion and vegetation dynamics. *Geomorphology*, *116*(3), pp.236-245.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J., 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions*, *13*(1), pp.126-139.

Roberts, D., Boon, R., Diederichs, N., Douwes, E., Govender, N., Mcinnes, A., Mclean, C., O'Donoghue, S. and Spires, M., 2012. Exploring ecosystem-based adaptation in Durban, South Africa:"learning-by-doing" at the local government coal face. *Environment and Urbanization*, *24*(1), pp.167-195.

Sahler, G., 2007. Biosphere reserves, a response to infrastructural challenges. *Model Regions with a Global Reputation*, *Journal of the German Commission for UNESCO*.

Sahu, P.K. and Singh, J.S., 2008. Structural attributes of lantana-invaded forest plots in Achanakmar–Amarkantak Biosphere Reserve, Central India. *Current Science*, *94*(4), 494–500.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A. and Leemans, R., 2000. Global biodiversity scenarios for the year 2100. *Science*, *287*(5459), pp.1770-1774.

Scholes, R.J. and Biggs, R., 2004. Ecosystem Services in Southern Africa: A Regional Assessment. A Contribution to the Millennium Ecosystem Assessment, (No. 33355 Caja (533)), CSIR.

Shackleton, C.M., Shackleton, S.E., Cousins, B., 2001. The role of land-based strategies in rural livelihoods: the contribution of arable production, animal husbandry and natural resource harvesting in communal areas in South Africa. *Development Southern Africa* 18, 581–604.

UNEP, 1994. United Nations Convention to Combat Desertification in Countries experiencing Serious Drought and/or Desertification, particularly in Africa.

UNESCO, 1996. Biosphere Reserves: The Seville Strategy and the Statutory Framework of the World Network.

Utuk, I.O. and Daniel, E.E., 2015. Land degradation: a threat to food security: a global assessment. *Journal of Environmental and Earth Science*, *5*(8), pp.13-21.

Van den Berg, E.C., Kotze, I. and Beukes, H., 2013. Detection, quantification and monitoring of Prosopis in the Northern Cape Province of South Africa using remote sensing and GIS. *South African Journal of Geomatics*, *2*(2), pp.68-81.

van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M. and Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management*, *89*(4), pp.336-349.

van Wilgen, B.W., Richardson, D.M., Le Maitre, D.C., Marais, C. and Magadlela, D., 2001. The economic consequences of alien plant invasions: examples of impacts and approaches to sustainable management in South Africa. *Environment, Development and Sustainability*, *3*(2), pp.145-168.

van Wilgen, B.W. and Wilson, J.R., 2018. The status of biological invasions and their management in South Africa in 2017. South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch. Accessed from: http://opus.sanbi.org/bitstream/20.500.12143/6231/1/National-Status-Report-web-6MB.pdf.

van Wilgen, B.W., Wannenburgh, A. and Foxcroft, L.C., 2020. The extent and effectiveness of alien plant control projects in South Africa.

van Wilgen, B.W., Measey, J., Richardson, D.M., Wilson, J.R. and Zengeya, T.A., 2020. Biological invasions in South Africa: an overview. *Biological Invasions in South Africa*, p.1.

Walker, B., and W. Steffen., 1997. An overview of the implications of global change for natural and managed terrestrial ecosystems. *Conservation Ecology* 1(2). *Conservation Ecology*, Resilience Alliance Inc.

Waterberg District Municipality, 2010. Waterberg District Municipality IDP annual report 2010/2011. Retrieved 23 May 2017, http://www.Waterberg.gov.za/docs/reports/FINAL.

Wells, M. P., and McShane, T. O., 2004. Integrating protected area management with local needs and aspirations. *Ambio,* 33, 513–519.

Wessels, K.J., Prince, S.D., Malherbe, J., Small, J., Frost, P.E. and van Zyl, D., 2007. Can human-induced land degradation be distinguished from the effects of rainfall variability? A case study in South Africa. *Journal of Arid Environments*, *68*(2), pp.271-297.

CHAPTER 2. LITERATURE REVIEW

2.1 Land degradation and land use/ land cover change

2.1.1 Land degradation overview

Land degradation is an environmental issue that is widespread globally. It is a complex concept that describes the modification of land resources such as vegetation, soil, water, and other natural resources to a degraded state. The phenomenon is receiving increasing levels of attention, particularly in the developing world, due to its severe environmental impacts (Webb et al., 2017; Moomen and Dewan, 2017; Meseret, 2016, Nkonya et al., 2016). The diverse environmental impacts of the problem are commonly assessed using temporal sequences of remotely sensed data such as satellite images. Land degradation patterns have been studied at several different scales, from global to local. For example, Nkonya et al. (2016), Bai et al. (2017), and Borrelli et al. (2017) assessed land degradation on a global scale. Meseret (2016) and Mapira (2017) studied land degradation on a national scale in Ethiopia and Zimbabwe, respectively. In contrast, Dube et al. (2017) applied land degradation assessment on a much smaller scale in the King Sabata Dalindyebo Local Municipality, Eastern Cape Province in South Africa.

The causes of land degradation remain poorly understood, particularly in dry areas (Herrmann and Hutchinson, 2016; Gibbs and Salmon, 2015; Ravi et al., 2010;). The problem is frequently associated with poverty (Barbier and Hochard, 2016; Gebreselassie et al., 2016; Way, 2016; Liu et al., 2008) which is perceived as a leading cause of land degradation and promote poverty itself. According to the Millennium Ecosystem Assessment, mitigating land degradation could be the first step in alleviating poverty, particularly in drylands (Millennium Ecosystem Assessment, 2005). However, Cowie et al. (2018) noted the challenges faced in quantifying the magnitude and severity of land degradation due to inadequate available quality data. Therefore, there is a need for reliable data that can assist in determining the land degradation problem to aid regional, national, and even international planning.

UNEP (1994, p.7) defines land degradation as "*the depletion or loss in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses*

or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as:

(i) soil erosion caused by wind and/or water;

(ii) deterioration of the physical, chemical and biological or economic properties of soil; and

(iii) long-term loss of natural vegetation;

Although the land degradation term is well defined above, it is most commonly applied to specific geographical locations (arid, semi-arid, and sub-humid areas). These are typically fragile habitats with minimal rainfall and high temperatures, prone to permanent biodiversity loss. The definition says that land degradation is caused by natural and anthropogenic activity, emphasizing the concept's complexity. Land degradation can be caused by various variables and interactions, such as inadequate land-use practices and management. The Ecosystem Assessment (2005) also defines land degradation as a reduction in productivity and is thus equivalent to a reduction in ecosystem provisioning services.

During the World Summit on Sustainable Development (WSSD) held in Johannesburg, South Africa, in 2002, land degradation was acknowledged as one of the leading global challenges to sustainable development in the 21st century (Gisladottir and Stocking, 2005). This position led to an agreement among countries that recognized land degradation as an environmental disaster, calling for “zero net land degradation” by reducing the rate of land deterioration and increasing the rate of land improvement (Higginbottom and Symeonakis, 2014). To achieve this goal, accurate and consistent scientific evidence is required to enable practical adaptation and mitigation measures to be taken (Grainger, 2015). In addition, the perceptions and views of people are crucial in understanding, mitigating, and preventing land degradation (Pulido and Bocco, 2014), particularly at the local scale.

2.1.2 The perception of land degradation

Environmental issues such as land degradation are of both global and local concern. However, a local understanding of dynamics is required (Wilbanks and Kates, 1999). Environmental problems usually attract more attention when they are seen to be occurring at a large spatial scale (Uzzell, 2000). For example, Meyer and Turner (1992) noted that environmental problems such as global warming, sea-level rise, and land degradation are perceived as more

important because they manifest at the global scale. Environmental problems only seem to receive more consideration at the local scale when immediate impacts directly impact human life (Measham et al., 2011; Bonaiuto et al., 1996), such as severe drought, can quickly lead to dehydration, malnutrition, and death. The conceptualization of environmental problems influences how they are prioritized (García-Mira et al., 2005) and may be affected by media coverage (Keinonen et al., 2016; García-Mira et al., 2005). The result is that local issues are often ignored at the expense of focusing on global environmental problems (Reed et al., 2016; Ostrom et al., 1999; García-Mira et al., 2005), a problem that Uzzell (2000) refers to as 'hyperopia.'

At the local scale, local perception and knowledge play a crucial role in providing the relevant and detailed information required for investigating the problem. Local perception of land degradation refers to how it is viewed by residents, depending on the status of the land and assumed causes (Pulido and Bocco, 2014). In many cases, local perceptions are not considered enough when addressing land degradation issues (Kiage, 2013). As a result, the community's ability to contribute to initiatives to mitigate the problem is limited. For example, in measuring land degradation using indicators such as the decline in crop yield, crucial first-hand information can be obtained from local farmers. Some studies revealed that indicators measured by external stakeholders are ineffective in detecting changes in degradation that are perceived by local land users (Pulido and Bocco, 2014). A study conducted by Gray and Morant (2003) revealed that some land degradation features identified by the local farmers in southwest Burkina Faso could not be detected by conventional scientific methods. The findings show that local perception and knowledge should be considered vital to understanding the extent and severity of land degradation issues locally (Kassa et al., 2013). Therefore, there is a need for local knowledge and perception to be integrated with the scientific findings for reliable results.

Pulido and Bocco (2014) argue that canvassing local perception is also a means of raising awareness and could help complement methods used in practice to curb land degradation. In Kushinga, Zimbabwe, farmers' perceptions based on traditional knowledge were combined with the researcher's views on the causes of land degradation within the region (Pulido and Bocco, 2014). This was done to identify factors determining farmers' decisions to adopt land

conservation practices in the local context (Pulido and Bocco, 2014). Traditional knowledge is based on local perceptions of land quality which, in most cases, means how the land is currently benefiting the community as compared to the past. Such knowledge is valuable in evaluating and categorising the land based on the nature and magnitude of degradation (Pulido and Bocco, 2003; 2014). For successful and long-term land degradation management, more trustworthy information on the many sources of land degradation is required.

2.1.3 Land use/ land cover change as a cause of land degradation

Land use/cover change can result in land degradation by reducing vegetation cover, soil erosion, and biodiversity loss. When measuring land degradation, a comparison of different land use or cover classes and biodiversity measures in the form of species abundance at the same sampling points are used (Breg Valjavec et al., 2014). Land cover refers to the physical elements of the surface and their biotic factors, and it is of interest to natural scientists (Meyer and Turner, 1992). Land use refers to how humans utilize or develop the land surface (Mwavu and Witkowski, 2008). Concerns over land use and land cover change developed with the increasing awareness that land surface transformation has diverse effects, including the global climate (Mustard et al., 2012). The land cover change was identified as the factor underlying changes in surface albedo in the 1970s, which resulted in changing climatic conditions (Cherubini et al., 2018; Deng et al., 2013). Terrestrial ecosystems (e.g., forests) are known to absorb carbon in the atmosphere. Therefore, in the 1980s, land use/ land cover change impacts were highlighted as having effects on the global climate through changes in the carbon cycle (Quesada et al., 2018; Reid et al., 2016; Prentice et al., 2001). Although Land use/cover change may be driven by natural factors (Plieninger et al., 2016), the primary cause of land surface transformation is people (Findell et al., 2017). Many studies highlight land use/ land cover change resulting from anthropogenic activities (Othow et al., 2017; Wang et al., 2015; Song et al., 2015; Foley et al., 2005).

When dealing with land cover change, the focus is on converting one type of land cover to another category or modifying land cover conditions within a category (Meyer and Turner, 1992). For example, converting woody forest into agricultural land is regarded as land conversion, whereas reducing tree cover due to firewood collection can be considered a land cover modification. In a land cover modification, the land cover character changes but not the

overall classification. However, changing the land cover may result in biodiversity loss, the leading cause of species extinction through habitat reduction, modification, and transformation (Pimm and Raven, 2000). Lambin et al. (2000) highlighted agriculture as the primary form of land cover conversion globally in recent decades. Croplands have expanded at an alarming rate due principally to population growth (Zumkehr and Campbell, 2015). However, detrimental impacts of land cover modification are also found in areas considered unsuitable for agriculture (Hoffman et al., 2018).

According to Meyer and Turner (1992), land cover conversion was previously afforded more attention than land cover modification because land conversion often has more obvious negative impacts. An example would be the draining of wetlands and their conversion to livestock pasture. Resource scarcity which may be driven by changing climatic conditions, can also increase pressure on land resources. When measuring land use/ land cover changes, the main categories are cultivation, forest, permanent pastures, and other lands, as identified by the Food and Agriculture Organisation (Meyer and Turener, 1992). To assess land degradation, reliable and accurate data on land use/ land cover is needed. Remote sensing and Geographical Information Systems (GIS) techniques improve the proficiency of handling spatial information on land surface characteristics and their diversity (Raju, 2006). Incorporating Remote Sensing and GIS for land degradation proved to be an efficient approach that has been successful in various land degradation investigations (AbdelRahman et al., 2018; Zerihun et al., 2018; Kotb et al., 2017).

2.1.4 Remote Sensing for land use/ land cover change and land degradation assessment

Land degradation is a complex process that is usually addressed using multidisciplinary approaches. Therefore, land degradation dynamics can be assessed under changing land covers. Land use/ land cover change studies assist in locating areas facing the deterioration of land resources (Gyawali et al., 2004). Remote sensing plays a crucial role in documenting the actual change in land cover on local, regional, and global spatial scales. The method applies the principles of monitoring and surveying, *viz;* allowing for repeat measurements, consistency, and objectivity (Hill et al., 2004). Remote sensing data further assist in mapping the land surface attributes, and it is of particular value in areas with limited accessibility (Al-

Mashreki et al., 2011). Various remote sensing products can detect, map, and monitor environmental changes (Hellden and Stern, 1980; Frederiksen, 1993, Raina et al., 1993; Sujatha et al., 2000) and are valuable tools for revealing the nature and extent of degradation at various scales (Hellden and Stern, 1980; Frederiksen, 1993, Raina et al., 1993; Sujatha et al., 2000). (Gao and Liu, 2008). The criteria used to assess land degradation may be physical, biological, or socio-economic (Salvati and Zitti, 2009). Sujatha et al. (2000) used Landsat pictures to map and monitor degraded land in Uttar Pradesh, India, as a result of water logging and salination, analyzing multitemporal imagery. Amissah-Arthur et al. (2000) applied SPOT data, collectively with biophysical (e.g., soil quality) and socioeconomic data (e.g., land-use intensity, population density and carrying capacity, and agricultural intensification) to assess land degradation status in African Sahel.

Gao and Liu (2008) argue that the success of remote sensing data to study land transformation will depend on the spectral uniqueness and the spatial resolution of the data used. More significant accuracy assessments are likely to be exhibited from the images with a high spatial resolution (Bozheva et al., 2005). For example, in mapping mangroves in the temperate zones, high spectral resolution proved to be more valuable in producing high accuracy levels in the assessment (Gao, 1999; Wang et al., 2018). Until recently, scientists relied on expert judgment to assess and monitor the environmental changes, rather than objective measurements with uncertain or variable results (see, for example, Middleton and Thomas, 1997; Nkonya et al., 2011). However, advances in the availability of remote sensing imagery and its analysis have created the foundation for synthesis across time. In the Sahel region, for example, remote sensing data helped in correcting the “myth” visualized as a marching desert by clarifying that degradation is not occurring within the entire Sahel, but only in specific areas of the desert (Lepers, 2005) and that there are significant temporal variations. Moreover, remote sensing has proven to be cost-effective and time-efficient as compared to traditional methods (Weng, 2002; Al-Mashreki et al., 2011).

An increasing number of remote sensing products are now available to assess land use/land cover changes (Buchanan et al., 2015). Some areas with substantial land use/land cover changes, such as Africa, North America, and European Russia, have such data with high spatial resolution depicting land use/land cover changes. For example, studies such as Peprah (2016), Moreau (2016); Paul (2016); Anselm et al., (2018) show the efficiency of remote

sensing in analyzing the land use/land cover change. High-quality available data provides opportunities for better characterizing, mapping, and monitoring changes (Giri, 2016).

2.1.4.1 Multitemporal image classification as a land degradation assessment method

One of the most extensively used change detection approaches is multitemporal image classification, assessing land degradation (Boori and Amaro, 2010; Gao and Liu, 2010). It provides valuable information for various supervised classification applications, such as long-term environmental monitoring and land cover change detection (Yang and Crawford, 2015). For decades, remote sensing researchers have been working on multi-temporal picture processing and change detection. For example, Almeida-Filho and Shirnabukuro (2002) used multitemporal TM data to map and monitor the evolution of degraded areas caused by independent gold miners, based on image segmentation/region classification techniques post-classification comparison, in the Roraima State, Brazilian Amazon. Sujatha et al. (2000) used Landsat MSS and TM data to map and monitor degraded lands caused by waterlogging and subsequent salinization/alkalinisation in Uttar Pradesh, India based on visual interpretation of multitemporal images. Although numerous successful application cases have been reported on monitoring and detecting environmental change, there are some challenges in applying multi-temporal imagery to derive timely information on the earth's environment and human activities (Jianya et al., 2008).

Maximum Likelihood Classification (MLC) is one of the algorithms widely used in the supervised classification approach due to its robustness and availability in almost all image processing software (Navalgund, 2001). MLC *quantitatively evaluates the variance and covariance of the category spectral response patterns when classifying an unknown pixel* (Su Mon et al., 2012 p.1046). It is deemed the best in terms of the statistical parameters for producing results (Choodarathnakara et al., 2012). It has traditionally been employed as a baseline to evaluate the classifier accuracy in remote sensing (Mining, 2006; Barandela and Juarez, 2002; Foody, 2002).

2.2 Alien invasive plants

2.2.1 Introduction and distribution

Globally, the movement of species from their native range into new environments has increased the occurrence of biological invasions over the years (Mack, 2003). The introduced species overcome several barriers and adapt to the conditions of the new environment, where they become established, reproduce and eventually spread to be treated as alien invasive (Pyšek et al., 2004). These alien species are introduced either intentionally or accidentally by humans (IUCN, 2000). However, natural pathways such as river channels are capable of spreading alien species through seeds dispersal. According to Hulme (2009), the biological invasion issue dates as far back as 1500 AD, although initially, the process would have been much less prevalent than today. The Industrial Revolution, which brought an increase in global trade, saw a continuous increase in invasive introductions due to the construction of canals, railways, and shipping that aided the movement of goods (Findlay and O'Rourke, 2009). In Europe and North America, a major increase in alien species (plants, mammals, and invertebrates) was observed in the 1800s (Hulme, 2009; Mack, 2003). The spread of alien invasives globally, particularly European species, was facilitated by the migration of Europeans in the 1820s and 1930s (McNeely, 2006). Since then, the problem has been amplified and is likely to remain an issue for the foreseeable future (Sutherland et al.,2008).

Several researchers have identified globalization as the primary source of alien species distribution (Hulme, 2009; Streftaris et al., 2005; McNeeley, 2006; Meyerson and Mooney, 2007). Other researchers, for example, Baier and Bergstrand (2001), point to related factors such as economic growth and transport efficiency as influencing biological invasions. Dispersal is substantially influenced by the distribution of suitable corridors (Mortensen et al., 2009), while habitat suitability is a constraint (Urban et al., 2008). Detailed studies of the characteristics of landscapes in which invasion is taking place have proved beneficial in documenting the nature and extent of alien invasive species and predicting future occurrences (Stohlgren et al., 2001).

South Africa has a lengthy history of alien species introduction (Richardson et al., 2011). For most introduction pathways, the rate of introduction of alien species has either remained

steady, fluctuated, or increased since the 1980s. The overall pattern suggests an increase over the years for most invasives (van Wilgen et al., 2018). Historical data on the pathways of introduction and dates on many species is either absent or poor quality (van Wilgen et al., 2018). Therefore, it is not conceivable to conclusively determine the exact time of introduction and rate of invasion. Once the species has been introduced, further dispersal or natural spread is highly probable if the species is invasive.

The spread of invasive alien species has been studied at various scales from the beginning of the science of invasion biology (Dyderski and Jagodziński, 2016). The spread of alien trees and shrubs has been studied by Tabacchi and Planty-Tabacchi (2003) along the River Valleys (Adour and Garonne Rivers) in France. Further, changes in alien trees and shrubs frequencies were zanalyzed by Dyderski et al. (2015) in Warta River Valley in Poznań. These studies emphasize the number of alien species increasing along the river valleys in particular. Similarly, roads commonly known to facilitate the spread of invasion are well studied at various scales (see Gelbard and Belnap, 2003; Brisson et al., 2010; Hayasaka et al., 2012).

2.2.2 Pathways of plants dispersal /Processes of invasion

There is a need to identify the dispersal pathways by which alien species are allowed entry into their non-native environment (Molnar et al., 2008). Both natural and artificial pathways are capable of spreading species across impressive distances. Air mass circulation and ocean currents play a significant role in species distribution through seed dispersal. For example, Mack (2003) highlighted plant seed dispersal via the Gulf Stream from the Caribbean to the British Isles due to ocean currents. Similar dispersal was noted by Murray (1986) in mangroves migrating through Oceania, and, in New Zealand, ocean currents were also identified as mechanisms by which plants invaded from Australia (Mack and Lonsdale, 2001).

Birds and other animals play an essential role in distributing alien invasive plant species. Animals can disperse plant seeds, usually locally, although birds (mainly migratory species) can facilitate long-distance dispersal. Animals typically use connected patches of habitat as avenues of movement (Bennett et al., 1994; Beier and Noss, 1998). Livestock is perceived as a major role player in local and regional distribution (Mack et al., 2003). Nevertheless, while natural forces play a significant role in species distribution, humans seem to be leading role

players in alien species distribution. People's dispersal of species had tended to mirror the natural pathways, especially when transport movements were largely dependent on wind and water currents. For example, ships were the main conveyors of plants from Africa to South America in the 18th century (Viola and Magolis, 1991). Canal construction around this time and, later, railways eventuated a massive increase in the spread of alien plants (Mills et al., 1999; Mack, 1991). In the United States, canals have created navigable canal networks and proved to be a standard dispersal mode for the deliberate and unintended spread of non-native plants (Mack, 2003; Mills et al., 1999). Exotic species' introduction may be accidental, for example, in ballast water in ships (Lavoie et al., 1999). Still, introductions are very frequently intentional, such as bio-control agents for agricultural use (Höfte and Altier 2010). With the rapid growth of shipping and the railway system in the 19th century, the transcontinental postal services provided opportunities for increased dispersal of plants (Mack et al., 2003). Eventually, with the introduction of air transport, the opportunities for plant movement created even greater chances of the natural movement of plants between continents (Mack et al., 2003).

Roads are principal pathways for alien plant species dispersal into protected areas and are especially effective for those species with high reproductive rates and short life cycles (Parendes and Jones, 2000; Trombulak and Frissell, 2000). Roads act as migration corridors that aid the dispersal of alien plants species and channel the population for subsequent expansion (Christen and Matlack, 2006). Roadsides are especially subject to disturbances that favor invasive species due to their linear structure (Forman et al., 2003). Less disturbing environments will not be so vulnerable to invasion (Pauchard and Alaback, 2004). Roadsides act as a corridor from which alien species spread from the boundaries to the interiors of less disturbed habitats (Cadenasso and Pickett, 2001) and act as reservoirs for alien plant propagules (Parendes and Jones, 2000). Long roads may create continuous, suitable habitats for invasive species for significant distances (Christen and Matlack, 2006). Species occurring alongside roads can spread long distances without encountering barriers that constrain their establishment or dispersal (Christen and Matlack, 2006). Roads may, therefore, serve as avenues of the invasion when they traverse different plant communities, allowing dispersal from one plant community to another (Christen and Matlack, 2006) and, in so doing, standardising the distribution of local and regional species. They can effectively reduce

dispersal distances (Gelbard and Belnap, 2003; Watkins et al., 2003) and thereby threaten biodiversity as non-native species displace those that are native. Due to their prominent role in facilitating alien plant invasion (Trombulak and Frissell, 2000).

2.2.3 Alien invasion and land transformation

Land-use change is the principal driver of alien plant invasion (Ficetola et al., 2010; McKinney et al., 2006). The extent of these land-use changes varies geographically, and rates vary through time, with periods of relatively little or slow change interrupted by rapid change (Holling and Gunderson, 1995). In any given ecosystem, species will respond to changes differently, and therefore, the composition of such ecosystems is subject to change over time. The movement or tolerance of species in a particular ecosystem depends on abiotic and biotic factors, including their interaction with other species. Suppose the ecosystems are dynamic, and species composition is variable. In that circumstance, depending on present conditions, the quantity and types of disturbance, and the composition of the regional species pool, the stage is set for species to arise and disappear (Hobbs, 2000).

Generally, changes in ecosystem properties provide opportunistic species (alien species) with the platform to expand. Any disturbance capable of increasing the supply of resources will provide opportunities for invasion (Hobbs, 2000). More generally, while disturbance is a natural part of ecosystem dynamics in many systems, human alteration of disturbance regimes and the introduction of novel disturbances alter the system setting and increase opportunities for invasion (Hobbs and Huenneke, 1992). Some species can take advantage of disturbances to colonize or expand their population (Hobbs, 2000). These are either disturbance specialists or generalists that can tolerate a wide range of conditions. The disturbance, which may only last a relatively short time, provides a window of opportunity for such species. Disturbances and land transformations allow these new species to colonize and spread, and they can often do so. As much as invasion can result from land modification, invasion by non-natives can be the driver of land modification. This happens when the invading species alters the ecosystem characteristics of an area (Hobbs, 2000). In such cases, the invading species may become dominant and completely transform the current vegetation type. For example, invading tree species can change grassland or shrubland into woodland, whereas the invasion by grasses can impact forests through modification and the fire regime

(Richardson et al., 1994; D'Antonio and Vitousek, 1992). Even moderate changes resulting from species invasion can alter structure and functioning that reduce ecosystem services, potentially leading to the need to develop another land use for that area or increase the degree of management needed to maintain current land use. This happens when forest or shrublands is impacted by pathogens (Hobbs, 2000) or when rangelands are invaded by woody perennials (Ludwig et al., 1996). Furthermore, invasion by non-native species may prevent the establishment of the desired native species in an area (Hobbs, 2000).

At the same time, some species invade an area with little or no impact on land uses. In contrast, some have proven to be from some perspectives, and the cause-and-effect relationship between invasion and land transformation is often complex. Changing land uses or continuous inappropriate use of land often may provide conditions that favor invading species. Whisenant (1990) and Ludwig et al. (1996) highlighted this problem in rangelands where an invasion by grasses and shrubs resulted from inappropriate grazing or fire regimes. Figure 2.1 illustrates the interlinkages between invasion and land transformation. Human activities introduce invasive species and further create opportunities to establish, affecting ecosystems that sustain human well-being.

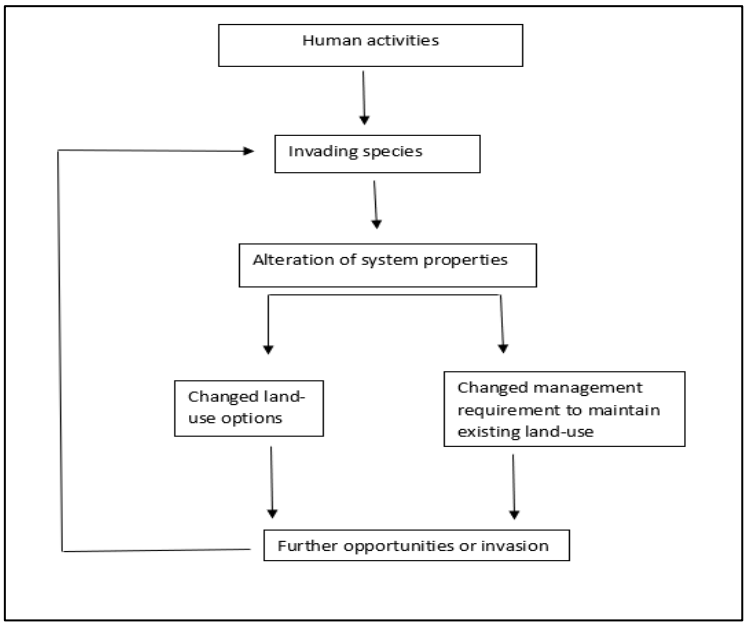


Figure 2.1. The interrelationship between human activities and invasion (Hobbs, 2000).

2.2.3.1 Common alien and invasive species in the Waterberg Savanna biome of South Africa

The characteristics of the alien and invasive species are summarised in the following sections and as listed in the National Environmental Management Biodiversity Act (NEMBA Act 104 of 2004).

2.2.3.1.1. Melia azedarach

Melia azedarach, commonly known as 'syringa' in South Africa, is a deciduous tree that can grow up to 23 meters in height. This species falls under category 1b in rural areas but is classified only as category 3 in urban areas. According to NEMBA, category 1b are species requiring compulsory control as they are considered to have high invasive potential. In contrast, category 3 species should not be propagated or sold, although existing plants do not need to be removed (Republic of South Africa, 2016). *Melia azedarach* originates from south Asia and northern Australia (Sarwar et al., 2015). In its non-native range, for instance, Africa, the Mediterranean basin, and North and South America, the species were introduced as ornamental trees to provide shade (Martínez and Manzano-García, 2019; van Wilgen et al., 2008; Mabberley, 1984). *Melia azedarach* produces thin fresh drupes that are green but turn yellow when they are ripe. These drupes can wrinkle and remain on the tree even after the branch has fallen, contributing to their invasiveness. The bark, leaves, flowers, and, mainly, the fruits of *Melia azedarach* are toxic to humans (Voigt et al., 2011) and means that significant invasion provides a threat rather than a benefit, even to people.

In South Africa, the tree was first recorded in Cape Town in 1800 (Smith, 1966). This tree has now been identified as highly invasive in the northern and eastern parts of South Africa (Henderson, 2001). It is particularly prominent in disturbed areas such as roads and forest margins and can also be found in riparian habitats (Henderson, 1991; Henderson and Musil, 1984). *Melia azedarach* can tolerate disturbance and can benefit and establish from disturbances such as fire. Based on its coverage in South Africa, Dean and Milton (2000) identified it as the second most invasive tree in the savanna biome, the third most invasive tree in the forest, and fourth in grassland in the savanna biome.

2.2.3.1.2. Eucalyptus camaldulensis

Eucalyptus camaldulensis, commonly known as river red gum, is an evergreen tree originating from Australia that occurs mostly along river channels and flood plains (Argent et al., 2004; Dzikiti et al., 2016). It is a species that adapts well to sub-tropical climates where rainfall and

humidity are high (Otim, 2008). Under favorable conditions, the species can grow very fast and tall (up to 45-55 metres in height and 1.2-2 metres diameter) (Boland et al., 2006). In South Africa, these were first introduced in the 1890s (Hirsch et al., 2020) and have now adapted well in the high rainfall zones of South Africa (Musengi and Archibald, 2017). Initially introduced for ornamental purposes, it is now used for timber, firewood, poles and provides a valuable source of nectar and pollen for producing honey (Musengi and Archibald, 2017; Forsyth et al., 2004; Allsopp and Cherry, 2004; Pirk et al., 2014).

In South Africa, this species is fast-growing and surpasses the growth rate of natural species. Under the NEMBA, *Eucalyptus* falls in different categories depending on where it is occurring. For instance, in riparian areas, protected areas, and within listed ecosystems, it falls under category 1b. In contrast, in plantations, woodlots, and bee-forage areas, it falls under category 2 (Republic of South Africa, 2016). However, suppose the species falls within cultivated lands that are at least 50 metres away from the non-transformed land and within 50 metres of farm properties or urban areas. In that case, it is not listed as a problematic species (Republic of South Africa, 2016).

2.2.3.1.3. Lantana camara

Lantana camara, also known as cherry pie or tickberry, is a weed native to South and Central America and known to be one of the world's most invasive plants (Taylor et al., 2012). This thicket-forming shrub can grow up to two to five metres high. However, with the support of other vegetation, it can climb up to 15 m and above (Totland et al., 2005; Gosper and Vivian-Smith, 2006). *Lantana* produces fruits that are purple-black drupes (van Oosterhout, 2004). According to the records, *Lantana camara* was introduced in South Africa in the 19th Century (Cowling et al., 2004). It is currently listed under category 1b in the NEMBA regulations. The species spread very slowly in the Western Cape due to unfavorable climatic conditions (Cowling et al., 2004). However, in the subtropical regions, such as the study area, it flourished since it was introduced in 1883 (Urban et al., 2011).

2.2.3.1.4. Jacaranda mimosifolia

Jacaranda mimosifolia, commonly known as jacaranda, is a tree indigenous to north-western Argentina (Olowoyo et al., 2009). With mauvish-blue tubular flowers, this tree is highly

invasive and can multiply up to a height of between 15 and 22 metres (Henderson, 1990). *Jacaranda mimosifolia* occurs mainly along riverbanks in warmer temperate and sub-humid conditions (Henderson, 1990). In South Africa, the tree was introduced in 1888 from Brazil (Olowoyo et al., 2009) for ornamental purposes and shade trees (Semenya et al., 2012). The tree can now be found in almost every part of South Africa except where frost is severe. Under NEMBA, *Jacaranda mimoifolia* is currently listed as category 1b in the Gauteng, KwaZulu-Natal, Limpopo, Mpumalanga and North-West Province. However, species is not listed in the urban areas of these regions or if it falls within 50 metres of the main house on a farm in the mentioned regions.

2.2.3.1.5. Cereus jamacaru

Cereus jamacaru, commonly known as the queen of the night, is a tree-like cactus originating from Brazil (Vatta et al., 2011; Torres et al., 2007). This species has segmented stems together with the main trunk and can grow up to 18 meters tall. In the late 1980s, the species was particularly prominent in Gauteng, North-West, and Limpopo (Klein, 1999). Substantial infestation is still recorded in Gauteng and Limpopo, but it has become a serious invasive species in many other parts of South Africa (Klein, 1999). In the NEMBA, *Cereus jamacaru* is listed under category 1b.

2.3 Ecosystem services

2.3.1 Degradation of ecosystems and impacts on human livelihoods

In the relatively recent past, ecosystems have been subject to extensive change rapidly to meet the increasing demand for goods and services (Hassan et al., 2005; Millennium Ecosystem Assessment, 2005). While billions of lives may be improved, the degradation of the ecosystems and loss of diversity of life on Earth is irreversible (Hassan et al., 2005; Millennium Ecosystem Assessment, 2005). Ecosystems provide a wide range of services, and their proper functioning inextricably limits human wellbeing. Their conversion is driven mainly by human activities such as agriculture and the establishment of alien invasive species, resulting in the depletion of native species. The impacts brought about by these factors are often impossible to reverse on any scale applicable to society. Therefore, continuous misuse may have dramatic consequences for humanity. To harness and manage ecosystems sustainably,

knowledge is required on the ecosystem's role in providing such services and understanding the skills, labor, and energy required to do that (Tengö et al., 2014).

The ecosystem services concept is increasingly being applied in contexts such as conservation and socio-economic development. Haines-Young et al. (2012) described these ecosystem services as gifts from nature to humankind. They are generated as linear flows from ecosystems to the people. They are generally greatly undervalued by society, perhaps because they are seen as complementary. Ecosystem services include a wide range of direct and indirect services (van Jaarsveld et al., 2005). Millennium Ecosystem Assessment (2005) distinguishes four categories of ecosystem services that sustain human well-being: provisional, regulating, supporting, and cultural services. Provisioning services include the material obtained from ecosystems such as fibre, freshwater, and food. Regulating services refer to the capacity of natural and semi-natural ecosystems to regulate essential ecological processes and life support systems through, for example, biogeochemical cycles (Millennium Ecosystem Assessment, 2005; Reid et al., 2002). Cultural services are the non-material benefits that human obtain from ecosystems, such as recreation, mental health, education, aesthetic appreciation, and spiritual enrichment. On the other hand, supporting services are those necessary for producing other ecosystem services such as soil formation, nutrient cycling, habitat for biodiversity, and maintenance of genetic pools.

Ecosystem services are widely considered to be deteriorating due to, among other disturbances, land degradation (Mueller et al., 2016). Indeed, land degradation is often synonymous with ecosystem services loss (Mueller et al., 2016). Millennium In many rural settlements, the main services that people derive from the ecosystems are provisioning (e.g., food, fuelwood), which relies on water availability and supporting services such as soil development and nutrient cycling (Millennium Ecosystem Assessment, 2005; Havstad et al., 2007). Therefore, when the land is degraded, when its ability to provide services reduces, people themselves are negatively impacted. Human perceptions can influence land degradation based on ecosystem performance as this may vary according to their personal experience and systematic knowledge (Roba and Oba, 2009). Therefore, when land degradation is assessed, it is essential to consider ecosystem services expected to be delivered (Riginos and Herrick, 2010).

Alien plant invasion also has significant impacts on the ecosystem services that sustain human livelihoods. Much of the literature on the impact of alien plants impact focuses, however, on hydrological effects. Nevertheless, alien invasive species can affect a wide range of ecosystem services such as altering water quality, nutrient cycling and can even elevate nutrient concentrations in groundwater (Chamier et al., 2012). Other effects include altering the fire regime. Therefore, communities need to develop strategies to cope with, adapt to, and mitigate such changes. People in arid areas have learned to cope with the spatial dynamics of ecosystems to continue benefiting from ecosystems services (Reynolds et al., 2007).

2.3.2. Ecosystem vulnerability

Ecosystem vulnerability is a concept applied in various research fields, although there seems to be no consensus as to a definition (Gallopín, 2006). It is defined by Adger (2006) as the state in which the system is susceptible or is at risk of harm from being exposed to various stressors with no or limited capacity to adapt. The link between ecosystem health and risk has been useful in eco-toxicology, where toxic compound effects are assessed on living organisms (De Lange et al., 2010; Ippolito et al., 2010). This field was then expanded to the species and population level, communities, ecosystems, and landscapes, requiring research to be conducted on the broader environment instead of limited to the laboratory (Depietri et al., 2015).

Vulnerability in this context is described based on parameters such as degree of exposure, species sensitivity, and the potential to recover, also known as resilience (De Lange et al., 2010). Disciplines such as engineering, economics, and anthropology, which are thought to have been initially used (Adger, 2006), also employ the term vulnerability. Human ecology and human geography have also contributed significantly to understanding the term in the socio-ecological context (Adger, 2013). The inability of the ecosystem to tolerate stressors (both natural and anthropogenic) in space and time is regarded as environmental vulnerability (Villa and McLeod, 2002). According to the IPCC, the definition of vulnerability includes climate change as the significant factor to ecosystem susceptibility (IPCC, 2013). Climate change represents multi-scale environmental change problems that involve multiple stressors and diverse actors (Adger, 2006).

Vulnerability does not exist in isolation, particularly in the environmental context. It is driven by natural and human actions-, such as poor land-use management, of which power distribution plays a role (Adger, 2006). Research on vulnerability is diverse (see Dercon and Krishnan, 2000; Janssen et al., 2006; Kamanou and Morduch, 2002). Cutter et al. (2003, p.243) define vulnerability as: "…the exposure (conditions that make people or places vulnerable to hazard) and as social condition (a measure of resilience to hazards) and the integration of potential exposures and societal resilience with a specific focus on places or regions". Exposure characteristics include duration, frequency, and magnitude of applied stress (Burton et al., 1993).

Vulnerability means that there is a risk that dependent livelihoods may be at risk. According to Adger (2003), more impoverished communities tend to settle in riskier or vulnerable areas, particularly in urban settlements where their lives are at risk from flooding, diseases, and other stresses. In coastal areas, for example, hurricane impacts can affect communities living in low-lying areas. People can, however, settle in an area that is originally capable of supporting livelihoods but that, as a consequence of their actions, becomes degraded and riskier. On the other hand, ecosystem vulnerability may typically be avoided or mitigated by avoiding or minimizing activities that enhance risk. When climatic conditions cause ecological vulnerability, policy adjustment can help keep the system in balance by adapting it (Adger, 2006).

2.3.3. Ecosystem-based Adaptation (EbA)

Ecosystem-based adaptation harnesses nature's potential to shield populations from the negative effects of climate change, for example, by offering ecosystem services in a sustainable manner (Munang et al., 2013). These approaches are defined as the adaptation of measures and policies that acknowledge the significant role of ecosystem services in minimizing communities' vulnerability to climate change (or other factors) impacts in a multi-sectoral and multiscale approach (Vignola et al., 2015). It brings together multiple sectors of government, from national to local, to address various pressures on ecosystem services, such as land use/land cover change, climate change, and how ecosystems may be better managed to boost people's resilience to climate change (Vignola et al., 2015). In a social-ecological system, how the community responds to changed conditions of the ecosystem and their

socio-economic driving forces determines the nature and composition of adaptive measures (Jodha, 1998). When EbA is explicit to a certain ecosystem to minimize impacts, it uses targeted management strategies in conservation and restoration (Munang et al., 2013). For example, mangroves and coastal marshes absorb the energy in storm surges. Therefore, protecting the coastal communities from the current and anticipated tropical storms requires restoring and conserving mangrove ecosystems (Alongi, 2008).

Ecosystems provide services that can assist in meeting adaptation needs across multiple human development sectors; such services include disaster risk reduction, food security (through agroforestry and fisheries), water management, and diversification of livelihoods (by increasing options on resources to use). Other economic, social and cultural benefits can be generated through EbA; thus, a well-maintained ecosystem can provide sustainable economic opportunities while safeguarding human lives. EbA has multiple co-benefits compared to other adaptive approaches and can also achieve multiple policy objectives for communities and the environment in the face of climate change (Munang et al., 2013). However, the potential of EbA is not yet fully recognized by many national governments despite the growing evidence of its benefits (Munang et al., 2013). In South Africa, several adaptation policies have been instigated, such as the Integrated Development Plan, the National Water Act, Working for Water, Working for Wetlands, among others (Department of Environmental Affairs, 2017). These policies aim to preserve the natural environment while improving people's lives, particularly in rural settings. The participatory has been widely applied to understand how policies assist in conserving the environment while benefiting the people and proves to be a suitable approach (Fernandez-Gimenez et al., 2008; Garcia and Cochrane, 2005; Pannell et al., 2006). This participatory approach was applied in this study, and the results are presented in Chapter Five.

2.4 References

AbdelRahman, M.A., Shalaby, A., Aboelsoud, M.H. and Moghanm, F.S., 2018. GIS spatial model-based for determining actual land degradation status in Kafr El-Sheikh Governorate, North Nile Delta. *Modeling Earth Systems and Environment*, *4*(1), pp.359-372.

Adger, W.N., 2003. Social capital, collective action and adaptation to climate change. *Economic Geography,* 79, pp. 387–404.

Adger, W.N., 2006. Vulnerability. *Global environmental change*, *16*(3), pp.268-281.

Adger, W.N., Barnett, J., Brown, K., Marshall, N. and O'brien, K., 2013. Cultural dimensions of climate change impacts and adaptation. *Nature climate change*, *3*(2), pp.112-117.

Allsopp, M.H. and Cherry, M., 2004. An assessment of the impact on the bee and agricultural industries in the Western Cape of the clearing of certain Eucalyptus species using questionnaire survey data. *Pretoria (South Africa): National Government of the Republic of South Africa, Department of Water Affairs, Internal Final Report*, *58*.

Almeida-Filho, R. and Shimabukuro, Y.E., 2002. Digital processing of a Landsat-TM time series for mapping and monitoring degraded areas caused by independent gold miners, Roraima State, Brazilian Amazon. *Remote sensing of Environment*, *79*(1), pp.42-50.

Al-Mashreki, M.H., Akhir, J.B.M., Rahim, S.A., Kadderi, D.M., Tukimat, L. and Haider, A.R., 2011. Land suitability evaluation for sorghum crop in the Ibb Governorate, Republic of Yemen using remote sensing and GIS techniques. *Australian Journal of Basic and Applied Sciences*, *5*(3), pp.359-368.

Alongi, D.M., 2008. Mangrove forests: resilience, protection from tsunamis, and responses to global climate change. *Estuarine, Coastal and Shelf Science*, *76*(1), pp.1-13.

Amissah-Arthur, A., Mougenot, B. and Loireau, M., 2000. Assessing farmland dynamics and land degradation on Sahelian landscapes using remotely sensed and socioeconomic data. *International Journal of Geographical Information Science*, *14*(6), pp.583-599.

Anselm, N., Brokamp, G. and Schütt, B., 2018. Assessment of land cover change in Peri-Urban high Andean environments south of Bogotá, Colombia. *Land*, 7(2), p.75.

Argent, R.M., McMahon, T.A., Bowler, J.M. and Finlayson, B.L., 2004. The dendroecological potential of Eucalyptus camaldulensis Dehnhardt (river red gum) from the Barmah Forest, Victoria, Australia. *Australian Geographical Studies*, *42*(1), pp.89-102.

Bai, Z.G., Dent, D.L., Olsson, L. and Schaepman, M.E., 2017. Global assessment of land degradation and improvement 1: identification by remote sensing. Report 2008/01, ISRIC – World Soil Information, Wageningen.

Baier, S.L. and Bergstrand, J.H., 2001. The growth of world trade: tariffs, transport costs, and income similarity. *Journal of international Economics*, *53*(1), pp.1-27.

Barandela, R. and Juarez, M., 2002. Supervised classification of remotely sensed data with ongoing learning capability. *International Journal of Remote Sensing*, 23(22), pp.4965-4970.

Barbier, E.B. and Hochard, J.P., 2016. Does land degradation increase poverty in developing countries? *PloS One*, *11*(5), p.e0152973.

Beier, P. and Noss, R.F., 1998. Do habitat corridors provide connectivity? *Conservation Biology*, *12*(6), pp.1241-1252.

Bennett, A.F., Henein, K. and Merriam, G., 1994. Corridor use and the elements of corridor quality: chipmunks and fencerows in a farmland mosaic. *Biological Conservation*, *68*(2), pp.155-165.

Boland, D.J., Brooker, M.I.H., Chippendale, G.M., Hall, N., Hyland, B.P.M., Johnston, R.D., Kleinig, D.A., McDonald, M.W. and Turner, J.D. eds., 2006. Forest Trees of Australia. CSIRO publishing.

Bonaiuto, M., Breakwell, G.M. and Cano, I., 1996. Identity processes and environmental threat: The effects of nationalism and local identity upon perception of beach pollution. *Journal of Community & Applied Social Psychology*, *6*(3), pp.157-175.

Boori, M.S. and Amaro, V.E., 2010. Land use change detection for environmental management: using multi-temporal, satellite data in the Apodi Valley of northeastern Brazil. Applied GIS, 6, pp. 1-15.

Borrelli, P., Robinson, D.A., Fleischer, L.R., Lugato, E., Ballabio, C., Alewell, C., Meusburger, K., Modugno, S., Schütt, B., Ferro, V. and Bagarello, V., 2017. An assessment of the global impact of 21st century land use change on soil erosion. *Nature Communications*, *8*(1), pp.1-13.

Bozheva, A.M., Petrov, A.N. and Sugumaran, R., 2005. The effect of spatial resolution of remotely sensed data in dasymetric mapping of residential areas. *GIScience & Remote Sensing*, *42*(2), pp.113-130.

Breg Valjavec, M., Zorn, M. and Čarni, A., 2018. Human-induced land degradation and biodiversity of Classical Karst landscape: On the example of enclosed karst depressions (dolines). *Land Degradation & Development*, *29*(10), pp.3823-3835.

Brisson, J., de Blois, S. and Lavoie, C., 2010. Roadside as invasion pathway for common reed (Phragmites australis). *Invasive Plant Science and Management*, 3(4), pp.506-514.

Buchanan, G.M., Brink, A.B., Leidner, A.K., Rose, R. and Wegmann, M., 2015. Advancing terrestrial conservation through remote sensing. *Ecological Informatics*, 30, pp.318-321.

Burton, I., Kates, R.W., White, G.F., 1993. The Environment as Hazard, Second ed. Guilford, New York.

Cadenasso, M.L. and Pickett, S.T., 2001. Effect of edge structure on the flux of species into forest interiors. *Conservation Biology*, *15*(1), pp.91-97.

Chamier, J., Schachtschneider, K., Le Maitre, D.C., Ashton, P.J. and van Wilgen, B.W., 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. *Water Research Commission*, *38*(2), pp.345-356.

Cherubini, F., Huang, B., Hu, X., Tölle, M.H. and Strømman, A.H., 2018. Quantifying the climate response to extreme land cover changes in Europe with a regional model. *Environmental Research Letters*, *13*(7), p.074002.

Choodarathnakara, A.L., Kumar, T.A. and Shivaprakash Koliwad, D.C., 2012. Satellite image classification with fuzzy logic: from hard to soft computing situation. *International Journal of Computer Science & Applications* (TIJCSA), 1(9), pp.2278 - 1080

Christen, D. and Matlack, G., 2006. The role of roadsides in plant invasions: a demographic approach. *Conservation Biology*, *20*(2), pp.385-391.

Cowie, A.L., Orr, B.J., Sanchez, V.M.C., Chasek, P., Crossman, N.D., Erlewein, A., Louwagie, G., Maron, M., Metternicht, G.I., Minelli, S. and Tengberg, A.E., 2018. Land in balance: The scientific conceptual framework for Land Degradation Neutrality. *Environmental Science & Policy*, *79*, pp.25-35.

Cowling, R.M., Richardson, D.M. and Pierce, S.M., 2004. Vegetation of Southern Africa. Cambridge University Press.

Cutter, S.L., 2003. The vulnerability of science and the science of vulnerability. *Annals of the Association of American Geographers*, *93*(1), pp.1-12.

D'Antonio, C.M. and Vitousek, P.M., 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*, *23*(1), pp.63-87.

Dean, W.R.J. and Milton, S.J., 2000. Directed dispersal of Opuntia species in the Karoo, South Africa: are crows the responsible agents? *Journal of Arid Environments*, *45*(4), pp.305-314.

Deng, X., Zhao, C. and Yan, H., 2013. Systematic modelling of impacts of land use and land cover changes on regional climate: a review. *Advances in Meteorology*, *2013,* pp.1-11.

Department of Environmental Affairs (DEA), 2017. Guidelines for Ecosystem-Based Adaptation (Eba) In South Africa. URL: https://www.sanbi.org/wp-content/uploads/2018/03/final-guidelines-ecosystem-based-adaptation-eba-south-africa.pdf, [Accessed May 2019].

Depietri, Y. and McPhearson, T., 2017. Integrating the grey, green, and blue in cities: Nature-based solutions for climate change adaptation and risk reduction. In *Nature-based solutions to climate change Adaptation in urban areas* (pp. 91-109).

Dercon, S. and Krishnan, P., 2000. Vulnerability, seasonality and poverty in Ethiopia. *The Journal of Development Studies*, *36*(6), pp.25-53.

De Lange, H.J., Sala, S., Vighi, M. and Faber, J.H., 2010. Ecological vulnerability in risk assessment—a review and perspectives. *Science of the Total Environment*, *408*(18), pp.3871-3879.

Dube, T., Mutanga, O., Sibanda, M., Seutloali, K. and Shoko, C., 2017. Use of Landsat series data to analyse the spatial and temporal variations of land degradation in a dispersive soil environment: A case of King Sabata Dalindyebo local municipality in the Eastern Cape Province, South Africa. *Physics and Chemistry of the Earth, Parts A/B/C*, *100*, pp.112-120.

Dyderski, M.K. and Jagodziński, A.M., 2016. Patterns of plant invasions at small spatial scale correspond with that at the whole country scale. *Urban Ecosystems*, *19*(2), pp.983-998.

Dyderski, M.K., Gdula, A.K. and Jagodziński, A.M., 2015. "The rich get richer" concept in riparian woody species–A case study of the Warta River Valley (Poznań, Poland). *Urban Forestry & Urban Greening*, *14*(1), pp.107-114.

Dzikiti, S., Gush, M.B., Le Maitre, D.C., Maherry, A., Jovanovic, N.Z., Ramoelo, A. and Cho, M.A., 2016. Quantifying potential water savings from clearing invasive alien Eucalyptus camaldulensis using in situ and high-resolution remote sensing data in the Berg River Catchment, Western Cape, South Africa. *Forest Ecology and Management*, *361*, pp.69-80.

Fernandez-Gimenez, M.E., Ballard, H.L. and Sturtevant, V.E., 2008. Adaptive management and social learning in collaborative and community-based monitoring: a study of five community-based forestry organizations in the western USA. *Ecology and Society*, *13*(2), pp.1-22.

Ficetola, G.F., Maiorano, L., Falcucci, A., Dendoncker, N., Boitani, L., Padoa-Schioppa, E., Miaud, C. and Thuiller, W., 2010. Knowing the past to predict the future: land-use change and the distribution of invasive bullfrogs. *Global Change Biology*, *16*(2), pp.528-537.

Findell, K.L., Berg, A., Gentine, P., Krasting, J.P., Lintner, B.R., Malyshev, S., Santanello, J.A. and Shevliakova, E., 2017. The impact of anthropogenic land use and land cover change on regional climate extremes. *Nature Communications*, *8*(1), pp.1-10.

Findlay, R. and O'rourke, K.H., 2009. Power and plenty: Trade, War, and the World Economy in the Second Millennium (Vol. 30). Princeton University Press, Columbia.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K. and Helkowski, J.H., 2005. Global consequences of land use. *Science*, *309*(5734), pp.570-574.

Foody, G.M., 2002. Hard and soft classifications by a neural network with a non-exhaustively defined set of classes. *International Journal of Remote Sensing*, 23(18), pp.3853-3864.

Foote, K.E. and Lynch, M., 1996. Geographic Information Systems as an Integrating Technology: Context, Concepts and Definitions. *The Geographer's Craft Project, Department of Geography, The University of Colorado at Boulder*.

Forman, R.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R.L., Heanue, K., Goldman, C.R. and Jones, J., 2003. Road Ecology: Science and Solutions. *Island press*.

Forsyth, G.G., Richardson, D.M., Brown, P.J. and Van Wilgen, B.W., 2004. A rapid assessment of the invasive status of Eucalyptus species in two South African provinces: working for water. *South African Journal of Science*, *100*(1-2), pp.75-77.

Frederiksen, P., 1993. Mapping and monitoring of soil degradation in semi-arid areas-satellite image methodology. In Proceedings from the Seminar on Satellite Remote Sensing of Environment and Agriculture in Developing Countries*. Institute of Geography, University of Copenhagen, Copenhagen, Denmark* (pp. 49-52).

Gallopín, G.C., 2006. Linkages between vulnerability, resilience, and adaptive capacity. *Global Environmental Change*, *16*(3), pp.293-303.

Gao, J. and Liu, Y., 2008. Mapping of land degradation from space: a comparative study of Landsat ETM+ and ASTER data. *International Journal of Remote Sensing*, *29*(14), pp.4029-4043.

Gao, J. and Liu, Y., 2010. Determination of land degradation causes in Tongyu County, Northeast China via land cover change detection. *International Journal of Applied Earth Observation and Geoinformation*, *12*(1), pp.9-16.

Gao, J., 1999. A comparative study on spatial and spectral resolutions of satellite data in mapping mangrove forests. *International Journal of Remote Sensing*, *20*(14), pp.2823-2833.

Garcia, S.M. and Cochrane, K.L., 2005. Ecosystem approach to fisheries: a review of implementation guidelines. *ICES Journal of Marine Science*, *62*(3), pp.311-318.

García-Mira, R., Real, J.E. and Romay, J., 2005. Temporal and spatial dimensions in the perception of environmental problems: An investigation of the concept of environmental hyperopia. *International Journal of Psychology*, *40*(1), pp.5-10.

Gebreselassie, S., Kirui, O.K. and Mirzabaev, A., 2016. Economics of land degradation and improvement in Ethiopia. In *Economics of land degradation and improvement–a global assessment for sustainable development* (pp. 401-430). Springer, Cham.

Gelbard, J.L. and Belnap, J., 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology*, *17*(2), pp.420-432.

Gibbs, H.K. and Salmon, J.M., 2015. Mapping the world's degraded lands. *Applied Geography*, *57*, pp.12-21.

Giri, C., 2016. Observation and monitoring of mangrove forests using remote sensing: opportunities and challenges. *Remote Sensing*, 8, 783

Gisladottir, G. and Stocking, M., 2005. Land degradation control and its global environmental benefits. *Land Degradation & Development*, *16*(2), pp.99-112.

Gökkaya, K., Budhathoki, M., Christopher, S.F., Hanrahan, B.R. and Tank, J.L., 2017. Subsurface tile drained area detection using GIS and remote sensing in an agricultural watershed. *Ecological Engineering*, *108*, pp.370-379.

Gosper, C.R. and Vivian-Smith, G., 2006. Selecting replacements for invasive plants to support frugivores in highly modified sites: a case study focusing on *Lantana camara*. *Ecological Management & Restoration*, *7*(3), pp.197-203.

Grainger, A., 2015. Is Land Degradation Neutrality feasible in dry areas? *Journal of Arid Environments*, *112*, pp.14-24.

Gray, L.C. and Morant, P., 2003. Reconciling indigenous knowledge with scientific assessment of soil fertility changes in southwestern Burkina Faso. *Geoderma*, *111*(3), pp.425-437.

Gyawali, B.R., Fraser, R., Wang, Y. and Bukenya, J.O., 2004. *Land cover and socio-economic characteristics in the eight counties of Alabama: a spatial analysis* (No. 377-2016-20592).

Haines-Young, R., Potschin, M. and Kienast, F., 2012. Indicators of ecosystem service potential at European scales: mapping marginal changes and trade-offs. *Ecological Indicators*, *21*, pp.39-53.

Hassan, R., Scholes, R. and Ash, N., 2005. Ecosystems and Human Well-Being: Current State and Trends. Island Press, Washington.

Havstad, K.M., Peters, D.P., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J. and Wright, J., 2007. Ecological services to and from rangelands of the United States. *Ecological Economics*, *64*(2), pp.261-268.

Hayasaka, D., Akasaka, M., Miyauchi, D., Box, E.O. and Uchida, T., 2012. Qualitative variation in roadside weed vegetation along an urban–rural road gradient. Flora-morphology, mistribution. *Functional Ecology of Plants*, 207(2), pp.126-132.

Helldén, U. and Stern, M., 1980. Monitoring land degradation in southern Tunisia: a test of Landsat imagery and digital data. *Lunds Universitets Naturgeografiska Institution 48*.

Henderson, L. and Musil, K.J., 1984. Exotic woody plant invaders of the Transvaal. *Bothalia*, *15*(1/2), pp.297-313.

Henderson, L., 1990. *Jacaranda*. *Plant Protection Research Institute. Department of Agricultural Development, Directorate of Agricultural Information, Pretoria*.

Henderson, L., 1991. Invasive alien woody plants of the Orange Free State. *Bothalia*, *21*(1), pp.73-89.

Henderson, L., 2001. Alien Weeds and invasive Plants. *Plant Protection Research Institute, Agricultural Research Council, Cape Town.*

Herrmann, S.M. and Hutchinson, C.F., 2016. The scientific basis: links between land degradation, drought and desertification. In: Johnson, P., Mayrand, K., Paquin, M., (Eds.) Governing Global Desertification: Linking Environmental Degradation, Poverty and Participation. *Ashgate Publishing: Hampshire, UK*.

Higginbottom, T.P. and Symeonakis, E., 2014. Assessing land degradation and desertification using vegetation index data: Current frameworks and future directions. *Remote Sensing*, *6*(10), pp.9552-9575.

Hill, J., Hostert, P., Röder, A., 2004. Long-term observation of Mediterranean ecosystems with satellite remote sensing. In: Mazzoleni, S., di Pasquale, G., Mulligan, M., di Martino, P., Rego, F. (Eds.), *Recent Dynamics of the Mediterranean Vegetation and Landscape*. John Wiley & Sons Ltd., Chichester, pp. 33–43.

Hobbs, R. J., 2000. Land use changes and invasions. in`; H. A. Mooney and R. J. Hobbs, (Eds.). *Invasive Species in a Changing World*. Island Press, Washington, D.C.

Hobbs, R.J. and Huenneke, L.F., 1992. Disturbance, diversity, and invasion: implications for conservation. *Conservation Biology*, *6*(3), pp.324-337.

Hoffman, M.T., Skowno, A., Bell, W. and Mashele, S., 2018. Long-term changes in land use, land cover and vegetation in the Karoo drylands of South Africa: implications for degradation monitoring. *African Journal of Range & Forage Science*, *35*(3-4), pp.209-221.

Höfte, M. and Altier, N., 2010. Fluorescent pseudomonads as biocontrol agents for sustainable agricultural systems. *Research in Microbiology*, *161*(6), pp.464-471.

Holling, C.S., Gunderson, L.H. and Light, S., 1995. Barriers and Bridges to the Renewal of Ecosystems. *New York: Columbia University Press. Cap*, *10*, pp.428-460.

Hulme, P.E., 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of applied ecology*, *46*(1), pp.10-18.

IPCC (2013), Climate Change 2013: the physical science basis. Working Group I contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. *Cambridge University Press, Cambridge.*

Ippolito, A., Sala, S., Faber, J.H. and Vighi, M., 2010. Ecological vulnerability analysis: A river basin case study. *Science of the Total Environment*, *408*(18), pp.3880-3890.

IUCN, 2000. Guidelines for the Prevention of Biodiversity Loss Caused by Alien Invasive Species. International Union for the Conservation of Nature, Switzerland.

Janssen, M.A. and Ostrom, E., 2006. Resilience, vulnerability, and adaptation: A cross-cutting theme of the International Human Dimensions Programme on Global Environmental Change.

Jianya, G., Haigang, S., Guorui, M. and Qiming, Z., 2008. A review of multi-temporal remote sensing data change detection algorithms. *The International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences*, *37*(B7), pp.757-762.

Jodha, N.S., 1998. Poverty and environmental resource degradation: An alternative explanation and possible solutions. *Economic and Political Weekly*, 33, pp.2384-2390.

Kamanou, G. and Morduch, J., 2002. Measuring vulnerability to poverty, WIDER Discussion Paper, No. 2002/58, ISBN 9291902411, The United Nations University World Institute for Development Economics Research (UNU-WIDER), Helsinki.

Kassa, Y., Beyene, F., Haji, J. and Legesse, B., 2013. Farmers' perception of the impact of land degradation and soil and water conservation measures in West Harerghe Zone of Oromia National Regional State, Ethiopia. *Journal of Biology, Agriculture and Healthcare*, 3(11): 13–18.

Keinonen, T., Palmberg, I., Kukkonen, J., Yli-Panula, E., Persson, C. and Vilkonis, R., 2016. Higher education students' perceptions of environmental issues and media coverage. *Discourse and Communication for Sustainable Education*, *7*(1), pp.5-22.

Kiage, L.M., 2013. Perspectives on the assumed causes of land degradation in the rangelands of Sub-Saharan Africa. *Progress in Physical Geography*, *37*(5), pp.664-684.

Klein, H., 1999. Biological control of three cactaceous weeds, Pereskia aculeata Miller, Harrisia martinii (Labouret) Britton and Cereus jamacaru De Candolle in South Africa. *African Entomology Memoir*, *1*, pp.3-14.

Kotb, M.M., Ali, R.R. and El Semary, M.A., 2017. Use of remote sensing and GIS for land degradation assessment of Qarun Lake Coastal Area, El-Fayoum, Egypt. In: In: M. Quessar, D. Gabriels, A. Tsunekawa & S. Evett (Eds.) *Water and Land Security in Drylands* (pp. 251-267). Springer, Cham.

Lambin, E.F., Rounsevell, M.D.A. and Geist, H.J., 2000. Are agricultural land-use models able to predict changes in land-use intensity? *Agriculture, Ecosystems & Environment*, *82*(1-3), pp.321-331.

Lavoie, D.M., Smith, L.D. and Ruiz, G.M., 1999. The potential for intracoastal transfer of non-indigenous species in the ballast water of ships. *Estuarine, Coastal and Shelf Science*, *48*(5), pp.551-564.

Lepers, E., Lambin, E.F., Janetos, A.C., DeFries, R., Achard, F., Ramankutty, N. and Scholes, R.J., 2005. A synthesis of information on rapid land-cover change for the period 1981–2000. *AIBS Bulletin*, *55*(2), pp.115-124.

Liu, Y., Wang, J. and Deng, X., 2008. Rocky land desertification and its driving forces in the karst areas of rural Guangxi, Southwest China. *Journal of Mountain Science*, *5*(4), pp.350-357.

Ludwig, J., Tongway, D., Hodgkinson, K., Freudenberger, D. and Noble, J., 1996. Landscape Ecology, Function and Management: Principles from Australia's Rangelands. CSIRO Publishing.

Mabberley, D.J., 1984. A monograph of Melia in Asia and the Pacific. The history of white cedar and Persian lilac. *Gardens' Bulletin*, *37*(1), pp.49-64.

Mack, R.N. and Lonsdale, W.M., 2001. Humans as Global Plant Dispersers: Getting More Than We Bargained For: Current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. *BioScience*, *51*(2), pp.95-102.

Mack, R.N., 1991. The commercial seed trade: an early disperser of weeds in the United States. *Economic Botany*, *45*(2), pp.257-273.

Mack, R.N., 2003. Plant naturalizations and invasions in the eastern United States: 1634-1860. *Annals of the Missouri Botanical Garden*, 90, pp.77-90.

Mack, R.N., Ruiz, G.M. and Carlton, J.T., 2003. Global plant dispersal, naturalization, and invasion: pathways, modes, and circumstances. In: Ruiz, G.M. and Carlton, J.T. (eds.) *Invasive Species Vectors and Management Strategies (eds). Island Press, Washington, pp. 3–30.*

Mapira, J., 2017. Land degradation and sustainable development in Zimbabwe: An historical perspective. *European Journal of Social Sciences Studies* 2(9), pp.1-15.

Martínez, G.J. and Manzano-García, J., 2019. Perception and use of non-native and invasive flora from Sierras de Córdoba in central Argentina. *Acta Botanica Brasilica*, *33*(2), pp.241-253.

McKinney, M.L., 2006. Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*(3), pp.247-260.

McNeely, J.A., 2006. As the world gets smaller, the chances of invasion grow. *Euphytica*, *148*(1-2), pp.5-15.

Measham, T.G., Preston, B.L., Smith, T.F., Brooke, C., Gorddard, R., Withycombe, G. and Morrison, C., 2011. Adapting to climate change through local municipal planning: barriers and challenges. *Mitigation and Adaptation Strategies for Global Change*, *16*(8), pp.889-909.

Meseret, D., 2016. Land degradation in Amhara region of Ethiopia: review on extent, impacts and rehabilitation practices. *Environmental Earth Sciences, 6, pp. 120-130*.

Meyer, W.B. and Turner, B.L., 1992. Human population growth and global land-use/cover change. *Annual Review of Ecology and Systematics*, *23*(1), pp.39-61.

Meyerson, L.A. and Mooney, H.A., 2007. Invasive alien species in an era of globalization. *Frontiers in Ecology and the Environment*, *5*(4), pp.199-208.

Middleton, N. and Thomas, D. (Eds.), 1997. World Atlas of Desertification. Chichester: Wiley & Sons.

Millennium Ecosystem Assessment., 2005. Ecosystems and Human Well-Being: Desertification Synthesis. World Resource Institute, Washington D.C.

Millennium Ecosystem Assessment., 2005. *Ecosystems and Human Well-Being: Global Assessment Reports*. *Island Press*, Washington, DC.

Mills, E.L., Chrisman, J.R. and Holeck, K.T., 1999. The role of canals in the spread of nonindigenous species in North America. *Nonindigenous Freshwater Organisms: Vectors, Biology and Impacts. Lewis Publishers, Boca Raton*.

Mining, W.I.D., 2006. Data Mining: Concepts and Techniques. Morgan Kaufinann.

Molnar, J.L., Gamboa, R.L., Revenga, C. and Spalding, M.D., 2008. Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, *6*(9), pp.485-492.

Moomen, A.W. and Dewan, A., 2017. Assessing the spatial relationships between mining and land degradation: evidence from Ghana. *International Journal of Mining, Reclamation and Environment*, *31*(7), pp.505-518.

Moreau, I., 2016. Evidencing land cover dynamics and tropical forest seasonality: the benefit of 13 years of daily global observation (SPOT-VEGETATION) (Doctoral dissertation, PhD thesis, Université Catholique de Louvain, Belgium).

Mortensen, D.A., Rauschert, E.S., Nord, A.N. and Jones, B.P., 2009. Forest roads facilitate the spread of invasive plants. *Invasive Plant Science and Management*, *2*(3), pp.191-199.

Mueller, H., Hamilton, D.P. and Doole, G.J., 2016. Evaluating services and damage costs of degradation of a major lake ecosystem. *Ecosystem Services*, *22*, pp.370-380.

Munang, R., Thiaw, I., Alverson, K., Mumba, M., Liu, J. and Rivington, M., 2013. Climate change and Ecosystem-based Adaptation: a new pragmatic approach to buffering climate change impacts. *Current Opinion in Environmental Sustainability*, *5*(1), pp.67-71.

Murray, D.R., 1986. Seed dispersal by water. In: Murray, D.G. *Seed Dispersal*, Academic Press, Sydney, pp.49-85.

Musengi, K. and Archibald, S., 2017. Demographics of Eucalyptus grandis and implications for invasion. *Koedoe: African Protected Area Conservation and Science*, *59*(1), pp.1-12.

Mustard, J.F., Defries, R.S., Fisher, T. and Moran, E., 2012. Land-use and land-cover change pathways and impacts. In *Land change science* (pp. 411-429). Springer, Dordrecht.

Mwavu, E.N. and Witkowski, E.T.F., 2008. Land-use and cover changes (1988–2002) around Budongo forest reserve, NW Uganda: Implications for forest and woodland sustainability. *Land Degradation & Development*, *19*(6), pp.606-622.

Navalgund, R.R., 2001. Remote sensing. *Resonance*, *6*(12), pp.51-60.

Nkonya, E., Anderson, W., Kato, E., Koo, J., Mirzabaev, A., von Braun, J. and Meyer, S., 2016. Global cost of land degradation. *Springer, Cham*. pp. 117-166.

Nkonya, E., Gerber, N., Baumgartner, P., von Braun, J., De Pinto, A., Graw, V., Kato, E., Kloos, J. and Walter, T., 2011. The economics of desertification, land degradation, and drought. Toward an Integrated Global Assessment IFPRI Discussion Paper, 1086.

Nkonya, E., Johnson, T., Kwon, H.Y. and Kato, E., 2016. Economics of land degradation in sub-Saharan Africa. *A Global Assessment for Sustainable Development* (pp. 215-259). Springer, Cham.

Olowoyo, J.O., van Heerden, E. and Fischer, J.L., 2010. Investigating *Jacaranda mimosifolia* tree as biomonitor of atmospheric trace metals. *Environmental Monitoring and Assessment*, *164*(1-4), pp.435-443.

Ostrom, E., Burger, J., Field, C.B., Norgaard, R.B. and Policansky, D., 1999. Revisiting the commons: local lessons, global challenges. *Science*, *284*(5412), pp.278-282.

Othow, O.O., Gebre, S.L. and Gemeda, D.O., 2017. Analyzing the rate of land use and land cover change and determining the causes of forest cover change in Gog district, Gambella regional state, Ethiopia. *Journal of Remote Sensing and GIS*, *6*(4), p.218.

Otim, C.K., 2008. Investigation of growth potential of alternative eucalyptus species for mid and high-altitude sites in the summer rainfall region in South Africa (Doctoral dissertation).

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F. and Wilkinson, R., 2006. Understanding and promoting adoption of conservation practices by rural landholders. *Australian Journal of Experimental Agriculture*, *46*(11), pp.1407-1424.

Parendes, L.A. and Jones, J.A., 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the HJ Andrews Experimental Forest, Oregon. *Conservation Biology*, *14*(1), pp.64-75.

Pauchard, A. and Alaback, P.B., 2004. Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. *Conservation Biology*, *18*(1), pp.238-248.

Paul, S., 2016. Mapping the Impact of Urbanization on Green Spaces: The Human Drivers of Land Use/Land Cover Change in Delhi. Doctoral dissertation, Manipal University, Ashoka Trust for Research in Ecology and the Environment (ATREE).

Peprah, P., 2016. Assessing land cover change resulting from surface mining development (a case study of Prestea and its environs in the Western Region of Ghana). Master's dissertation, Kwame Nkrumah University of Science and Technology.

Pimm, S.L. and Raven, P., 2000. Biodiversity: extinction by numbers. *Nature*, *403*(6772), p.843.

Pirk, C.W., Human, H., Crewe, R.M. and Van Engelsdorp, D., 2014. A survey of managed honeybee colony losses in the Republic of South Africa–2009 to 2011. *Journal of Apicultural Research*, *53*(1), pp.35-42.

Plieninger, T., Draux, H., Fagerholm, N., Bieling, C., Bürgi, M., Kizos, T., Kuemmerle, T., Primdahl, J. and Verburg, P.H., 2016. The driving forces of landscape change in Europe: A systematic review of the evidence. *Land Use Policy*, *57*, pp.204-214.

Prentice, I.C., Farquhar, G.D., Fasham, M.J.R., Goulden, M.L., Heimann, M., Jaramillo, V.J., Kheshgi, H.S., LeQuéré, C., Scholes, R.J. and Wallace, D.W., 2001. The carbon cycle and atmospheric carbon dioxide. *Cambridge University Press*.

Pulido, J. and Bocco, G., 2014. Local perception of land degradation in developing countries: a simplified analytical framework of driving forces, processes, indicators and coping strategies. *Living Reviews in Landscape Research*, *8* (4).

Pulido, J.S. and Bocco, G., 2003. The traditional farming system of a Mexican indigenous community: the case of Nuevo San Juan Parangaricutiro, Michoacán, Mexico. *Geoderma*, *111*(3-4), pp.249-265.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M. and Kirschner, J., 2004. Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. *Taxon*, *53*(1), pp.131-143.

Quesada, B., Arneth, A., Robertson, E. and de Noblet-Ducoudré, N., 2018. Potential strong contribution of future anthropogenic land-use and land-cover change to the terrestrial carbon cycle. *Environmental Research Letters*, *13*(6), p.064023.

Raina, P., Joshi, D.C. and Kolarkar, A.S., 1993. Mapping of soil degradation by using remote sensing on alluvial plain, Rajasthan, India. *Arid Land Research and Management*, *7*(2), pp.145-161.

Rajitha, K., Mukherjee, C.K. and Chandran, R.V., 2007. Applications of remote sensing and GIS for sustainable management of shrimp culture in India. *Aquacultural Engineering*, *36*(1), pp.1-17.

Raju, P.L.N., 2006. Fundamentals of geographical information system. *Satellite Remote Sensing and GIS Applications in Agricultural Meteorology*, p.103.

Ravi, S., Breshears, D.D., Huxman, T.E. and D'Odorico, P., 2010. Land degradation in drylands: Interactions among hydrologic–aeolian erosion and vegetation dynamics. *Geomorphology*, *116*(3), pp.236-245.

Reed, J., Van Vianen, J., Deakin, E.L., Barlow, J. and Sunderland, T., 2016. Integrated landscape approaches to managing social and environmental issues in the tropics: learning from the past to guide the future. *Global Change Biology*, *22*(7), pp.2540-2554.

Reid, P.C., Hari, R.E., Beaugrand, G., Livingstone, D.M., Marty, C., Straile, D., Barichivich, J., Goberville, E., Adrian, R., Aono, Y. and Brown, R., 2016. Global impacts of the 1980s regime shift. *Global Change Biology*, *22*(2), pp.682-703.

Reid, W.V., Mooney, H.A., Cropper, A., Capistrano, D., Carpenter, S.R., Chopra, K., Dasgupta, P., Dietz, T., Duraiappah, A.K., Hassan, R., Kasperson, R., Leemans, R., May, R.M., McMichael, A.J., Pingali, P., Samper, C., Scholes, R., Watson, R.T., Zakri, A.H., Shidong, Z., Ash, N.J., Bennett, E., Kumar, P., Lee, M.J., RaudseppHearne, C., Simons, H., Thonell, J., Zurek, N.B., 2005. Millennium Ecosystem Assessment Synthesis report. Island Press, Washington DC.

Republic of South Africa (RSA), 2016. 'Government Notice 40166', Government Gazette No. 32090. Accessed from https://www.environment.gov.za/sites/default/files/gazetted_notices/nemba10of2004_alienandinvasive_specieslists2016_0.pdf

Reynolds, J.F., Smith, D.M.S., Lambin, E.F., Turner, B.L., Mortimore, M., Batterbury, S.P., Downing, T.E., Dowlatabadi, H., Fernández, R.J., Herrick, J.E. and Huber-Sannwald, E., 2007. Global desertification: building a science for dryland development. *Science*, *316*(5826), pp.847-851.

Richardson, D.M., Williams, P.A. and Hobbs, R.J., 1994. Pine invasions in the Southern Hemisphere: determinants of spread and invadability. *Journal of Biogeography*, pp.511-527.

Richardson, D.M., Wilson, J.R.U. and Weyl, O.L.F., 2011. South Africa: Invasions. In: D. Simberloff, M. Rejmánek (Eds.), *Encyclopedia of Biological Invasions*, University of California Press, Berkeley 2011, pp. 643-651

Riginos, C., Herrick, J.E. 2010. Monitoring rangeland health: A guide for pastoralists and other land managers in Eastern Africa. Version II. Nairobi, Kenya: ELMT-USAID/East Africa.

Roba, H.G. and Oba, G., 2009. Efficacy of integrating herder knowledge and ecological methods for monitoring rangeland degradation in northern Kenya. *Human Ecology*, *37*(5), pp.589-612.

Rokach, L. and Maimon, O., 2005. Top-down induction of decision trees classifiers-a survey. *IEEE Transactions on Systems, Man, and Cybernetics, Part C (Applications and Reviews)*, *35*(4), pp.476-487.

Salvati, L. and Zitti, M., 2009. The environmental "risky" region: identifying land degradation processes through integration of socio-economic and ecological indicators in a multivariate regionalization model. *Environmental Management*, *44*(5), p.888.

Sarwar, A., Mahmood, Q., Bilal, M., Bhatti, Z.A., Pervez, A., Saqib, A.N.S., Khan, A.R. and Sultan, S., 2015. Investigation on Melia azedarach biomass for arsenic remediation from contaminated water. *Desalination and Water Treatment*, *53*(6), pp.1632-1640.

Semenya, S.S., Tshisikhawe, M.P. and Potgieter, M.T., 2012. Invasive alien plant species: A case study of their use in the Thulamela Local Municipality, Limpopo Province, South Africa. *Scientific Research and Essays*, 7, pp. 2363-2369.

Smith, C.A., 1966. Common Names of South African Plants. (Botanical Survey Memoir 35). Botanical Research Institute, Pretoria.

Song, W., Deng, X., Yuan, Y., Wang, Z. and Li, Z., 2015. Impacts of land-use change on valued ecosystem service in rapidly urbanized North China Plain. *Ecological Modelling*, *318*, pp.245-253.

Stohlgren, T.J., Otsuki, Y., Villa, C.A., Lee, M. and Belnap, J., 2001. Patterns of plant invasions: a case example in native species hotspots and rare habitats. *Biological Invasions*, *3*(1), pp.37-50.

Streftaris, N., Zenetos, A. and Papathanassiou, E., 2005. Globalisation in marine ecosystems: the story of non-indigenous marine species across European seas. In: R. N. Gibson, R. J. A. Atkinson, and J. D. M. Gordon (Eds.) *Oceanography and Marine Biology,* Taylor & Francis (pp. 429-464).

Su Mon, M., Mizoue, N., Htun, N.Z., Kajisa, T. and Yoshida, Y. 2012. Estimating Forest canopy density of tropical mixed deciduous vegetation using Landsat data: a comparison of three classification approaches. *International Journal of Remote Sensing* 33, 4, 1042-1057.

Sujatha, G., Dwivedi, R.S., Sreenivas, K. and Venkataratnam, L., 2000. Mapping and monitoring of degraded lands in part of Jaunpur district of Uttar Pradesh using temporal spaceborne multispectral data. *International Journal of Remote Sensing*, *21*(3), pp.519-531.

Sutherland, W.J., Bailey, M.J., Bainbridge, I.P., Brereton, T., Dick, J.T., Drewitt, J., Dulvy, N.K., Dusic, N.R., Freckleton, R.P., Gaston, K.J. and Gilder, P.M., 2008. Future novel threats and opportunities facing UK biodiversity identified by horizon scanning. *Journal of Applied Ecology*, *45*(3), pp.821-833.

Tabacchi, E. and Planty-Tabacchi, A.M., 2003. Recent changes in riparian vegetation: possible consequences on dead wood processing along rivers. *River Research and Applications*, *19*(3), pp.251-263.

Taylor, S., Kumar, L., Reid, N. and Kriticos, D.J., 2012. Climate change and the potential distribution of an invasive shrub, Lantana camara L. *PloS One*, *7*(4), p.e35565.

Tengö, M., Brondizio, E.S., Elmqvist, T., Malmer, P. and Spierenburg, M., 2014. Connecting diverse knowledge systems for enhanced ecosystem governance: the multiple evidence base approach. *Ambio*, *43*(5), pp.579-591.

Torres, L.B.V., Brito Primo, D.M., Andrade, M.G.S., Silva, S.M. and Lopes, M.F., 2007, October. Quality of plated Cereus (Cereus jamacaru DC) fruit harvested in different maturity stages. In *VI International Congress on Cactus Pear and Cochineal 811* (pp. 179-184).

Totland, Ø., Nyeko, P., Bjerknes, A.L., Hegland, S.J. and Nielsen, A., 2005. Does forest gap size affects population size, plant size, reproductive success and pollinator visitation in Lantana camara, a tropical invasive shrub? *Forest Ecology and Management*, *215*(1-3), pp.329-338.

Trombulak, S.C. and Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*, *14*(1), pp.18-30.

UNEP., 1994. United Nations Convention to Combat Desertification in Countries experiencing Serious Drought and/or Desertification, particularly in Africa*. Geneva: United Nations*.

Urban, A.J., Simelane, D.O., Retief, E., Heystek, F., Williams, H.E. and Madire, L.G., 2011. The invasive 'Lantana camara L.'hybrid complex (Verbenaceae): a review of research into its identity and biological control in South Africa. *African Entomology*, *19*(2), pp.315-348.

Urban, M.C., Phillips, B.L., Skelly, D.K. and Shine, R., 2008. A toad more traveled: the heterogeneous invasion dynamics of cane toads in Australia. *The American Naturalist*, *171*(3), pp. E134-E148.

Uzzell, D.L., 2000. The psycho-spatial dimension of global environmental problems. *Journal of Environmental Psychology*, *20*(4), pp.307-318.

van Jaarsveld, A.S., Biggs, R., Scholes, R.J., Bohensky, E., Reyers, B., Lynam, T., Musvoto, C. and Fabricius, C., 2005. Measuring conditions and trends in ecosystem services at multiple scales: the Southern African Millennium Ecosystem Assessment (SA f MA) experience. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *360*(1454), pp.425-441.

van Oosterhout E., 2004 *Lantana Control Manual*. Department of Natural Resources, Mines and Energy, Brisbane.

van Wilgen, B.W. and Wilson, J.R., 2018. The status of biological invasions and their management in South Africa in 2017. South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch. Accessed from: http://opus.sanbi.org/bitstream/20.500.12143/6231/1/National-Status-Report-web-6MB.pdf.

van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M. and Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management*, *89*(4), pp.336-349.

Vatta, A.F., Kandu-Lelo, C., Ademola, I.O. and Eloff, J.N., 2011. Direct anthelmintic effects of Cereus jamacaru (Cactaceae) on trichostrongylid nematodes of sheep: in vivo studies. *Veterinary Parasitology*, *180*(3-4), pp.279-286.

Vignola, R., Harvey, C.A., Bautista-Solis, P., Avelino, J., Rapidel, B., Donatti, C. and Martinez, R., 2015. Ecosystem-based adaptation for smallholder farmers: Definitions, opportunities and constraints. *Agriculture, Ecosystems & Environment*, *211*, pp.126-132.

Villa, F. and McLeod, Helena., 2002. Environmental vulnerability indicators for environmental planning and decision-making: guidelines and applications. *Environmental Management*, *29*(3), pp.335-348.

Viola, H.J. and Margolis, C., 1991. Seeds of Change: a quincentennial commemoration. Washington, DC: Smithsonian Institution Press.

Voigt, F.A., Farwig, N. and Johnson, S.D., 2011. Interactions between the invasive tree Melia azedarach (Meliaceae) and native frugivores in South Africa. *Journal of Tropical Ecology*, *27*(4), pp.355-363.

Wang, D., Wan, B., Qiu, P., Su, Y., Guo, Q. and Wu, X., 2018. Artificial mangrove species mapping using pléiades-1: An evaluation of pixel-based and object-based classifications with selected machine learning algorithms. *Remote Sensing*, *10*(2), p.294.

Wang, J., Wang, K., Zhang, M. and Zhang, C., 2015. Impacts of climate change and human activities on vegetation cover in hilly southern China. *Ecological Engineering*, *81*, pp.451-461.

Watkins, R.Z., Chen, J., Pickens, J. and Brosofske, K.D., 2003. Effects of forest roads on understory plants in a managed hardwood landscape. *Conservation Biology*, *17*(2), pp.411-419.

Way, S.A., 2016. Examining the links between poverty and land degradation: From blaming the poor toward recognising the rights of the poor (pp. 47-62). Routledge.

Webb, N.P., Marshall, N.A., Stringer, L.C., Reed, M.S., Chappell, A. and Herrick, J.E., 2017. Land degradation and climate change: building climate resilience in agriculture. *Frontiers in Ecology and the Environment*, *15*(8), pp.450-459.

Weng, Q., 2002. Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. *Journal of Environmental Management*, *64*(3), pp.273-284.

Wessels, K.J., Prince, S.D., Malherbe, J., Small, J., Frost, P.E. and VanZyl, D., 2007. Can human-induced land degradation be distinguished from the effects of rainfall variability? A case study in South Africa. *Journal of Arid Environments*, *68*(2), pp.271-297.

Whisenant, S.G., 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implications. In: *Proceedings from the Symposium on Cheatgrass Invasion,*

Shrub Dieoff and Other Aspects of Shrub Biology and Management. USFS Gen. Tech. Rep. INT 4-10.

Wilbanks, T.J. and Kates, R.W., 1999. Global change in local places: how scale matters. *Climatic Change*, *43*(3), pp.601-628.

Yang, H.L. and Crawford, M.M., 2015. Spectral and spatial proximity-based manifold alignment for multitemporal hyperspectral image classification. *IEEE Transactions on Geoscience and Remote Sensing*, *54*(1), pp.51-64.

Zerihun, M., Mohammedyasin, M.S., Sewnet, D., Adem, A.A. and Lakew, M., 2018. Assessment of soil erosion using RUSLE, GIS and remote sensing in NW Ethiopia. *Geoderma Regional*, *12*, pp.83-90.

Zumkehr, A. and Campbell, J.E., 2015. The potential for local croplands to meet US food demand. *Frontiers in Ecology and the Environment*, *13*(5), pp.244-248.

CHAPTER 3. LAND USE/ LAND COVER CHANGE AND LAND DEGRADATION MAPPING

3.1. Introduction

Land use/cover change is one of the main factors underlying land degradation. Numerous studies show that a detailed understanding of land use/cover change is needed in land degradation assessment (for example, Symeonakis et al., 2007; Bajocco et al., 2012; Alemu, 2015; Reed and Stringer, 2016). This chapter presents the analysis of land use/cover change and land degradation mapping within and outside the WBR. The methods applied to obtain the results are detailed and presented in maps, figures, and tables. The chapter aims to address objective (1) to document the nature, scale, distribution, and temporal trends in land degradation using remote sensing techniques.

3.2. Methods

3.2.1. Geo-statistical data preparation for land use /land cover change assessment

Remote sensing was the key technique applied in Land use/cover change assessment. Remotely sensed satellite images of the same season (winter) with different bands (Landsat TM and Landsat ETM+) were downloaded from Earth Explorer (https://earthexplorer.usgs.gov/). Winter images were chosen for the following reasons: minimal cloud cover during that season and overgrazing effects which are more apparent. The acquired images used were of Landsat 4 - 5 TM C2 L2 and Landsat 8 OLI/TIRS C2 L2 (Table 3. 1) with a spatial resolution of 30 meters and WGS 84S UTM zone projection. USGS Landsat Collection 2 Level-2 includes Surface Reflectance and Surface Temperature. Landsat Collection 2 offers improved processing, geometric accuracy, and radiometric calibration compared to previous Collection 1 products https://docs.digitalearthafrica.org).

The Landsat images were already corrected, and therefore geometric and atmospheric corrections were not necessary. Satellite imagery, including Landsat products, are corrected for geometric distortions (desktop.arcgis.com), and as such, it was not necessary to perform geometric and atmospheric correction for this study.

The study area fell within two scenes designated by paths 170 and 171 and rows 76 and 77 across all the years. The layers of each image downloaded were stacked together using ERDAS IMAGINE 10.2 to form one layer with different bands (e..,g. see Figure 3.1A & B). These layers cover both parts of the study areas. Image sub-setting and layer stacking to remove the null values (visible black borders on the images). The images were cropped using the shapefiles of both study areas. Mosaicking was performed after image cropping to form one image covering the entire study area for each sample year (e.g., see Figure 3.1C). The images were analysed to ascertain the Spatio-temporal pattern of land use/cover change within and outside the WBR. The analysis was carried out using ArcMap 10.4 software. Three bands were used in the visible spectral range and near-infrared (NIR) (e.g., band 2, 3, 4, and 5, which are for blue, green, red, and NIR respectively in Landsat 8 ORI, whereas for Landsat TM and ETM+ band 1, 2, 3 and 4 which are blue, green, red and NIR respectively) to form false-color composite images using ArcGIS software. Figure 3.1.D shows an example of the final output after performing all the tasks. This process was followed by image classification, accuracy assessment, change detection, and analysis of the results.

Table 3.1. The satellite images were used for land degradation assessment.

Landsat Satellite	**Image date**	**Landsat sensor**	**Path/Row**	**Bands and wavelength**	**Radiometric resolution**
4-5	1989 (7,23 & 30 August) 2000 (21 & 28 August)	Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+)	170,171; 076,077	1- Blue (0,45 - 0,52) 2- Green (0,52 -0,60) 3- Red (0,63-0,69) 4- NIR (0,76-0,90)	8 bits
8	2013 (16 & 25 August) 2017 (11 & 20 August)	Operational Land Imager (OLI) and the Thermal Infrared Sensor (TIRS)	170,171; 076,077	2- Blue (0,45 - 0,51) 3- Green (0,53 -0,59) 4- Red (0,64-0,67) 5- NIR (0,85-0,88)	16 bits

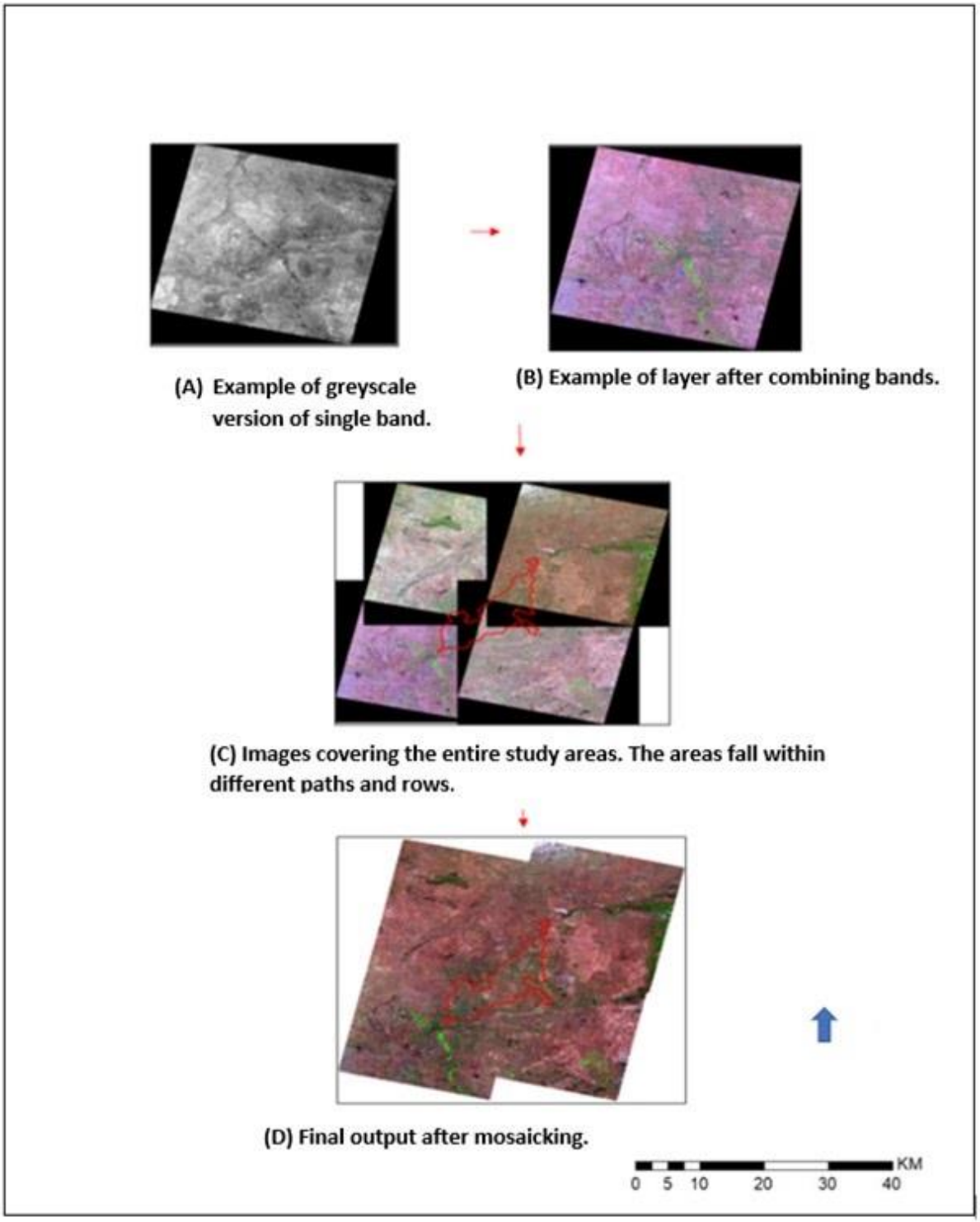


Figure 3.1. Screenshots showing an example of layer stacking performed to have one image covering the entire study area. Image B-D used band combinations 2, 3, and 4, and image A only shows band 1.

For this study, digital classification was performed for land use cover change as it yields better results than traditional visual classification (Yiran et al., 2012). Digital classification is generally preferred by scientists as it allows quantitative analyses to be carried out, and it processes

images automatically (Puig et al., 2002). The Maximum Likelihood Classification (MLC) technique was used for supervised image classification in this study for the following two reasons: (1) The analyst is knowledgeable with the research field and has gathered data; (2) supervised classification produces better outcomes than unsupervised classification (Lacroix et al., 2017; Chunjing et al., 2017).MLC was chosen as it assigns the unknown measurement vector to the class with the highest probability of belonging (Otukei and Blaschke, 2010). It has proved to perform better than other classifiers (Topaloğlu et al., 2016).

Under supervised classification, features can be accurately classified as the image is trained to assign a specific code to particular features. However, some different features can still be given the same code during image training due to the problem of similar spectral reflectance highlighted above. This problem was encountered when analysing the results of this dissertation. However, re-training the image using more points increased the accuracy of the classification and, in cases where misclassification was identified, re-classification was performed manually under supervised classification. Numerous applications have identified shadows to be a constraint in remote sensing with specific reference to change detection and image classifications (Dare, 2005; Zhou et al., 2009), as they can affect the accuracy of the results (Paul, 1997). This problem is especially difficult in mountainous areas with steep slopes, and the accuracy level may be lower for classification in such cases (Shahtahmassebi et al., 2013).

Before performing the supervised classification, a ground-truthing field visit was conducted, and GPS coordinates for areas sampled as training points were recorded. The following classes were used: water bodies, cultivated fields, woody vegetation, grassland, bare land, and settlements. These classes are described in Table 3.2. During the image classification, bare land and settlement were classified under the same category due to both identified classes having the same spectral reflectance. As a result, the settlements were manually digitized for all the years. The total landcover for the settlements class was subtracted from the total landcover of the bare land class to avoid duplication.

Table 3.2. Description of land use/ land cover classes used for the study area.

Class	Description
Water bodies	Includes all surface water from the rivers, dams, wetlands, streams
Cultivated fields	Areas cultivated with crops, weeds, vegetables, or fruits
Woody vegetation	Includes healthy woody vegetation and shrubs/scrubs
Grasslands	Vegetation dominated by grasses
Bare land	Includes areas of exposed soil surfaces due to natural causes (e.g., floods) or human activities (e.g., road building, overgrazing, burnt scars). In this area, sparse vegetation can be present.
Settlements	Includes build-up areas where people live or large buildings.

3.2.2. Spectral separability and accuracy assessment

Spectral separability analysis was performed using ArcMap 10.4. This process allows the researcher to assess how features can be spectrally separable from the Landsat images. The process was performed on Landsat 4 - 5 TM C2 L2 of 1989 and Landsat 8 OLI/TIRS C2 L2 of 2017. The assessment was based on the mean reflectance spectra obtained from the pixels of both Landsat images.

Accuracy assessment was performed to quantify the accuracy and reliability of the images classified. Accuracy assessment compares the image classified to the data source deemed to be accurate (e.g., high-resolution imagery) or with the ground-truthed data (Rwanga and Ndambuki, 2017). For the 2017 classification, ground-truthed data were used as a reference. In contrast, for other years, Google Earth images and Google Earth Engine images of 1989, 2000, and 2013 timelapse obtainable from https://earthengine.google.com/timelapse/ were used as reference. In total, 290 points were used for accuracy assessment for each image classified (Figure 3.2; Table 3.3). All these points were randomly and unevenly distributed to avoid bias. The classification errors, as indicated in Table 3.4, were performed using ArcMap 10.4. It should also be noted that accuracy assessment for settlement was not performed as

this class was manually digitised. Appendix 1 gives an illustration of error metrics performed for other images.

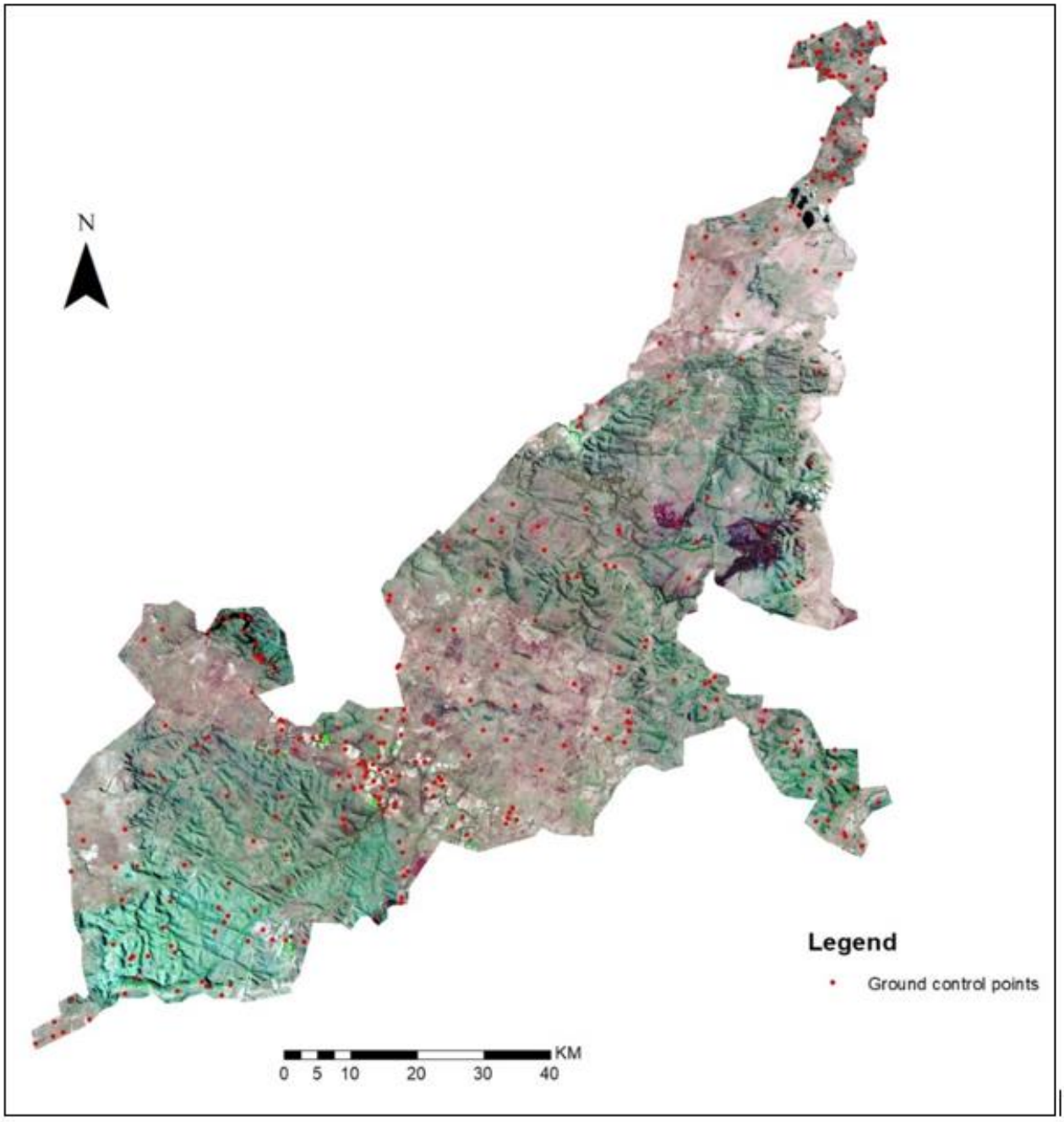


Figure 3.2. Map showing 290 ground control points used in the error matrix for 2017 image classification.

Table 3.3. Error (confusion) matrix showing the number of ground control points and accuracy of the 2017 Landsat OLI image classification.

	Bare land	Grassland	Woody vegetation	Cultivated	Water	**Total ground control points**
Bare land	**69**	0	0	0	1	**70**
Grass land	1	**64**	7	1		**73**
Woody veg	0	2	**61**	0	0	**63**
cultivated	0	4	2	**39**	0	**45**
Water	0	0	0	0	**39**	**39**
Total	**70**	**70**	**70**	**40**	**40**	**290**

Table 3.4. Classification errors for classification performed on the 2017 Landsat image.

Classified Raster	Commissions	Row total	Percentage
Bare land	**1**	**70**	**1**
Grassland	**9**	**73**	**12**
Woody veg	**2**	**63**	**3**
cultivated	**6**	**45**	**13**
Water	**0**	**39**	**0**

Classified Raster	Omissions	Column total	Percentage
Bare land	**1**	**70**	**1**
Grassland	**6**	**70**	**9**
Woody veg	**9**	**70**	**13**
cultivated	**1**	**40**	**3**
Water	**1**	**40**	**3**

The overall accuracy, Kappa coefficient statistics, user and producer's accuracy were calculated using the confusion matrix (Adam et al., 2014; Congalton and Green, 2008). The overall accuracy represents the probability that randomly selected points are classified correctly on the produced land use/cover map (Richards and Jia, 2013). The overall accuracy was determined using the formula below:

Overall accuracy = $\frac{\Sigma\,(classes\ correctly\ classified\ along\ diagonal)}{\Sigma(Row\ total\ or\ Column\ total)}$

The producer's accuracy, which is also called omission error, indicates how well reference pixels of the certain ground cover type are classified and was determined using the following formula:

Producer accuracy = $\frac{Number\ of\ the\ correctly\ classified\ class\ in\ a\ column}{Verified\ items\ total\ number\ in\ that\ column}$

The user's accuracy, also called the commission error, represents the likelihood that pixels classified into a certain category truly represent that category on the ground. It was determined using the following formula:

User accuracy = $\frac{Correctly\ classified\ number\ of\ item\ in\ a\ row}{Verified\ items\ total\ number\ in\ that\ row}$

Both producer's and User's accuracy help demonstrate an error's even or uneven distribution (Congalton, 1991).

The computed Kappa coefficient lies between the value of 0 and 1 and gives a measure of the difference between the actual and expected agreement (Congalton, 1991). The value of 1 symbolises complete agreement, whereas a zero symbolises no agreement (Congalton, 1991). The Kappa coefficient was calculated using the following formula:

$$\kappa = \frac{N\sum_{i=1}^{n} m_{i,i} - \sum_{i=1}^{n}(G_i\, C_i)}{N^2 - \sum_{i=1}^{n}(G_i\, C_i)}$$

N is the total number of pixels classified that are compared to ground-truthed observations, ***i*** represents the class number, ***mi, i*** is the number of pixels belonging to the ground class ***i***, ***Gi*** represents the total number of ground truth pixels that belong to class ***i***, ***Ci*** represents the total number of classified pixels that belong to class ***i*** (Gashu and Gebre-Egziabher, 2018).

3.2.3. Framework developed for assessing land degradation severity at a local scale

Four indicators were identified as crucial for assessing land degradation severity in the study areas: vegetation loss, invasive alien plants distribution (abundance and occurrence), population pressure, and land use intensity. The vegetation loss based on the 2018 Vegetation map, the loss of vegetation due to land use intensity, the abundance of alien plants distribution, and population distribution were overlaid to determine the severity of degradation using ranges. The vegetation cover representing habitat loss, land use intensity, and alien plants distribution are the key indicators of land degradation that can be derived from satellite images. However, for this study, alien plants distribution is solely based on data collected in the field. Population density is a crucial factor to include in severity assessment as it indicates the population stress on the environment. The following describes how different indicators were applied to assess land degradation severity within and outside the WBR.

3.2.3.1. Vegetation loss

The 2018 National Vegetation Map (VegMap) obtained from SANBI (https://bgis.sanbi.org/) was overlaid with the 2017 land use/ land cover map in ArcMap to determine the amount of vegetation lost to different land uses/ land cover. The VegMap classifies the diverse vegetation of South Africa, Lesotho, and Swaziland, currently known as eSwatini. The VegMap data originated in 1936, while more recent versions have improved scale and accuracy (https://bgis.sanbi.org). Although vegetation mapping does not change, land use/ land cover change indicates the types of vegetation lost in a certain area at a certain year. The computation was performed using ArcGIS and Microsoft Excel to determine the percentages of vegetation types lost in 1989, 2000, 2013, and 2017. According to the National VegMap, the following six vegetation types are found within the WBR: Central Sandy Bushveld, Limpopo Sweet Bushveld, Makhado Sweet Bushveld, Roodeberg Bushveld, Waterberg Mountain Bushveld, and Waterberg-Magaliesberg Summit Sourveld. The study area outside the WBR is comprised of the following vegetation types: Central Sandy Bushveld, Dwaalboom Thornveld, Madikwe Dolomite Bushveld, Springbokvlakte Thornveld, Subtropical Alluvial Vegetation, Waterberg Mountain Bushveld, Waterberg-Magaliesberg Summit Sourveld, Western Sandy Bushveld. Vegetation loss was calculated in the following classes: bare lands, cultivated fields, water bodies, and settlements. According to Jabbar and Chen (2006),

vegetation loss higher than 40% is considered severe, 40-25 is considered high, 10-25 is considered medium, and less than 10% is considered moderate. Jabbar and Chen (2006) assigned the threshold to the indicator following the United Nation's indices for desertification assessment and using experts' opinions.

3.2.3.2. Land-use intensity

Intensity analysis is a numerical framework that interprets differences among categories that have been summarized by the transition matrix tables as shown in Table 3.22 and 3.23, section 3.4, in which the row and column's categories are similar (Aldwaik and Pontius, 2012). Thus, land use cover classification and change detection matrix need to be performed to analyze land-use intensity. This method allows the researcher to analyze the land use in more detail (Niya et al., 2019). The intensity analysis can be carried out at time interval level, category level, and transition level. The observed changes at each level are compared to the hypothetical uniform change, as shown in section 3.3.4.2 of the uniform line's land-use intensity study results. The transition level was used in this study to examine the transformation of woody vegetation and grasslands from 1989 to 2017 into other groups. Researchers such as Niya et al. (2019), Pontius et al. (2013), Aldwaik and Pontius (2012) applied this method for assessing the land-use intensity in respective areas. The following formulae were used to calculate the land-use intensity by transition level, focusing on woody vegetation and grassland with the description of mathematical notation in Table 3.5.

$$R_{tin}\frac{Annual\ transition\ from\ i\ to\ n\ during\ [Y_t, Y_{t+1}]}{Size\ of\ i\ at\ Y_t}100\% = \frac{C_{tin}/(Y_{t+1}-Y_t)}{\sum_{i=1}^{J} C_{tij}}100\%$$

$$W_{tn}\frac{Annual\ gain\ of\ n\ during\ [Y_t, Y_{t+1}]}{Size\ of\ non-n\ at\ Y_t}100\% = \frac{\left[\left(\sum_{i=1}^{J} C_{tin}\right) - C_{tnn}\right]/(Y_{t+1}-Y_t)}{\sum_{j=1}^{J}\left[\left(\sum_{i=1}^{J} C_{tij}\right) - C_{tnj}\right]}100\%$$

Table 3.5. Description of mathematical notation in the formulae used (Aldwaik and Pontius 2012 in Niya et al., 2019).

Symbol	Description
T	number of time points
Y_t	year at time point t
t	index for the initial time point of an interval $[Y_t - Y_{t+1}]$, where t ranges from 1 to $T-1$
J	number of categories
i	index for a category at the initial time point of an interval
j	index for a category at the latter time point of an interval
n	index of the gaining category for the selected transition
C_{tij}	size of transition from category i to category j during interval $[Y_t - Y_{t+1}]$
S_t	annual change during interval $[Y_t - Y_{t+1}]$
G_{tj}	intensity of annual gain of category j during interval $[Y_t - Y_{t+1}]$ relative to size of category j at time $t+1$
L_{ti}	intensity of annual loss of category i during interval $[Y_t - Y_{t+1}]$ relative to size of category i at time t
R_{tin}	intensity of annual transition from category i to category n during interval $[Y_t - Y_{t+1}]$ relative to size of category i at time t
W_{tn}	uniform intensity of annual transition from all non-n categories to category n during interval $[Y_t - Y_{t+1}]$ relative to size of all non-n categories at time t

3.2.3.3. Alien plants distribution

This element follows Henderson's approach to ranking alien invasive species based on the distribution (abundance) per transects as described and illustrated in Chapter 4.

3.2.3.4. Population pressure

The study areas fall within the Waterberg District Municipality, comprising the six local district municipalities (see Chapter 1). The total area of each municipality was calculated using ArcMap 10.4, and the total population per local municipality was obtained from Statistics South Africa (StatsSA). Because the study areas were cut across several local municipalities, the ArcMap tool was used to clip parts of the local municipalities falling within our study areas. In calculating population density, the portion of the local municipality falling within our study area was divided by the actual area of the local municipality and multiplied by the total population of that area. This method assumes that the populations per local municipality are distributed across the municipality in a similar and reasonably even pattern. According to Jabbar and Chen (2006), the population density was categorized, who also calculated density based on the total area and population. The category is as follows: greater than 50 persons km^{-2} is severe, 30-50 persons km^{-2} as high, 0-30 persons km^{-2} as a medium, and 0 as low.

3.2.4 Statistical analysis

In assessing the statistical difference between the level of degradation within and outside the biosphere reserve, the chi-square test was performed using the land use/land cover change values in table 3.8 and 3.10 with respect to woody vegetation, grasslands, and bare lands. The formula used to calculate chi-square is as follows: $\chi^2 = \sum (O_i - E_i)^2/E_i$, where O_i is the observed value and E_i is the expected value.

3.3. Results

3.3.1 Accuracy assessment and spectral separability

Tables 3.6 and 3.7 illustrate the results of accuracy assessment for the classified images. The overall accuracy of the classified photos in the WBR spans all years, ranging from 94.5 to 98.6%, with Kappa coefficient values ranging from 0.93 to 0.98 percent. Outside the WBR, the overall accuracy achieved ranges between 86.2 and 96.5 %, with Kappa coefficient ranging between 0.82 and 0.96 %. Tables 3.6 and 3.7 also include the User's and producer's accuracy results, which vary between 82 and 100 percent, indicating that most image pixels were correctly classified (Richards and Jia, 2013).

The accuracy achieved when performing the error matrix technique within and outside the WBR is greater than 85 percent, with a kappa coefficient close to 1. According to Anderson et al., this shows that the obtained results meet the minimum criterion (1976).

Table 3.6 Confusion matrices performed for the area within WBR.

Year	OA (%)	KC	AT (%)	WB (%)	CF (%)	WV (%)	G (%)	BL (%)
1989	98.6	0.98	Producer's	100	100	99	99	97
			User's	95	98	99	100	100
2000	94.8	0.93	Producer's	100	93	94	94	94
			User's	100	82	97	93	100
2013	96.5	0.96	Producer's	100	90	97	96	99
			User's	100	92	97	93	100
2017	94.5	0.93	Producer's	98	95	89	94	99
			User's	100	88	97	88	100

OA= Overall Accuracy, KC= Kappa Coefficient, AT= Accuracy Type, WB= Water bodies, CF= Cultivated fields, WV= Woody vegetation G= Grassland, BL= Bare land.

Table 3.7 Confusion matrices for the area outside the WBR.

Year	OA (%)	KC	AT (%)	WB (%)	CF (%)	WV (%)	G (%)	BL (%)
1989	86.2	0.82	Producer's	83	88	71	97	91
			User's	100	83	100	72	90
2000	96.5	0.96	Producer's	95	93	99	96	99
			User's	97	90	99	97	97
2013	94.4	0.93	Producer's	93	90	91	99	97
			User's	95	92	98	91	96
2017	93.7	0.92	Producer's	98	98	87	91	99
			User's	100	87	97	88	99

OA= Overall Accuracy, KC= Kappa Coefficient, AT= Accuracy Type, WB= Water bodies, CF= Cultivated fields, WV= Woody vegetation G= Grassland, BL= Bare land.

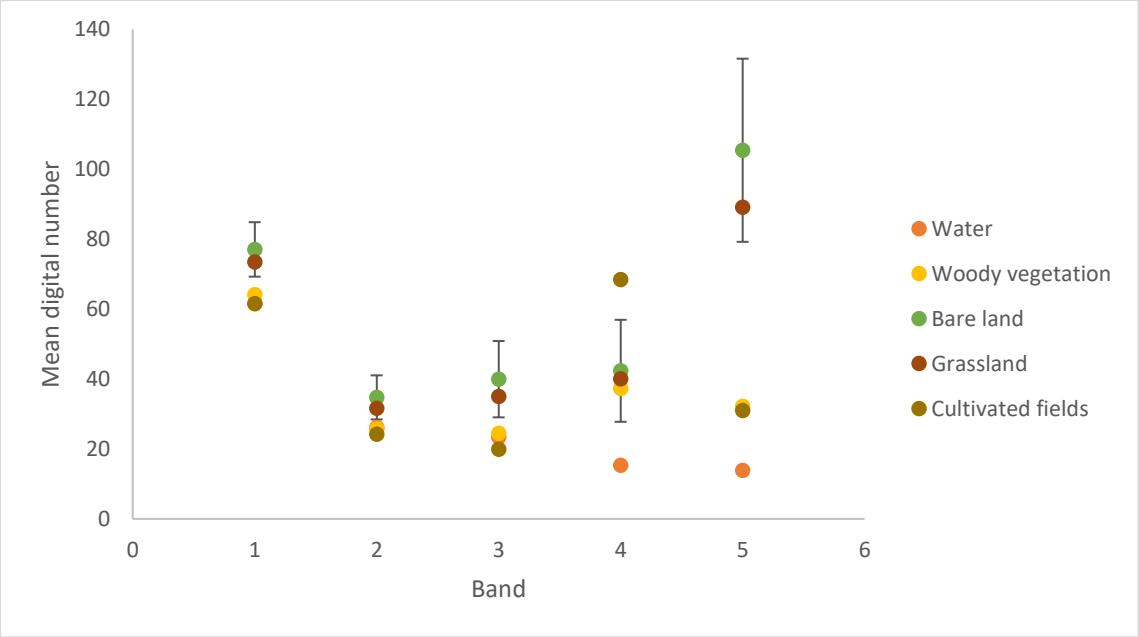


Figure 3.3. Spectral separability was performed on Landsat 8 OLI/TIRS C2 L2 of 2017.

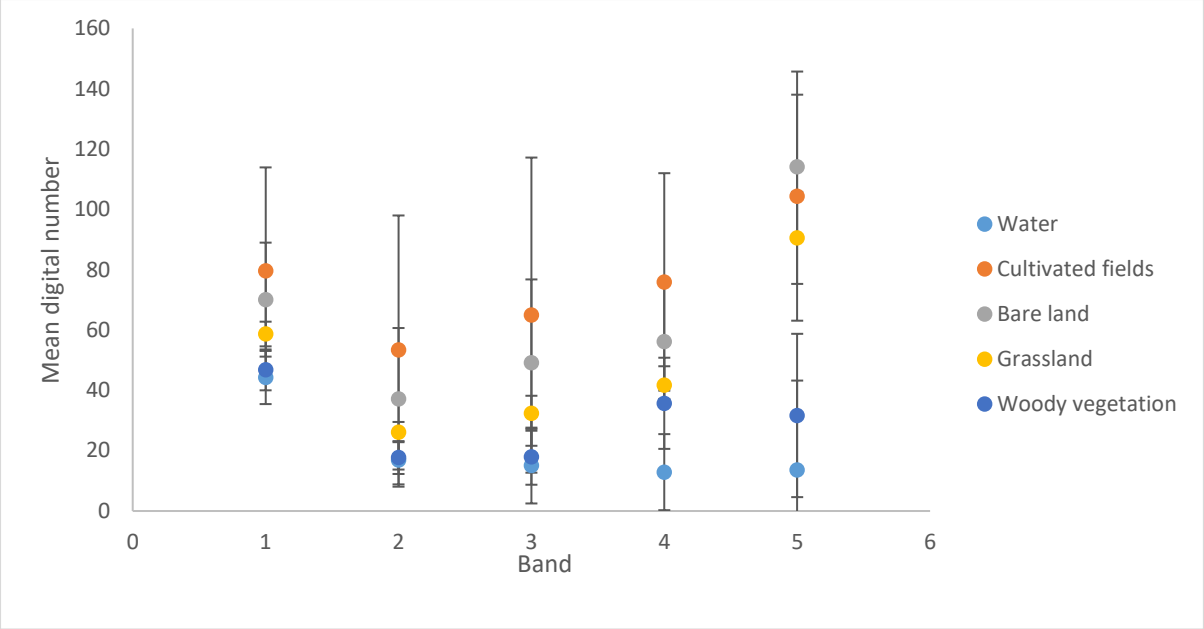


Figure 3.4. Spectral separability was performed on Landsat 4 - 5 TM C2 L2 of 1989.

From the spectral reflectance assessment, the results indicate that most land cover classes could be separated with Landsat 8 OLI/TIRS C2 L2 of 2017. However, other classes overlap. From Figures 3.3 and 3.4, it could be seen that bands 4 and 5 tend to perform better in terms of separating the classes. Where the classes overlap, re-classification was performed to improve the accuracy of the results.

3.3.2. Land use/cover change within the WBR

The Land use/cover map for assessing changes within the WBR is presented in Figure 3.5, and the percentage of land use values of land cover is in Table 3.8. The area classified as bare land was higher in 1989 with 68.89 % compared to the other years. This period predated the establishment of the WBR. By 2017, the area deemed bare land was at the lowest compared to other years with 48.9 %. This marked decline in bare land cover may be attributed to various factors such as increasing human settlement, the transition of areas into agricultural lands, and some areas that are now protected. Woody vegetation and grassland reached a peak in 2013 and 2017 with 8.5 and 43.6 %, respectively.

The year 2013 onwards saw a significant increase in cultivated fields and settlement, although the water bodies declined. The increase in cultivated fields may be attributed to the increase

in human settlements, which can be linked to the increasing population in the area. Based on the available STATSA census data, the area's population increased from 468 141 to 632 807 between 1996 and 2016 (Table 3.15). Water bodies were lower in 2013, probably due to less rainfall than 2000 (Figure 3.6); dam constructions are evident in the area and visible on the classified satellite images in proximity to cultivated sites. The highest water bodies for the year 2000 (Table 3.8) coincide with the highest rainfall received that year (Figure 3.6).

Table 3.8. Percentage of land use/land cover type within the WBR 1989 -2017.

Class	1989	%	2000	%	2013	%	2017	%
Water bodies	19,2	0,3	35,8	0,5	16,3	0,2	21,7	0,3
Cultivated fields	7,9	0,1	11,9	0,2	26,8	0,4	21	0,3
Woody vegetation	275,8	4,2	479,7	7,2	563,9	8,5	412,3	6,2
Grasslands	1748,7	26,3	1638,2	24,7	2619	39,4	2898,2	43,6
Bare lands	4581,8	68,9	4460,1	67,1	3388,1	50,9	3254,1	48,9
Settlements	11,2	0,16	18,9	0,3	30,5	0,5	37,3	0,6
Total	6644,6		6644,6		6644,6		6644,6	

Table 3.9. Percentage change of land use/ land cover between different years within the WBR.

Year	**WB (%)**	**CF (%)**	**WV (%)**	**G (%)**	**BL (%)**	**S (%)**
1989-2000	0.25	0.60	3.06	-1.73	-1.83	0.12
2000-2013	-0.29	0.22	1.27	14.76	-16.13	0.17
2013-2017	0.08	-0.09	-2.30	4.20	-2.01	0.10
1989-2017	0.04	0.20	2.05	17.29	-19.98	0.39

WB= Water bodies CF= Cultivated fields WV= Woody vegetation G= Grassland BL= Bare land S= Settlements

Within 11 years, between 1989 and 2000, bare land cover decreased slightly by 1.83 % and further decreased by 16.13 % between 2000 and 2013, as shown in Table 3.9. Over 28 years, between 1989 and 2017, the amount of bare land continued to decline markedly with an

overall reduction of almost 20 %. Woody vegetation and grassland increased 2.05 and 17.29 %, respectively, between 1989 and 2017. Although the overall woody vegetation cover increased over the years (Table 3.9, Figure 3.7), the trend between 2013 and 2017 shows a decline of 2.3 %. Although the total findings from 1989 to 2017 show a 0.2 percent rise in cultivated fields, this is equivalent to the loss in cultivated fields between 2013 and 2017.

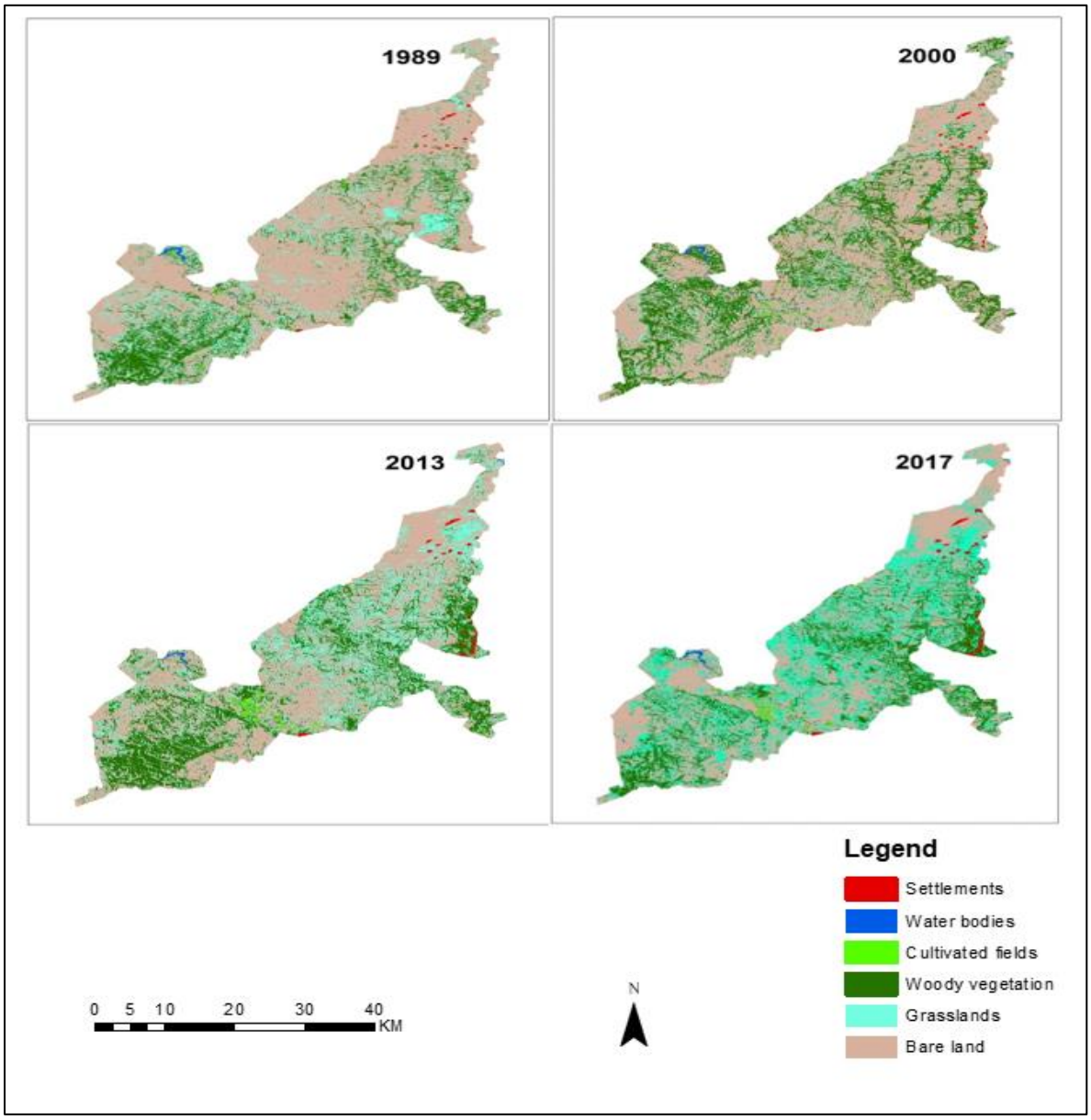


Figure 3.5. Land use/cover types within the WBR from 1989 to 2017.

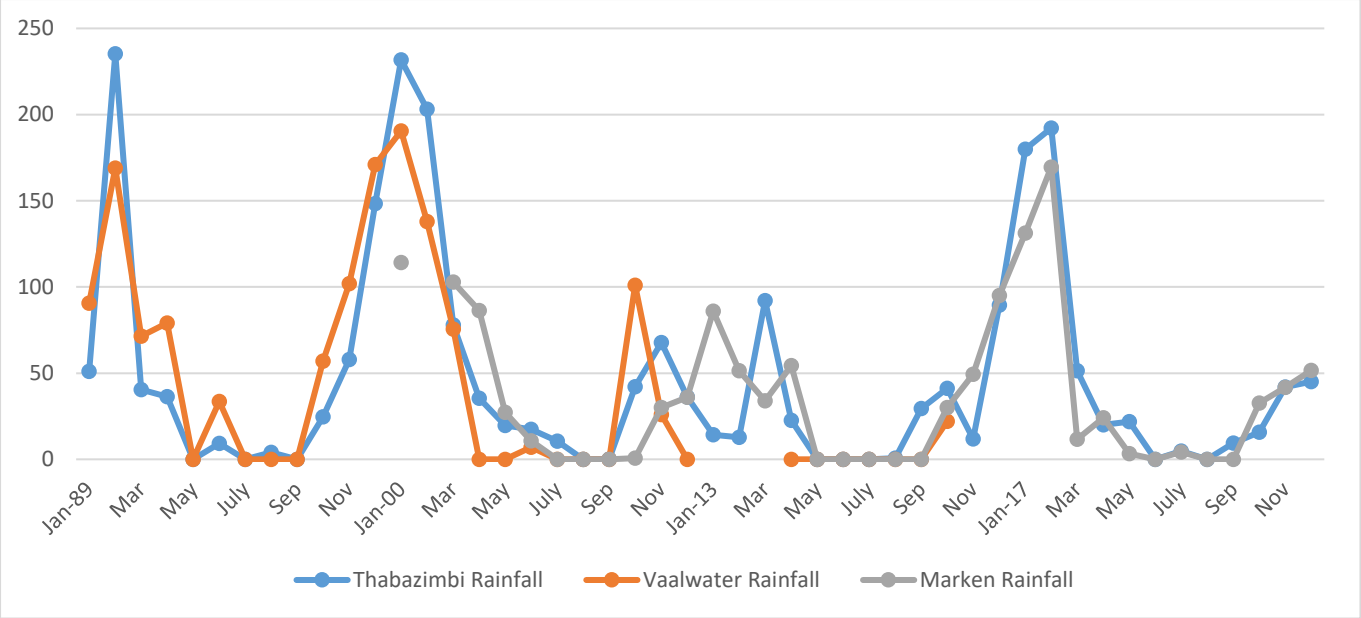


Figure 3.6. Rainfall trend (in mm) within the WBR between 1989 and 2017 for the three weather stations that fall within the WBR (Data obtained from the South African Weather Service).

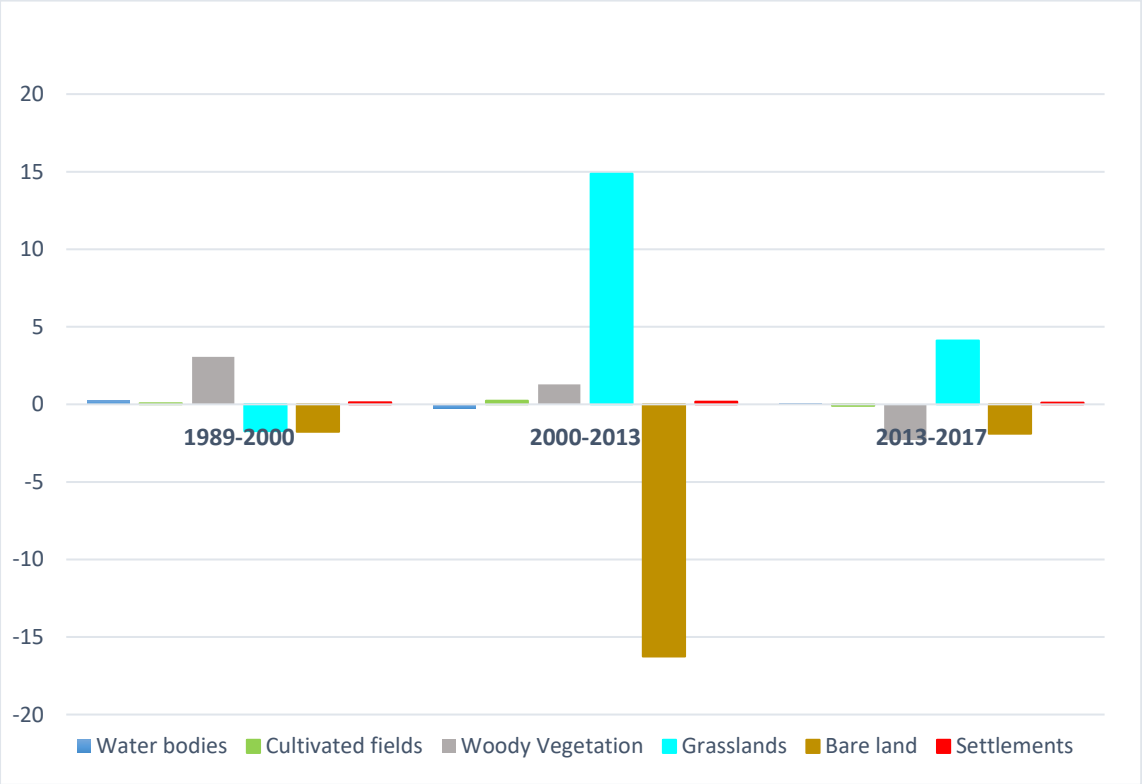


Figure 3.7. Land use/cover percentage change by time period within the WBR.

3.3.2 Land use/cover changes outside the WBR

Figure 3.8 presents the Land use/cover map for assessing the changes for the area outside the WBR, with Table 3.10 presenting the land use values attained. The amount of bare land has been increasing at an alarming rate over the years, as presented in Table 3.10. Compared with the WBR results, the initial percentage of bare land cover was lower with 21.5 % in 1989. However, this amount has since increased enormously over the years and reached a peak of 42.6 % in 2017. Compared with the WBR, mainly dominated by rural and traditional settlements, informal settlements and small towns characterize the outside area. As people flock to the small towns to seek employment coming from the rural villages, more spaces are needed for settlements. Woody vegetation cover reached a peak in 2000 with 37.4 % while the highest percentage cover for the grassland was in 1989. The woody vegetation has been declining since 2000, while the grassland increased in 2013 and declined again in 2017.

Table 3.10. Percentage of land use cover type of the area outside the WBR 1989 -2017.

Class	1989	%	2000	%	2013	%	2017	%
Water bodies	7,8	0,12	27,4	0,4	8,8	0,14	10,9	0,17
Cultivated fields	26,01	0,4	70,5	1,1	49,2	0,78	77,1	1,2
Woody vegetation	2147,8	33,9	2370,01	37,4	1566,2	24,7	1476,4	23,3
Grasslands	2782,6	43,9	2332,5	36,8	2351,6	37,2	2033,7	32,1
Bare lands	1359,7	21,5	1521,59	24	2316,7	36,6	2693,1	42,6
Settlements	5,09	0,08	7	0,11	36,5	0,5	37,8	0,59
Total	6329		6329		6329		6329	

The area outside the WBR recorded more water bodies with 0.4 % in 2000 (Table 3.10). During that year, the area experienced the highest rainfall compared to other years. Although the area recorded the highest rainfall (Figure 3.9), the percentage cover of cultivated fields was slightly less compared to 2017, where the water bodies were less than the year 2000. According to the local communities, extensive flooding made it difficult to practice agriculture during that year. These views are supported by Khandlhela and May (2006) on the impacts

of 2000 floods in communities of Limpopo and how people's livelihoods were affected. Settlements also expanded markedly during 2017 from 0.11 % in 2000 to 0.59 % in 2017.

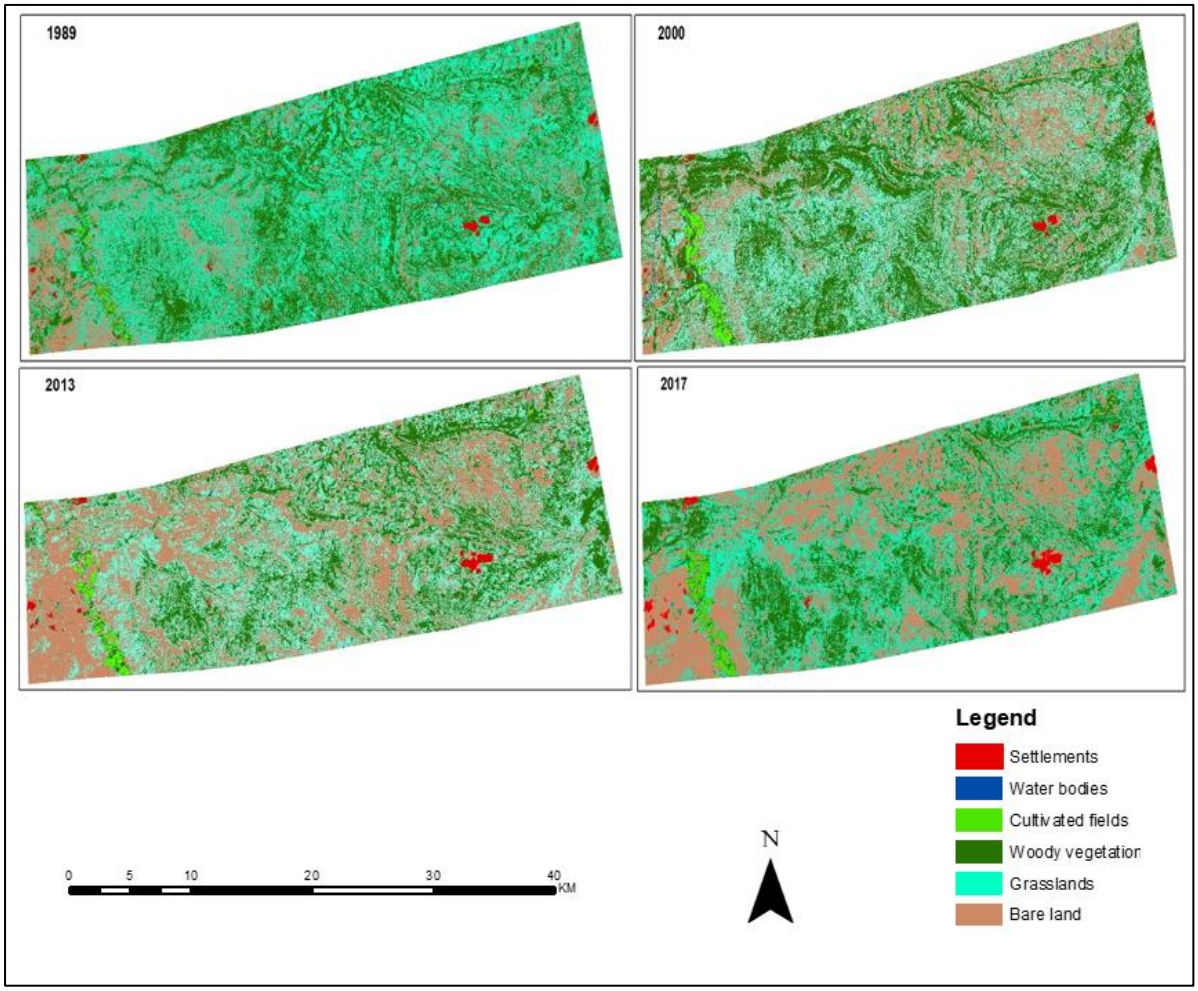


Figure 3.8. Land use/cover types outside the WBR from 1989 to 2017.

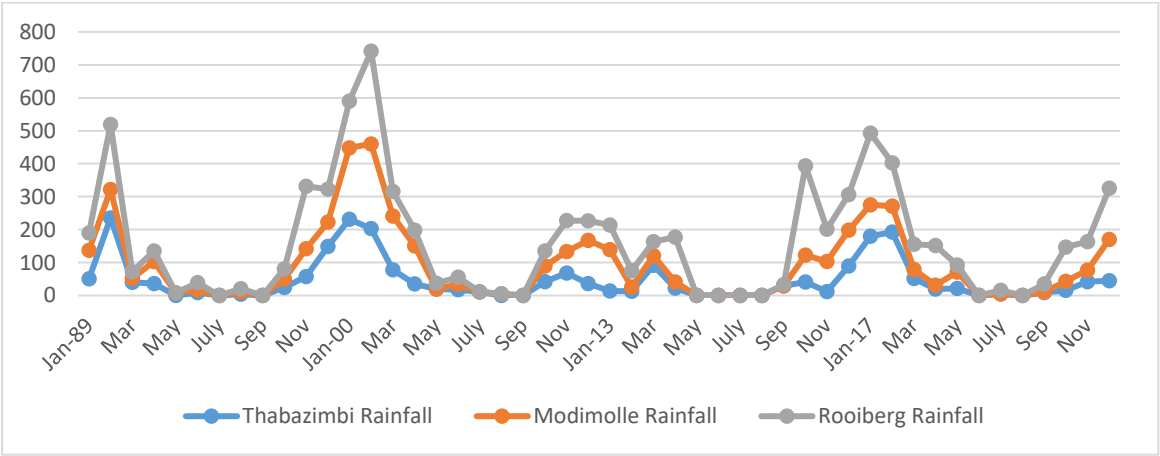


Figure 3.9. Rainfall trend (in mm) of the study area outside the WBR between 1989 and 2017 (Data obtained from the South African Weather Service).

Over 28 years, the area of bare land increased by 21.18 % between 1989 and 2017 (Table 3.11; Figure 3.10) outside the WBR. The highest change occurred between 2000 and 2013, during which time the amount of bare land increased by 12.49 %. During this period, woody vegetation also reduced by 12.57 %, although the area experienced heavy rainfalls (Figure 3.9). While the vegetation can help regulate the volume of water runoff and vegetation growth, extreme rainfalls can stress the vegetation (see Abera et al., 2018). The overall results for woody vegetation and grasslands reveal that this area experienced a substantial loss in vegetation cover between 1989 and 2017 (Table 3.11). The cultivated fields, which are occurring mainly adjacent to the rivers, increased between 1989 and 2000. This increase is also noticeable in water bodies between those years. The overall results show that the area outside the WBR experienced a high loss of woody vegetation and grassland and a significant increase in the amount of bare land over the study period.

Table 3.11 Percentage change of Land use/cover between different years outside the Waterberg biosphere.

Year	WB (%)	CF (%)	WV (%)	G (%)	BL (%)	S (%)
1989-2000	0.31	0.70	3.38	-7.24	2.75	0.10
2000-2013	-0.29	-0.33	-12.57	0.43	12.49	0.28
2013-2017	0.03	0.44	-1.42	-5.02	5.95	0.02
1989-2017	0.05	0.81	-10.61	-11.83	21.18	0.40

WB= Water bodies CF= Cultivated fields WV= Woody vegetation G= Grassland BL= Bare land S= Settlements

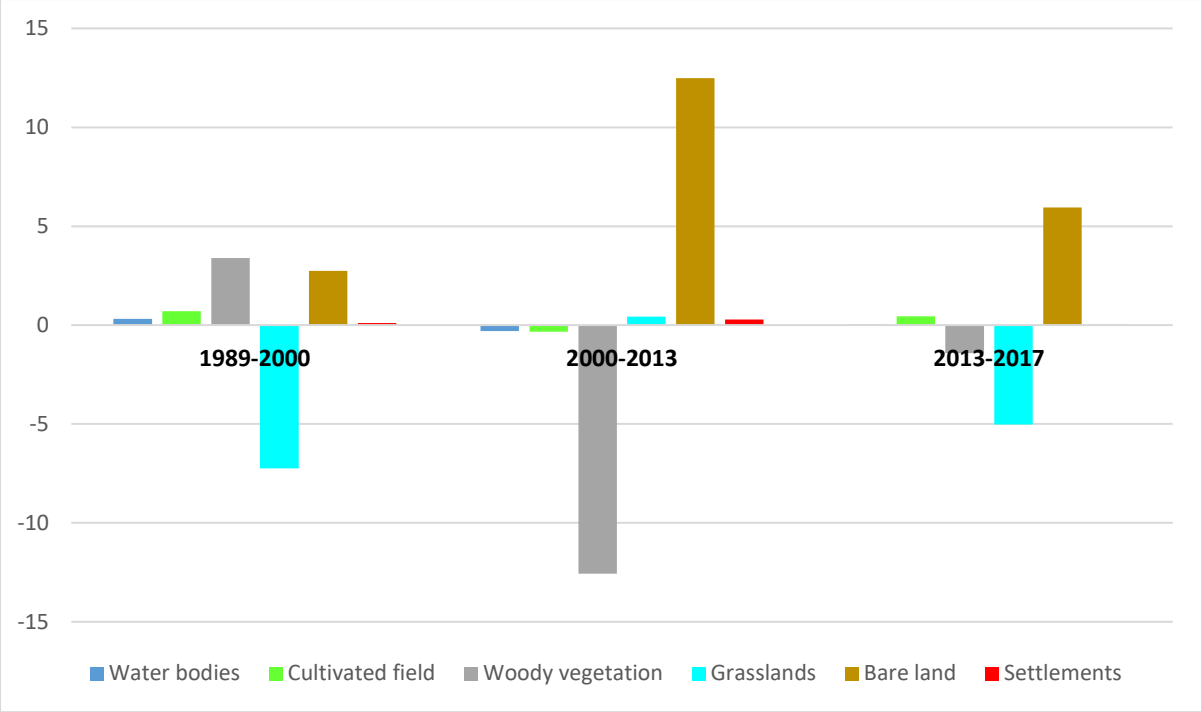


Figure 3.10. Land use/cover percentage change by time period outside the WBR.

3.3.4 Land degradation assessment within and outside the WBR

3.3.4.1. Vegetation loss within the WBR

Table 3.12 presents the vegetation types as per the 2018 VegMap within the WBR. Limpopo Sweet Bushveld, which had the least vegetation cover, experienced the highest vegetation loss. The vegetation was lost and solely to bare land. The Western Sandy Bush was the second most impacted vegetation type, with a loss of more than a quarter of the initial cover. Much of the vegetation was lost to bare land, cultivated fields, and in some instances to water bodies.

Similarly, the Roodeberg Bushveld, Central Sandy Bushveld, and Waterberg-Magaliesberg Summit vegetation types lost almost a quarter of the initial cover to bare land, cultivated fields, water bodies, and settlements. Makhado Sweet Bushveld and Waterberg Mountain Bushveld experienced no vegetation loss but gained fair amounts of vegetation. Apart from the vegetation falling within the highly protected part of the WBR (core zones), some vegetation that is still intact is found in areas not easily accessible due to the landscape.

Table 3.12. Percentage of vegetation change within the WBR.

Vegetation type	% Initial vegetation cover	% Remaining vegetation cover	% Vegetation loss	Severity level
Central Sandy Bushveld	20.39	17.92	2.47	Low
Limpopo Sweet Bushveld	0.059	0.001	0.057	Severe
Makhado Sweet Bushveld	1.57	2.38	0	None
Roodeberg Bushveld	14.64	11.14	3.50	Low
Waterberg Mountain Bushveld	53.08	61.76	0	None
Waterberg-Magaliesberg Summit Sourveld	2.61	2.16	0.46	Low
Western Sandy Bushveld	7.63	4.63	3.0	Low

3.3.4.2. Land-use intensity within the WBR

Figure 3.11 and Table 3.13 demonstrate the intensity of the observed transitions of woody vegetation and grassland to other land uses, based on the gained percentage between 1989 and 2017 illustrated in Table 3.22 and 3.23, section 3.4. The bar that passes the uniform line indicates the gain of the category targeting the losing category. In contrast, the bar that stops before the uniform line indicates that the gaining category avoids the losing category. As demonstrated in Figure 3.11, the transformation of woody vegetation was active at cultivated fields, whereas all classes were dormant at the grassland. Intensity Analysis indicates that woody vegetation was systematically targeted by cultivated fields while being avoided by other classes. Figure 3.11 illustrates that much of the grassland areas were transitioned into woody vegetation between 1989 and 2017, although all classes are dormant.

Table 3.13. Percentages of land-use intensity by transition level within the WBR, 1989-2017.

From woody vegetation to others	
Class	%
Bare land	0.04
Cultivated fields	0.26
Grasslands	0.16
Water bodies	0.001
From grassland to others	
Bare land	0.72
Cultivated fields	0.74
Woody vegetation	1.36
Water bodies	0.64



Figure 3.11. Transition intensity analysis within the WBR 1989 - 2017.

3.3.4.3. Invasive alien plants abundance within the WBR

The species abundance ratings (number of species per surveyed plot) followed Henderson (2007). However, Henderson's survey was on the national scale applying the Quarter Degree Square. Each abundance was expressed in a number of individual alien plants per 5 km transect. A total number of 66 plots recorded different categories of invasions. The highest number of plots contains only one species per plot, with 28 plots recording up to five individual species and only two plots recording more than five species per plot (Table 3.14).

Table 3.14. The abundance of surveyed invasive alien plants per plot within the WBR (method adapted from Henderson (2002)).

No. of invasive plants per plot	Total no. of plots with species	Category of invasion
One species	33	Light invasion
Up to 5 species	28	Moderate invasion
Greater than 5 species	2	High invasion
Total number of plots with species	66	

3.3.4.4. Population density within the WBR

The population density within the Waterberg Biosphere (Table 3.15) shows Mogalakwena Local Municipality as the most densely populated. Thabazimbi, Lephalale, and Modimolle are less densely populated based on the total population and area occupied. Table 3.16 illustrates the population change for all local municipalities falling within our study areas between 1996 and 2016. It should be noted that the municipality boundaries have been changing over the years. Therefore, the population density was calculated based on the current (2016) municipal boundary.

Table 3.15. Percentage of population density within the WBR based on 2016 census data (StatsSA).

Local Municipality	Area 1 (sq_km)	Total no. of people per area	Area 2 (sq_km)	Population density	% Population density
Thabazimbi	9847	96232	130	1270	1.02
Lephalale	19595	136626	3226	22493	18.02
Modimolle	4271	36655	1635	14032	11.24
Mogalakwena	6184	328905	1636	87013	69.72
Total				124809	100

Area 1= the total area of each local municipality; **Area 2**=the total area of the portion of the local municipality falling within the study area.

Table 3.16. Total population per local municipality falling within the WBR, 1996-2016 (StatsSA).

	Population				
Local Municipality	1996	2001	2011	2016	Total area (sq_km) per local municipality boundary
Thabazimbi	60175	65533	85234	96232	9847
Lephalale	78715	85272	115767	136626	19595
Mogalakwena	282534	298439	307682	328905	6184
Modimolle	46717	69027	68513	71044	6226
Total area	**468141**	**518271**	**577196**	**632807**	

3.3.4.5. Vegetation loss outside the WBR

Outside the WBR, the Dwaalboom Thornveld vegetation type was the most strongly affected, losing almost half of the vegetation cover (Table 3.17). The vegetation was lost to bare land, cultivated fields, settlements, and water bodies. Springbokvlakte Thornveld, Waterberg-Magaliesberg Summit Sourveld, and Western Sandy Bushveld were the second most affected vegetation types with a loss of less than a quarter of the initial cover. Similarly, with the Dwaalboom Thornveld, the Springbokvlakte, Thornveld, and Western Sandy Bushveld were

lost to bare land, cultivated fields, settlements, and water bodies. The Waterberg-Magaliesberg Summit Sourveld vegetation, on the other hand, was merely lost to bare land. Several vegetation types such as the Central Sandy Bushveld, Madikwe Dolomite Bushveld, Subtropical Alluvial Vegetation, and the Waterberg Mountain Bushveld experienced no degradation. The private nature and game reserves in the area could be the factor behind the vegetation that is still intact in other areas.

Table 3.17. Percentage of vegetation change outside the WBR.

Vegetation type	% Initial vegetation cover	% Remaining vegetation cover	% Vegetation loss	Severity level
Central Sandy Bushveld	36.46	38.91	0	none
Dwaalboom Thornveld	11.16	6.23	4.93	severe
Madikwe Dolomite Bushveld	0.07	0.10	0	none
Springbokvlakte Thornveld	4.43	3.38	1.04	low
Subtropical Alluvial Vegetation	0.10	0.14	0	none
Waterberg Mountain Bushveld	33.77	37.50	0	none
Waterberg-Magaliesberg Summit Sourveld	1.17	0.94	0.24	low
Western Sandy Bushveld	12.82	12.79	0.03	low

3.3.4.6. Land-use intensity outside the WBR

Figure 3.12 and Table 3.18 show that all land uses play an active role in transforming woody vegetation. However, grassland, bare land, woody vegetation, and cultivated fields are more active with only the dormant waterbodies.

Table 3.18. Percentages of land-use intensity by transition level outside the WBR, 1989-2017.

From woody vegetation to others	
Class	%
Bare land	0.94
Cultivated fields	0.91
Grasslands	1.19
Water bodies	1.37
From grassland to others	
Bare land	1.59
Cultivated fields	1.14
Woody vegetation	1.39
Water bodies	0.01

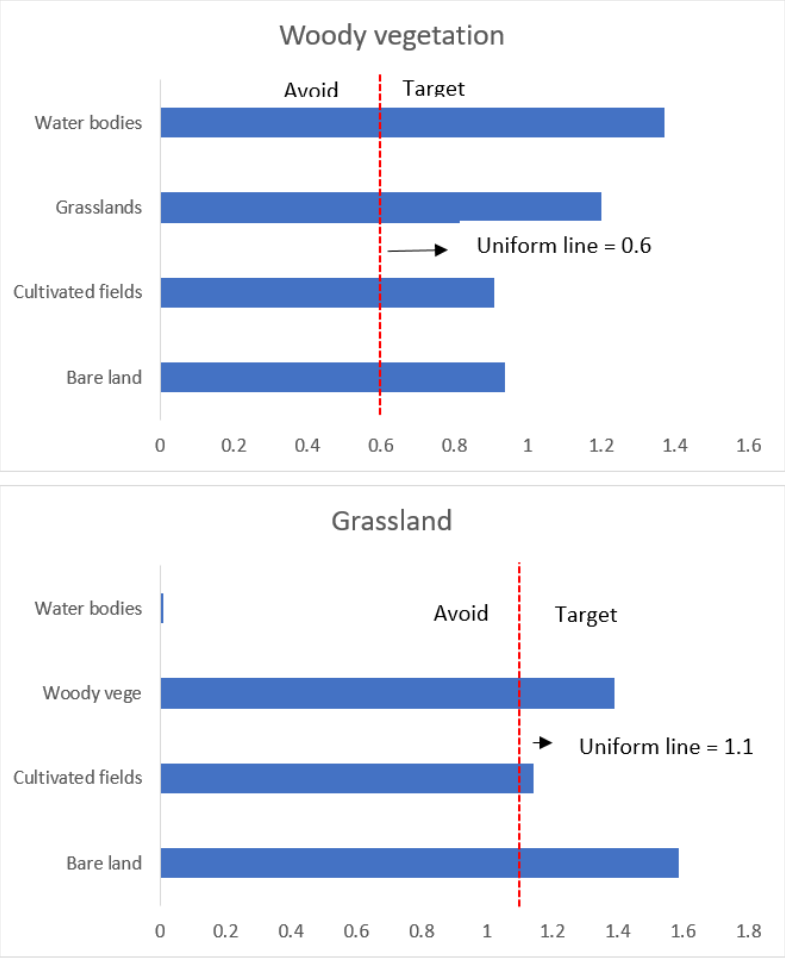


Figure 3.12. Transition intensity analysis outside the WBR 1989 - 2017.

3.3.4.7. Invasive alien plants abundance outside the WBR

Outside the WBR, a total number of 87 plots recorded different categories of invasions. The highest number of plots contains up to five species per plot, followed by only one species. Only seven plots have recorded more than five species per plot (Table 3.19).

Table 3.19. The abundance of surveyed alien plants per plot outside WBR (method adapted from Henderson (2002).

No. of invasive plants per plot	Total no. of plots with species	Category of invasion
One species	27	Light invasion
Up to 5 species invasion	53	Moderate
Greater than 5 species	07	High invasion
Total number of plots with species	8 7	

3.3.4.8. Population density outside the WBR

Outside the WBR, Table 3.20 indicates that Modimolle, Bela-Bela, and Thabazimbi have a higher population density than Mookgopong. The changes in population over the years are illustrated in Table 3.21. Similar to the WBR, this area's population density was calculated based on the current municipal boundary.

Table 3.20. Population density outside the WBR based on 2016 census data (StatsSA).

Local Municipality	Area 1 (sq_km)	Total no. of people per area	Area 2 (sq_km)	Population density	% Population density
Thabazimbi	9847	96232	2218	21676	28.0
Mookgopong	4271	36655	717	6154	7.9
Modimolle	6226	71044	2416	27569	35.6
Bela-Bela	3375	76296	978	22109	28.5
Total				77507	100

Area 1= the total area of each local municipality; **Area 2**=the total area of the portion of the local municipality falling within the study area.

Table 3.21. Total population per local municipality falling within our study areas outside the WBR, 1996-2016 (StatsSA).

	Population				
Local Municipality	1996	2001	2011	2016	Total area (sq_km) per local municipality boundary
Thabazimbi	60175	65533	85234	96232	9847
Mookgopong	15674	34541	35640	36655	4271
Modimolle	46717	69027	68513	71044	6226
Bela-Bela	47592	52124	66500	76296	3375
Total area	**170158**	**221225**	**255887**	**280227**	

3.3.4.9. Chi-squared tests results

The computed value (Chi-square) for the significant difference between inside and outside the biosphere reserve with specific to degradation level was 967500.08. Using the Chi-square distribution table, a P-value of 0,05 and degree of freedom of 2 shows a significant relationship between the land degradation level within and outside the biosphere reserve with respect to woody vegetation, grasslands, and bare land.

3.4 Discussion

3.4.1 Land use/ land cover change pattern

It is commonly perceived that biodiversity is well protected from human activities once an area has been designated as formally protected (Liu et al., 2001). However, this common perception does not always hold true, as protected areas are also susceptible to change over time due to various factors. In the Vhembe Biosphere Reserve of Limpopo Province, Evans (2017) applied the landcover map of South Africa to map Land use/cover for 23 years (1990 - 2013). The results revealed that much vegetation loss arose from human activities such as fuelwood collection, commercial mining, and infrastructure development. Coetzer et al. (2010) applied a similar approach in the Kruger to Canyon Biosphere Reserve, with the results also indicating human activities resulting in vegetation loss in parts of the reserve.

The results for this study indicate that parts of the WBR experienced vegetation loss at some point but still recovered. While woody vegetation and grassland underwent regeneration over 28 years, the grassland seems to have fared better. Approximately 17.23 % of grassland and 2.04 % of woody vegetation were regained over 28 years. This positive trend of increasing vegetation is commonly referred to as greening, while browning refers to diminishing

vegetation (Melvin, 2019; de Jong et al., 2012). Recent studies have estimated that substantial drylands areas globally are undergoing greening (see Fensholt et al., 2012; Bai et al., 2008) with a correspondingly marked decline in browning (Saha et al., 2015). Results presented here for the area within the WBR concur with these studies, although the situation outside the reserve boundary is less favourable. Rapid greening within the WBR was detected within the core and buffer zones, whereas greening was only associated with private nature reserves outside the reserve. The increase in browning outside the WBR is especially evident in the Thabazimbi area (western part of Figure 3.8), where industrial mining activities are located. Waste heaps can be observed in areas where mining activities are no longer occurring, indicating that land rehabilitation has not been undertaken after mine closure. Loss of vegetation due to human settlement and other activities, e.g., removing vegetation for road constructions, was evident during the field survey in both areas. The expansion of towns into rural landscapes typically results in rapid alteration of the surrounding natural environment (Kjelland et al., 2007). These changes are notable in our study areas, especially where human settlements are growing rapidly. This is not surprising as the government has been providing housing through Reconstruction Development Programme since 1994. Formal settlements within the WBR, such as Leseding, Raphuti, and Rooiberg, were established through these subsidy houses from the government (Marcatelli, 2015). With the increasing population of the Waterberg (Tables 3.20 and 3.21), it can be expected that such settlements will impact the environment (e.g., Brandt et al., 2017).

The temporal analysis demonstrates that over 28 years, the areas of browning increased outside the WBR (Figure 3.8, Table 3.11), while areas of greening decreased. Specifically, some 716 hectares of the area covered with woody vegetation and 1213 hectares of grassland transitioned into bare land. Human activities such as fuelwood, harvesting grass for roof thatching, and branches for the manufacture of brooms may underlie the loss of grassland and woody vegetation in the area. Fuelwood is the main energy source in many rural households of Limpopo, which is harvested from far and close surroundings for cooking, lighting, and space heating (Scheepers et al., 2013). Twine et al. (2003) conducted a study in the rural villages of Limpopo, which showed that savanna grasses and woody vegetation are still harvested and used for roofing and for producing hand and twig brooms.

Nonetheless, human activities cannot be singled out as the only driver of vegetation loss in the semi-arid savanna. Natural stressors such as climate (rainfall or recurrent drought) can also influence the greening or browning of an area. The above-average rainfall is likely to increase woody vegetation cover at a local scale (Kgosikoma and Mogotsi, 2013). Mucina and Rutherford (2006) found the woody vegetation of the semi-arid savanna of South Africa to be abundant where there is greater availability of water. The statements support findings in this study, where heavy rainfall in 2000 may have contributed to the increase in woody vegetation in both study areas. However, rainfall can have different effects on vegetation, depending on environmental conditions such as slope and soil texture (Tietjen, 2016). Although such conditions were not explicitly measured in this study, more woody vegetation cover was observed on steeper slopes. Of course, it could be argued that accessibility may be restricting human activities from occurring in those areas. Muller et al. (2016) also found events such as a severe frost to affect the abundance of savanna woody vegetation, subject to the area's topography. Woody vegetation loss was severe at the base of the slope, with 60 -100 percent loss within the broader savanna and Nama-Karoo biomes of South Africa (Muller et al., 2016). With no impacts recorded at the highest elevation, such impacts declined with elevation (Muller et al., 2016).

Drought affects vegetation cover by suppressing the grassland cover while stimulating woody vegetation cover (Abate and Angassa, 2016; Skarpe, 1990). Such occurrences are most likely to contribute to a reduction in grassland cover, thereby stimulating bush encroachment. Bush encroachment and degradation of rangelands area are associated with a large density of cattle in Africa (Kgosikoma and Mogotsi, 2013). They can affect the structure and functioning of the ecosystem (Van Auken, 2009). Herbivory and disturbance events such as fire are also recognized as mechanisms that woody vegetation responds to (Sebata, 2017). Bush encroachment is a severe problem that farmers are concerned about within the WBR, particularly private farms and nature reserves (Henderson, 2013). Sicklebush (*Dichrostachys cinerea)* is one the most common species that encroach most areas of the Waterberg (Henderson, 2013). Figure 3.13 displays dense Sicklebush in the D'Nyala Game Reserve in the Waterberg. Sicklebush has many benefits to human livelihoods, including use for fuelwood and medicine. Evans (2017) highlighted that bush encroachment encourages overexploitation of resources for fuelwood use in the Vhembe Biosphere Reserve. Moleele et al. (2002) found

bush encroachment in the savanna of Botswana to be in areas concentrated with a high density of grazers. The possible explanation could be the dispersal of seeds from the encroacher plants by grazers (Kgosikoma and Mogotsi, 2013). Within the region of the Waterberg, various wild grazers and browsers such as wildebeest, giraffe, warthog, kudu, impala, and klipspringer characterize the area form part of the game farming industry. De Klerk (2003) highlighted livestock farming as one of the causes of vegetation loss in the Waterberg due to overgrazing. Although game farming provides fewer employment opportunities (Spierenburg and Brooks, 2014), in the 1970s, farmers began shifting their interest from crop farming due to a lack of government subsidies which only favored livestock production (Marcatelli, 2015; Krug, 2001). Ecotourism in the Waterberg contributes to the local economy through game farming, with the annual income increase from R9 million to R303 million between 1991 and 2010 (Snijders, 2012; Farmers Weekly, 2012). Water shortages in the area are also another factor that stimulated the shift from crop to game farming (Seaman et al., 2013).



Figure 3.13. Area encroached by Sicklebush (*Dichrostachys cinerea)* plants in the D'Nyala Game Reserve, Waterberg District (Mangani et al. 2020).

Yet again, the change detection matrix (Tables 3.22 and 3.23) illustrates that substantial amount of cultivated fields within and outside the WBR transitioned into bare land. The observation above regarding the shift from crop to animal farming is a possible observation for these results. Natural vegetation was the second most common landcover type that cultivated fields were lost to, with gain in grassland and woody vegetation within and outside

the WBR, respectively. Cultivated fields are often ploughed after harvesting, leaving only exposed soil. However, vegetation is likely to develop on the abandoned land but can struggle to establish (Chaudhary et al., 2018; Drake, 1991; Baeten et al., 2010). Jacovac et al. (2021) identified this process of vegetation regrowth as secondary succession, which is likely to depend on the intensity of land-use history (Dawson et al., 2016). Pioneer species such as grasses and weeds are likely to be the first to grow and, under favourable conditions, such fields can transform into shrublands and possibly develop into woodlands (Nije-Bijvank, 2015). It should be noted that native and alien vegetation were not distinguished regarding land use and vegetation mapping.

Vegetation was also established in areas previously covered with water bodies, while some areas that occupied surface water were converted for cultivation in both study areas. Freshwater sources such as wetlands, lakes, and rivers play an important role in sustaining human livelihoods, yet many are still threatened due to human activities. The studies by Phethi and Gumbo (2019); Golele and Gumbo (2016) emphasized the threat to wetlands in Limpopo Province due to agricultural operations. The conversion was primarily prompted by population increase and rising food consumption (Phethi and Gumbo, 2019). Alien plant invasion is another danger to water supplies and biodiversity (Brundu, 2015).). The alien plants are widely distributed, particularly within the farms and water bodies in the Waterberg (Henderson, 2013). According to Henderson (2013), farmers are less concerned with these invasive alien plants but more concerned with bush encroachment on the grazing sites.

Table 3.22. Change detection matrix of Land use/cover change outside the WBR between 1989 and 2017.

		1989					
	Class	BL (Ha)	CF (Ha)	G (Ha)	WB (Ha)	WV (Ha)	Row Total (Ha)
2017	Bare land	791.93	20.7	380.3	1.8	171.9	13664.63
	Cultivated fields	6.6	11.9	2.7	0.05	4.8	26.0
	Grassland	1213.5	24.7	966.03	3.3	574.0	2781.5
	Water bodies	4.1	0.2	1.3	1.5	0.8	7.8
	Woody vegetation	716.2	19.6	682.4	4.19	724.5	2146.9
	Class Total	**2732.33**	**77.2**	**2032.8**	**10.9**	**1475.9**	**6328.83**

WB= Water bodies, CF= Cultivated fields, WV= Woody vegetation G= Grassland, BL= Bare land

In comparison, the WBR shows an overall recovery in vegetation from 1989 to 2017 and a decline in bare land (Figure 3.5 and Table 3.9). Although activities such as stone digging and sand mining have resulted in the loss of vegetation, particularly in the transition zone, the trend in vegetation cover indicates that many of the areas within the WBR have been recovering. While the causes of greening remain vague, some studies broadly associate increase in greening with water availability (Saha et al., 2015; Hickler et al., 2005), changes in atmospheric CO_2 concentration (Donohue et al., 2013), and in some instances changing composition of plant communities influenced by other drivers (Bai et al., 2008). The increase in greening within the WBR can be attributed to some of the policies implemented for biodiversity protection, such as the National Environmental Management: Protected Areas Act 2003, the National Environmental Management: Biodiversity Act 10 of 2004, which includes the invasive Alien Species regulations that deal with exotic species management.

According to regulations, it is compulsory for the WBR to meet the principles of sustainable development (land use.unesco.org/mab/doc/faq/brs.pdf), thus balancing conservation while improving the socio-economic conditions of the adjacent community.
It should be noted that some of the activities that result in vegetation loss, such as stone digging and sand mining, generate income for many households in the area. In interaction with the local communities of the Waterberg, unemployment was identified as one of the most important problems facing the community. The residents highlighted that scarce jobs offered by farm owners are seasonal and only for a limited number of people. Therefore, such mining activities are the primary source of income for most residents. The major employer in the area is the agricultural sector. However, some residents find employment in the commercial mining sectors mainly located outside the WBR (Waterberg District Municipality, 2017). The increasing commercial mining raises concerns about an area already experiencing water shortages (Ruiters and Matji M.P., 2015). Tables 3.22 and 3.23 demonstrate that large areas of the WBR retained and gained waterbodies (largely from manmade dams) as opposed to outside where industrial mining practices are growing.

The settlement class that was manually digitized due to having the same spectral reflectance as bare land was not included in the change detection matrix analysis. This means the bare land in the change detection matrix includes the human settlements. In many rural areas of

Limpopo, the vegetation is usually removed in the yards for safety concerns. It is believed that dense vegetation offers cover for criminal activities (Kuo and Sullivan, 2001). The low-lying vegetation is also removed for safety from the creeping animals, and in most instances, this symbolises cleanliness, a point raised by residence during our interaction. As a result, the cleared spaces and some r-painted roofing influenced pixel separation during digital classification. Hence settlements were manually digitized.

Table 3.23. Change detection matrix of land use/land cover within the WBR between 1989 and 2017.

		1989					
	Class	**BL (Ha)**	**CF (Ha)**	**G (Ha)**	**WB (Ha)**	**WV (Ha)**	**Row Total (Ha)**
2017	**Bare land**	2580.2	13.9	1839.1	5.6	148.8	4587.6
	Cultivated fields	3.5	0.8	3.05	0.02	1.1	8.6
	Grasslands	666.6	4.29	920.4	3.9	156.8	1752.1
	Water bodies	4.03	0.28	2.48	11.5	1.1	19.4
	Woody vegetation	38.0	1.5	132.9	0.7	104.0	277.1
	Class Total	**3292.33**	**20.8**	**2897.9**	**21.7**	**411.8**	**6644.8**

WB= Water bodies, CF= Cultivated fields, WV= Woody vegetation G= Grassland, BL= Bare land

3.4.2 Mapping the land degradation severity at a local scale within and outside the WBR

Vegetation or soil degradation in drylands is widely believed to be a severe threat to sustainable human livelihoods. However, maps showing the extent and severity of degradation at a local scale are limited. In South Africa, Hoffman et al. (1999) mapped the extent of land degradation. Since then, several studies have been analysing the underlying effects of land degradation at a national scale (see Hoffman and Ashwell, 2001; Meadows and Hoffman, 2002; Meadows and Hoffman, 2003; Hoffman and Todd, 2000). In Limpopo Province, the assessment and mapping of land degradation on a local scale were performed by Botha and Fouche (2000), applying remote sensing of vegetation and local knowledge in the Sekgosese District. The findings suggested that people's movement concerning government policy was the driving force of land degradation in the Sekgosese District.

Land degradation or improvement is mainly the consequence of interlinked direct and underlying causes, including natural, socio-economic, and other related activities. Various factors play a significant role in the land degradation occurring within the Waterberg region, including direct and indirect causes. The underlying causes include population pressure, poverty, and a lack of education or awareness. The primary component of land degradation in this study is vegetation loss. The area with the most severe land degradation (Figure 3.14) is linked to more intensive human activity. The large areas showing degradation, particularly within the WBR, are mainly concentrated near settlements and land previously used for subsistence farming. Old, abandoned fields still exist in the Waterberg with varying impacts (de Klerk, 2003).

Most of these areas were established in the colonial period when the European settlers appropriated the land for commercial farming, and indigenous people were confined to what is currently known as communal lands (Marcatelli, 2015; Wessels et al., 2008). This probably means a high population in a limited area exerts pressure on the surrounding natural environment. However, after the colonial period, a series of new acts dismantled movement restrictions and opened a number of opportunities and land tenure (Robinson, 2003). Furthermore, substantial changes were made throughout South Africa after 1994, resulting in the adjustment of demarcations (Robinson, 2003), thus, allowing more people to occupy larger areas.

Higher population density is not always coincident with greater levels of degradation. Large areas of the WBR experienced medium degradation irrespective of population density in that area. Outside the reserve, the area with medium population density is associated with severe degradation, while other areas with low and high populations exhibit medium degradation. This indicates that population pressure is not the only driver of resources exploitation. People travel to exploit resources where they are available and accessible. Overexploitation of resources in areas of lower population density can occur, for instance, where vegetation resources are impacted for commercial purposes or under grazing pressure. The assumption was that population is evenly distributed within both study areas when associated with land degradation severity (amount of vegetation lost).

Although exploited vegetation can recover, the level of threat varies with the intensity of land use. Intensity analysis in this study reveals that cultivated fields played an active role in the loss of woody vegetation within and outside the WBR. Commercial and subsistence crop farming has been practised over the years in the area. As previously indicated, the crop farming practice has been shifting over the years in Waterberg (Marcatelli, 2015) and Southern Africa (Bryceson, 2000; Bryceson, 2002), though it is still the largest employer in the Waterberg area, followed by commercial or industrial mining (Waterberg District Municipality, 2017). The Waterberg area has important mineral zones and remains the supplier of coal for generating fossil fuel electricity in South Africa and other neighbouring countries (Department of Mineral Resources, 2009). According to the Department of Mineral Resources (2009), mining in this area is considered as a replacement for South Africa's depleted Mpumalanga Highveld coalfield electricity supply. These mining activities are particularly occurring within the periphery of the area outside the WBR.

While mining is the most significant contributor to the Gross Domestic Product within the Waterberg area (Waterberg District Municipality, 2017), the increasing mining industry in the area raises concerns regarding the ecological impacts and social well-being (de Klerk, 2003). The large area outside the WBR is experiencing severe degradation where industrial mining is the major activity. Severe scars are left on the fragile environment, such as compaction of topsoil outside the WBR. Due to intensive mining, the decline in vegetation covers exposes the topsoil to adverse soil erosion, making it impossible to grow (Huang et al., 2015). Further, the toxins released from coal mining can negatively affect vegetation near the mining areas, making it impossible to absorb carbon (Abdul-Wahab and Marikar, 2012).
Moreover, the burning of spoil heaps also causes vegetation damage. Other negative impacts of coal mining are on both ground and surface water. The Oliphant River in Mpumalanga was severely impacted by mining activities (de Klerk, 2003). Similar impacts, such as soil compaction and vegetation loss, are experienced within the WBR, where small-scale mining (sand mining and stone digging) is practiced. Although small-scale mining is recognised as promoting local and rural development and providing social relief to many (Department of Mineral Resources, 2009), some challenges are still faced within the WBR where communities practice mining without mining rights and appropriate tools. Illegal sand and stone mining

without proper planning causes further harm to the environment by removing indigenous species that keep soil intact and causing erosion.

Protected areas, including private nature reserves, exhibited lower vegetation loss both within and outside the WBR (Personal observation during all field visits between 2017 and 2019). The natural vegetation in these areas is preserved as it forms part of the ecological management for tourism in the nature reserves (Bredenkamp and Brown, 2001). However, invasive alien weeds still threaten some of these conservation sites. One of the conservation sites visited during the survey was highly infested with pompom weed, *Campuloclinium macrocephalum*. This weed threatens the conservation of grasslands, wetlands and open savanna (arc.agric.za/arc-ppri/Pages/Pompom%20weed/Pompom-Weed.aspx). The survey on the abundance of invasive alien plants at a finer scale shows that most species were recorded at the highly disturbed sites, in this case, roadsides at the major roads connecting towns and near human settlements. Invasive alien plants are more aggressive on disturbed land with exposed soil than on areas with good vegetation cover (Hoffman and Ashwell, 2001). Moreover, invasive alien plants can reduce land quality, exposing it to degradation (Carter et al., 2015).

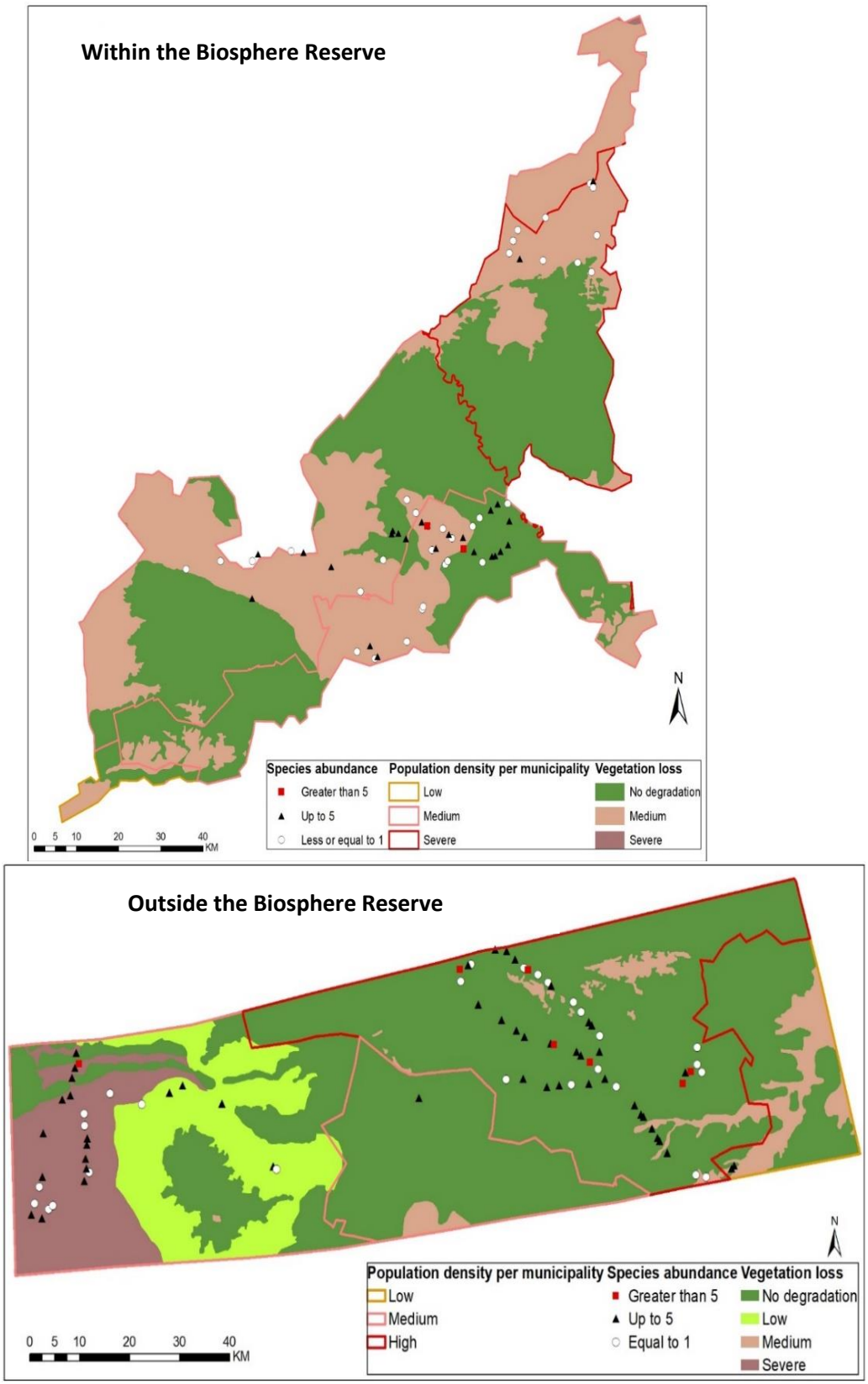


Figure 3.14. The map shows land degradation and the diverse alien plant species recorded within and outside the WBR.

3.4.3. Comparing land degradation within and outside the WBR

The evaluation of land degradation severity within and outside the WBR is based on the land use/cover change analysis performed for both study areas. The medium level of degradation in reserve is consistent with the conservation initiatives undertaken in the area. According to Marcatelli (2015), the conservation practices in the Waterberg were initiated in the 1980s by wealthy whites businesspersons and farmers who were self-proclaimed conservationists. The initiative was established to rescue land damaged by overgrazing and return the bush to its natural status. These efforts resulted in the proclamation of certain areas as WBR. However, returning the area back to its natural context had advantages only for some community members. Although the WBR contributed to increasing international recognition of the Waterberg, some locals are still questioning the work done by Non-Governmental Organisations who are administering it (Marcatelli, 2015). Residents within the WBR also raised this concern during our field survey. The construction of fences and introduction and protection of, in some cases, dangerous wild animals, particularly on private properties, limited the mobility of people and their ability to practice livestock farming due to limited access to grazing sites (Marcatelli, 2015). The fences also separated farms and towns/formal settlements. People were removed and relocated due to poor living conditions within their farms, which supported the philosophy of separating humans and nature (Hall et al., 2013).

Both within and outside the WBR, areas exhibiting no degradation (greening) are mostly private and public natural reserves. Medium levels of degradation are exhibited more frequently around human settlements in the WBR. Outside the reserve, severe degradation is exhibited more where intense industrial mining is associated with settlements. This pattern is also demonstrated by the water bodies where visible manmade dams used for conserving and managing water are in private farms and nature reserves. According to Marcatelli (2015), this unregulated practice allows farmers to secure water supply while poor rural communities experience water shortages. However, the decrease in browning within the WBR did not result only from the proclamation of the Biosphere Reserve. The conservation initiatives that commenced in the 1980s may have played a role. Mundalamo's (2019) analysis of the WBR over a 32-year period indicated that vegetation growth happened in the WBR even before it was classified as a UNESCO protected area, supporting this claim. Human activity (both legal and illicit) outside the reserve, on the other hand, may have accelerated catastrophic

vegetation loss, particularly in rural towns, where young people from rural communities rush for work prospects.

3.5. Conclusion

Changes have been detected between the years in both study areas from the Land use/cover maps produced in this chapter. The study showed that there had been considerable changes in land use/cover change in both areas. Although the results show that land degradation was higher within the WBR than the outside area initially, over the years, the WBR recovered. In contrast, the quality of land in the area outside the WBR has been deteriorating. Conclusion on land recovery within the WBR could be made on the proclamation of the area as the Biosphere Reserve by UNESCO in 2001, where other parts of the Biosphere Reserve were zoned as core, with high restriction of human activities. Therefore, certain activities such as sand mining, stone digging, fuelwood collection, and human settlements are only permitted in the transition zone where the degradation is still visible. Outside the WBR, commercial mining activities seem to be the primary role player in vegetation loss. Additionally, are growing informal settlements as people flock to such areas for employment. Therefore, population pressure could also be concluded as the contributor to vegetation loss outside the WBR.

3.6. References

Abate, T. and Angassa, A., 2016. Conversion of savanna rangelands to bush dominated landscape in Borana, Southern Ethiopia. *Ecological Processes,* 5 (1): 6.

Abdul-Wahab, S. and Marikar, F., 2012. The environmental impact of gold mines: pollution by heavy metals. *Open engineering*, *2*(2), pp.304-313.

Abera, T.A., Heiskanen, J., Pellikka, P. and Maeda, E.E., 2018. Rainfall–vegetation interaction regulates temperature anomalies during extreme dry events in the Horn of Africa. *Global and Planetary Change*, *167*, pp.35-45.

Adam, E., Mutanga, O., Odindi, J. and Abdel-Rahman, E.M., 2014. Land-use/cover classification in a heterogeneous coastal landscape using RapidEye imagery: Evaluating the performance of random forest and support vector machines classifiers. *International Journal of Remote Sensing*, *35*(10), pp.3440-3458.

Alemu, B., 2015. The effect of land use land cover change on land degradation in the highlands of Ethiopia. *Journal of Environmental Earth Sciences*, *5*(1), pp.1-12.

Aldwaik, S.Z. and Pontius Jr, R.G., 2012. Intensity analysis to unify measurements of size and stationarity of land changes by interval, category, and transition. *Landscape and Urban Planning*, *106*(1), pp.103-114.

Anderson, J.R., 1976. *A Land Use and Land Cover Classification System for Use with Remote Sensor Data* (Vol. 964). US Government Printing Office.

Baeten, L., Velghe, D., Vanhellemont, M., De Frenne, P., Hermy, M. and Verheyen, K., 2010. Early trajectories of spontaneous vegetation recovery after intensive agricultural land use. *Restoration Ecology*, *18*, pp.379-386.

Bai, Z.G., Dent, D.L., Olsson, L. and Schaepman, M.E., 2008. Proxy global assessment of land degradation. *Soil Use and Management*, *24*(3), pp.223-234.

Bajocco, S., De Angelis, A., Perini, L., Ferrara, A. and Salvati, L., 2012. The impact of land use/land cover changes on land degradation dynamics: a Mediterranean case study. *Environmental Management*, *49*(5), pp.980-989.

Botha, J.H. and Fouche, P.S., 2000. An assessment of land degradation in the Northern Province from satellite remote sensing and community perception. *South African Geographical Journal*, *82*(2), pp.70-79.

Brandt, M., Rasmussen, K., Peñuelas, J., Tian, F., Schurgers, G., Verger, A., Mertz, O., Palmer, J.R. and Fensholt, R., 2017. Human population growth offsets climate-driven increase in woody vegetation in sub-Saharan Africa. *Nature, Ecology & Evolution*, *1*(4), pp.1-6.

Bredenkamp, G.J. and Brown, L.R., 2001. Vegetation–A reliable ecological basis for environmental planning. *Urban Greenfile Nov-Dec*, *2001*, pp.38-39.

Brundu, G., 2015. Plant invaders in European and Mediterranean inland waters: profiles, distribution, and threats. *Hydrobiologia*, 746(1), pp.61-79.

Bryceson, D., 2000. Rural Africa at the Crossroads: Livelihood Practices and Policies. London: Overseas Development Institute.

Bryceson, D.F., 2002. The scramble in Africa: reorienting rural livelihoods. *World Development*, *30*(5), pp.725-739.

Carter, E.T., Eads, B.C., Ravesi, M.J. and Kingsbury, B.A., 2015. Exotic invasive plants alter thermal regimes: implications for management using a case study of a native ectotherm. *Functional Ecology*, 29(5), pp.683-693.

Chaudhary, S., Wang, Y., Khanal, N.R., Xu, P., Fu, B., Dixit, A.M., Yan, K., Liu, Q. and Lu, Y., 2018. Social impact of farmland abandonment and its eco-environmental vulnerability in the high mountain region of nepal: A case study of dordi river basin. *Sustainability*, *10*(7), p.2331.

Chunjing, Y., Yueyao, Z., Yaxuan, Z. and Liu, H., 2017. Application of convolutional neural network in classification of high resolution agricultural remote sensing images. *The International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences*, *42*, pp 989-992.

Coetzer, K.L., Erasmus, B.F., Witkowski, E.T.F. and Bachoo, A.K., 2010. Land-cover change in the Kruger to Canyons Biosphere Reserve (1993-2006): a first step towards creating a conservation plan for the subregion. *South African Journal of Science*, *106*(7-8), pp.1-10.

Congalton, R.G., 1991. A review of assessing the accuracy of classifications of remotely sensed data. *Remote Sensing of Environment*, *37*(1), pp.35-46.

Congalton, R. G and Green, K., 2008. *Assessing the Accuracy of Remotely Sensed Data: Principles and Practices.* New York: Taylor and Francis.

Dare, P.M., 2005. Shadow analysis in high-resolution satellite imagery of urban areas. *Photogrammetric Engineering & Remote Sensing,* 71(2), pp.169-177.

Dawson, S.K., Fisher, A., Lucas, R., Hutchinson, D.K., Berney, P., Keith, D., Catford, J.A. and Kingsford, R.T., 2016. Remote sensing measures restoration successes, but canopy heights lag in restoring floodplain vegetation. *Remote Sensing*, *8*(7), p.542-.

de Jong, R., Verbesselt, J., Schaepman, M.E. and De Bruin, S., 2012. Trend changes in global greening and browning: contribution of short-term trends to longer-term change. *Global Change Biology*, *18*(2), pp.642-655.

de Klerk, A., 2003. *The Waterberg Biosphere Reserve:* a Land Use Model for Ecotourism Development (Doctoral dissertation, University of Pretoria PRETORIA).

Department of Mineral Resources (DMR), 2009. Developments in the Economic Contribution of Hydrocarbons, Natural Gas and Coal. Report No. 78/2009. Department of Mineral Resources, Pretoria, South Africa.

Donohue, R.J., Roderick, M.L., McVicar, T.R. and Farquhar, G.D., 2013. Impact of Carbon-dioxide (CO^2) fertilization on maximum foliage cover across the globe's warm, arid environments. *Geophysical Research Letters*, *40*(12), pp.3031-3035.

Drake, J.A., 1991. Community-assembly mechanics and the structure of an experimental species ensemble. *The American Naturalist*, *137*(1), pp.1-26.

Evans, S.W., 2017. An assessment of land cover change as a source of information for conservation planning in the Vhembe Biosphere Reserve. *Applied Geography*, *82*, pp.35-47.

Evans, J. and Geerken, R., 2004. Discrimination between climate and human-induced dryland degradation. *Journal of Arid Environments*, *57*(4), pp.535-554.

Farmer's Weekly., 2012. http://www.farmersweekly.co.za/article.aspx?id=22178&h=Talking-bullwith-Cyril-Ramaphosa.

Fensholt, R., Langanke, T., Rasmussen, K., Reenberg, A., Prince, S.D., Tucker, C., Scholes, R.J., Le, Q.B., Bondeau, A., Eastman, R. and Epstein, H., 2012. Greenness in semi-arid areas across the globe 1981–2007—an Earth Observing Satellite based analysis of trends and drivers. *Remote Sensing of Environment*, *121*, pp.144-158.

Gashu, K. and Gebre-Egziabher, T., 2018. Spatiotemporal trends of urban land use/land cover and green infrastructure change in two Ethiopian cities: Bahir Dar and Hawassa. *Environmental Systems Research*, *7*(1), pp.1-15.

Golele, F. and Gumbo, J.R., 2016. Investigation of Ecological Impacts of Agricultural Activities in Tshitavhadudlu wetland, Limpopo Province. International Conference on Advances in Science, Engineering, Technology and Natural Resources (ICASETNR-16) Nov. 24-25, 2016.

Hall, R., Wisborg, P., Shirinda, S. and Zamchiya, P., 2013. Farm Workers and Farm Dwellers in Limpopo Province, South Africa. *Journal of Agrarian Change*, *13*(1), pp.47-70.

Henderson, L., 2007. Invasive, naturalized and casual alien plants in southern Africa: a summary based on the Southern African Plant Invaders Atlas (SAPIA). *Bothalia*, *37*(2), pp.215-248.

Henderson, L., 2013. SAPIA newsletters are posted at the ARC website: www.arc.agric.za. Volume No. 28.

Hickler, T., Eklundh, L., Seaquist, J.W., Smith, B., Ardö, J., Olsson, L., Sykes, M.T. and Sjöström, M., 2005. Precipitation controls Sahel greening trend. *Geophysical Research Letters*, *32*(21), pp. 21415-

Hoffman, M.T. and Ashwell, A., 2001. Nature Divided: Land Degradation in South Africa. University of Cape Town Press.

Hoffman, M.T. and Todd, S., 2000. A national review of land degradation in South Africa: the influence of biophysical and socio-economic factors. *Journal of Southern African Studies*, *26*(4), pp.743-758.

Hoffman M.T, Todd S, Ntshona Z and Turner S., 1999. Land Degradation in South Africa. Department of Environment Affairs and Tourism, Pretoria.

Jabbar, M.T. and Chen, X., 2006. Land degradation assessment with the aid of geo-information techniques. *Earth Surface Processes and Landforms*, *31*(6), pp.777-784.

Jakovac, C.C., Junqueira, A.B., Crouzeilles, R., Peña-Claros, M., Mesquita, R.C. and Bongers, F., 2021. The role of land-use history in driving successional pathways and its implications for the restoration of tropical forests. *Biological Reviews*, 94(4), pp. 1114-1134.

Kgosikoma, O.E. and Mogotsi, K., 2013. Understanding the causes of bush encroachment in Africa: The key to effective management of savanna grasslands. T*ropical Grasslands-Forrajes Tropicales*, 1(2), pp.215-219.

Khandlhela, M. and May, J., 2006. Poverty, vulnerability and the impact of flooding in the Limpopo Province, South Africa. *Natural Hazards*, 39(2), pp.275-287.

Kjelland, M.E., Kreuter, U.P., Clendenin, G.A., Wilkins, R.N., Wu, X.B., Afanador, E.G. and Grant, W.E., 2007. Factors related to spatial patterns of rural land fragmentation in Texas. *Environmental Management*, 40(2), pp.231-244.

Krug, W., 2001. Private supply of protected land in southern Africa: a review of markets, approaches, barriers and issues. World Bank/OECD International Workshop on Market Creation for Biodiversity Products and Services, Paris (Vol. 25).

Kuo, F.E. and Sullivan, W.C., 2001. Environment and crime in the inner city: Does vegetation reduce crime?. *Environment and Behavior*, *33*(3), pp.343-367.

Lacroix, A., Hortobagyi, T., Beurskens, R. and Granacher, U., 2017. Effects of supervised vs. unsupervised training programs on balance and muscle strength in older adults: a systematic review and meta-analysis. *Sports Medicine*, *47*(11), pp.2341-2361.

Liu, J., Linderman, M., Ouyang, Z., An, L., Yang, J. and Zhang, H., 2001. Ecological degradation in protected areas: the case of Wolong Nature Reserve for giant pandas. *Science*, *292*(5514), pp.98-101.

Mangani, T., Coetzee, H., Kellner, K. and Chirima, G., 2020. Socio-Economic Benefits Stemming from Bush Clearing and Restoration Projects Conducted in the D'Nyala Nature Reserve and Shongoane Village, Lephalale, South Africa. *Sustainability*, *12*(12), p.5133.

Marcatelli, M., 2015. Suspended redistribution: 'green economy 'and water inequality in the Waterberg, South Africa. *Third World Quarterly*, 36(12), pp.2244-2258.

Meadows, M.E. and Hoffman, T.M., 2003. Land degradation and climate change in South Africa. *Geographical Journal*, *169*(2), pp.168-177.

Meadows, M.E. and Hoffman, M.T., 2002. The nature, extent and causes of land degradation in South Africa: legacy of the past, lessons for the future?. *Area*, *34*(4), pp.428-437.

Melvin A., 2019. Understanding Northern Latitude Vegetation Greening and Browning: Proceeding of Workshop. National Academies of Science, Engineering and Medicine Washington, DC: The National Academies Press.

Moleele, N.M., Ringrose, S., Matheson, W. and Vanderpost, C., 2002. More woody plants? The status of bush encroachment in Botswana's grazing areas. *Journal of Environmental Management*, 64(1), pp.3-11.

Mucina, L., Rutherford, M.C. and Powrie, L.W., 2006. Vegetation Atlas of South Africa, Lesotho and Swaziland. The Vegetation of South Africa, Lesotho and Swaziland. *Strelitzia* 19, pp, pp.748-789.

Muller, K., O'Connor, T.G. and Henschel, J.R., 2016. Impact of a severe frost event in 2014 on woody vegetation within the Nama-Karoo and semi-arid savanna biomes of South Africa. *Journal of Arid Environments*, 133, pp.112-121.

Mundalamo, T.D., 2019. Characterizing historic land use and land cover change using Landsat earth observation data in the Waterberg biosphere reserve. Master's dissertation, *University of Witwatersrand.*

Nije-Bijvank, M.J.M., 2015. Natural revegetation of abandoned cultivated land in the Spanish Pyrenees and its effect on streamflow. Master's dissertation, University Utrecht.

Niya, A. K, Huang, J., Karimi, H., Keshtkar, H. and Naimi, B., 2019. Use of intensity analysis to characterize land use/cover change in the biggest island of Persian Gulf, Qeshm Island, Iran. *Sustainability*, *11*(16), p.4396.

Otukei, J.R. and Blaschke, T., 2010. Land cover change assessment using decision trees, support vector machines and maximum likelihood classification algorithms. *International Journal of Applied Earth Observation and Geoinformation*, *12*, pp.S27-S31.

Paul, A., 1997. Rule-based classification of water in Landsat MSS images using the variance filter. *Photogrammetric Engineering & Remote Sensing*, *63*(5), p.485491.

Phethi, M.D. and Gumbo, J.R., 2019. Assessment of impact of land use change on the wetland in Makhitha village, Limpopo province, South Africa. *Jamba: Journal of Disaster Risk Studies*, *11*(2), pp.1-6.

Pontius, R.G., Gao, Y., Giner, N.M., Kohyama, T., Osaki, M. and Hirose, K., 2013. Design and interpretation of intensity analysis illustrated by land change in Central Kalimantan, Indonesia. *Land*, *2*(3), pp.351-369.

Puig, C.J., Hyman, G. and Bolaños, S., 2002. Digital classification vs visual interpretation: a case study in humid tropical forests of the Peruvian Amazon. *Proceedings of 29th International Symposium on Remote Sensing of the Environment, pp. 8-12 Buenos Aires, Argentina.*

Reed, M.S. and Stringer, L.C., 2016. Land Degradation, Desertification and Climate Change: Anticipating, Assessing and Adapting to Future Change. London: Routledge.

Richards, J.A and Jia, X., 2013. Supervised classification techniques. Supervised Classification Techniques, Remote Sensing Digital Image Analysis: an Introduction (pp. 247-318). Springer, Berlin, Heidelberg.

Robinson, P.S., 2003. Implications of rural settlement patterns for development: A historical case study in Qaukeni, Eastern Cape, South Africa. *Development Southern Africa*, *20*(3), pp.405-421.

Ruiters, C. and Matji, M.P., 2015. Water institutions and governance models for the funding, financing and management of water infrastructure in South Africa. *Water SA*, *41*(5), pp.660-676.

Rwanga, S.S. and Ndambuki, J.M., 2017. Accuracy assessment of land use/land cover classification using remote sensing and GIS. *International Journal of Geosciences*, *8*(04), p.611.

Saha, M.V., Scanlon, T.M. and D'odorico, P., 2015. Examining the linkage between shrub encroachment and recent greening in water-limited southern Africa. *Ecosphere*, *6*(9), pp.1-16.

Scheepers, H., Kotze, N. and Scheepers, L., 2013. The environmental impact of fuelwood use by households in Bela-Bela Township, South Africa. *Life in a Changing Urban Landscape*, *1*, p.328.

Seaman, M.T., Watson, M., Avenant, M.F., Joubert, A.R., King, J.M., Barker, C.H., Esterhuyse, S., Graham, D., Kemp, M.E., Le Roux, P.A. and Prucha, B., 2013. *The Mokolo River Case Study*. Water Resource Commission, South Africa.

Sebata, A., 2017. Ecology of woody plants in African savanna ecosystems. *Plant Ecology-Traditional Approaches to Recent Trends*, pp.25-35. DOI: 10.5772/intechopen.69865.

Shahtahmassebi, A., Yang, N., Wang, K., Moore, N. and Shen, Z., 2013. Review of shadow detection and de-shadowing methods in remote sensing. *Chinese Geographical Science*, *23*(4), pp.403-420.

Skarpe, C., 1990. Shrub layer dynamics under different herbivore densities in an arid savanna, Botswana. *Journal of Applied Ecology*, 27(3), pp.873-885.

Spierenburg, M. and Brooks, S., 2014. Private game farming and its social consequences in post-apartheid South Africa: contestations over wildlife, property and agrarian futures. *Journal of Contemporary African Studies*, 32(2), pp.151-172.

Snijders, D., 2012. Wild property and its boundaries–on wildlife policy and rural consequences in South Africa. *Journal of Peasant Studies*, 39(2), pp.503-520.

Symeonakis, E. and Drake, N., 2004. Monitoring desertification and land degradation over sub-Saharan Africa. *International Journal of Remote Sensing*, *25*(3), pp.573-592.

Tietjen, B., 2016. Same rainfall amount different vegetation—How environmental conditions and their interactions influence savanna dynamics. *Ecological Modelling*, 326, pp.13-22.

Topaloğlu, R.H., Sertel, E. And Musaoğlu, N., 2016. Assessment of classification accuracies of Sentinel-2 and Landsat-8 Data for land cover/use mapping. *International Archives of the Photogrammetry, Remote Sensing & Spatial Information Sciences*, *41,* pp. 1055 -1059.

Twine, W., Moshe, D., Netshiluvhi, T. and Siphugu, V., 2003. Consumption and direct-use values of savanna bio-resources used by rural households in Mametja, a semi-arid area of Limpopo province, South Africa. *South African Journal of Science*, *99*(9), pp.467-473.

Van Auken, O.W., 2009. Causes and consequences of woody plant encroachment into western North American grasslands. *Journal of Environmental Management*, *90*(10), pp.2931-2942.

Waterberg District Municipality., 2017. *Integrated Development Plan* (2017-2022).

Wessels, K.J., Prince, S.D. and Reshef, I., 2008. Mapping land degradation by comparison of vegetation production to spatially derived estimates of potential production. *Journal of Arid Environments*, *72*(10), pp.1940-1949.

Yiran, G.A.B., Kusimi, J.M. and Kufogbe, S.K., 2012. A synthesis of remote sensing and local knowledge approaches in land degradation assessment in the Bawku East District, Ghana. *International Journal of Applied Earth Observation and Geoinformation*, *14*(1), pp.204-213.

Zhou, W., Huang, G., Troy, A. and Cadenasso, M.L., 2009. Object-based land cover classification of shaded areas in high spatial resolution imagery of urban areas: A comparison study. *Remote Sensing of Environment*, *113*(8), pp.1769-1777.

CHAPTER 4. ALIEN PLANTS DISTRIBUTION WITHIN AND OUTSIDE THE WBR

4.1 Introduction

Alien invasive plants are considered to be a threat to the conservation of biodiversity worldwide. In South Africa, alien plants are abundant and especially noticeable along roadsides. In rural areas, gravel and paved roads are bordered by natural and semi-natural vegetation, including various land use/cover types (Dawson, 1991). Therefore, it is crucial to understand the distribution and abundance of these species to assess their impacts on ecosystems. The relative frequency of alien plants along the roadsides could also indicate the potential of alien plants to naturalise in various land uses or favourable climatic conditions (Milton and Dean, 1998). This chapter explores the problem of alien plant invasion in the study area, initially through a description of methods and presenting results and discussion. The chapter addresses objective two of the thesis, *viz*. to document and assess the distribution (occurrence and abundance) and types of invasive and alien plant species within and outside the WBR, focusing on selected main roads and Land use/cover types.

4.2 Methods

4.2.1 Fieldwork phase

Due to their high level of disturbance, roads are a primary facilitator of the ongoing spread of alien plants and are principal entry ports and corridors for invasives to colonise adjacent natural vegetation (Mortensen et al., 2009; Slodowicz et al., 2018). Roads were selected as indicators of the magnitude of the alien invasive vegetation problem in the study area, as not only are they important vector pathways for such species but also are easily accessible for sampling (see Rahlao et al., 2010; Kalwij et al., 2008; Milton and Dean, 1998). The data used for this chapter were collected during autumn 2018 and 2019, within and outside the WBR.

Prior to the commencement of fieldwork, a desktop study was conducted using Google Earth and the 1:50 000 topographical maps to record the various road types in the area. The site visit was followed by field observation of the planned roadways for assessment. The frequency of vehicle traffic was observed, selecting areas for assessment and dividing them

into divisions as part of the desktop task. The distance traveled was computed on each route, and the number of plots to be sampled was determined. On a portable GPS, the GPS coordinates for each suggested starting point and probable sampling sites were marked to identify their locations in the field. The plot size was calculated to be 15 meters by 15 meters. The key invasive species in the research area are distribution and abundance, namely *Lantana camara, Jacaranda mimosifolia, Cereus jamacaru, Melia azedarach,* and *Eucalyptus camaldulensis*, as first reported by Henderson (2013) for the Waterberg, were recorded as primary data. These species belong to the following families: Verbenaceae, Bignoniaceae, Cactaceae, Meliaceae, and Myrtaceae, respectively. Nevertheless, other alien plant species found along the roadside and within the plots were also recorded. Others identified along the road verges even if not falling within the sampled plots.

A single observer drive-by method was used to survey the road and record the alien plants present, initially at a fixed interval of 5 km. A similar method was applied by Kalwij et al. (2018) when assessing the road verges as invasion corridors in the Karoo region of South Africa. Rahlao et al. (2010) also applied the drive-by method when assessing the alien weed fountain grass (*Pennisetum setaceum)*. Due to the species occurrence frequency, it was decided during field sampling to reduce the survey interval to 2 km along some roads to improve resolution and accuracy. In all, sampling was conducted along 17 roads (10 roads within the WBR and seven roads outside the WBR). Of all the roads surveyed, eight were paved/tarred roads, and nine were gravel roads. Figure 4.1 provides a schematic representation of the roadside survey method applied. Plots were established on both sides of the road using the same units of measurement.

Following the setting out of plot size with a measuring tape (Figure 4.2), GPS coordinates of the centre of the road were recorded. Land use /land cover types adjoining road verges were categorised as either natural vegetation, cultivation fields, or settlements. The cultivated fields category includes both cultivated and uncultivated areas in the survey (they had similar functions). Vegetation type was observed as well as the condition of the roads in terms of maintenance. Finally, any indications of alien plant clearance were noted since the authorities had been engaged in alien vegetation removal in the area. Within the WBR, most of the roads surveyed were located within the transition zone due to the greater density of main roads

found within that zone. Most roads located within the buffer zone were either not main roads or were only infrequently used. The core zones of the Biosphere Reserve also had limited main roads and are protected for conservation purposes, environmental education activities, and research. There were no accessibility constraints encountered in sampling plots outside the Biosphere Reserve.

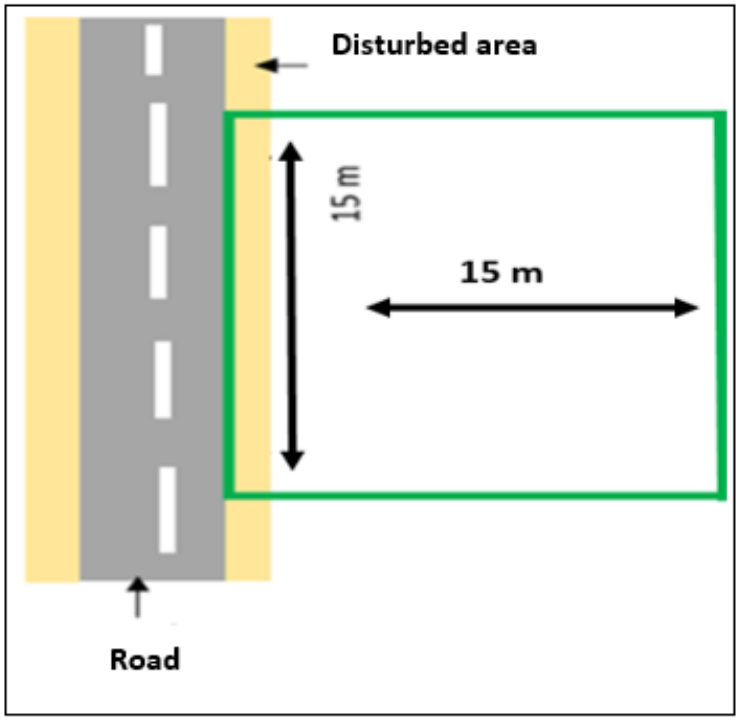


Figure 4.1. Schematic representation of the plot setup. Each surveyed plot was 15m × 15m.



Figure 4.2. The measuring tape was used in the field to measure the plots to be surveyed.

4.2.2 Data analysis

Because the data sets for different sampling points were limited, alien plant species were recorded both inside and outside the WBR. The datasets were combined and statistically analyzed to see any significant differences (P 0.05) in the frequency of alien plant species records between different classification criteria. The frequencies of alien plant species were investigated using a Generalised Linear Model with a multinomial probit link function. As an ordinal answer variable, the number of alien plant species was stated. In SPSS Version 20 (Appendix 2), the location of sampling points – either inside or outside the WBR, road type – either gravel or tar road, and land use type were provided as predictor variables.

4.3 Results

4.3.1. Alien invasive plants distribution by road type

Overall, there are significant interspecies differences in the frequencies of different alien plant species and road types ($\chi^2 = 19.9$; df = 4; $p < 0.001$; Table 4.1). Also, overall, there were significant interactions in frequencies of different alien plant species and road types. The most dominating species on both gravel and paved roads was *Melia azedarach* ($\chi^2 = 25.3$; df = 4; $p < 0.001$). *Eucalyptus camaldulensis*, *Cereus jamacaru,* and *Lantana camara* follow with the highest count, respectively, on a gravel road, whereas *Jacaranda mimosifolia* was the second dominating species on tarred road, followed by *Eucalyptus camaldulensis, Lantana camara,* and *Cereus jamacaru,* respectively (See Table 4.1; Figure 4.3).

Table 4.1. Frequency of different alien invasive plant species per road type within and outside the WBR.

		Scientific name					
		Cereus jamacaru	*Eucalyptus camaldulensis*	*Jacaranda mimosifolia*	*Lantana camara*	*Melia azedarach*	Total
Road type	Gravel	15	17	6	11	44	93
	Paved/ Tarred	2	5	9	3	65	84

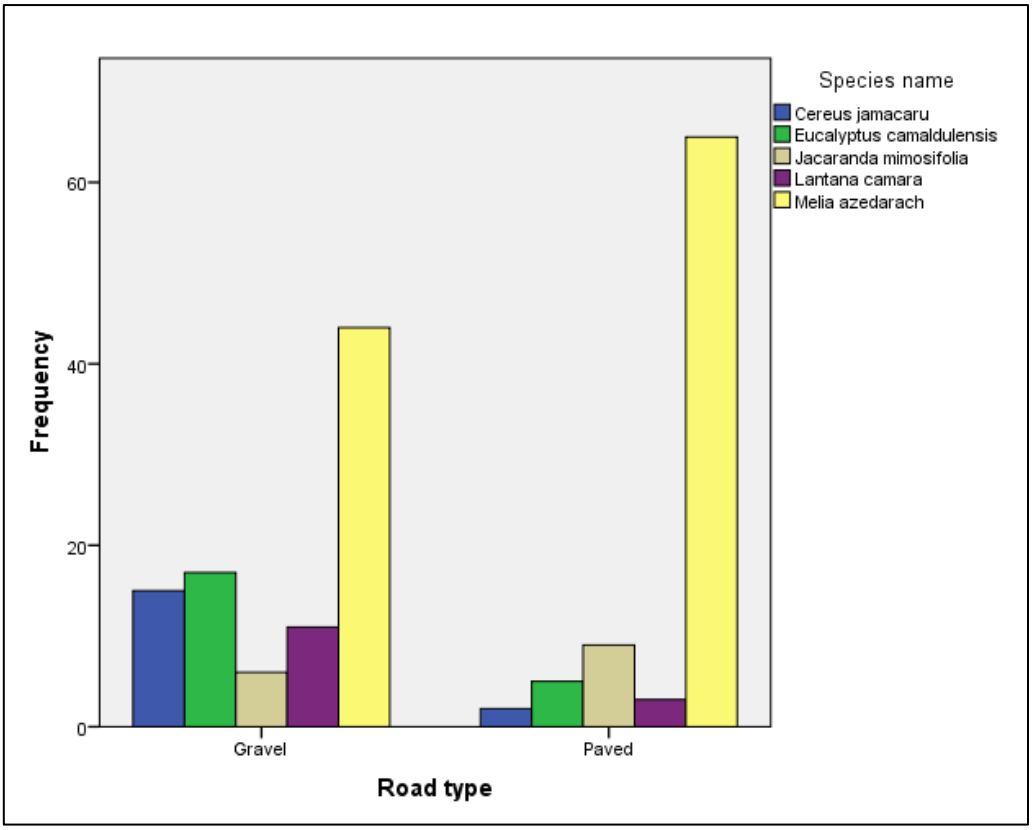


Figure 4.3. Alien plants distribution per road type within and outside the WBR.

4.3.2. Alien invasive plants distribution in different land use/ land cover types

Overall, there was a significant relationship between the occurrence of different alien plant species and different land use/cover types (χ^2 = 18.7; df = 8; p = 0.016; Table 4.4). *Melia azedarach* is the dominant species across all land uses/cover types, according to species occurrence throughout distinct land use/cover types (Table 4.2; Figure 4.2;). *Eucalyptus camaldulensis* was the second most prominent species with the highest counts only in cultivated fields, whereas *Cereus jamacaru* and *Lantana camara* follow natural vegetation. *Jacaranda mimosifolia* follows the count within the settlements, where it is the second most prominently occurring species.

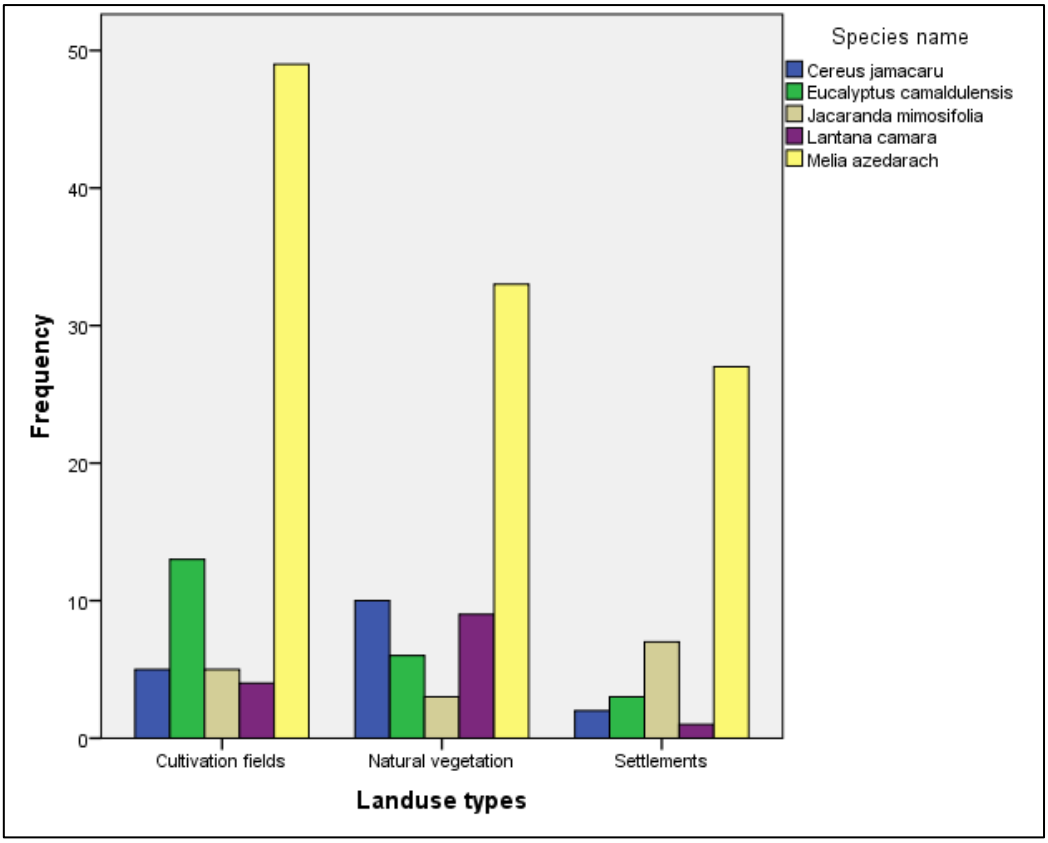


Figure 4.4. Frequency of different plant species per land use /land cover type.

Table 4.2 Frequency of occurrence of each invasive alien plant species recorded in the field by land use/land cover type

		Scientific name					
		Cereus jamacaru	*Eucalyptus camaldulensis*	*Jacaranda mimosifolia*	*Lantana camara*	*Melia azedarach*	Total
Landuse type	Cultivated fields	5	13	5	4	49	76
	Natural vegetation	10	6	3	9	33	61
	Settlements	2	3	7	9	28	49

4.3.3. Alien invasive plants distributed by area within and outside the WBR

Overall, there was a significant difference in the number of alien plant species occurring within and outside the WBR ($\chi^2 = 9.6$; df = 4; $p < 0.05$; Table 4.3). It is particularly true in the

case of *Melia azedarach*, which was found both inside and outside the WBR. Outside the WBR, the second most prevalent species is *Eucalyptus camaldulensis*, followed by *Jacanda mimosifolia* and *Lantana camara*. Outside the WBR, *Eucalyptus camaldulensis*, *Jacanda mimosifolia*, and *Lantana camara* were the least dominant species, whereas *Eucalyptus camaldulensis*, *Jacanda mimosifolia*, and *Lantana camara* were the least dominant within the WBR (Table 4.3; Figure 4.5).

Table 4.3. Frequency of occurrence of each invasive alien plant species recorded in the field within and outside WBR.

		Scientific name					
		Cereus jamacaru	*Eucalyptus camaldulensis*	*Jacaranda mimosifolia*	*Lantana camara*	*Melia azedarach*	Total
Location	inside	12	7	4	4	39	66
	outside	5	15	11	10	70	111

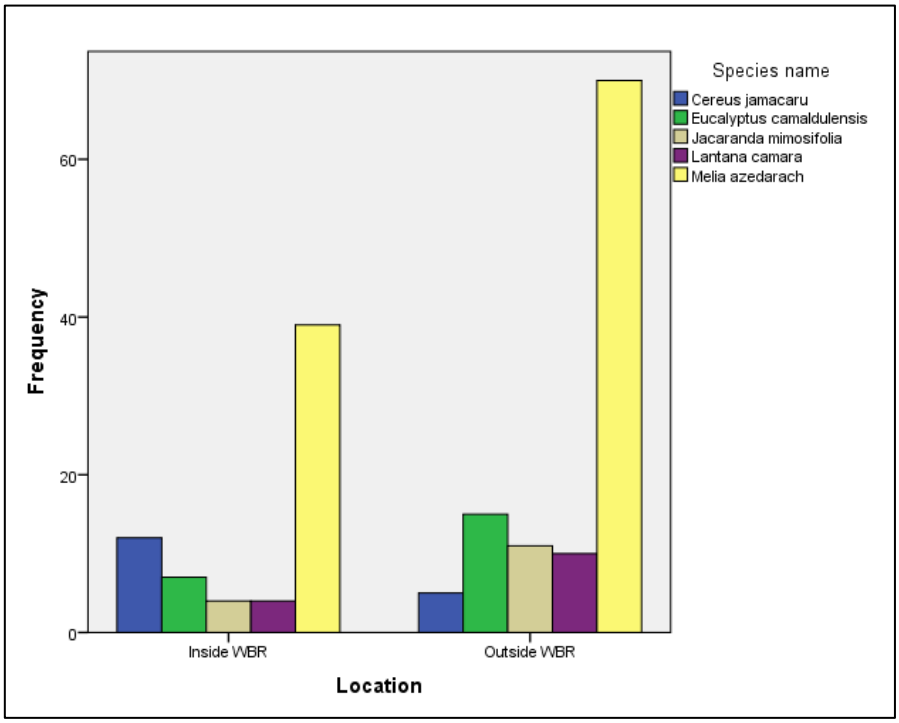


Figure 4.5. Frequency of alien plant species recorded within and outside the WBR.

The species distribution and invasion severity were ranked based on the abundance of species per plot (Figure 4.6). This approach followed Henderson (2007), where the distribution of alien plants in South Africa, Lesotho, and Swaziland was assessed and ranked based on their abundance and distribution. Most plots appear to exhibit light invasion, with around one species per plot in both study areas (Table 3.14 and 3.19). The gravel roads exhibited more alien invasive plant species than paved/ tarred roads inside and outside the WBR. The plots with zero alien plants were not mapped in terms of distribution and severity.

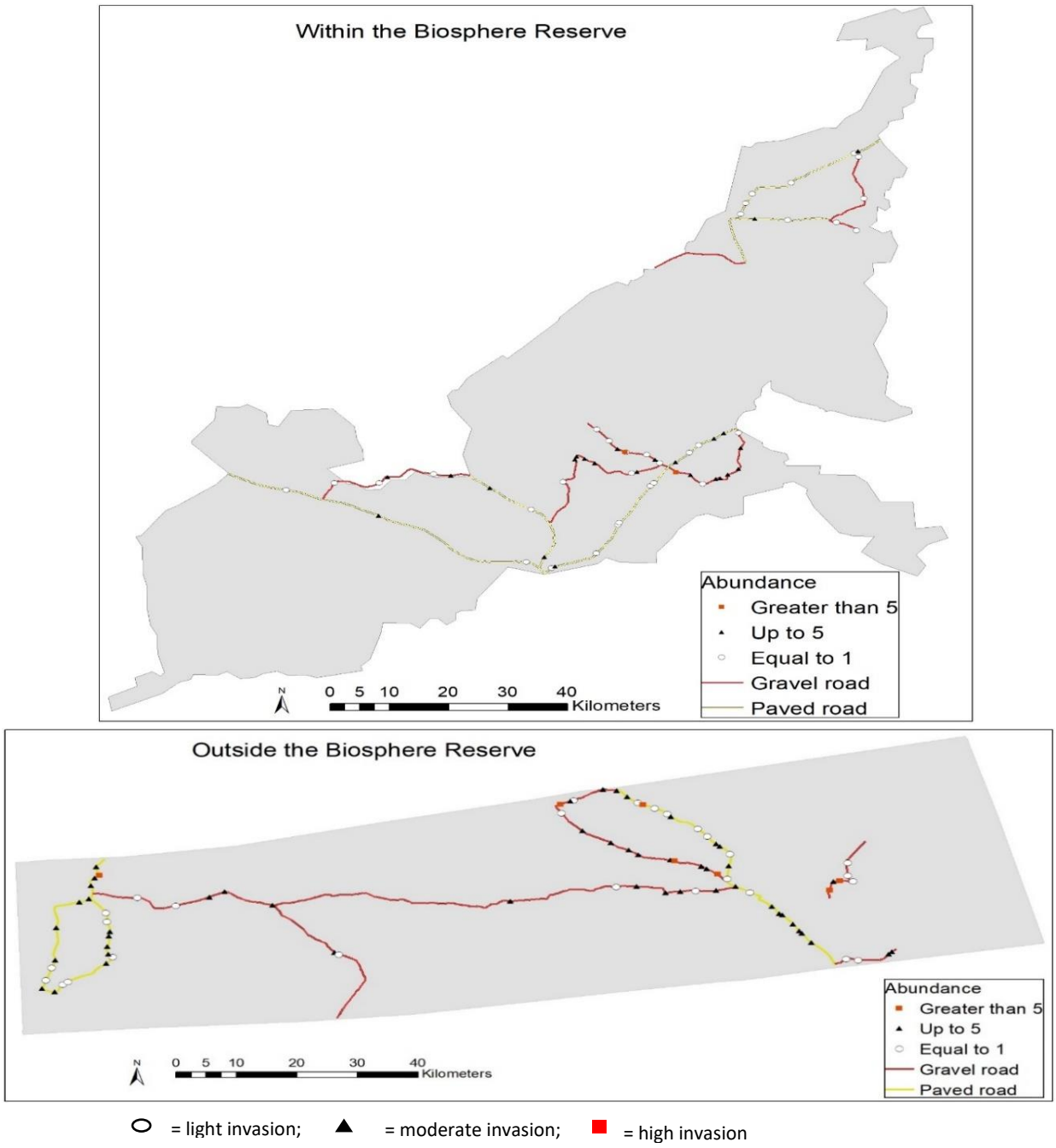


Figure 4.6. Invasive alien plant species distribution and severity within and outside the WBR.

4.4. Discussion

4.4.1. Species abundance by road types and area within and outside the WBR

Roads can significantly contribute to invasive alien plants species spread and establishment even within the protected area owing to the high levels of disturbance that create conducive conditions for many alien species (see Foxcroft et al., 2011; Pauchard and Alaback, 2004; Larson, 2003; Catford et al., 2012). The increased diversity of invasive alien plants species and dispersal potentials along certain road types indicates differences in disturbance regimes. Larger road verges are associated with greater disturbance and often occur on roads with greater traffic volumes, thus creating favourable conditions for invasion (Joly et al., 2011). Be that as it may, little research has been conducted on the influence of road type in spreading the alien invaders. More emphasis is usually placed on Land use/cover types such as agriculture and settlement, despite roads being significant seed dispersal (Joly et al., 2011). Where control and management of invasive alien plants are conducted alongside roads, more emphasis is placed on major highways usually paved (Kalwij et al., 2008).

The frequency of occurrence of invasive alien plants, following the approach by Henderson (2007), the frequency of invasive alien plants was recorded as significantly higher on gravel roads than on paved roads (Figure 4.6). Henderson's (2007) survey was conducted on a larger (national) scale along the road verge. In contrast, the survey was conducted at a local scale for this study. Henderson (2007) expressed the abundance rating as light, moderate, or high invasion based on the number of individuals/groups per distance along the transect., These results concur with Kalwij et al. (2008), who observed a strong correlation between alien species frequency and richness and gravel roads in the Nama-Karoo region of South Africa. However, the results of this study contrast with those of Joly et al. (2011), as highlighted above. According to Kalwij et al. (2008), the disturbance of gravel roads such as grading, which disturbs the soil of the road reserves, is the main contributor to alien plant's establishment. Road disturbances create bare areas and deeper soil that can favour the establishment of non-native seeds (Trombulak and Frissell 2000). Gelbard and Belnap identify road maintenance, construction, and traffic volumeGelbard and Belnap (2003) identified road maintenance, construction, and traffic volume as some of the disturbance factors contributing to alien plants richness along various roads. Kalwij et al. (2008) highlighted the

issue of heterogeneous drainage and broken ground arising from the construction of drainage systems during the gravel roads maintenance as contributing to a greater diversity of alien plants species. On the contrary, the drainage lines on the paved roads are generally directed into the formal stormwater system (Kalwij et al., 2008). Gravel road maintenance usually involves the application of water to the road surface such that propagules may become incorporated in the wet soil attached to the vehicle after that, transported and dispersed (Von der Lippe and Kowarik, 2007). Vehicles collect and transfer seeds of alien plants along roads (Zwaenepoel et al., 2006; Lonsdale and Lane, 1994; Schmidt, 1989). Although paved roads are typically more frequently used, gravel roads may have heavier total traffic volumes in rural areas due to short journeys connecting local places. During the survey, we were often directed to the shorter-distance gravel route that could connect us to the nearest town during the interaction with some game lodge owners. In terms of species composition, both road types appear to have the same species dominating (Table 4.1; Figure 4.3), suggesting that these species can become established at disturbed edges of either paved or gravel roads.

The success of the establishment of the exotic species can also be associated with a regular disturbance of paved road reserves (Truscott et al., 2005). For this study, road disturbances such as grass cutting, and herbicides were visible along the surveyed paved roads within and outside the WBR than along gravel roads. In some instances, Expanded Public Works Programmes (EPWP) workers were mowing vegetation along the paved roads. Mowing road verges could favour non-native plants that are not sensitive to clipping (Benefield et al., 1999; Forman and Alexander, 1998). Equally, using herbicides to reduce invasive alien vegetation cover (observed in both study areas) may favour non-native species while reducing native vegetation cover (Tyser et al., 1998). Fire has also proven to be successful in promoting invasions (Richardson et al., 1995). Along verges of some paved roads, fire scars were evident, suggesting that burning is often done to control vegetation growth. It is possibly done to prevent the growing vegetation from obstructing vehicle movement or blocking road signs on frequently used roads.

Outside the WBR, the frequency of alien plant species was significantly higher than within the WBR (Table 4.3; Figures 4.5 & 4.6). Furthermore, this could be attributed to the fact that the area outside the WBR is largely dominated by small towns where the area's main economic

activities occur (See Chapter 1). Although there are efforts to clear alien plants in both study areas (Personal communication with the residents), these results can be qualified because land disturbance is severe outside than within the WBR (see Chapter 3). In general, Land disturbance can strongly favour the establishment of non-native plant species over native species (see Kulmatiski et al., 2006; Woitke and Dietz, 2002; D'Antonio et al., 1999). The towns where commercial activity occurs are subjected to more regular road maintenance, allowing alien plant species to spread more quickly than smaller rural villages, such as the WBR. It is also crucial to note that, unlike the area outside the WBR, some activities are regulated within the WBR, particularly within the core zones, limiting the spread of alien invasive plants. Moreover, the area within the WBR has areas that are highly protected and reserved for conservation purposes (Marcatelli, 2015).

4.4.2. Land use/cover type and dominating species

Land use is also a highly influential factor determining alien species distribution (Hobbs, 2000; Sax and Brown, 2000). Different land uses modify disturbance regimes differently, thereby directly affecting the invasion processes by creating sources of propagules (Pauchard and Alaback, 2004). Propagule pressure from sources such as agriculture can influence how land uses affect the abundance of alien plants species along the roadsides (Pauchard and Alaback, 2004; Parendes and Jones, 2000). Thus, different roadside habitat types can also explain the increased diversity of alien plant species near different habitat types. For this study, roads near cultivated fields recorded the highest species frequency, with *Melia azedarach* as the dominating species and across other land uses (Figure 4.4).

Cultivated fields, including gardens, are recognised as sources from which invasive alien plant species escape (Wilson et al., 1992; Flory and Clay, 2006; Milton, 2004). A study by Pauchard and Alaback (2004) on alien species within the protected area of South-central Chile indicated that road verges near agricultural fields and secondary forests had more alien plant species due to the flow of propagules from the neighbouring areas. Roads were considered as the corridors that influenced the invasion processes (Pauchard and Alaback, 2004). In this study, a higher frequency of alien plants is most strongly associated with cultivated land, with *Melia azedarach* as the dominating species followed by *Eucalyptus camaldulensis* (Table 4.2). *Melia azedarach* appeared to be growing randomly along the roadsides where it was observed. The

spread and success process of *Melia azedarach* is highly influenced by the abundance and variety of propagators such as birds and bats over a longer distance or small mammals over a shorter distance that feed on drupes (Badalamenti et al., 2013).

Similarly, the inappropriate disposal of garden refuse may form sites from which exotic plants species spread, particularly in areas such as Waterberg, where illegal dumping is often noticeable along roadsides. Once the alien plant establishes along the road verges, it is likely to persist due to ongoing human disturbance and seed persistence (Milton, 2004; Milton and Dean, 1998). *Eucalyptus camaldulensis,* on the other hand, appears to have been planted where they were observed in large numbers along the roads. Eucalypt species are often introduced to practice bee-keeping (Mswazi and Poole, 2015) and produce timber (Mamvura et al., 2018). Such cultivation practices play a primary role in increasing the probability that invasives will disperse into the surrounding environment (Foxcroft et al., 2008; Reichard and White, 2001).

Natural areas are less prone to invasion by alien species; however, roads remain the important distribution vector (Rejmánek et al., 2005; Hobbs, 2000). Human activities, including ecologically acceptable ones (e.g., ecotourism), also promote invasion even in the pristine vegetation (Parendes and Jones, 2000). As more roads are extended into natural vegetation free of exotic species (Forman et al., 2003), the resultant disturbances and the frequently associated movement expose these natural areas to receiving propagules and being colonised by exotic species. Nonetheless, habitat suitability remains a powerful influence of species establishment and abundance. The results further show that common invaders are consistently distributed across different land-use types in both study areas (Figure 4.4), indicating that similar land-use types in both areas are susceptible to invasion by the same species types.

Human settlements are also recognised as essential sources from which alien species originate. Most alien species are moved around as ornamentals due to increased movement ability between different places (Pauchard and Alaback, 2004; Arévalo et al., 2005; Mokotjomela et al., in press). Most frequent species that were also abundant in roadsides were dominating in people's households. *Melia azedarach* and *Jacaranda mimosifolia* are

mainly planted in many households to provide shade or protect properties against strong winds. *Lantana camara* is often planted for fencing or for decorating due to the beautifully colourful flowers they produce (Figure 4.7A and B).

Similarly, *Cereus jamacaru* was often observed in households, planted for decoration (Figure 4.7C). As a result, the likelihood of the seeds being dispersed along the road verges via wind, animals, people, or vehicles is high. The alien plants' species in the Nama-Karoo found to occur in proximity to houses probably escaped from the gardens (Kalwij et al., 2008). Therefore, houses and related human activity located along roads appear to have acted as propagule sources for alien plant species (Gelbard and Belnap, 2003).



Figure 4.7. Alien and invasive plants are planted in the villages within the study areas for various services they offer. (A & B) *Lantana camara* (C) *Cereus jamacaru* planted for decoration (D) *Jacaranda mimosifolia*.

4.5. Conclusion

The results confirm that roads can markedly facilitate the spread of alien and invasive plants species. Overall, there is a significant difference in the number of alien plant species occurring within and outside the WBR. Furthermore, this area has experienced land degradation

indicated by increased browning over the years due to human activities. Disturbed areas such as road verge favour the establishment of exotic species, and these exotic species are unevenly distributed across different road types, and land uses. The most common species recorded along the roads in the study area include the species commonly found in human settlements. Although some of these alien species were introduced unintentionally, some were, and are still, deliberately introduced as they offer various services to people. Species such as *Melia azedarach* and *Jacaranda mimosifolia* deliver essential services such as providing shade, particularly useful in the hot summer temperatures of the Waterberg. With the increasing population and the need for areas to create settlements combined with the effects of climate change, the problem of alien infestation is likely to intensify. It warrants ongoing intervention, at the very least through raising awareness of their impacts among the local community, but preferably through intensified efforts to remove them.

4.6 References

Arévalo, J.R., Delgado, J.D., Otto, R., Naranjo, A., Salas, M. and Fernández-Palacios, J.M., 2005. Distribution of alien vs. native plant species in roadside communities along an altitudinal gradient in Tenerife and Gran Canaria (Canary Islands). *Perspectives in Plant Ecology, Evolution and Systematics*, *7*(3), pp.185-202.

Badalamenti, E., Cusimano, D., La Mantia, T. and Pasta, S., 2013. The recent spread of the invasive woody alien plant Melia azedarach L. (Meliaceae) in Sicily. Naturalista Siciliano, 2, pp. 505-513.

Benefield, C., DiTomaso, J., Kyser, G., Orloff, S., Churches, K., Marcum, D. and Nader, G., 1999. Success of mowing to control yellow starthistle depends on timing and plant's branching form. *California Agriculture*, *53*(2), pp.17-21.

Catford, J.A., Vesk, P.A., Richardson, D.M. and Pyšek, P., 2012. Quantifying levels of biological invasion: towards the objective classification of invaded and invasible ecosystems. *Global Change Biology*, *18*(1), pp.44-62.

Dawson, B.L., 1991. South African road reserves: valuable conservation areas?. In: Saunders, A., Hobbs, R.J. (Eds.), *Nature Conservation 2: the Role of Corridors*. Surrey Beatty, Chipping, Norton, Australia, pp. 119–130.

D'Antonio CM Dudley TL Mack M. 1999. Disturbance and biological invasions: Direct effects and feedbacks. Pages. 143-452. in Walker L, ed. Ecosystems of Disturbed Ground. Amsterdam: Elsevier.

Flory, S.L. and Clay, K., 2006. Invasive shrub distribution varies with distance to roads and stand age in eastern deciduous forests in Indiana, USA. *Plant Ecology*, *184*(1), pp.131-141.

Forman, R.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R.L., Heanue, K., Goldman, C.R. and Jones, J., 2003. *Road Ecology: Science and Solutions*. Island press.

Forman, R.T. and Alexander, L.E., 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics*, *29*(1), pp.207-231.

Foxcroft, L.C., Jarošík, V., Pyšek, P., Richardson, D.M. and Rouget, M., 2011. Protected-area boundaries as filters of plant invasions. *Conservation Biology*, *25*(2), pp.400-405.

Foxcroft, L.C, Richardson, D.M, Wilson, J.R.U., 2008. Ornamental plants as invasive aliens: problems and solutions in Kruger National Park, South Africa. *Environmental Management 41:32–51.*

Gelbard, J.L. and Belnap, J., 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology*, *17*(2), pp.420-432.

Henderson, L., 2013. SAPIA newsletters are posted at the ARC website: www.arc.agric.za. Volume No. 28.

Henderson, L., 2007. Invasive, naturalized and casual alien plants in southern Africa: a summary based on the Southern African Plant Invaders Atlas (SAPIA). *Bothalia*, *37*(2), pp.215-248.

Hobbs, R. J., 2000. Land use changes and invasions. *Invasive Species in a Changing World*. Island Press, Washington, D.C.

Joly, M., Bertrand, P., Gbangou, R.Y., White, M.C., Dubé, J. and Lavoie, C., 2011. Paving the way for invasive species: road type and the spread of common ragweed (*Ambrosia artemisiifolia*). *Environmental Management*, *48*(3), pp.514-522.

Kalwij, J.M., Milton, S.J. and McGeoch, M.A., 2008. Road verges as invasion corridors? A spatial hierarchical test in an arid ecosystem. *Landscape Ecology*, *23*(4), pp.439-451.

Kulmatiski A, Beard KH, Stark JM., 2006. Soil history as a primary control on plant invasion in abandoned agricultural fields. Journal of Applied Ecology 43(5):868–876.

Larson, D.L., 2003. Native weeds and exotic plants: relationships to disturbance in mixed-grass prairie. *Plant Ecology*, *169*(2), pp.317-333.

Lonsdale, W.M. and Lane, A.M., 1994. Tourist vehicles as vectors of weed seeds in Kakadu National Park, Northern Australia. *Biological Conservation*, *69*(3), pp.277-283.

Mamvura, T.A., Pahla, G. and Muzenda, E., 2018. Torrefaction of waste biomass for application in energy production in South Africa. *South African Journal of Chemical Engineering*, *25*, pp.1-12.

Marcatelli, M., 2015. Suspended redistribution: 'green economy 'and water inequality in the Waterberg, South Africa. *Third World Quarterly*, 36(12), pp.2244-2258.

Milton, S.J., 2004. Grasses as invasive alien plants in South Africa: working for water. *South African Journal of Science*, *100*(1-2), pp.69-75.

Milton, S.J. and Dean, W.R.J., 1998. Alien plant assemblages near roads in arid and semi-arid South Africa. *Diversity and Distributions*, *4*(4), pp.175-187.

Mokotjomela, T.M, Nemurangoni, T, Mundalamo, T, Jaca T. and Kuhudzai A.G., (In press). The value of dump sites in monitoring biological invasions in South Africa. Biological Invasions.

Mortensen, D.A., Rauschert, E.S., Nord, A.N. and Jones, B.P., 2009. Forest roads facilitate the spread of invasive plants. *Invasive Plant Science and Management*, *2*(3), pp.191-199.

Mswazi, M. and Poole, C., 2015. Eucalypts and other honeybee forage plants-what you can do to help our honeybees: our natural resources. *Agriprobe*, *12*(2), pp.33-35.

Parendes, L.A. and Jones, J.A., 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the HJ Andrews Experimental Forest, Oregon. *Conservation Biology*, *14*(1), pp.64-75.

Pauchard, A. and Alaback, P.B., 2004. Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. *Conservation Biology*, *18*(1), pp.238-248.

Rahlao, S.J., Milton, S.J., Esler, K.J. and Barnard, P., 2010. The distribution of invasive Pennisetum setaceum along roadsides in western South Africa: the role of corridor interchanges. *Weed Research*, *50*(6), pp.537-543.

Reichard, S.H. and White, P., 2001. Horticulture as a pathway of invasive plant introductions in the United States: most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. *BioScience*, *51*(2), pp.103-113.

Rejmánek, M., Richardson, D.M. and Pyšek, P., 2005. Plant invasions and invasibility of plant communities. *Vegetation Ecology*, *20*, pp.332-355.

Richardson, D.M., Macdonald, I.A.W., Hoffmann, J.H. & Henderson, L., 1995. Alien plant invasions. In: Cowling, R.M. and Richardson, D.M. (Eds.) Cambridge University Press, Cambridge (pp 535-570).

Sax, D. F., and J. H. Brown., 2000. The paradox of invasion. *Global Ecology and Biogeography* 9:363–371.

Schmidt, W., 1989. Plant dispersal by motor cars. *Vegetatio*, *80*(2), pp.147-152.

Slodowicz, D., Kikodze, D., Khutsishvili, M., Kalatozishvili, L. and Müller-Schärer, H., 2018. Monitoring invasive alien plants in Protected Areas in Georgia. *Bulletin of the Georgian National Academy of Sciences,* 12 (2), pp. 111-116.

Trombulak, S.C. and Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*, *14*(1), pp.18-30.

Truscott, A.M., Palmer, S.C.F., McGowan, G.M., Cape, J.N. and Smart, S., 2005. Vegetation composition of roadside verges in Scotland: the effects of nitrogen deposition, disturbance and management. *Environmental Pollution*, *136*(1), pp.109-118.

Tyser, R.W., Asebrook, J.M., Potter, R.W. and Kurth, L.L., 1998. Roadside revegetation in Glacier National Park, USA: effects of herbicide and seeding treatments. *Restoration Ecology*, *6*(2), pp.197-206.

Von der Lippe, M. and Kowarik, I., 2007. Long-distance dispersal of plants by vehicles as a driver of plant invasions. *Conservation Biology*, *21*(4), pp.986-996.

Wilson, J.B., Rapson, G.L., Sykes, M.T., Watkins, A.J. and Williams, P.A., 1992. Distributions and climatic correlations of some exotic species along roadsides in South Island, New Zealand. *Journal of Biogeography*, 19(2), pp.183-193.

Woitke M, Dietz H., 2002. Shifts in dominance of native and invasive plants in experimental patches of vegetation. *Perspectives in Plant Ecology & Evolutionary Systemaatics* 5(3):165–184.

Zwaenepoel, A., Roovers, P. and Hermy, M., 2006. Motor vehicles as vectors of plant species from road verges in a suburban environment. *Basic and Applied Ecology*, *7*(1), pp.83-93.

CHAPTER 5. INTEGRATED IMPACTS OF LAND DEGRADATION AND ALIEN PLANTS INVASION ON ECOSYSTEM SERVICES

5.1. Introduction

This chapter seeks to integrate the identified impacts (negative and positive) on ecosystem services identified by the local communities during the focus group discussions. The chapter attempts to address the study's objectives 3 and 4, which are:

a) to investigate local knowledge on land degradation and invasive alien plant impacts, as well as how they affect community well-being.
b) to investigate the integration of scientific knowledge and EbA for land management and human well-being.

5.2. Methods

The information was gathered in both study areas using a qualitative approach. A focus group discussion method was applied to facilitate information about invasive alien species and land degradation in the study areas. A focus group method is a qualitative approach commonly used for gaining an in-depth understanding of social issues affecting local communities (Nyumba et al., 2018). It is an approach that attempts to bridge the gap between scientific research and local knowledge (Cornwall and Jewkes, 1995). This method collects data from a selected group (or groups) of people towards a particular purpose. The method is widely applied and has been used to conduct similar studies elsewhere (e.g., Reed et al., 2010; Stringer and Reed, 2007; Shrestha et al., 2019; Raymond et al., 2010). However, there are no readily available procedures or guidelines when using this approach, particularly in the conservation sector (Nyumba et al., 2018).

From Google Earth, villages falling within the study area were grouped into two sections in both study areas, depending on the location (see Figure 5.1). Fifteen settlements were chosen within and outside the WBR (nine within and six outside the WBR). The community leaders and other community members were given a verbal invitation. The focus group discussion included nine settlements (five within the WBR and four outside). Ga-Tshokwe, Ga-Nkidikitlana, Magagamatala, Mosuka, Ga-Rapadi, Thabazimbi, Regorogile, Phagameng, and Modimolle are the villages in question. The invites were sent to both older and younger

residents of the area. The verbal invitation was viewed as the best option for the following reasons: a) to reach people who cannot read and write and b) in most villages, particularly within the WBR, any announcement by the community leaders is made through community gathering. One community member (usually the leader's right-hand man) rings the bell, and the community gathers the following day to hear an announcement. Two areas in each study area were selected as the meeting centre point. They were identified with the help of the community leaders and officials from the National Park (Marakelele) and the Department of Environment Forestry and Fisheries (DEFF). People were briefed before the discussion commenced regarding the issues under consideration, their willingness to participate was ascertained, and informed consent was given. Approval from the Science Faculty Ethics in Research Committee, whose name was changed to Faculty of Science Research Ethics Committee (FSREC), was obtained before the discussions. All the participants who took up the invitation resided in the area for at least ten years.

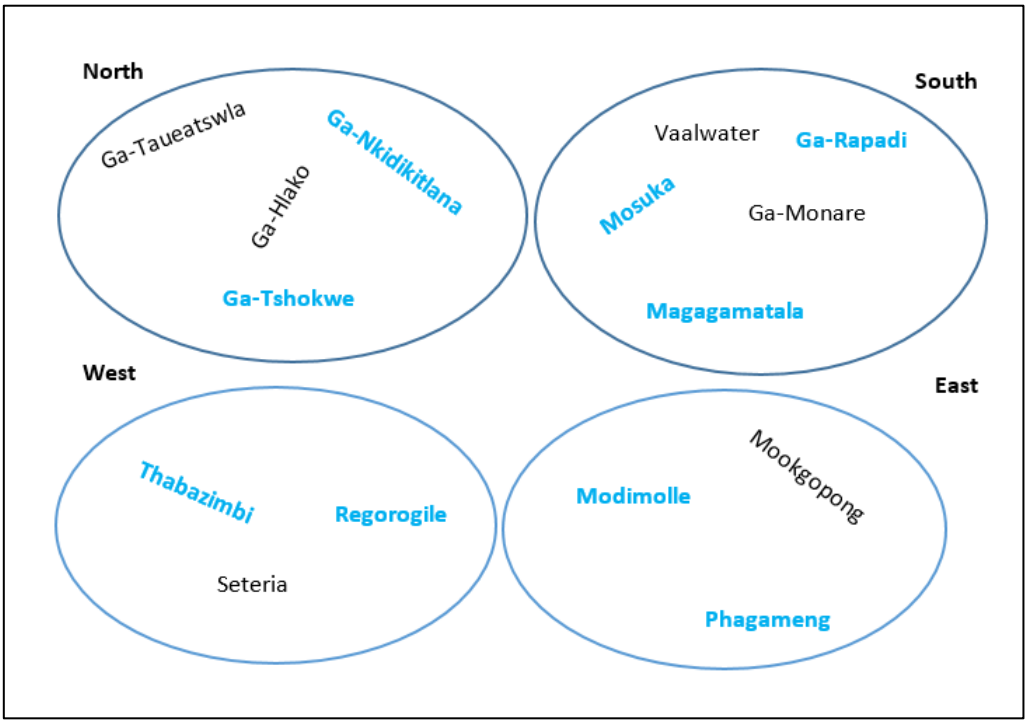


Figure 5.1. Villages that were invited and the ones highlighted took part in the discussion. North and south are villages within the WBR, and west and east are villages outside the WBR.

The discussion commenced with a brief presentation by the researcher regarding the aims and objectives of the meeting, which provided the participants with the background to the

study. The researcher was accompanied by officials from the South African National Parks, The Waterberg Biosphere Reserve, Masebe Nature Reserve, and the Department of Environment, Forestry, and Fisheries (Working for Water). The researcher further explained the meaning of the terms ecosystem, ecosystem services, land degradation, and invasive alien species in the indigenous languages, viz. Sepedi and Setswana. During the discussions, alien plants species recorded in the area were displayed on the laptop and shown to the participants for identification. The discussion aimed to obtain information regarding the following questions:

- What ecosystem services does the community derive from the area, which is most valuable, and why?
- What are the community views about the nature and severity of land degradation and invasive alien plants?
- What ecosystem-related problems does the community encounter?
- What are the introduction pathways of invasive alien species?
- How should land degradation be prevented or resolved?
- How can invasive alien plant introduction be avoided?
- What is the community view of climate-related disasters such as drought, flooding?
- What possible options are available for protecting the ecosystems and ecosystem services?

5.3. Results

5.3.1. Information gathered from the participants

In total, 28 and 22 people who honoured the invitation (within and outside WBR, respectively) participated in the focus group discussions. Among the participants, the age range was between 32 and 71 years old. From the participants, only 64% of the population had formal education, while 22% exited at secondary, 42% at primary level, and 36% had no formal education. Table 5.1 provides information on the participants, and Table 5.2 provides information on the economic challenges in the Waterberg area. Although the approach targeted both elders and the youth, most of the participants were elders who also seemed to be aware of the historical changes.

Table 5.1. Participants profile within and outside the WBR.

Area	Total no. of participants	Gender		Age group			Occupation			Education level		
		M	F	30-49	50-59	60-75	Employed	Unemployed	Students	No education	Primary	Secondary
Within WBR	28	21	7	6	10	12	1	26	1	6	14	8
Outside the WBR	22	9	13	2	13	7	2	20	0	12	7	3

Table 5.2. Challenges faced by people in the Waterberg area. The table was adopted from Lyon et al. (2019).

Economic	lack of job opportunities, high poverty, poor sectoral communication
Land use	lack of access to land and tenure
Infrastructure	water (shortage and provision, sanitation, poor service maintenance); waste (unlicensed landfill sites and illegal dumping, lack of recycling facilities); electricity capacity (old infrastructure, illegal connections); roads and transport (deteriorated roads, poor public transport)
Education	Illiteracy, unskilled labour force, lack of training opportunities
Biodiversity	urban development, destruction of habitats, invasive alien species and habits management, pollution
Tourism	tourism controlled by white people, lack community involvement in the value chain, shortage of small and medium-sized companies
Local government	lack of financial management, lack of skills, and staff commitment.

5.3.2. Identification of key ecosystem services by local communities within and outside the WBR

Several ecosystem services were identified and listed as being valuable by the participants. The types of ecosystem services were grouped into different categories defined in the Millennium Ecosystem Assessment (MEA, 2005), as shown in Table 5.3. All the participants identified **provision services** as the most valuable service with fresh water, food, healthy soil,

and sand for building houses rated as very important. Medicinal plants were rated by 76 % of the participants as being valuable. Some 34 % indicated that they did not believe in medicinal plants because of their religious background. Fuelwood was rated by 84 % as an important source of energy for cooking and heating water. Although all participants had electricity in their homes, affordability remains an issue. As a result, fuelwood is regarded as the better option for cooking, while electricity is preferably used for lighting.

In terms of **cultural services**, 88 % of the participants highlighted the importance of rivers for cultural and religious purposes (e.g., used for baptism or cleansing ceremonies).

74 % of participants highlighted flood control, as a **regulating service**, as being important. 10 % highlighted the importance of using plants for fencing as they cannot afford proper fencing wires. Some participants mentioned that such plants prevent livestock (goats) from accessing their gardens to feed on fruit-bearing trees and other home-grown vegetables. In particular, trees such as *Melia azedarach* and *Jacaranda mimosifolia* were acknowledged as crucial by all the participants for providing shade, particularly during the summer season. 10% of discussants indicated the importance of having clear roads for safety as they connect them to the nearest villages. Services such as sand mining and stone digging for commercial purposes and gaming were categorised as **supporting services** and identified as important by 52 % for generating income. A smaller percentage of participants (16%) highlighted the pleasure of visiting game reserves to see wildlife, particularly when offered free tours.

Table 5.3. Response from the participants on the value of the ecosystem services in their areas.

Ecosystem services category	Types of service to the people	% of respondents who see services as most valuable
Provision	Freshwater	100
	Food	100
	Healthy soil for food	100
	Medicinal plants	76
	Fuelwood	84
	Sand for building houses (proper shelters)	100
Cultural	Rivers (baptism and traditional ceremonies)	88
Regulating	Streams that channel water to the rivers and prevent flooding	74
	Use of plants for fencing	10
	Clear roads for travelling	10
	Shades from the trees	100
Supporting	Stone digging and sand mining for commercial purposes	52
	Wildlife from game reserves	16

5.3.3. Perspectives gained from interacting with communities on land degradation and invasive alien plants and their impacts

In general, the participants exhibited a good understanding of which plants are alien invasives. The participants were able to identify the species displayed on the laptop screen and recorded as occurring in the area by their vernacular or common names. The participants appeared to know the identified species' impacts and the consequence of land disturbance on their environment and human well-being (Table 5.4). However, in general, the participants did not thoroughly understand the pathways or dispersal vectors. Nevertheless, some participants allude to knowing how species such as *Melia azedarach* and *jacaranda mimosifolia* were introduced in the areas. It was mentioned that such species were initially planted in churches and schools to provide shades, but they are now found even in places where they were not introduced. One example was of the Mogalakwena riverbank, where *Jacaranda mimosifolia* and *Melia azedarach* species were occurring. This river transverses the

WBR and is used to supply the residents with water. The mentioned species were also observed when the river was visited during the survey.
Interestingly, one participant highlighted that birds could have introduced some species along the riverbank. The suggestion provoked a lively debate, and participants acknowledged that some introductions may have occurred during water collection at the newly constructed dam near the river, which connects communities to water.

In contrast, others assumed the seeds might have been introduced from other areas through waterways. Some alien species were identified as occurring in the wetlands and are known to invade such habitats. In communication with one of the managers from the local nature reserve Masebe, which is located northeast of the WBR, it was indicated that clearing occurs as part of the Working for Water programme, and priority is given to removal from areas associated with water resources.

Participants identified activities that may have adverse environmental and socio-economic impacts, including land degradation, while beneficial or necessary to sustain their livelihoods. For example, some women mentioned how stone digging, which has traditionally supplemented their livelihoods over the past 30 years through selling the stones for ornamental purposes (see Figure 5.2), has resulted in land degradation through loss of vegetation. Several discussants mentioned that reducing vegetation cover near their homes has a security benefit as it is easier to observe problematic animals and intruders. Reduced visibility from dust, especially during windy days, was one of the negative impacts of a loss of vegetation cover due to road construction, stone excavation, and sand mining. Reduction in the occurrence of food and medicinal plants, such as *Lippia javanica* and *Drimia elata* knows as “mosunkwane” and “sekanama” in the indigenous language (*Sepedi*), was also identified as a problem. These medicinal plants are used to treat influenza, repel mosquitos, pain relief, and combat feverish colds. The participants noted that the distance they had to travel to obtain such services and the hilly terrain were significant constraints.

Sand mining was identified as an activity that provides people with economic benefits as this resource may be sold directly or used in building houses. Some of the negative impacts identified as a result of sand mining include open pits that pose a danger to children,

particularly on rainy days as they become flooded. These land disturbances can affect the water channels, resulting in the downstream areas being flooded during heavy rains. Unreliable rainfall has led to fewer residents relying on subsistence farming practices and resorting instead to other alternatives (sand mining and stone digging in particular), contributing to land disturbance.

Table 5.4. Summary of common ecosystem services and impacts arising from land degradation and invasive alien plants within and outside the WBR as provided by the participants in the focus group discussion.

Ecosystem services from invasive alien plants in the areas	***Ecosystem services arising from the land disturbance in the areas***	***Impacts identified from invasive alien plants***	***Impacts identified from land degradation***
• Provide shade • Fuelwood • Fruits (prickly pear) • Used for fencing (*Lantana camara*).	• Stones digging for commercial purposes • Sand mining for commercial purposes and for building houses • Clear roads for travelling.	• Loss of indigenous plants • Deprivation of food and medicinal plants • Loss of water • Toxic to humans and animals • Loss of grazing lands • Damage to properties.	• Loss of vegetation • The holes pose a threat to people and livestock as they remain open after digging and sometimes become flooded during rainy seasons • Diverts water direction and floods people's property • Reduced visibility during windy days as due to exposed soil • Loss of grazing lands • Loss of livestock as cars travel on the constructed roads and knock down their livestock.

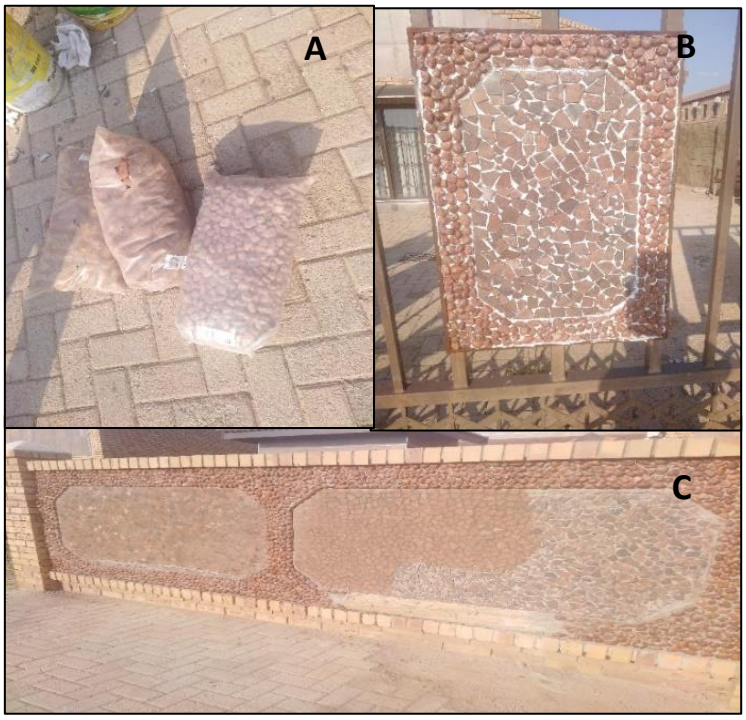


Figure 5.2. The packed stones that were purchased in the WBR villages and utilized for decoration in Modimolle.

The discussants seemed largely unaware of possible adaptation strategies to deal with climate changes that manifest as changing rainfall patterns. For example, there were plans by the Department of Environment, Forestry, and Fisheries (DEFF) to adjust agricultural practices or projects that conserve water by removing invasive alien plants, particularly at the riparian zones. The Working for Water initiative is an example where the main objective is to tackle the alien invasive plants in the country and improve water resources availability while creating employment opportunities (Mudavanhu et al., 2018). Be that as it may, community members in these areas are critical of the government. They feel that insufficient attention is paid to their needs, particularly in relation to access to water and employment opportunities.

On the other hand, some settlements have a common borehole for drinking water, which some residents have taken advantage of by planting food gardens. One example is shown in Figure 5.3, showing elders working on their food garden project called “Itsoseng Batsofe.” The vegetables feed their families during the harvesting season, and street vendors sell any surplus at the nearest shopping centre to generate income.



Figure 5.3 (left). A woman is fetching water at the communal borehole. Behind her is the food garden project started by the community. (Right) a man is pushing a trolley containing harvested vegetables for selling.

5.4. Discussion

5.4.1. Impacts of recorded invasive alien plants on ecosystem services and human well-being

Melia azedarach is a threat to the native vegetation, including the riparian communities, particularly in the tropical and subtropical regions of South Africa (Henderson, 1990). Along with *Jacaranda mimosifolia*, these species spread within the semi-arid savanna in South Africa, invading the perennial rivers, particularly where water is freely available to survive the drought season (Van Wilgen et al., 2001; Henderson, 2001). *Melia azedarach* can form dense stands that can limit the growth of native species and invade open spaces (Van Wilgen et al., 2001; Henderson, 2001). The species can persist in the environment by re-sprouting through stumps and roots after clearance (Tourn et al., 1999). *Melia azedarach* is also toxic, especially following ingestion of fruits (Steyn and Rindl, 1929; Oelrichs et al., 1983; Phua et al., 2008). The toxicity of *Melia azedarach* seeds was highlighted as a problem in the study areas. Some of the identified issues were choking, forcing seeds into the noses by toddlers, leading to suffocation. Such impacts were highlighted in Oelrichs et al. (1983); Steyn and Rindl (1929). In South Africa, fatal cases of poisoning from *Melia azedarach* fruit ingestion were recorded in King Williams Town in the 1910s (Steyn and Rindl, 1929).

Eucalyptus camaldulensis is a species found frequently in places where roads and water channels intersect. According to Dzikiti et al. (2016), *eucalypts* and other species such as *Acacia mearnsii* and *Populus* dominate mainly in the riparian zones. These species are capable of consuming large amounts of water. In Mpumalanga, for example, afforestation of catchments with *eucalypts* had marked hydrological impacts and resulted in the drying up of streams and rivers (Forsyth et al., 2004). In the Waterberg area, large numbers of people rely on water from the rivers, streams, and wells for drinking and other uses. As such, such species pose a threat to their most basic need (water).

Lantana camara is a shrub that multiplies and is an aggressive invader. This shrub can cause ecological destruction through loss in the diversity of native species and through its ability to dominate the understorey (Totland et al., 2005). This weed was introduced for ornamental purposes but has excellent ability to escape cultivation and negatively affect grazing lands and native species richness (Islam et al., 2001; Totland et al., 2005). *Lantana* is also known to affect plant community composition, particularly in forested habitats (Islam et al., 2001). Several studies have highlighted the impacts of this species on livestock (Seawright, 1964; Sharma et al., 1988; Shackleton et al., 2017). These findings support the community's views on the toxicity of *Lantana camara* to their livestock in the WBR (see Table 5.4).

Cereus jamacaru is a cactus known for forming dense stands that can displace native species and limit access to grazing for animals (Witt and Luke, 2017). This species can obstruct the movement of livestock and other herbivores and has spines that cause injuries to people and animals (Witt and Luke, 2017). Additionally, *cereus jamacaru* can prevent the livestock from having access to the shades and consequently contribute to heat stress in animals.

5.4.2. Ecosystem services and livelihoods

The Millennium Ecosystem Assessment, with its emphasis on human well-being and poverty reduction, highlights the significance of ecosystem services to improve the human well-being of poor communities (Adams et al., 2004). Loss of services resulting from ecosystem degradation could significantly reduce human well-being and intensify poverty (Pinho et al., 2014). This relationship between ecosystem services and their role in poverty alleviation has been heavily documented (e.g., Millennium Ecosystem Assessment, 2005; Daw et al., 2011;

Suich et al., 2015). When ecosystem services are subject to change, they can directly affect human well-being either negatively or positively. For example, by carefully maintaining ecosystem processes, an increase in ecosystem services may result in poverty alleviation and prevention (Daw et al., 2011), whereas neglecting or destroying ecosystems may lead to deteriorating human livelihoods as services decline.

In the Waterberg District, various socio-ecological stressors are considered to be a threat to livelihoods. Lack of access to a clean and reliable water supply, limitations on grazing areas for those who still practice livestock farming, and toxic weeds were identified as the most significant threats by the local people. These threats are related to the food supply and health and well-being rather than economics. The paucity of such services and infestation by exotic weeds is caused by several pressures, including land disturbance and invasion by alien plants. Most participants identify human activities as the cause of changing climatic conditions, resulting in the aforementioned threats. It is known that activities such as deforestation, the overuse of land, and technological and socio-economic changes can intensify climate change (Millennium Ecosystem Assessment, 2005).

Local communities are essential agents in preventing or mitigating activities that contribute to climate change, such as deforestation. However, they are often not considered in decision-making or action planning (Jan and Anja, 2007). While much of the research effort involved in climate change is based within the physical sciences, the needed policies must be focused on communities which, in cases like the Waterberg, means engaging with local perceptions and indigenous knowledge (Ishaya and Abaje, 2008). This is because the local community can observe and plan how to deal with the consequences of changes in their environment.

In South Africa, stakeholders in different sectors have implemented various adaptation projects to deal with climate change. The key stakeholders are the national, provincial, and local government, state entities, and NGOs (Department of Environmental Affairs, 2017). While some projects may not always be labelled as climate adaptation projects *per se*, they contribute towards adaptation by reducing vulnerability, for example, Working for Water and Working for Wetlands (Mukheibir, 2008). These programmes attempt to protect valuable water resources within South Africa by removing alien plants along the riparian zones

(Department of Environmental Affairs, 2017). Other projects in the agricultural sector assist the farmers in preparing for future climate change (Department of Environmental Affairs, 2017). Ecosystem restoration is also applied in ecosystem-based agriculture to conserve biodiversity (Vignola et al., 2015). Biodiversity stewardship is another adaptation approach that involves custodians of the resources such as local communities and landowners (Department of Environmental Affairs, 2019). The agreement is reached with all the parties on managing and protecting biodiversity priority areas (Cumming and Read 2016). Many other adaptation projects focus on the association between climate change and human health, including food security.

When ecosystem health is improved, communities can build resilience that will better assist them in coping with the impacts associated with climate change (Department of Environmental Affairs, 2016). Some adaptation projects have been implemented in the disaster sector, such as climate-proofing settlements and the development of local early warning systems (Department of Environmental Affairs, 2019). These are the adaptation measures needed to be applied in an area for the benefit of the people and the protection of the environment. In the Waterberg area, there are challenges to implementing available adaptation strategies, particularly in relation to invasive alien plants, that include not consulting with and involving the local communities resulting in the ineffectiveness of the methods used for controlling invasive alien plants. Without awareness, some residents still introduce alien species (e.g., for gardening, fencing, providing shades) as they are unaware of the threats they pose to their environment and well-being. In terms of land degradation in the area, particularly arising from activities such as sand mining and stone digging, the government is failing to work with the people to develop sustainable long-term plans for the benefit of the people while protecting the environment.

5.5 References

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B. and Wolmer, W., 2004. Biodiversity conservation and the eradication of poverty. *Science*, *306*(5699), pp.1146-1149.

Cornwall, A. and Jewkes, R., 1995. What is participatory research? *Social Science & Medicine*, *41*(12), pp.1667-1676.

Cummings, A.R. and Read, J.M., 2016. Drawing on traditional knowledge to identify and describe ecosystem services associated with Northern Amazon's multiple-use plants. *International Journal of Biodiversity Science, Ecosystem Services & Management*, *12*(1-2), pp.39-56.

Daw, T.I.M., Brown, K., Rosendo, S. and Pomeroy, R., 2011. Applying the ecosystem services concept to poverty alleviation: the need to disaggregate human well-being. *Environmental Conservation*, *38*(4), pp.370-379.

Department of Environmental Affairs (DEA), 2016. South Africa's 1st Annual Climate Change Report, Theme E: Monitoring the Adaptation Landscape in South Africa: Desired Adaptation Outcomes, Adaptation Projects and the Intended Nationally Determined Contribution. Available at: https://www.environment.gov.za/sites/default/files/reports/themeE_adaptation_landscape.pdf, [Accessed May 2019].

Department of Environmental Affairs (DEA), 2017. South Africa's 2nd Biennial Update Report to the United Nations Framework Convention on Climate Change, Pretoria: Department of Environmental Affairs. Available at: https://unfccc.int/files/national_reports/nonannex_i_parties/biennial_update_reports/application/pdf/south_africa_2nd_bur.pdf, [Accessed May 2019].

Department of Environmental Affairs (DEA), 2019. Draft National Climate Change Adaptation Strategy. Government Gazette, No.42446. Pretoria: Department of Environmental Affairs. Available at: https://www.gov.za/sites/default/files/gcis_document/201905/42446gon644.pdf, [Accessed May 2019].

Dzikiti, S., Gush, M.B., Le Maitre, D.C., Maherry, A., Jovanovic, N.Z., Ramoelo, A. and Cho, M.A., 2016. Quantifying potential water savings from clearing invasive alien *Eucalyptus camaldulensis* using in situ and high-resolution remote sensing data in the Berg River Catchment, Western Cape, South Africa. *Forest Ecology and Management*, *361*, pp.69-80.

Forsyth, G.G., Richardson, D.M., Brown, P.J. and Van Wilgen, B.W., 2004. A rapid assessment of the invasive status of Eucalyptus species in two South African provinces: working for water. *South African Journal of Science*, *100*(1-2), pp.75-77.

Henderson, L., 1990. *Jacaranda*. Plant Protection Research Institute. Department of Agricultural Development, Directorate of Agricultural Information, Pretoria.

Henderson, L., 2001. Alien Weeds and invasive Plants. Cape Town: Plant Protection Research Institute, Agricultural Research Council.

Ishaya, S. and Abaje, I.B., 2008. Indigenous people's perception on climate change and adaptation strategies in Jema'a local government area of Kaduna State, Nigeria. *Journal of Geography and Regional Planning*, *1*(8), p.138-143.

Islam, K.R., Ahmed, M.R., Bhuiyan, M.K. and Badruddin, A., 2001. Deforestation effects on vegetative regeneration and soil quality in tropical semi-evergreen degraded and protected forests of Bangladesh. *Land Degradation & Development*, *12*(1), pp.45-56.

Jan, S., Anja, B., 2007. Indigenous Peoples and Climate Change. University of Oxford and Missouri Botanical Garden.

Lyon, A. and Hunter-Jones, P., 2019. Critical discourse analysis and the questioning of dominant, hegemonic discourses of sustainable tourism in the Waterberg Biosphere Reserve, South Africa. *Journal of Sustainable Tourism*.

Millennium Ecosystem Assessment., 2005. *Ecosystems and Human Well-Being: Global Assessment Reports*. Island Press, Washington, DC.

Mudavanhu, S., Blignaut, J. and Vink, N., 2018. A system dynamics modelling approach to restoration of natural capital: An economic assessment of costs and benefits of different land use options and value added products to assist in the control of invasive alien plants in the fynbos biome of the Western Cape Province, South Africa. *Proceedings of the 56th International Conference of the Agricultural Economics Association of South Africa,* Cape Town South Africa, September 2018.

Mukheibir, P., 2008. Water resources management strategies for adaptation to climate-induced impacts in South Africa. *Water Resources Management*, *22*(9), pp.1259-1276.

Nyumba, 0.T., Wilson, K., Derrick, C.J. and Mukherjee, N., 2018. The use of focus group discussion methodology: Insights from two decades of application in conservation. *Methods in Ecology and Evolution*, *9*(1), pp.20-32.

Oelrichs, P.B., Hill, M.W., Vallely, P.J., MacLeod, J.K. and Molinski, T.F., 1983. Toxic tetranortriterpenes of the fruit of Melia azedarach. *Phytochemistry*, *22*(2), pp.531-534.

Phua, D.H., Tsai, W.J., Ger, J., Deng, J.F. and Yang, C.C., 2008. Human *Melia azedarach* poisoning. *Clinical Toxicology*, *46*(10), pp.1067-1070.

Pinho, P.F., Patenaude, G., Ometto, J.P., Meir, P., Toledo, P.M., Coelho, A. and Young, C.E.F., 2014. Ecosystem protection and poverty alleviation in the tropics: Perspective from a historical evolution of policy-making in the Brazilian Amazon. *Ecosystem Services*, *8*, pp.97-109.

Raymond, C.M., Fazey, I., Reed, M.S., Stringer, L.C., Robinson, G.M. and Evely, A.C., 2010. Integrating local and scientific knowledge for Environmental Management. *Journal of environmental management*, *91*(8), pp.1766-1777.

Reed, J., Van Vianen, J., Deakin, E.L., Barlow, J. and Sunderland, T., 2016. Integrated landscape approaches to managing social and environmental issues in the tropics: learning from the past to guide the future. *Global Change Biology*, *22*(7), pp.2540-2554.

Reed, M.S., Evely, A.C., Cundill, G., Fazey, I., Glass, J., Laing, A., Newig, J., Parrish, B., Prell, C., Raymond, C. and Stringer, L.C., 2010. What is social learning?. *Ecology and society*, *15*(4), pp. 1-10.

Seawright, A.A., 1964. Studies on the pathology of experimental lantana (*Lantana camara* L.) poisoning of sheep. *Pathologia Veterinaria*, *1*(6), pp.504-529.

Shackleton, R.T., Witt, A.B., Aool, W. and Pratt, C.F., 2017. Distribution of the invasive alien weed, *Lantana camara*, and its ecological and livelihood impacts in eastern Africa. *African Journal of Range & Forage Science*, *34*(1), pp.1-11.

Sharma, O.P., Makkar, H.P.S. and Dawra, R.K., 1988. A review of the noxious plant *Lantana camara. Toxicon*, *26*(11), pp.975-987.

Shrestha, B.B., Shrestha, U.B., Sharma, K.P., Thapa-Parajuli, R.B., Devkota, A. and Siwakoti, M., 2019. Community perception and prioritization of invasive alien plants in Chitwan-Annapurna Landscape, Nepal. *Journal of Environmental Management*, *229*, pp.38-47.

Steyn, D.G. and Rindl, M., 1929. Preliminary report on the toxicity of the fruit of *Melia azedarach* (syringa berries). *Transactions of the Royal Society of South Africa*, *17*(4), pp.295-308.

Stringer, L.C. and Reed, M.S., 2007. Land degradation assessment in southern Africa: integrating local and scientific knowledge bases. *Land Degradation & Development*, *18*(1), pp.99-116.

Suich, H., Howe, C. and Mace, G., 2015. Ecosystem services and poverty alleviation: a review of the empirical links. *Ecosystem Services*, *12*, pp.137-147.

Totland, Ø., Nyeko, P., Bjerknes, A.L., Hegland, S.J. and Nielsen, A., 2005. Does forest gap size affects population size, plant size, reproductive success and pollinator visitation in *Lantana camara*, a tropical invasive shrub?. *Forest Ecology and Management*, *215*(1-3), pp.329-338.

Tourn GM, Menvielle MF, Scopel AL, Pidal B. 1999. Clonal strategies of a woody weed: Melia azedarach. *Plant and Soil*, 217(1-2), pp.111-117.

van Wilgen B.W., Richardson D.M., Le Maitre D.C., Marais C. and Magadlela D., 2001. The economic consequences of alien plant invasions: examples of impacts and approaches to sustainable management in South Africa.

Vignola, R., Harvey, C.A., Bautista-Solis, P., Avelino, J., Rapidel, B., Donatti, C. and Martinez, R., 2015. Ecosystem-based adaptation for smallholder farmers: Definitions, opportunities and constraints. *Agriculture, Ecosystems & Environment*, *211*, pp.126-132.

Witt A and Luke Q., 2017. guide to the naturalized and invasive plants of Eastern Africa. Wallingford, UK: CABI. vi + 601 pp.

CHAPTER 6. CONCLUSION

This chapter summarises the conclusions based on the methodologies applied to attain the results and findings.

6.1 Land use/cover change and land degradation

Using remote sensing, Landsat images of 1989, 2000, 2013 and 2017 were used to quantify land use and cover change within and outside the WBR. The land use cover maps demonstrate that changes occurred in both sections of the study area. The changes mainly result from human activities. Human settlements, cultivation fields, and other activities such as commercial and small-scale mining resulted in the loss of vegetation cover. Within the WBR, bare land areas decreased by 1.8% between 1989 and 2000, 16.1% between 2000 and 2013, and 2.0% between 2013 and 2017. However, there is still evidence of vegetation loss, particularly in the transition zone. The increased greening results from conservation measures in the protected areas of the Biosphere Reserve as some activities with negative impacts on the environment are restricted.

Conversely, the bare land outside the WBR increased by 2.75% between 1989 and 2000, 12.49% between 2000 and 2013, and 5.95% between 2013 and 2017. Although some areas outside the WBR are also preserved for conservation purposes, the expansion of commercial mining resulted in informal and formal settlements. These formal and informal settlements are still expanding as people seek dwellings in proximity to their workplaces. The government purely forms some formal settlements such as Leseding through its Reconstruction Development Programme subsidy housing, which came into effect after 1994, to provide proper housing for people (Marcatelli, 2015). Between 1989 and 2017, increased human settlements within and outside the WBR increased by 0.39 percent and 0.40 percent, respectively, due to the rise of subsidized housing and accompanying in-migration of people. The area covered by agricultural activities increased and declined between the years in both study parts of the study area. The change detection matrix demonstrated that 3.5% and 6.6% of cultivated fields within and outside the WBR, respectively, transitioned into bare land between 1989 and 2017. These changes result mainly from inconsistency and the shift from crop to game farming due to government subsidies favouring game farming (Marcatelli,

2015). Game farming practices play a crucial role as it contributes to the local economy of the Waterberg.

The amount of vegetation loss demonstrated the degree of degradation in both areas, where parts were rated as low, medium, high, or severe degradation. Severely degraded areas signified that intense human activities resulted in the complete loss of vegetation. This was largely evident at the periphery of the area outside the WBR, where commercial mining is occurring. While the mining industry also contributes to the local economy, some concerns are evident over the negative impacts on the natural environment and the strain on the water as a scarce resource in the area (van Niekerk and du Plessis, 2013). The current situation of land degradation in the study area outside the WBR is alarming and requires proper and effective land use planning and management. The situation is similar in the transition zone of the WBR, where activities such as sand and stone mining can escalate land degradation and create deleterious impacts, especially given the fact that much of the activities are conducted without proper planning and management.

6.2 Invasive alien plants distribution

Invasive alien plants often rapidly colonize disturbed areas (Kalwij et al., 2008), and roads are dispersal pathways. Overall, there is a significant difference in the number of alien plant species occurring within and outside the WBR. The distribution of alien plant species was estimated along different selected main roads within and outside the WBR and, variables such as the adjacent land use/cover type were recorded. The results indicate that alien plants species within the WBR are abundant along the gravel roads. While more species were recorded along paved than gravel roads, cultivated fields are characterized by more alien species than natural vegetation and settlements. The differences are attributed to habitat suitability and the behaviour of road use, given the fact that WBR is mainly a rural, traditional dwelling, while the area outside the reserve is associated with more small towns, formal and informal settlements and, moreover, is the economic hub of the region. The results suggest no differences in alien plant species composition along different road types and land uses. Species such as *Melia azedarach* dominate across different road types, and land uses in both areas. The relationship between alien invasive species occurrence and land disturbance was

noted, and several studies address this association (see Ficetola et al., 2010; McKinney et al., 2006; Pauchard and Alaback, 2004).

6.3 Ecosystem services

Invasive alien species may be problematic and offer some ecosystem services to the people ranging from provision (food, fuelwood) to regulating services (shade). The focus group discussion assisted in gaining insightful information on how the two pressures compromise human livelihoods. Land degradation and alien plants invasion collectively negatively impact the functioning of the ecosystems in providing the basic needs by affecting multiple roles of ecosystems such as provisioning, cultural, regulating, and supporting services (Millennium Ecosystem Assessment, 2005). In most instances, the impacts are exacerbated by changing climatic conditions. EbA, which focuses on developing resilience, particularly on the climate change-related impacts, is seen as the most cost-effective way of protecting people and helping them in meeting their basic needs (Mooney et al., 2009). For EbA to be effectively implemented requires understanding how land-use decisions are made by people and their influences (Geist and Lambin, 2004). Therefore, to manage land effectively, it is crucial to integrate local knowledge through community participation. This will lead to the creation of a database that could be used for effective land-use planning. This study has demonstrated the effective use of remote sensing to detect land use/cover change and the inclusion of local communities in identifying the local issues in relation to natural resources.

6.4. References

Ficetola, G.F., Maiorano, L., Falcucci, A., Dendoncker, N., Boitani, L., Padoa-Schioppa, E.M.I.L.I.O., Miaud, C. and Thuiller, W., 2010. Knowing the past to predict the future: land-use change and the distribution of invasive bullfrogs. *Global Change Biology*, *16*(2), pp.528-537.

Geist, H.J. and Lambin, E.F., 2004. Dynamic causal patterns of desertification. *Bioscience*, *54*(9), pp.817-829.

Kalwij, J.M., Milton, S.J. and McGeoch, M.A., 2008. Road verges as invasion corridors? A spatial hierarchical test in an arid ecosystem. *Landscape Ecology*, *23*(4), pp.439-451.

Marcatelli, M., 2015. Suspended redistribution:'green economy'and water inequality in the Waterberg, South Africa. *Third World Quarterly*, 36(12), pp.2244-2258.

McKinney, M.L., 2006. Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*(3), pp.247-260.

Millennium Ecosystem Assessment., 2005. *Ecosystems and Human Well-Being: Global Assessment Reports*. Island Press, Washington, DC.

Mooney, H., Larigauderie, A., Cesario, M., Elmquist, T., Hoegh-Guldberg, O., Lavorel, S., Mace, G.M., Palmer, M., Scholes, R. and Yahara, T., 2009. Biodiversity, climate change, and ecosystem services. *Current Opinion in Environmental Sustainability*, *1*(1), pp.46-54.

Pauchard, A. and Alaback, P.B., 2004. Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. *Conservation Biology*, *18*(1), pp.238-248.

Ruiters, C. and Matji, M.P., 2015. Water institutions and governance models for the funding, financing and management of water infrastructure in South Africa. *Water SA*, *41*(5), pp.660-676.

APPENDIX 1. ERROR MATRICES AND CLASSIFICATION ERROR PERFORMED FOR OTHER YEARS.

1.

Overall accuracy 1989 0,986207

Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**68**	0	0	0	0	**68**
Grass land	0	**69**	0	0	0	**69**
Woody veg	0	1	**69**	0	0	**70**
cultivated	0	0	1	**40**	0	**41**
Water	2	0	0	0	**40**	**42**
Total	**70**	**70**	**70**	**40**	**40**	**290**

2. Commission and Ommision error

Commission 1989

Classified Raster	Commision	Raw total	Percent
Bare land	**0**	**68**	**0**
Grass land	**0**	**69**	**0**
Woody veg	**1**	**70**	**1**
cultivated	**1**	**41**	**2**
Water	**2**	**42**	**5**

Ommision 1989

Classified Raster	Ommision	Column total	Percent
Bare land	**2**	**70**	**3**
Grass land	**1**	**70**	**1**
Woody veg	**1**	**70**	**1**
cultivated	**0**	**40**	**0**
Water	**0**	**40**	**0**

2A. 2000

Overall accuracy 2000 0,948276

Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**66**	0	0	0	0	**66**
Grass land	2	**66**	1	2	0	**71**
Woody veg	0	1	**66**	1	0	**68**
cultivated	2	3	3	**37**	0	**45**
Water	0	0	0	0	**40**	**40**
Total	**70**	**70**	**70**	**40**	**40**	**290**

2B

Commission 2000

Classified Raster	Commision	Raw total	Percent
Bare land	**0**	**66**	**0**
Grass land	**5**	**71**	**7**
Woody veg	**2**	**68**	**3**
cultivated	**8**	**45**	**18**
Water	**0**	**40**	**0**

Ommision 2000

Classified Raster	Ommision	Column total	Percent
Bare land	**4**	**70**	**6**
Grass land	**4**	**70**	**6**
Woody veg	**4**	**70**	**6**
cultivated	**3**	**40**	**8**
Water	**0**	**40**	**0**

3A 2013

Overall accuracy
2013 0,965517

Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**69**	0	0	0	0	**69**
Grass land	1	**67**	1	3	0	**72**
Woody veg	0	1	**68**	1	0	**70**
cultivated	0	2	1	**36**	0	**39**
Water	0	0	0	0	**40**	**40**
Total	**70**	**70**	**70**	**40**	**40**	**290**

3B

Commission 2013

Classified Raster	Commision	Raw total	Percent
Bare land	**0**	**69**	**0**
Grass land	**5**	**72**	**7**
Woody veg	**2**	**70**	**3**
cultivated	**3**	**39**	**8**
Water	**0**	**40**	**0**

Ommision 2013

Classified Raster	Ommision	Column total	Percent
Bare land	**1**	**70**	**1**
Grass land	**3**	**70**	**4**
Woody veg	**2**	**70**	**3**
cultivated	**4**	**40**	**10**
Water	**0**	**40**	**0**

OUTSIDE THE WBR

1A 1989

Overall accuracy 1989	0,862069					
Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**64**	0	0	0	7	**71**
Grass land	3	**68**	18	5	0	**94**
Woody veg	0	0	**50**	0	0	**50**
cultivated	3	2	2	**35**	0	**42**
Water	0	0	0	0	**33**	**33**
Total	**70**	**70**	**70**	**40**	**40**	**290**

1B

Commission 1989

Classified Raster	Commision	Raw total	Percent
Bare land	**7**	**71**	**10**
Grass land	**26**	**94**	**28**
Woody veg	**0**	**50**	**0**
cultivated	**7**	**42**	**17**
Water	**0**	**33**	**0**

Ommision 1989

Classified Raster	Ommision	Column total	Percent
Bare land	**6**	**70**	**9**
Grass land	**2**	**70**	**3**
Woody veg	**20**	**70**	**29**
cultivated	**5**	**40**	**13**
Water	**7**	**40**	**18**

2A 2000

Overall accuracy 2000	0,965517					
Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**69**	0	0	0	2	**71**
Grass land	0	**67**	0	2	0	**69**
Woody veg	0	0	**69**	1	0	**70**
cultivated	0	3	1	**37**	0	**41**
Water	1	0	0	0	**38**	**39**
Total	**70**	**70**	**70**	**40**	**40**	**290**

2B

Commission 2000			
Classified Raster	Commision	Raw total	Percent
Bare land	**2**	**71**	**3**
Grass land	**2**	**69**	**3**
Woody veg	**1**	**70**	**1**
cultivated	**4**	**41**	**10**
Water	**1**	**39**	**3**

Ommision 2000			
Classified Raster	Ommision	Column total	Percent
Bare land	**1**	**70**	**1**
Grass land	**3**	**70**	**4**
Woody veg	**1**	**70**	**1**
cultivated	**3**	**40**	**8**
Water	**2**	**40**	**5**

3A 2013

Overall accuracy 2013	0,944828					
Classified Raster	Bare	Grassland	Woody	Cultivated	Water	**Ground truth points**
Bare land	**68**	0	0	0	3	**71**
Grass land	0	**69**	4	3	0	**76**
Woody veg	0	0	**64**	1	0	**65**
cultivated	0	1	2	**36**	0	**39**
Water	2	0	0	0	**37**	**39**
Total	**70**	**70**	**70**	**40**	**40**	**290**

3B

Commission 2013			
Classified Raster	Commision	Raw total	Percent
Bare land	**3**	**71**	**4**
Grass land	**7**	**76**	**9**
Woody veg	**1**	**65**	**2**
cultivated	**3**	**39**	**8**
Water	**2**	**39**	**5**

Ommision 2013

Classified Raster	Ommision	Column total	Percent
Bare land	**2**	**70**	**3**
Grass land	**1**	**70**	**1**
Woody veg	**6**	**70**	**9**
cultivated	**4**	**40**	**10**
Water	**3**	**40**	**8**

	Bare land	Grassland	Woody vegetation	Cultivated	Water	Ground control points
Bare land	**69**	0	0	0	1	**70**
Grass land	1	**64**	7	1		**73**
Woody veg	0	2	**61**	0	0	**63**
cultivated	0	4	2	**39**	0	**45**
Water	0	0	0	0	**39**	**39**
Total	**70**	**70**	**70**	**40**	**40**	**290**

Classified Raster	Commision	Raw total	Percent
Bare land	**1**	**70**	**1**
Grass land	**9**	**73**	**12**
Woody veg	**2**	**63**	**3**
cultivated	**6**	**45**	**13**
Water	**0**	**39**	**0**

Classified Raster	Ommision	Column total	Percent
Bare land	**1**	**70**	**1**
Grass land	**6**	**70**	**9**
Woody veg	**9**	**70**	**13**
cultivated	**1**	**40**	**3**
Water	**1**	**40**	**3**

APPENDIX 2. PARAMETER ESTIMATES OF ALIEN INVASIVE PLANTS RECORDS WITHIN AND OUTSIDE THE WBR.

Parameter Estimates								
Parameter		B	Std. Error	95% Wald Confidence Interval		Hypothesis Test		
				Lower	Upper	Wald Chi-Square	df	Sig.
Plant species	*Cereus jamacaru*	-1.971	.2545	-2.470	-1.472	59.948	1	0.000
	Eucalyptus camaldulensis	-1.373	.2296	-1.823	-.923	35.732	1	0.000
	Jacaranda mimosifolia	-1.089	.2236	-1.527	-.650	23.710	1	0.000
	Lantana camara	-.859	.2213	-1.292	-.425	15.055	1	0.000
	Melia azedarach	0[a]						
Location: Inside BWR		-.238	.1958	-.621	.146	1.472	1	0.225
Location: Outside BWR		0[a]	.	.	.	.	.	.
Road type: Gravel		-.846	.2026	-1.243	-.449	17.432	1	0.000
Road type: Paved		0[a]	.	.	.	.	.	.
Landuse type: Cultivation fields		-.052	.2422	-.526	.423	.045	1	0.831
Landuse type: Natural vegetation		.043	.2673	-.481	.567	.025	1	0.873
Landuse type: Settlements		0[a]	.	.	.	.	.	.
(Scale)		1[b]						

Dependent Variable: Scientific.name

Model: (Threshold), Location, Road.type, Landuse.type

a. Set to zero because this parameter is redundant.

b. Fixed at the displayed value.