Conserving living landscapes: Investigating the impacts of livestock grazing and assessing rangeland restoration potential in Overberg Renosterveld, South Africa

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Conserving living landscapes: Investigating impacts of livestock grazing and assessing rangeland restoration potential in Overberg Renosterveld

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Abstract

Biodiversity is declining faster than at any other time in the Earth's history, driven mostly by land use change and degradation. Overberg Renosterveld, some of the most species diverse mediterranean type shrublands, are no exception with about 95% of their original extent lost to agriculture. Historically, large herds of indigenous grazing mammals roamed these landscapes. Today the Overberg's agricultural lands are fragmented by land cover change and divided by fences. In the contemporary landscape animals, largely domestic livestock, and plant resources are closely coupled, and overgrazing of remaining renosterveld fragments a significant threat, with potential to cause irreversible damage. The Conservation of Agricultural Resources Act (CARA) (Act 43 of 1983) states that farmers must not exceed the grazing capacity of the veld unless it is protected against deterioration and destruction, and that any land that is degraded or denuded must be effectively restored or reclaimed. Despite this legislation, there is little empirical research on the impacts of livestock grazing on renosterveld, as well as on restoration of overgrazed areas. It was the aim of this thesis to contribute to this gap in understanding. The thesis assessed the role of grazing by different livestock types, namely cattle and sheep, on biodiversity, the soil seed bank, and the restoration potential of renosterveld vegetation from resting the veld.

The effect of livestock grazing by sheep and cattle on plant species richness and diversity and growth form diversity was assessed using Modified Whittaker plots and presented in Chapter 3. It was hypothesised that livestock grazing by cattle would have less effect on species richness and diversity and growth form diversity than sheep grazing and that both cattle and sheep grazing would lead to a reduction in species richness and diversity in comparison to renosterveld sites with a treatment of no grazing. Thirty sites where either no grazing has taken place or that have been grazed by cattle or sheep were selected with sites being evenly distributed between Eastern, Central and Western Rûens Shale Renosterveld. At each of the thirty sites, vegetation data were collected from a series of nested subplots of ten 1 m^2, two 10 m^2 and one 100 m^2 subplots nested within a 1 000 m^2 plot. One soil sample was also collected from each 1 000 m^2 plot to a depth of 10 cm for nutrient analysis. Findings revealed that sites grazed by sheep had significantly lower plant species richness (median richness = 29 species, mean Shannon-Weiner = 3.39) and diversity when compared to sites with a treatment of no grazing (median richness = 49 species, mean Shannon Weiner = 3.83). Sites with a treatment of no grazing had significantly higher richness of geophyte species (mean = 14.7) than sites grazed by cattle (mean = 7.0) and sheep (mean = 7.1) during the study. The results obtained were in line with the hypothesis that livestock grazing by sheep resulted in a reduction in species richness and diversity and vegetation cover in Overberg Renosterveld in comparison to sites where no grazing has taken place. Sites with a treatment of no grazing showed higher species richness and vegetation cover of non-succulent shrubs, annual forbs and perennial forbs than sites grazed by sheep. It was concluded that livestock grazing of Overberg Renosterveld by sheep needs to be done with

care. This can be done by adopting a passive adaptive management approach. Here one set of management protocols can be developed and implemented and through science-based monitoring to inform management, these can be adapted as needed based on the key findings.

Chapter 4 investigated ecosystem resilience and the restoration potential of Overberg Renosterveld through an exploration of its soil seed bank as a source for potential recovery. A glasshouse germination experiment investigated the effect of livestock grazing by cattle and by sheep in comparison with a grazing treatment of no grazing on the soil seed bank in Overberg Renosterveld, as well as the similarity between the standing vegetation and the soil seed bank. It was hypothesised that cattle and sheep grazing would reduce species richness, species diversity and growth form diversity in the soil seed bank in comparison with sites with a treatment of no grazing. Soil samples were collected from 30 sites that were also used in Chapter 3. The soil was then spread on top of a 6 cm layer of compost in seed trays, and smoke treated to enhance germination. Seedlings were assigned to growth form categories including forbs, geophytes, annuals, graminoids, succulent shrubs and non-succulent shrubs and then identified to family, genus or species level. The results of the soil seed bank study were correlated with the vegetation results from Chapter 3 to examine the relationship between the standing vegetation and the soil seed bank. A total of 48% of taxa in the standing vegetation had seed present in the germinable seed bank. However, there were no differences in species richness, species diversity or number of individuals between grazing treatments. The results indicated that livestock grazing has a far less significant impact on the composition, species diversity and growth form diversity of the soil seed bank in Overberg Renosterveld than hypothesised. Instead, the results showed that there was a well-developed seed bank comprising mainly indigenous species with a variety of different growth forms including palatable grasses and shrubs. This indicates that Overberg Renosterveld vegetation has high restoration potential.

Chapter 5 showed results on the effects of livestock grazing by cattle and sheep over time on plant species richness, diversity and growth form diversity in comparison with sites protected from grazing. Following collection of a baseline dataset, four years of follow up data were collected. A total of 22 fenced plots across Western, Central and Eastern Rûens Shale Renosterveld had a baseline dataset collected prior to being monitored on an annual basis over four years in grazed/ungrazed paired plots. Results on vegetation recovery from the fenced exclosures showed a significant increase in plant cover over time at sites that were not grazed. Mean species richness increased from 20.6 species to 25.4 species at sites with no grazing. Mean vegetation cover increased from 71% at T0 (the baseline time step) to 120% at T4 (the final time step) at the end of the study. Sites grazed by sheep had a decrease in vegetation cover over time each year from T0 to T4 from 75% to 50%. Results from a linear mixed model revealed that species richness between grazing treatments was significantly different at all time steps in the study. However, the significant differences were primarily due to comparisons between grazed sites and sites with a treatment of no

grazing. Therefore, livestock grazing by sheep has a significant effect on renosterveld vegetation over time. Findings from this component of the study indicates that Overberg Renosterveld degraded by continuous heavy grazing has significant passive restoration potential by fencing renosterveld patches to facilitate more effective grazing management.

Most of the renosterveld of the Overberg has been lost through habitat transformation for agriculture, and the future of that which remains is uncertain. This thesis affirms concerns around the impact of livestock grazing and shows the importance of improved ecological understanding around grazing management. Grazing by sheep was shown to cause greater impacts on renosterveld than other domestic livestock studied and is therefore a threat to renosterveld. These findings warrant closer attention to management practices around sheep grazing. However, the state of renosterveld soil seed banks offer considerable hope. Findings revealed a diverse indigenous seed bank, showing that renosterveld degraded by overgrazing has high restoration potential. Furthermore, fencing renosterveld to exclude livestock improves species richness and diversity over time. These findings highlight the need for caution when grazing renosterveld. However, where the damage has been done, the potential for recovery is high. Harnessing the soil seed bank in combination with excluding livestock grazing by fencing are effective tools in this critically endangered vegetation for achieving restoration and conservation goals.

Keywords: *grazing; passive restoration; Overberg Renosterveld; Mediterranean type shrublands; rangeland ecology; soil seed banks*

Dedication

This thesis is dedicated to my Mother Jeany Caroline Poulsen (1952-2016) who always loved the natural world, its wildlife and wildflowers.

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Chapter 1: Introduction

1.1. Background and Rationale

The world's biodiversity is declining at a rate faster than at any other time in the earth's history, driven predominantly by land use change and habitat degradation (Alkemade et al. 2013; IBPES, 2019). The average abundance of indigenous species in most terrestrial ecosystems has fallen by at least 20% (IBPES, 2019), with severe impacts on global ecosystem structure and function (Gonthier et al. 2014). Human activities have threatened as many as 1 million plant and animal species across the globe, which represents about 25% of all species (IBPES, 2019). These statistics only represent the species that have been assessed as threatened with extinction and therefore are on the IUCN Red List, with many, particularly plants and insects, still yet to be assessed (Callmander et al. 2005; Eggleton, 2020). It is widely recognised that despite efforts to increase the rate at which plant species are assessed, many may become extinct prior to being assessed, and in some cases, even before they have been discovered and described (Willis, 2017). Nature across the globe is becoming significantly altered primarily as a result of land transformation which now affects more than 75% of the Earth's surface (IBPES, 2019). Over 85% of the world's wetlands have already been lost (IPBES, 2019). Land transformation represents some of the most pressing environmental issues today (Segan et al. 2016). The spread and intensification of agriculture since 1970 has been the most significant driver of land changes, with over one third of the terrestrial land surface now used for cropping and animal agriculture (IPBES, 2019). Grazing occurs on 25% of the world's ice-free lands and approximately 70% of drylands (IPBES, 2019). Vegetation and wild herbivores co-evolved in many of these rangeland systems, with some of these among the most species diverse ecosystems in the world (Fornoni, 2011; Sayre et al. 2013). Loss of wildlife, particularly in African rangelands, is occurring both from habitat transformation for growing crops and from competition with grazing livestock (Alkemade et al. 2013). Furthermore, the impacts of a changing climate are further exacerbating the impact of other pressures on ecosystems around the world (IPBES, 2019).

The National Biodiversity Assessment (NBA) (Skowno et al. 2019) highlights South Africa's global biodiversity importance. South Africa has been identified as one of the world's 17 megadiverse nations (Thuiller et al. 2006). These nations combined contain two thirds of the world's plant diversity (Skowno et al. 2019). South Africa ranks among the top ten nations in the world for plant species richness (Skowno et al. 2019). As a result of bioclimatic variation and a highly varied topography and geology, South Africa has exceptionally high ecosystem diversity, species diversity and levels of endemism (Skowno et al. 2019). The country's terrestrial realm can be divided into nine different biomes with 458 ecosystem types, of which 80% of their species are endemic (Skowno et al. 2019). However, many of South

Africa's highly biodiverse ecosystems are under threat for a variety of different reasons including habitat loss from urban development, transformation for agriculture and alien plant invasion (Skowno et al. 2019). At present, 22% of South Africa's ecosystems are considered threatened and 25% currently do not fall within any form of protected area (Skowno et al. 2019). A total of 20 401 plant species across the country have been assessed through the Red List of South African Plants and 14% are currently assessed as threatened species (Skowno et al. 2019). The NBA calls for better monitoring protocols for rangelands, because at present most untransformed natural vegetation is mapped as 'natural/near natural' (Skowno et al. 2019). This has led to levels of ecosystem modification being underestimated. This increases the likelihood that an ecosystem may be at greater risk of collapse than the available data suggests (Skowno et al. 2019).

Renosterveld is one of several distinct vegetation types found within South Africa's Fynbos Biome (Krug, 2004). It is characterised by a dominance of Asteraceous shrubs including Renosterbos (*Dicerothamnus rhinocerotis*) (Asteraceae) and C_3 grasses and occurs on fertile soils mainly derived from shales (Rebelo et al. 2006). It is also known for its diversity of geophytes, many of which are local endemics (Paterson-Jones, 1998). Renosterveld forms part of the vegetation of the Cape Floristic Region (CFR) which is recognised as one of the world's biodiversity hotspots (Myers et al. 2000; Goldblatt & Manning, 2012). It is also a UNESCO World Heritage Site owing to exceptional plant diversity at all taxonomic levels (Van Wilgen et al. 2016). The fynbos vegetation of the CFR has been studied extensively from a variety of perspectives but until recently renosterveld has been neglected by ecological researchers (Curtis, 2013a). Renosterveld was commonly perceived to be a homogenous vegetation type with few localised rare species (Rebelo, 1995). Further research undertaken by Newton and Knight (2010) has shown that this is not the case. For example, renosterveld fragments less than 2 km apart have been shown to differ by as much as 80% in species composition (Curtis, 2013a, Curtis, 2013b). Alpha diversity of renosterveld is now known to be at least equal to, if not richer than comparable fynbos habitats as well as other Mediterranean type shrublands in other parts of the world (Curtis, 2013a). Owing to its extensive transformation for agriculture, renosterveld has now become a Critically Endangered vegetation type and only 4-6% of its original extent remains (Winter et al. 2007; Curtis, 2013a; Moncrieff, 2021). It is regarded as one of South Africa's most under conserved ecosystems (Krug, 2004). Renosterveld has also become highly fragmented with a total of 12 296 patches remaining, only 72 of which are greater than 80 ha in size (Topp and Loos, 2019). Most renosterveld fragments are on private farmland with farmers as the custodians of this unique and highly biodiverse vegetation type (Winter et al. 2007). How they decide to use and manage their veld is critical and makes the difference between contributing to habitat loss or effective renosterveld conservation (Meadows, 2003).

A further challenge of conserving this Critically Endangered vegetation type is that there is currently very little known about renosterveld ecology (Rebelo, 1995; Cowan, 2019) with associated uncertainty around best management practices (Chandra et al. 2009). In many cases the only information available relies on comparative studies undertaken in adjacent karroid or fynbos habitats (Curtis, 2013a). The degradation of renosterveld fragments due to mismanagement is common and widespread (Meadows, 2003). Management quandaries are commonplace in renosterveld and further applied research is needed to inform management practices (Chandra et al. 2009). There is currently almost no quantitative data available to inform grazing practice and whether the presence of grazing livestock is either beneficial or detrimental to renosterveld (Curtis, 2009; Curtis, 2013a). Consultation with conservation managers and farmers in the Overberg suggested that key priorities for further research to inform management in renosterveld were first fire, with grazing listed as the second most important (Curtis, 2009). Recent research by Curtis (2013a) suggested that although species diversity and vegetation cover were not affected by grazing, plant productivity of palatable species was significantly reduced. Findings from recent research suggested that at present we do not yet have the capacity to restore a renosterveld ecosystem to its original state after destruction by ploughing or mismanagement (Heelemann, 2010). Results have shown that although the vegetation structure and keystone shrub species such as *Dicerothamnus rhinocerotis* will return, the species diversity and complexity of the system will not return despite active restoration interventions (Heelemann, 2010). However, the majority of empirical research in renosterveld restoration has been concentrated in old fields (Heelemann, 2010). Ploughing has brought about low restoration potential as a result of changes in soil chemistry, soil structure and loss of the soil seed bank with alien grass competition also acting as a barrier to recruitment (Heelemann, 2010; Midoko-Iponga et al. 2005).

It is clear that there is a need to act now to reduce the acceleration in the global rate of species extinctions and habitat loss (IBPES, 2019). To do this it is vital to conserve the ecosystems that still remain. This is particularly important within rangelands as livestock production grows with increasing population growth (Alkemade et al. 2013). As demand for agricultural production grows, conserving the biodiversity adjacent to agricultural lands has become a key focus of conservation policy (Gonthier et al. 2014). It is crucial to maintain biodiversity within agricultural landscapes by incorporating more natural and semi-natural landscapes and ecosystems on farms into conservation and maintaining high habitat diversity in agricultural landscapes (Gonthier et al. 2014). Despite this, the impacts on rangeland biodiversity from increasing intensity of livestock production still remain poorly understood (Alkemade et al. 2014). Nowhere could this be truer than the renosterveld vegetation of the Overberg (Curtis, 2013a). To improve understanding and to undertake effective conservation of this Critically Endangered vegetation, there is a significant need to develop more effective protocols around ecosystem management and ecological restoration (Forbes, 2014).

1.2. Attending to gaps in the knowledge base

This research focused on establishing the role of livestock grazing by cattle and sheep in Overberg Renosterveld with a view to inform conservation-appropriate land management practices. Furthermore, this research has led to better understanding of the restoration ecology of renosterveld degraded through overgrazing by domestic livestock. According to the Society of Ecological Restoration (SER), ecological restoration is defined as "an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its its health, integrity and sustainability. Frequently, the ecosystem that requires restoration has been degraded, damaged, transformed or entirely destroyed as the direct or indirect result of human activities" (SER, 2004: pp. 1). Ecological restoration aims to recover the characteristics of an ecosystem such as its biodiversity and function, which have been degraded or destroyed, primarily as a result of human activities (Benayas and Bullock, 2012). Restoration interventions can greatly increase the conservation value of degraded renosterveld fragments. This is important given the paucity of renosterveld which remains and the associated difficulty in meeting conservation targets (Waller, 2013). However, active restoration interventions can be expensive and without substantial external funding can often be beyond the capacity of most private landowners (Pressey et al. 2003; Waller, 2013). In consequence, it is critical that active restoration efforts are appropriately targeted within areas where there are no other alternatives for rehabilitation (Holmes et al. 2020). Further research was therefore needed to better understand the extent to which overgrazed renosterveld will recover as a result of passive restoration.

This research examines the impacts of livestock grazing by cattle and sheep on renosterveld vegetation with a key focus on vegetation species richness, species diversity and growth form diversity. Modified Whittaker Plots were used, for the first time in Overberg Renosterveld, to create a baseline dataset to better understand the role of grazing by cattle and sheep on Western, Central and Eastern Rûens Shale Renosterveld (Mucina and Rutherford, 2006). This method allowed for thorough vegetation surveying, encompassing a series of nested subplots that allow species diversity to be analysed (Stohlgren, 1995; Stohlgren et al. 1998). Another key area for further research was to assess the impact of grazing by cattle and sheep on the soil seed bank. The soil seed bank is an important component of ecosystem resilience and a key determinant of restoration potential in degraded systems (Wang et al. 2013). The majority of research in this field encompasses grassland vegetation (Jacquemyn et al. 2011) but there has also been some attention paid to shrubland ecosystems. A study undertaken in the calcareous grasslands of arid central Australia, for example, found that heavy grazing significantly reduced the size and diversity of the germinable seed bank but light to moderate grazing had no significant effect (Kinloch and Freidel, 2005). In contrast, a 36 year field experiment in the semiarid grasslands of Texas found no significant effect of grazing on the soil seed bank (Kinucan and Smiens, 1992). In contrast, numerous studies undertaken in calcareous grasslands in Europe have found a significant decrease in species diversity in the soil seed bank as a result of the cessation of grazing (Barbaro et al. 2001; Bossuyt et al. 2006; Jacquemyn et al. 2011). However, the effects of this often only became apparent more than 50 years after livestock removal (Bakker et al. 1996b; Jacquemyn et al. 2011).

Another key aim of this research was to provide a baseline dataset on trajectories of vegetation recovery over time following livestock grazing in Overberg Renosterveld to begin to fill this gap in the literature. This study using paired fenced/unfenced grazing plots upscaled from the initial work undertaken on livestock grazing in Overberg Renosterveld by Curtis (2013a), with a larger number of replicate paired plots used across sites grazed both by cattle and by sheep across a larger study area. This upscaling is in line with recommendations by Stohlgren et al. (1999) that highlights that many of the exclosure-based grazing studies undertaken to date have an insufficient number of replicates to draw the most meaningful conclusions possible. This study is one of few studies that recorded changes in grazing over a timeline of several years. Research over longer timescales is important to attain a more detailed picture of the effects of livestock grazing and rest from grazing in this vegetation type. Research findings from this study will be used to improve conservation guidelines on appropriate grazing management for Overberg Renosterveld. This project brings together collaborators from the University of Cape Town and the Overberg Renosterveld Conservation Trust to conduct research into the role of grazing in renosterveld vegetation.

1.3. Location and Regional Context

The study area is located within South Africa's Overberg Region which lies from 19-21°E in the Western Cape at the southern tip of Africa (Shaw et al. 2010). The study area encompasses the range of Overberg Renosterveld vegetation (also known as South Coast Renosterveld). This runs from north of Botrivier in the east to Heidelberg in the west and southwards to Bredasdorp bounded by the Soetmuisberg Mountains above the village of Napier and the Akkedisberg Mountains to the west (Curtis et al. 2013). Field sites are highlighted in Figure 1.1. in red circles.

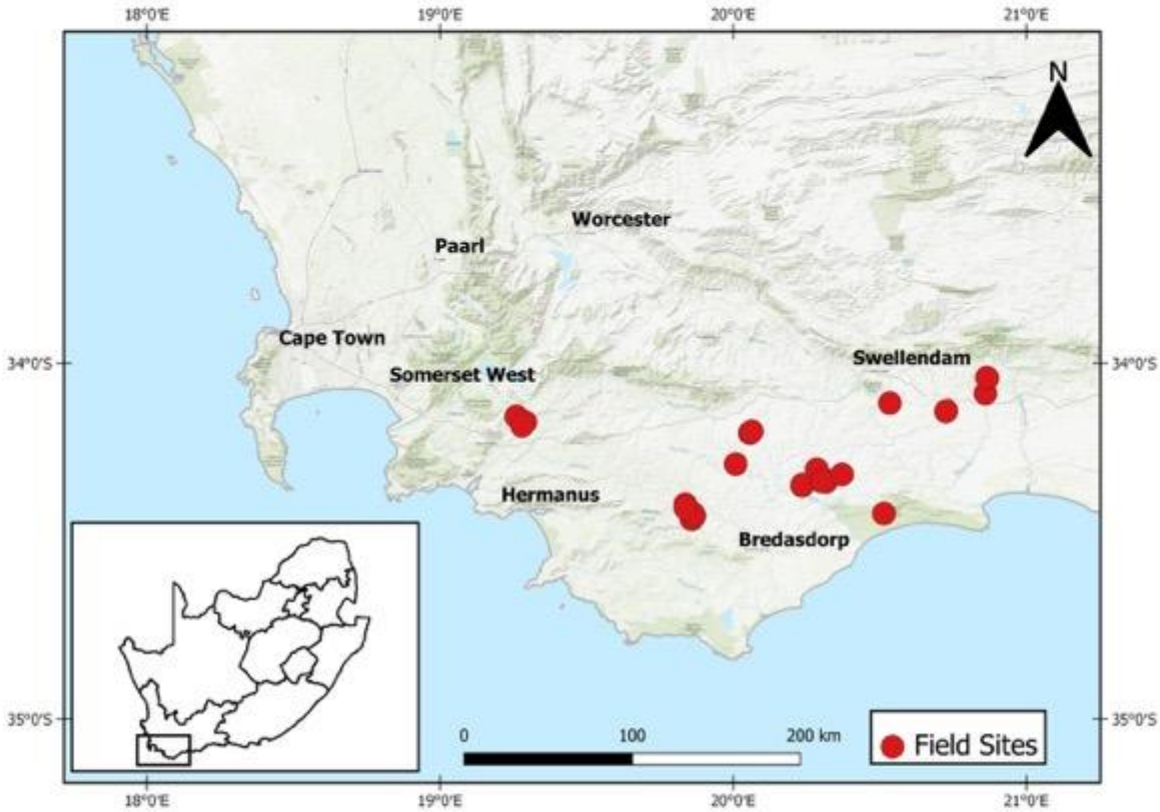

Figure 1.1: Location map showing study sites for the project in the Overberg area.

1.4. Topography and Hydrology

The majority of the study area is comprised of gently rolling hills derived from shale geology which varies from around 100-350 m asl. This is a landscape of relative heterogeneity and there is a very high turnover of geology and soils across the region (Cowling et al. 2009; Curtis, 2013a; Curtis et al. 2013). The largest river in the region and also the Western Cape is the Breede River which has its source in the Skurweberg Mountains near Ceres (Steynor et al. 2009). It has a catchment of 12 600 km² and drains the eastern Overberg, reaching the sea at Witsand and Cape Infanta (Steynor et al. 2009). The Sout River is the other major river in the region which runs across the region in a south-easterly direction, draining into De Hoop Vlei at De Hoop Nature Reserve (Haddad et al. 2009).

1.5. Geology and Soils

Overberg Renosterveld is dominated by deep clay soils derived from shales from the Devonian Bokkeveld Group which forms part of the Bidouw Subgroup (Raitt, 2005; Groenewald, 2014; Radloff et al. 2014). The region has a complex geomorphological history which contributes to the species diversity of the area (Cowling et al. 2009). In the Overberg, the old African Surface was capped by silcrete duricrusts (Cowling et al. 2009; Curtis et al. 2013). These were deposited throughout the area during the Palaeocene (Cowling et al. 2009; Curtis et al. 2013). These later eroded to reveal the Devonian and Cretaceous sediments underneath, from which the fertile soils that make this area so significant for agriculture are derived (Cowling et al. 2009; Curtis et al. 2013). However, small patches of silcrete duricrusts still remain, predominantly in the eastern parts of the Overberg (Curtis, 2013a; Curtis et al. 2013). These patches take the form of quartz-silcrete and often found on the tops of koppies, literally translated as ‘small hills’ in English, in this area (Curtis et al. 2013; Magee et al. 2016). These quartz-silcrete patches are known for their succulent flora and high degree of endemism, in some cases up to 91% (Curtis et al. 2013). There are also island-like deposits of silcrete in proximity to the Breede River (Groenewald, 2014), which were formed during the Eocene and Miocene (Alley, 1977). These form the habitat of Rûens Silcrete Renosterveld which like the other three renosterveld types in the region is classed as ‘Critically Endangered’ (Groenewald, 2014). Along the rivers in the area, the Quaternary Bredasdorp Group is characterised by light grey to red sandy soils (Raitt, 2005).

1.6. Landscape History

Historically the rolling lowland landscapes of the Overberg would have looked completely different to the predominantly agricultural landscape of today complete with its wheat, barley and canola fields (Cowan, 2019). The small fragments of renosterveld vegetation, now mainly restricted to kloofs and klipbank, are all that remains of a vegetation that once occurred across the whole of the Overberg Rûens northwards to the Langeberg and Riviersonderend Mountains (Molteno & Smith, 2019). This landscape would have supported a far more diverse fauna than the crop monoculture it does today (Curtis, 2013a; Boshoff & Kerley, 2001). Historical records provide some idea of the richness of this landscape at the start of colonial settlement, with far more extensive renosterveld vegetation teeming with large herds of game and other wildlife (Skead, 2011; Curtis-Scott et al. 2020). This wild landscape was not, however, uniformly appreciated. In 1842, C.J.F. Bunbury wrote: “…the country that we traversed in this long day’s journey…was drearily monotonous; wide plains and low round hills, uniformly covered with stunted bushes, without trees or cultivation, offering nothing either to please the eye or excite the imagination” (Bunbury, 1842: pp. 17). Few travellers moving eastwards from Cape Town journeyed south into the Bredasdorp District. Much of this region lay south of the main trek route between Caledon and Swellendam. Written accounts of the historic landscapes of this part of the Overberg are therefore few in number (Skead, 2011).

The Western Cape was first occupied by European settlers in the mid-1600s although it is thought that prior to this the Khoe pastoralists had been living in the region for approximately 2 000 years (Humphreys, 1981). Prior to the arrival of the European settlers, the landscape was home to large herds of medium sized ungulates and their associated predators. Most of these were soon hunted to extinction in this region (Skead, 2011; Boshoff and Kerley, 2001). The Blue Buck (*Hippotragus leucophaeus*) and the Quagga (*Equus quagga quagga*) which are thought to have occurred here historically are both extinct and are known today only from illustrations and photos (Rookmaaker, 1992; Raitt, 2005;

Heywood, 2013). Research has suggested that larger herbivores would have avoided fynbos vegetation which occurs on nutrient-poor sandstone and limestone derived soils, instead favouring more nutrient-rich shale renosterveld habitats (Radloff, 2008). It is thought that forage would have been restricted in fynbos vegetation during the hot and dry summer months (Radloff, 2008). Anders Sparrman passed over the Hottentots Holland Mountains on the 26 July, 1775. Upon reaching the other side, he describes riding over plains, hills and dales and seeing large herds of plains game and ostriches (*Struthio camelus*) which suggest the presence of favourable grazing conditions and a lack of disturbance (Skead, 2011). In addition to the Blue Buck and the Quagga, the following grazing mammals are known to have historically occurred across the region: Cape Buffalo (*Syncerus caffer*), Cape Mountain Zebra (*Equus zebra*), Bontebok (*Damaliscus pygargus*), Steenbok (*Raphicerus campestris*), Blue Duiker (*Philantomba monticola*), Springbok (*Antidorcas marsupialis*), Common Duiker (*Sylvicapra grimmia*), Cape Grysbok (*Raphicerus melanotis*), Klipspringer (*Oreotragus oreotragus*), Grey Rhebok (*Pelea capreolus*), Bushbuck (*Tragelaphus scriptus*) and Red Hartebeest (*Alcelaphus buselaphus caama*) (Raitt, 2005). The Cape Buffalo was extinct in the region by 1819 (Raitt, 2005).

It is not known how far elephant (*Loxodonta africana*) occurred towards the coastline in the Caledon and Bredasdorp Districts. There is little evidence on this (Skead, 2011). In 1819 JWD Moodie commented that elephants did not appear to have been found near Swellendam originally (Skead, 2011). Although written accounts about Black Rhinoceros (*Diceros bicornis*) in the Overberg are notably absent, archaeological evidence suggests that they were present in the region until about 500 BP (Skead, 2011). A complete skeleton was found at Pearly Beach south of Caledon (Skead, 2011). Fragments of teeth were found exposed in eroded calcrete along the coast near Struisbaai and four discoveries of skeletal material were also found in the dunefield between De Hoop Vlei and the coast of De Hoop Nature Reserve (Skead, 2011). While larger predators, such as lion (*Panthera leo*) used to occur in the region, Sparrmann commented that they were virtually extinct in the Caledon District by 1775 (Skead, 2011). Other predatory mammals remained longer in the region with Sparrman making reference to jackals (*Canis mesomelas*), “wolves” (Hyena) (*Hyaena brunnea*) and leopard (*Panthera pardus pardus*) in Caledon as late as 1816 (Skead, 2011). It is possible that the decline in leopard populations may have led to an increase in baboon (*Papio ursinis*) and porcupine populations (*Hystrix africaeaustralis*) in the area (Raitt, 2005). Baboons are omnivorous and often target grasses, seeds, fruit, pods, flowers, bark, shoots and leaves (Hamilton et al. 1978). Porcupines eat bulbs, roots and tubers (Pillay et al. 2015). It is possible that an increase in these mammals would have resulted in a decrease in population of certain geophytes in renosterveld within recent historic time (Raitt, 2005). Today the large herds of ungulates that used to roam the landscapes of the Overberg are long since gone. Some populations of antelopes remain (Curtis et al. 2020). Small herds of Grey Rhebok are often seen as well as smaller antelope such Steenbok and Klipspringer (Kraaij and Novellie, 2010). But vast herds of game and their large predators no longer occur in this landscape, now mainly ploughed and replaced by a monoculture of grain and canola (Cowan, 2019). These selective grazers adapted to life in the renosterveld of the Overberg that formerly occurred have now been replaced by sheep and cattle which are non-selective grazers and not adapted to survive in renosterveld (Voeten and Prins, 1999; Brand, 2000). We are only now beginning to understand exactly what impact these farmed mammals are having on what little remains of the renosterveld vegetation of the Overberg Rûens.

1.7. Regional Biodiversity

Renosterveld vegetation is considered part of the Fynbos Biome which in turn comprises part of South Africa's CFR (Rebelo et al. 2006). However, some researchers have demonstrated that renosterveld vegetation should in fact be considered a distinct vegetation type (Bergh et al. 2014). The CFR is internationally known for its extraordinary plant diversity at all taxonomic levels and is classified as a UNESCO World Heritage Site (Tassone et al. 2021). It is also recognised as a biodiversity hotspot (Rebelo et al. 2010). Recently the CFR was expanded and became recognised as the Greater Cape Floristic Region, with the former extent of the CFR becoming known as the Core Cape Subregion (CCR) (Manning and Goldblatt, 2012). The Core Cape Subregion encompasses an area of 90 760 km^2 and is home to around 9 383 species of vascular plants, of which 68% are endemic to the area (Brown and Botha, 2004; Manning and Goldblatt, 2012). This constitutes 46% of that of the whole of the southern African region (Manning and Goldblatt, 2012). There are four different biomes recognised to occur within the area, each with its own distinct vegetation types (Mucina and Rutherford, 2006; Manning and Goldblatt, 2012). This study focuses on the renosterveld vegetation of the area which is described in more detail in section 1.8.

1.8. Overberg Renosterveld Vegetation Types

There are four types of renosterveld vegetation that occur in the Overberg, collectively known as South Coast Renosterveld (Von Hase et al. 2003). All four of these vegetation types are described by Rebelo et al. (2006) as being "moderately undulating hills and plains supporting cupressoid and small-leaved low to moderately tall grassy shrubland, dominated by renosterbos" (Rebelo et al. 2006: pp. 184). Western, Central and Eastern Rûens Shale Renosterveld collectively comprise the majority of renosterveld vegetation in the region (Rebelo et al. 2006) but there are also smaller 'islands' of Rûens Silcrete Renosterveld which occur from Botrivier eastwards across the region. However, the majority of Rûens Silcrete Renosterveld is concentrated within the vicinity of the banks of the Breede River (Groenewald, 2014). All renosterveld types in the Overberg are Critically Endangered and are classed as 100% irreplaceable (Von Hase et al. 2003). For the purposes of this project, the study focused on the three main shale renosterveld types which are shown in Figure 1.2. The fourth renosterveld vegetation type found in the Overberg is located on silcrete derived soils and was not included to reduce the compounding effect of soil types. Dominant soils for all three vegetation types are Glenrosa and Mispah Forms (Rebelo et al. 2006; Curtis, 2013a). Rebelo et al. (2006) also state that *Hyparrhenia hirta* (L.) Stapf (Poaceae) is the most common palatable grass within all three of these vegetation types, but it has only been seen growing along road verges in the Overberg (Curtis, 2013a).

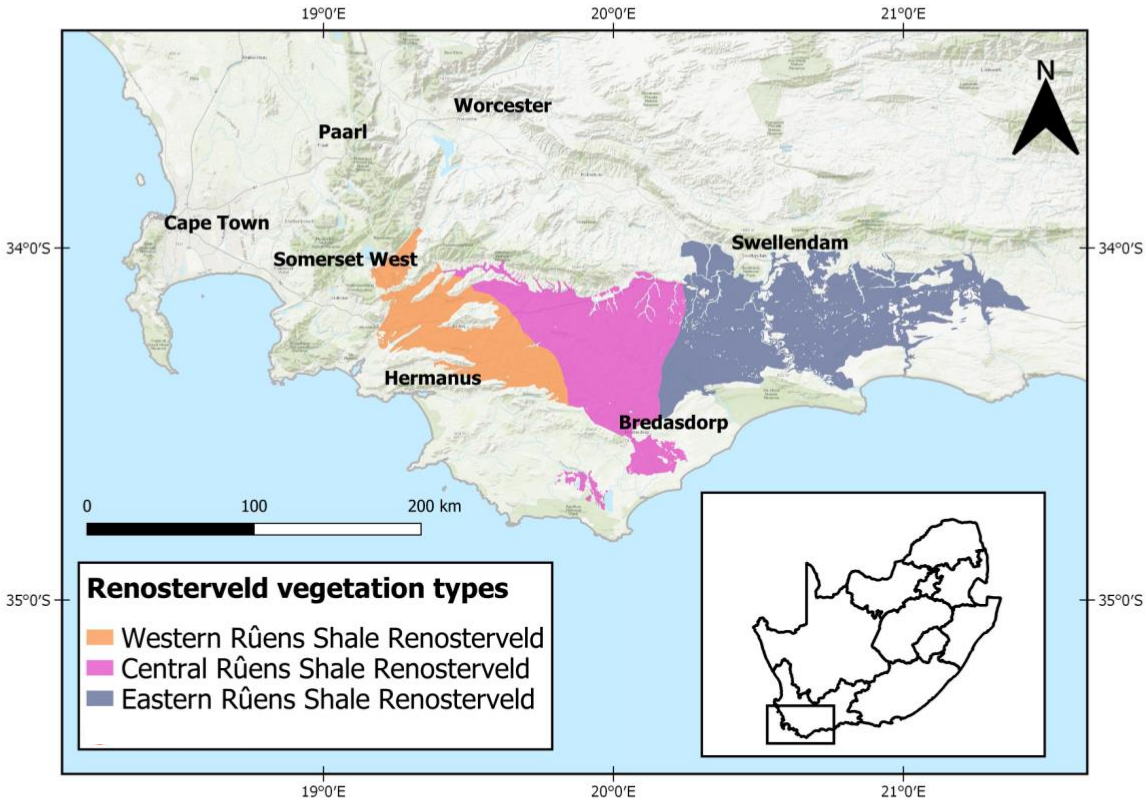

Figure 1.2: Vegetation map showing field sites and the three Overberg Renosterveld vegetation types that form the focus of the study.

Western Rûens Shale Renosterveld (Figure 1.3) is distributed from Botrivier and Villiersdorp, with some of the largest fragments occurring in the Shaw's River Pass area, eastwards towards Napier (Rebelo et al. 2006). According to Rebelo et al. (2006), it is distinguished from Central and Eastern Rûens Shale Renosterveld by the absence of *Hermannia flammea* Jacq. (Malvaceae) and rarity of *Aloe ferox* Mill. (Asphodelaceae) and *Vachellia karoo* (Hayne) Banfi and Gallaso. (Fabaceae) It tends to be a grassy renosterveld type with a significant geophyte component (Curtis, 2013a). Central Rûens Shale Renosterveld (Figure 1.3) occurs across the central Overberg from Stormsvlei to Napier and Bredasdorp (Rebelo et al. 2006). This vegetation type is delimited from Western and Eastern Rûens Shale Renosterveld by Rebelo et al. (2006) solely by the absence of *Aloe ferox*. Curtis (2013a) also adds that *Pteronia incana* (Burm.) DC. (Asteraceae) and *Galenia africana* L. (Aizoaceae) are also predominantly absent from this vegetation type. It is a grassier renosterveld type than Eastern Rûens Renosterveld (Figure 1.3) and is also richer in geophyte species (Curtis 2013a). Eastern Rûens Shale Renosterveld is found eastwards from Bredasdorp and northwards towards Swellendam as far as the foothills of the Langeberg Mountains (Rebelo et al. 2006; Curtis, 2013a). It occurs across the Malgas and Heidelberg districts as far as the Goukou River near Riversdale (Rebelo et al. 2006; Curtis, 2013a). It is a highly varied vegetation which has led to the suggestion that it may merit subdivision into several different renosterveld types (Curtis et al. 2013; Curtis, 2013a). It has an overall lower grass component than Western and Central Rûens Shale Renosterveld but becomes more grassy in proximity to the foothills of the Langeberg where *Themeda triandra* Forssk. (Rooigras) (Poaceae) becomes dominant (Raitt, 2005; Curtis, 2013a). Eastern Rûens Shale Renosterveld is also characterised by the presence of quartz-silcrete patches (Fig. 1.3) which are known for their diverse succulent flora and extraordinarily high levels of plant endemism (Curtis et al. 2013). Within recent years several new species endemic to this habitat have been discovered and described, including *Aspalathus quartzicola* C.H.Stirt. and Muasya (Fabaceae) and *Ficinia overbergensis* Muasya and C.H.Stirt. (Cyperaceae) (Curtis et al. 2013; Curtis, 2013a). Furthermore, taxonomic research into these under-researched ecosystems have contributed to multiple revisions of key genera and taxonomic novelties present in renosterveld, such as research by Zhigila et al. (2020) on the genus *Thesium* (Santalaceae).

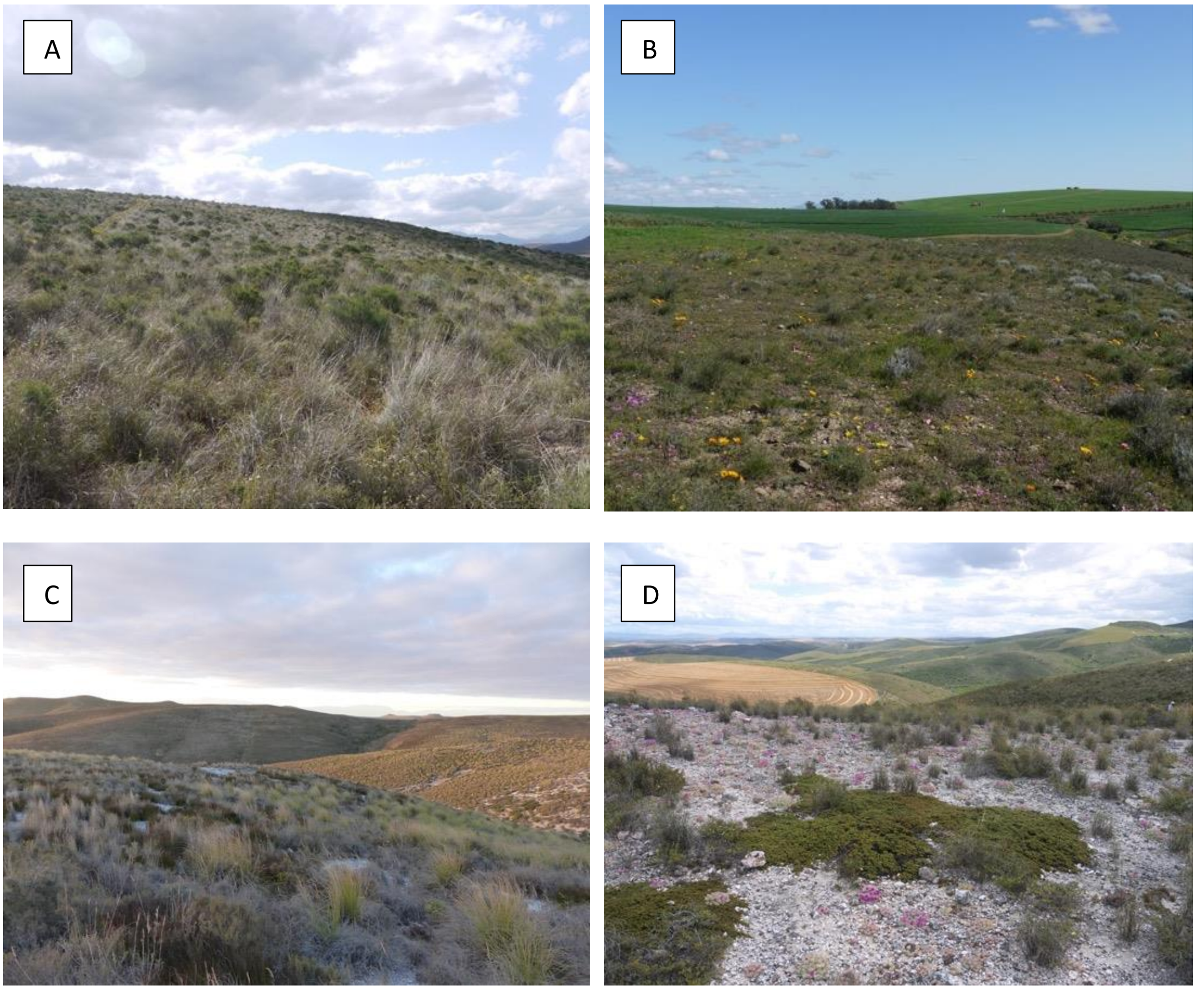

Figure 1.3: A: Western Rûens Shale Renosterveld on Fisantekraal farm near Botrivier in the western Overberg. B: Central Rûens Shale Renosterveld on Volmoed farm in the central Overberg. C: Eastern Rûens Shale Renosterveld at Haarwegskloof Renosterveld Reserve in the eastern Overberg. D: Quartz-Silcrete Patch within Eastern Rûens Shale Renosterveld in the Overberg Region, showing *Aspalathus quarticola* forming a green mat-like covering, one of six new species described in Curtis et al. (2013).

1.9. Regional Climate

Table 1.1 shows the climate of the three main shale renosterveld types found in the Overberg. The wettest Overberg weather is found in the Western Rûens, which has a mean annual precipitation of 490 mm. Both the Central and Eastern Rûens have a significantly drier climate, receiving a mean annual precipitation of 380 mm and 385 mm respectively mm (Rebelo et al. 2006). Western Rûens Shale Renosterveld receives the majority of rainfall during the winter months, but further east the area experiences year-round rainfall with an even distribution throughout the year (Rebelo et al. 2006; Curtis, 2013a). All three of these Overberg vegetation types experience their lowest mean daily minimum temperatures during July. There is little variation in minimum temperature between the three vegetation types (Table 1.1). All three vegetation types experience three days of frost per year on average. The highest mean summer temperatures occur in the Western Rûens, with the mean maximum summer temperature being lower for the Central Rûens and lowest for the Eastern Rûens (Rebelo et al. 2006).

Table 1.1: Table showing climatic variation between the three Overberg Renosterveld vegetation types encompassed in the study (data from Rebelo et al. 2006). (min = minimum, max = maximum)

	Western Rûens Shale Renosterveld	Central Rûens Shale Renosterveld	Eastern Rûens Shale Renosterveld
Mean annual precipitation	490 mm	380 mm	385 mm
Precipitation distribution	Peak May-August	Peak August	Even distribution
Mean daily min temp.	6.1 °C	5.6 °C	5.9 °C
Month of mean daily min temp.	July	July	July
Mean daily max temp.	29.9 °C	27.3 °C	26.9 °C
Month of mean daily max temp	February	January	January
Frost Incidence	3 days per year	3 days per year	3 days per year

1.10. Renosterveld and Fire

Like fynbos, renosterveld vegetation is also prone to fire (Cousins et al. 2017). Fire and grazing are both important ecological drivers in renosterveld vegetation, but their relative importance is currently unclear (Cousins et al. 2017). However, following several recent studies (e.g. Van De Merwe and Van Rooyen, 2011; Cousins et al. 2017) a small but growing body of literature has emerged concerning the role of fire in renosterveld (Cousins et al. 2017). It was originally thought that renosterveld was formerly a grassland with increasing shrub cover caused by overgrazing (Cowling et al. 1986). It was also considered to be adapted to short fire intervals of three to five years (Boucher, 1983; Beukes, 1987). These earlier ideas were attributed to renosterveld having a considerable number of plant species that have short maturation times (Kraaij, 2010; Kraaij and Van Wilgen, 2014). It has also been suggested that the shorter fire frequencies that may happen as a result of changing climate will not be a problem for this vegetation (Forsyth and Van Wilgen, 2008). However, more recent reports have revealed that this is unlikely to be the case (Cousins et al. 2017). Renosterveld in more arid areas will likely have a far longer natural fire interval (Bergh et al. 2014). This is particularly the case in areas of Overberg Renosterveld which receive lower rainfall and have a high succulent component to the vegetation such as quartz patches that occur within a matrix of Eastern Rûens Shale Renosterveld (Schmiedel et al. 2006). It is likely that this habitat would not have burnt as a result of its more rocky, open structure (Kraaij and Van Wilgen, 2014). These quartz patches are also home to some slow-maturing plant species, such as *Relhania garnottii* (Less.) K.Bremer (Asteraceae) that is thought to take up to twenty years to reach maturity (Curtis et al. 2013). However, palaeoecological research by Forbes et al. (2018) in the West Coast Renosterveld at Elandsberg Nature Reserve reveals an increase in fire frequency over time after the early colonists arrived, followed by a decrease in fire frequency after land use transition of the reserve from predominantly agriculture to ecological conservation. It is likely that this change is driven by more frequent burning of renosterveld vegetation to improve grazing for livestock (Forbes et al. 2018). This highlights the variability in renosterveld vegetation communities across the Cape lowlands and it is therefore important not to generalise with regard to recommendations around fire frequency in the context of fire management (Curtis, 2013a).

Another key question which can be asked about the role of fire in renosterveld is the extent to which fire create gaps in the landscape to facilitate germination and flowering of understorey species (Topp et al. 2021). Furthermore, there is the question of the impact of fire on plant species richness and diversity (Topp and Loos, 2019; Kraaij and Novellie, 2010). Fire plays an important role in creating space for plant recruitment, maintains spatial heterogeneity in vegetation at the landscape scale and avoids competitive exclusion, thus maintaining species richness (Van de Merwe and Van Rooyen, 2011). Due to its long history of being home to large herds of grazing megaherbivores to a far greater extent than fynbos, it is likely that grazing mammals also played an important part in modifying vegetation structure and plant species diversity by driving gap formation in renosterveld (Skead, 2011; Cousins et al. 2017). In fact, Krug et al. (2004) argued that herbivory was the primary driver of diversity in renosterveld with fire playing a far lesser role. However, there have been several studies that have shown that after fire renosterveld vegetation has an increase in species richness in comparison with unburnt renosterveld. Van de Merwe and Van Rooyen (2011) examined changes in vegetation species richness and diversity over time in Roggeveld Mountain Renosterveld after fire. They found that vegetation cover began to re-establish in the first nine months after fire, with cover remaining high throughout the rest of the study. Species richness was highest three years after fire and remained high from year three to year ten after the fire. The lowest species diversity was recorded in the ninth and tenth year after fire (Van de Merwe and Van Rooyen, 2011). Research by Curtis (2013a) in Overberg Renosterveld found that burnt renosterveld had higher species richness as well as higher numbers of flowering geophytes and annuals when compared with unburnt vegetation. More recent research by Cousins et al. (2017) in Swartland Renosterveld revealed significant increase in species richness and cover of annual and graminoid growth forms and higher overall plant species diversity when compared to unburnt Swartland Renosterveld. Palaeoecological research by Forbes et al. (2018) revealed that despite increasing fire frequencies in the West Coast Renosterveld at Elandsberg Nature Reserve, the vegetation remained surprisingly resilient to changes in fire frequency as well as increasing fire frequency. Despite far higher grazing pressure and increasing fire frequency, the renosterveld at Elandsberg still remains highly biodiverse with high ecological and landscape heterogeneity today (Forbes et al. 2018). This small but significant body of research clearly demonstrates that fire plays an important role in renosterveld as an ecological driver and more research is needed to improve understanding in this field (Curtis, 2013a, Cousins et al. 2017).

1.10. Project Aim

Given the importance of land use activities and their influence on species richness, diversity, composition and vegetation cover in Overberg Renosterveld, the main aim of this project was to investigate the impacts of grazing by cattle and sheep on the Western, Central and Eastern Rûens Shale Renosterveld and to assess rangeland restoration potential in the Overberg region.

1.11. Specific Objectives and Hypotheses

The project has the following specific objectives:

- To assess the impact of livestock grazing by sheep and cattle on plant species richness and diversity and growth form diversity over time in Overberg Renosterveld.
- To assess the impact of livestock grazing by cattle and sheep on the renosterveld soil seed bank.

- To examine the impact of livestock grazing by cattle and sheep over time on total vegetation cover and vegetation cover by growth form in comparison with sites protected from grazing. Following collection of a baseline dataset, four years of follow up data were collected to address this objective.

This project has the following hypotheses:

- It was hypothesised that livestock grazing by cattle would have less effect on species richness and diversity and growth form diversity than sheep grazing and that both cattle and sheep grazing would lead to a reduction in species richness and diversity in comparison to renosterveld sites with a treatment of no grazing.
- It was also hypothesised that cattle and sheep grazing would reduce species richness, species diversity and growth form diversity in the soil seed bank in comparison with sites with a treatment of no grazing.

1.12. Thesis Structure

Chapter 1: Introduction

The first chapter of the thesis presents the background and rationale, introduce the aims and objectives and present an outline of the study area. The latter encompasses the location and regional context, topography and hydrology, geology and soils, landscape history, regional biodiversity, types of renosterveld vegetation, regional climate as well as a synthesis of the literature on renosterveld and fire.

Chapter 2: Literature Review

This chapter presents a synthesis and detailed literature review on rangeland and restoration ecology, including agricultural rangelands as a global land use, importance of conservation in rangelands and rangelands and livestock grazing in Mediterranean type ecosystems. It also includes a synthesis on the debates around whether renosterveld is a shrubby grassland or grassy shrubland, including the influence of livestock grazing in driving these putative changes. The conservation status and threats facing renosterveld are also discussed, as well as current renosterveld management practices. This synthesis concludes with a look at renosterveld restoration, including restoration potential and the role of the soil seed bank, as well as the role of fencing in passive restoration.

Chapter 3: Effects of livestock grazing on plant species richness and diversity in Overberg Renosterveld using Modified Whittaker plots.

An assessment was undertaken examining the impacts of grazing by cattle and grazing by sheep in comparison with sites with a treatment of no grazing on plant species richness and diversity and growth form diversity. Vegetation data was collected across thirty sites using Modified Whittaker plots, a series of nested subplots within a larger plot to examine species diversity at different spatial scales. This data was used to generate species area curves, as well as variation in richness, diversity and cover of vegetation and different growth forms across sites with a treatment of no grazing, grazed by cattle and grazed by sheep.

Chapter 4: Ecosystem resilience and restoration potential: Investigating the effects of grazing on diversity and composition of the soil seed bank in Overberg Renosterveld

The effects of grazing by cattle and grazing by sheep on the soil seed bank in Overberg Renosterveld in comparison to sites with a treatment of no grazing was also investigated. Data was collected on presence of viable seeds in the soil using germination trials. A total of 300 soil samples were collected, sieved and germinated in the glasshouse from the same thirty sites used for Modified Whittaker plot vegetation surveys from Chapter 3, which had treatments of no grazing, grazing by cattle and grazing by sheep.

Chapter 5: The effect of protection and grazing by cattle and sheep on plant species richness and growth form diversity and cover in Overberg Renosterveld vegetation.

This component of the project examined the impact of rest from grazing (passive restoration) in comparison to treatments of grazing by cattle and grazing by sheep through use of paired fenced exclosures and unfenced survey plots. This task was undertaken by monitoring changes in plant species richness, plant species diversity and growth form diversity annually for four years following collection of an initial baseline dataset.

Chapter 6: Synthesis and Research Conclusions

The final chapter presents the synthesis and research conclusions, reflects upon the project aims and objectives and details directions for future research. Implications of research findings for restoration and conservation of Overberg Renosterveld will also be discussed.

Chapter 2: Literature Review

2.1. Agricultural rangelands as a global land use

Agricultural rangelands today occupy as much as one third of the earth's ice-free land (Herrick et al. 2012). The term rangeland is internationally used, but there are a wide variety of different definitions in circulation (Grice and Hodgekinson, 2002). One of the most widely cited definitions was written by Pratt et al. (1966), which defines rangelands as "land carrying natural or semi natural vegetation which provides a habitat suitable for herds of wild or domestic ungulates". Meanwhile, the Society of Rangeland Management defines rangelands as "a type of land on which the natural vegetation is dominated by grasses, forbs and shrubs and the land is managed as a natural ecosystem" (Herrick et al. 2012). Land areas that are now known as rangelands have a very long history of human use, dating back more than 11 000 years (Grice and Hodgekinson, 2002). Ecosystems that typically constitute rangelands such as grassland, savanna and woodland have played a significant part in the shaping of human development (Hruska et al. 2017). The earliest pastoral systems were subsistence based, with commercial pastoralism developing later with a focus on goods and services beyond the rangelands themselves (Grice and Hodgekinson, 2002). The use of the term 'range' dates to the 1400s in England and was used in reference to extensive areas of grassland or woodland (Grice and Hodgekinson, 2002). The term was used later by early colonists to the USA to refer to areas of natural vegetation that were used for livestock grazing (Grice and Hodgekinson, 2002).

Today rangelands as a global land use tend to be limited to areas of the world that are climatically unsuitable for other forms of agriculture (Menke and Bradford, 1992). They are predominantly limited to agricultural lands that are too cold, too wet, too dry, steep, or infertile for cultivation of crops (Herrick et al. 2012). With the growth in population, demand for meat production has increased (Espinosa et al. 2020). This in turn has increased the demand for marginal croplands to be brought into grain production to grow feed for growing livestock numbers (Hasanpori et al. 2020). Many of the world's most historically productive rangelands have now been converted into agricultural croplands (Richard et al. 2014). Combine harvesters have now replaced ruminants in many parts of North America and rangelands, which have been used for livestock have given way to fields of wheat in the iconic Pampas region of Argentina (Herrick et al. 2012). In South Africa, areas where elephants and lion once roamed have been converted to intensive commercial cultivation of crops such as wheat, barley, and canola (Skead, 2009; Cowan, 2019; Moncrieff, 2021). Furthermore, changing climate is making livestock farming in many dryland areas more marginal than ever before (Dougill et al. 2010; Holechek et al. 2020). One might think in the face of accelerating land cover and global change that rangeland science is becoming less relevant in today's world (Godde et al. 2020). In fact, it plays a crucial role (Herrick et al.

2012; Godde et al. 2020). Whereas historically rangeland science has had a stronger focus on maximising agricultural productivity, today even more so there is a need to ensure long term sustainability of pastoral systems in agricultural rangelands (Scogings et al. 1999; Reid et al. 2021). A key growth sector for rangeland science and management is development of sustainable adaptive management strategies for rangeland areas that have been degraded through long term unsustainable livestock farming (Wang et al. 2020). Rangeland science should play a role in better understanding and identifying areas where ecosystems are at risk of crossing irreversible thresholds, as well as better understanding levels of ecosystem resilience (Liao et al. 2020). This in turn should feed into developing sustainable grazing management practices that are in line with restoring degraded ecosystems and achieving conservation goals (Herrick et al. 2012; Virgilio et al. 2019).

2.2. Importance of conservation in rangelands

As the science of rangeland management has changed and evolved through the years, conservation has played an increasingly important role in the discipline (Fuhlendorf et al. 2012; Bestelmeyer et al. 2019). In the earlier days of rangeland science and management, during the late 19th and early 20th centuries, there was a far stronger utilitarian focus, with the sole aim of achieving sustainable livestock production predominantly outside the context of other ecosystem services (Freese et al. 2014). The key goals were effective livestock production through sustainable forage use, with the first and foremost aim of maintaining economic stability within the agricultural sector to the benefit of the many (Fuhlendorf et al. 2012). This economy and production driven thinking were later joined by the protectionist "fortress conservation" paradigm during the early 20th Century, with a primary focus on setting aside often large tracts of land to become protected areas such as national parks and wilderness areas where they were to remain distant from human influence (Montgomery et al. 2020). Human influence on these lands as part of this paradigm is seen as harmful to biodiversity conservation (Beymer-Farris and Bassett, 2012). In more recent years, the idea that rangelands are not only a natural resource for livestock production has gained increasing traction, leading to a more multi-disciplinary approach with biodiversity conservation now seen as a key ecosystem service that rangelands can provide (Huntsinger and Oviedo, 2014; Bestelmeyer et al. 2019). This has brought about the third paradigm to influence the development of the science of rangeland management, that of ecosystem management (Fuhlendorf et al. 2012). This paradigm is driven by the need to conserve rangelands as ecosystems by conserving processes such as fire and grazing (McGranahan and Kirkman, 2013). The ecosystem management paradigm can be originally attributed to ecologist Aldo Leopold, who developed it to counter a land management system that he believed to be exploitative and without science at its core (Leopold, 1949; Fuhlendorf et al. 2012). However, despite this, utilitarian principles remain at the centre of rangeland science, often to the detriment of best management practice in terms of achieving conservation goals in rangelands (Fuhlendorf et al. 2012). The focus has sometimes meant that livestock production and wildlife habitat are seen as competing rather than as being parts of the whole picture (Butt and Turner, 2012). This has meant that management has often focused on maintenance of palatable plant species and the minimisation of soil degradation (Fuhlendorf et al. 2012). Management goals to achieve this have focused on maintaining climax vegetation in the context of light to medium disturbance, without taking into consideration the roles of ecological drivers such as fire and grazing as key ecosystem processes rather than only as management tools (Fuhlendorf et al. 2012).

Population growth has meant that rangelands across the world have become increasingly transformed for the cultivation of crops to be used predominantly as animal feed in increasingly industrialised agricultural systems (Richard et al. 2014). This has meant that many of the world's rangelands have become subject to fragmentation which has emerged as a primary component of global change (Hobbs et al. 2008). This transformation and fragmentation of rangelands, particularly in areas of the world with high biodiversity, has meant that conserving ecosystems used as rangelands has become increasingly important (Western et al. 2020). Today one of the greatest threats to rangeland ecosystems is habitat transformation and loss, be it as a result of crop cultivation, urban development, alien plant invasions or woody plant encroachment (Fulhlendorf et al. 2012; Cameron et al. 2014). These land cover changes have made rangeland management increasingly complex as plant community structure, biodiversity and ecosystem function have changed significantly from historical ecosystem patterns and processes (Hobbs et al. 2008). Fragmentation of rangeland ecosystems does not only happen as a result of land cover change, it can also happen as a result of physical barriers in the landscape such as fences (Fuhlendorf et al. 2017). The presence of fences can be functionally significant, even if they do not result in an overall loss of habitat (Hobbs et al. 2008). At smaller grazing scales when the landscape has been divided into smaller units, there is a very tight coupling between animal and plant resources (Baker and Hoffman, 2006). As agricultural landscapes have become increasingly industrialised and more fragmented through land cover change and fencing, livestock grazing often continues through use of artificial pastures and additional supplemental feeding within these rangeland areas (Hobbs et al. 2008). This makes it possible to continue to farm livestock in situations where there would otherwise be insufficient natural resources available (Yan et al. 2011). In effect, the plant-animal system becomes decoupled, leaving the chances of degradation of any remaining natural rangeland habitat particularly high (Hobbs et al. 2008). Furthermore, in semiarid and arid ecosystems there is high climatic variability with more frequent droughts occurring in the face of changing climate (Szejner et al. 2020). This makes it increasingly difficult to determine appropriate livestock carrying capacity and further increases the likelihood of overgrazing taking place (McKeon et al. 2008). These challenges increase the necessity of effective conservation-focused rangeland management in these fragmented ecosystems, an important area for further research.

2.3. Rangelands and livestock grazing in mediterranean type ecosystems

Historically, the majority of research into rangelands has been concentrated in temperate grasslands, and to this day there are many differing opinions around the role of grazing as a driver in rangelands in mediterranean ecosystems worldwide (Sternberg et al. 2000). Rangelands in mediterranean ecosystems have in common high seasonality with regards to resource availability, significant inter-annual rainfall variation, a large component of annual plants in their respective floras as well as a long history of grazing and other forms of disturbance (Sternberg et al. 2000; Merdas et al. 2021). There is a long history of livestock grazing in mediterranean type ecosystems, and this is particularly the case in ecosystems in the Mediterranean region (Papanastasis, 2009). In fact, domestic livestock have grazed the rangelands of the Mediterranean region for over 5 000 years, in particular in western Asia, north Africa and the Middle East (Louhaichi et al. 2009). Owing to this long history of livestock grazing, ecosystems within the Mediterranean region are highly resilient to heavy grazing pressure (Sternberg et al. 2000). Grazing of domestic livestock is often perceived to be a driver of degradation in these ecosystems (Kosmas et al. 2015). This view was expressed as early as the fourth century by Plato who commented that the hills around Athens were "like the skeleton of an old man, all the fat and soft earth wasted away and only the bare framework of the land being left" (Perevolotski and Seligman, 1998). This perceived ecological degradation has been predominantly attributed to grazing of sheep and goats, with expanding demand for meat and milk leading to increases in herd sizes in many rural areas beyond the carrying capacity of the vegetation (Alhamad, 2006; Louhaichi et al. 2009).

However, the perception of livestock grazing as a driver of ecological degradation has led to the cessation of traditional livestock grazing in many protected areas across the Mediterranean region (Verdu et al. 2000; Tardella et al. 2020). Grazing in the ecosystems of the Mediterranean region has been shown to increase biodiversity through modifying competition between woody species and herbaceous plants, creating a mosaic of open and more densely vegetated areas (Perevolotski and Seligman, 1998; Perevolotski, 2005). A reduction in grazing load within these protected areas has led to a reduction in biodiversity through an increase in the cover of woody vegetation (Verdu et al. 2000; Bashan and Bar-Massada, 2017). These resultant changes have led to increasing recognition of the role of grazing in maintaining biodiversity in Mediterranean region ecosystems through the maintenance of habitat heterogeneity (Verdu et al. 2000; Silva et al. 2019). The long history of livestock grazing has in fact created highly resilient ecosystems that persist under intensive use (Perevolotski and Seligman, 1998). This role of grazing as a driver which maintains heterogeneity in the landscape is far from unique to only the Mediterranean region but is also present in other mediterranean type ecosystems elsewhere in the world (Perevolotski and Seligman, 1998; Gao and Carmel, 2020).

The role of grazing mammals, both wildlife and livestock grazing, is a topic that remains under researched within South Africa's Fynbos Biome, another of the world's mediterranean shrubland ecosystems (Radloff, 2008). Much of the region has low soil nutrient status, winter rainfall and unusually species diverse vegetation (Rebelo et al. 2006). Historically the vegetation of the Fynbos Biome supported a diverse assemblage of herbivores, but many of these have become locally extinct due to hunting following European colonisation in 1652 (Skead, 2009). However, despite extensive information being available to researchers from early written accounts, knowledge of the functional role of herbivorous mammals in the functional ecology of the vegetation of the Fynbos Biome is still limited (Radloff, 2008). Fynbos vegetation in the CFR has often been regarded as being of limited value to grazing mammals due to the relative absence of C_4 grasses in comparison with renosterveld vegetation (Radloff et al. 2014). Hendley (1983) suggested that the vegetation of the Fynbos Biome was more grassy in the past, with the widespread local extinction of grazing and browsing mammals leading to an increase in shrubby vegetation cover at the expense of more grassy open areas. In contrast, Rebelo (1992) suggested that large herbivores contributed little to the dynamics of fynbos vegetation.

There is also a long history of livestock grazing in the Fynbos Biome, with sheep and cattle being introduced by the Khoe more than 2 000 years ago (Radloff, 2008). Following European colonisation, livestock numbers increased and grazing intensified, with regular burning of fynbos vegetation to increase grass cover and forage yield within the vegetation (Joubert, 1991). However, the fynbos was never adapted to carry these grazing loads or the increased fire frequencies from burning to increase forage. As a consequence, these practices eventually led to extensive loss of overstorey vegetation and soil erosion (Adamson and Salter, 1950). This is further illustrated by the work of Le Roux (1988) who studied the effects of livestock grazing on fynbos vegetation in the Riviersonderend mountains. The conclusion of this research was that intensive livestock grazing and associated frequent burning would lead to a reduction in plant species diversity. Le Roux (1988) reached the conclusion that all livestock grazing in fynbos vegetation should be stopped due to its impact on the vegetation (Van Wilgen et al. 2016). In contrast, the renosterveld vegetation of the Fynbos Biome has a long history of grazing both by indigenous large mammals and by domestic livestock. This is discussed in more detail in section 2.5 of the literature review.

2.4. Links to theoretical literature framing the study

This thesis is framed by the following theoretical literature. Non-equilibrium theory is sometimes considered to be a new paradigm of contemporary rangeland ecology. It contrasts with the equilibrium theory, that underpinned rangeland ecology prior to the 1970s (Wessels et al. 2007). The equilibrium theory is built upon the idea that ecosystems are linear in nature, exhibiting a succession of changes over time (Briske et al. 2017). It was thought that ecosystems exhibited a series of negative feedback mechanisms, that maintained their stability (Wessels et al. 2007). If changes to the ecosystem driven by disturbances such as grazing or fire took place, it was believed that the vegetation would eventually return to the same pre-climax state prior to when the disturbance took place (Wessels et al. 2007). Non-equilibrium theory emerged as part of "the new ecology" in the 1990s (Gillson and Hoffman, 2007). Non-equilibrium theory is associated with the metaphor: "The flux of nature", acknowledging that ecosystems have a limited ability to remain in equilibrium following a certain level of disturbance and may have multiple different stable states (Briske et al. 2017). Furthermore, non-equilibrium theory poses the idea that vegetation will not exceed its carrying capacity because drought and other abiotic factors will limit the number of livestock that are able to survive there (Wessels et al. 2007). However, it has been highlighted that the mechanism through which drought determines livestock numbers is unclear and also that there is no fixed 'carrying capacity' due to variability in climatic conditions over time (Gillson and Hoffman, 2007). This variability will only increase in the face of a changing climate (Nhamo et al. 2019). It also does not take into account the presence of artificial watering points; the presence of artificial fodder crops adjacent to the veld and division of the landscape through use of fences (Hobbs et al. 2008). The latter plays a particularly key role in modifying the degree of impact of livestock on the vegetation in a rangeland context (Hobbs et al. 2008).

Resilience theory speaks to the ability of ecosystems to be dynamic while remaining persistent as self-organised systems (Briske et al. 2017). It recognises that ecosystems can exhibit significant variation around factors such as species composition but can still remain resilient (Briske et al. 2017). Resilience thinking also acknowledges the role of ecological thresholds contributing to the formation of alternate stable states (Briske et al. 2017). An ecological threshold is defined as the point at which there is an abrupt change in the state of an ecosystem or ecological process or a larger change has taken place in response to a small variation in an ecological process (Groffman et al. 2006). Human impacts on ecosystems can increase the likelihood of ecological communities being shifted into states that are difficult if not impossible to reverse, making the concept highly applicable to habitat management and ecological restoration (Suding and Hobbs, 2009). This is potentially highly likely in Overberg Renosterveld vegetation in response to livestock grazing, but further research around where ecological thresholds lie and what ecological drivers are influencing these changes are needed to better understand whether irreversible changes have taken place. However, empirically identifying the presence of a threshold that has been exceeded is more challenging, with most relying on inferring that there has been an ecological regime shift from time series of monitoring data rather than controlled experiments (Andersen et al. 2008). The concept of alternate stable states is also one that has significant application in the context of exploring the rangeland and restoration ecology of Overberg Renosterveld. When an ecological threshold has been exceeded and an ecosystem has become degraded, it may enter a persistent and resilient alternate stable state that requires a specific set of pathways to recover (Suding et al. 2004). This has application when investigating the restoration potential of renosterveld vegetation following grazing through exclusion of livestock.

2.5. What is renosterveld? Grassy shrubland or shrubby grassland?

Renosterveld is one of the world's most species diverse mediterranean type shrublands. It is also known as Cape transitional small-leaved shrubland (Cowling, 1984). It is both floristically and ecologically distinct from fynbos (Bergh et al. 2014). In contrast to fynbos vegetation, renosterveld is defined by a lack of dominants from the Proteaceae and Ericaceae families and having not more than 5-10 % cover of Restionaceae (Rebelo et al. 2006). As a result of extensive habitat transformation for agriculture and local extinction of large herbivores, there is limited knowledge of the ecology of 'pristine' renosterveld (Waller, 2013). We know little about renosterveld landscapes prior to colonial settlement in the mid 17th Century (Forbes et al. 2018). There is a long history of anthropogenic influence that adds further challenge to understanding historical landscape change in renosterveld vegetation (Forbes, 2014). There has been considerable debate in the literature around what renosterveld is, how it has changed over time and therefore what we should be managing towards in terms of best conservation practice for what little of this Critically Endangered and 'irreplaceable' vegetation remains (Von Hase et al. 2003; Newton and Knight, 2005; Newton et al, 2005; Newton and Knight, 2010).

Cowling (1986) proposed the hypothesis that South Coast Renosterveld was historically a grassland dominated by *Themeda triandra*, and that Asteraceous shrubs such as *Dicerothamnus rhinocerotis* (L.f.) (Asteraceae) Less. and *Metalasia* spp. (Asteraceae) then started to dominate the landscape following overgrazing by domestic livestock. This triggered a debate that continues to this day about whether renosterveld is a 'shrubby grassland' or 'grassy shrubland' (Curtis and Bond, 2013; Radloff et al. 2014; Forbes et al. 2018). Boucher (1983) refers to coastal renosterveld as a product of recent, regular burning and overgrazing, a product of a disturbed environment similar to coastal sagebrush in California. Newton and Knight (2004) further built on this theory, asserting that South Coast Renosterveld likely evolved as a grassland over the last half million years. This was until the last 200 years when the loss of indigenous grazing and browsing herbivores and the intensification of domestic livestock grazing following European colonisation led to transformation of the vegetation structure from a 'shrubby grassland' to a 'grassy shrubland' (Newton and Knight, 2004). Severe and continuous overgrazing of newly burnt renosterveld and the loss of large grazing mammals was blamed for the putative shift from grassland to shrubland, with this hypothesis supported by various authors in the historical literature (Krug et al. 2004; Rebelo et al. 2006). In 1772, Thunberg wrote about Buffeljagsrivier, east of Swellendam in the eastern Overberg and indicated that "The plains now begin to abound more in grass and looked something like meadows…In the same proportion the herds of cattle became larger and occurred more frequently" (Skead, 2009: pp. 116). In 1786 Sparrman wrote in reference to the area between Swellendam and the

Gouritz River: "...without a shadow of a doubt as such places as before abounded in grass…. are now falling off considerably…The rhinoceros bush, a dry shrub which otherwise used to thrive on bare tracts of land, now began to encroach more on such places as have been thoroughly cleared and cultivated. Asked why this is so, the colonists said they were to blame; they did not know how to look after the soil. Whereas the Hottentots kept their cattle on the move, the colonists grazed them on the same ground year after year." (Skead, 2009: pp. 119). By 1822, Burchell wrote in reference to the area between Caledon and Genadendal: "The face of the country was open, and its surface varied with smooth hills covered almost exclusively with a neat pale bushy shrub of the height of 3 or 4 feet called renosterbosch and said to have formerly been the food of the huge rhinoceros till those animals fled before the colonists as they gradually advanced over the country where the shrub grows" (Skead, 2009: pp. 125). The question of whether renosterveld was historically more grassy or shrubby at the landscape scale may never be answered conclusively (Radloff et al. 2014).

However, a few researchers have sought answers to these questions through use of palaeoecological proxy data. Curtis (2013a) used stable carbon isotope analysis to test the hypothesis of whether Overberg Renosterveld historically supported a higher C_4 grass component. Results from this study revealed that C_3 grasses and shrubs were more prevalent in the past and that Overberg Renosterveld likely always comprised a mix of C_3 and C_4 grasses and shrubs as it does today. Very few of the results from this research suggest the presence of any pure C_4 grasslands in the Overberg at any time in the past. The results also confirmed increasing presence of C_4 grasses across a west to east gradient across the Overberg, in line with current renosterveld vegetation communities today (Curtis, 2013a; Curtis and Bond, 2013). These renosterveld vegetation communities have always been highly heterogenous and more research is needed to better understand the ecological drivers that lead to increased grasses or shrubs in renosterveld (Radloff et al. 2014). Building on these findings, Forbes et al. (2018) used a variety of different palaeoecological proxies including macrocharcoal, pollen and coprophilous fungal spores to examine vegetation change over time from a core taken in West Coast Renosterveld at Elandsberg Private Nature Reserve. Palaeoecological evidence showed remarkable vegetation stability at Elandsberg from the 8th Century to the mid 20th Century. In the late 1800s there was a subtle increase in Asteraceae/*Dicerothamnus*/*Stoebe* type pollen, followed by a greater increase from the 1950s onwards. Statistical analysis revealed this to have coincided with an increase in the presence of macrocharcoal and coprophilous fungal spores, suggesting an associated increase in fire and herbivory in the area. This is likely to represent local land use change including increased grazing intensity and burning, increased agricultural intensity at the site through livestock farming, crop cultivation and associated burning taking place at the site from the mid 20th Century (Forbes et al. 2018). Another important finding from this research is that historical renosterveld at the site was historically not as grassy as has been suggested by other authors (Newton and Knight, 2004) and the historical literature (Skead, 2009). The findings from this study support recent research such as Curtis (2013a), who reported a historical compositional state of a combination of C_3 and C_4 grasses and shrubs rather than a pure C_4 grassland (Forbes et al. 2014).

2.6. Conservation status and threats to renosterveld

The lowlands of the CFR contain some of the most transformed habitats in South Africa as a result of agriculture, urbanisation and the spread of alien invasive plants (Von Hase et al. 2003). Renosterveld vegetation is no exception, being one of the most critically endangered habitats in the CFR (Topp and Loos, 2019). All four renosterveld types in the Overberg are listed as Critically Endangered. Despite renosterveld being used for livestock grazing for over 300 years, we know little about its ecology (Rebelo et al. 2006). One of the most significant threats to this vegetation type is the illegal transformation of habitat for agriculture, which is in contravention of the Conservation of Agricultural Resources Act (Landcare South Africa. National Department of Agriculture, 2002; Curtis et al. 2013). Across the CFR more than 90% of lowland renosterveld vegetation has been ploughed up for agricultural production, representing an estimated 160 000 ha lost to the plough between 1918 and 1990 (Kemper et al. 1999; Newton and Knight, 2005; Newton et al. 2005; Newton and Knight, 2010). The conservation importance of renosterveld and other lowland vegetation types of the Fynbos Biome has long been acknowledged, but detailed GIS mapping and planning has translated into little conservation action (Curtis, 2013a; Topp and Loos, 2019).

The first map to identify priority ecosystems for conservation in the CFR was undertaken by Jarmin (1986). Metadata compiled with this map took note of the presence of some of the largest and best examples of South Coast Renosterveld being in the Bredasdorp-Swellendam area, as well as their high recorded numbers of threatened species present. Jarmin (1986) also highlighted illegal ploughing for agriculture and intensive livestock grazing as major threats to the ecosystem (Landcare South Africa. National Department of Agriculture, 2002). In 2003, the Cape Lowlands Renosterveld Conservation Project compiled a technical report on conservation planning for renosterveld with associated GIS tools (Von Hase et al. 2003). This came with ambitious five- and twenty-year targets with a goal of effective protection by 2020 in line with the Aichi Biodiversity Targets, but to date this has not yet been achieved. Renosterveld remnants continue to be at risk and conservation targets are not being met (Curtis, 2013a; Topp and Loos, 2019). Land cover change mapping by Moncrieff (2021) using satellite imagery revealed that the extent of surviving renosterveld vegetation in the Overberg decreased by 0.72% between 2016 and 2020. Furthermore, these landcover change events were often found to precede the planting and harvesting seasons of rainfed annual grain crops (Moncrieff, 2021).

One of the most significant barriers to conservation for lowland renosterveld is the fact that the majority of land is privately owned (Von Hase et al. 2003). Often the only fragments of renosterveld that have survived are those that are on land that is too steep or too rocky to plough and are therefore not representative of the biological communities that used to occur there (Rebelo et al. 2006). The fragmentation of renosterveld has significant ramifications for its biodiversity in ways we are only just starting to understand (Ruwanza, 2017). Around 18 000 fragments of lowland renosterveld remain across the CFR, with the majority of fragments smaller than 1 ha in size (Von Hase et al. 2003). Larger tracts of renosterveld are often used as grazing camps, with smaller renosterveld fragments having different disturbance regimes (Kemper et al. 1999). Smaller patches of renosterveld are often more heavily grazed, more frequently burnt and subjected to contamination from herbicides and pesticide drift from neighbouring cultivated fields (Kemper et al. 1999). The latter often happens when renosterveld patches are downslope from agricultural lands with river courses and seepages at risk of becoming eutrophic (Rebelo et al. 2006).

We are only just starting to understand the effects of renosterveld fragmentation on pollinators with just a handful of community and landscape scale studies having been undertaken (Cowan and Anderson, 2019). Research by Donaldson et al. (2002) revealed a decrease in pollinator abundance in Overberg Renosterveld with increasing distance from larger patches of veld. Perennial plants growing on smaller renosterveld fragments were found to be more vulnerable to pollination failure than on those patches that were larger or well-connected in the landscape (Donaldson et al. 2002). In addition, research by Cowan and Anderson (2019) revealed the presence of highly specialised pollinator networks in Overberg Renosterveld vegetation in comparison to others globally. These pollinator network structures were found to differ on previously ploughed old lands in comparison with pristine areas of renosterveld (Cowan and Anderson, 2019). Like other ecosystems within the CFR, renosterveld is also under threat as a result of changing climate with invertebrates and their associated ecosystem functions being identified as being at risk (Topp and Loos, 2019). A key example are termites, which play an important role as ecosystem engineers and in renosterveld species diversity. It is highly likely that they could be impacted by changing rainfall and vegetation patterns. (Picker et al. 2007). Climate change may also impact on fire cycles and drought, both of which are key ecological drivers in renosterveld vegetation (Topp and Loos, 2019).

However, despite the plethora of threats facing lowland renosterveld vegetation across the CFR, some progress has been made in terms of increasing the area under conservation. The Biodiversity Stewardship Programme continues to be an effective tool in securing biodiversity on private land outside formal protected areas (Waller, 2013). Since its inception in 2012, the Overberg Renosterveld Conservation Trust (ORCT) has also made significant strides in bringing renosterveld on private land under conservation. In 2013 WWF-SA purchased the world's largest area of lowland renosterveld on the farm Haarwegskloof in the Overberg's Eastern Rûens which is now managed for conservation by the ORCT (Curtis-Scott et al. 2020). Learning from other mediterranean type ecosystems such as in California, conservation easements are used to conserve indigenous species and their habitats in private working landscapes with many notable successes (Topp and Loos, 2019).

2.7. Renosterveld management: Current practice and views

In comparison to other vegetation types of South Africa's Fynbos Biome, renosterveld vegetation has received little attention from researchers and only a handful of applied research studies have taken place to better understand how these vegetation types should be managed (Radloff et al. 2014). Effective management to best conserve what little remains of these Critically Endangered vegetation types is crucial (Topp and Loos, 2019). Also, because the majority of surviving renosterveld vegetation is located on private land, it is vital that conservation practitioners work with landowners as custodians (Curtis and Bond, 2013). In line with meeting this urgent need, the Overberg Renosterveld Conservation Trust published the booklet the 'Overberg Rûens Renosterveld' to be distributed to landowners and other key stakeholders. This booklet, written in an accessible format, provides information on the history, wildlife and flora of the region and outlines key management guidelines for conserving Overberg Renosterveld. Management guidelines are also accessible to landowners in a free downloadable document made available on the Overberg Renosterveld Conservation Trust's (ORCT) website at https://overbergrenosterveld.org.za/management/.

Guidelines that are currently issued to landowners are as follows. Firstly, regarding fire management, ORCT advice is that renosterveld vegetation should be burnt every 7-12 years with ecological burns recommended to take place during autumn. Landowners are advised never to graze their veld immediately after a fire, but to rather rest the veld for 18-24 months after burning as grazing livestock are likely to negatively impact upon the growth of resprouting vegetation as well as newly germinated seedlings. Furthermore, landowners are advised to avoid feeding livestock with supplementary fodder in their renosterveld where possible, as this can lead to overgrazing, excessive trampling and to the growth of alien weeds in the renosterveld. Grazing is advised to rather take place in renosterveld during the summer months and for renosterveld to be considered a reserve food source. It is also advised to continually monitor the veld during periods of livestock grazing for signs of overgrazing and immediately removal of livestock if it is observed.

These guidelines are supported by government legislation as part of the Conservation of Agricultural Resources Act (CARA, Act 43 of 1983), which is enforceable by the National Department of Agriculture. Regulation 2 states that without authority from written permission, no land user may cultivate any virgin soil, where virgin soil is defined as any land that has not been cultivated during the preceding 10 years. Regulation 3 states that no land with a slope steeper than 12% may be cultivated without authority from written permission. Regulation 9 states that every land user must protect the veld on their farm unit effectively against deterioration and destruction. Regulations 10 and 11 state that the grazing capacity of the veld may not be exceeded unless the veld is sufficiently protected against deterioration and destruction. Finally, Regulation 14 states that a land user must effectively restore or reclaim any disturbed or denuded land on their farm unit (Landcare South Africa and National Department of Agriculture, 2002). Despite these detailed, forward-looking and conservation focused guidelines and regulations, illegal ploughing, ploughing of watercourses and continued heavy livestock grazing continue to be significant threats facing renosterveld vegetation (Curtis, 2013a). More applied research is needed around renosterveld management to better inform and refine management guidelines for landowners, conservation practitioners and other stakeholders.

2.8. Restoration in renosterveld

Despite the renosterveld vegetation of the Cape lowlands being used for livestock grazing for more than 300 years, we know very little about its ecology (Rebelo et al. 2006). Insufficient renosterveld vegetation survives in the Cape lowlands to meet international conservation targets and all of that which remains is categorised as 'Critically Endangered and 'irreplaceable' (Von Hase et al. 2003). It is therefore crucial that the amount of renosterveld vegetation under formal protection is increased. This can be achieved through schemes to conserve threatened habitats on private land such as stewardship and conservation easements, such as the kind of work being undertaken by the Overberg Renosterveld Conservation Trust (Topp and Loos, 2019). Ecological restoration has also been highlighted by several authors as a key priority for lowland renosterveld vegetation (Rebelo et al. 2006; Waller, 2013; Heelemann, 2010). This is vital given the extensive transformation that has taken place and the remaining extant remnants being insufficient to meet conservation targets (Rebelo et al. 2006; Waller, 2013).

Ecological restoration is defined as the process of assisting the recovery of degraded, damaged or transformed ecosystems (Society for Ecological Restoration, 2004). Research into restoration in the Fynbos Biome has until now been focussed predominantly on fynbos (Heelemann, 2010), with restoration in renosterveld vegetation currently limited to just a handful of studies (Waller, 2013; Topp and Loos, 2019). Most of this research has also been undertaken in West Coast Renosterveld with a focus on restoring old lands where renosterveld vegetation has been previously ploughed for agriculture and then no longer cultivated for varying lengths of time (Midoko-Iponga, 2004; Shiponeni et al. 2005). These areas of old lands have been identified as being crucial for restoration to improve the conservation status of renosterveld vegetation by increasing the size of existing fragments and creating corridors and links between fragments (Midoko-Iponga et al. 2005; Krug, 2004). However, research by Heelemann (2010) revealed that old lands have very low restoration potential, often having significantly different species composition and altered ecosystem processes in comparison with uncultivated areas (Hobbs et al. 2006). Heelemann (2010) undertook research in West Coast Renosterveld at Tygerberg Nature Reserve in the greater Cape Town area, comparing 'pristine' renosterveld with vegetation on old lands with vegetation under pine plantations. These old lands investigated in this study were found to have highly depleted seed banks, high levels of nutrient enrichment and extensive invasion by alien grasses. Heelemann's (2010) conclusion was that old lands had very low restoration potential and little likelihood of reverting to their historical landscape state over time (Heelemann, 2010).

Midoko-Iponga (2004) undertook a field experiment at Elandsberg Private Nature Reserve to examine the effects of herbivory and plant establishment on restoration of West Coast Renosterveld vegetation on old lands. Five different shrub species were planted in blocks with different treatments with grass competition, herbivory or both being manipulated. Survival, growth and canopy cover were measured monthly over 14 months and results between different treatments were compared. Seedlings in blocks with grass competition were found to have higher mortality and reduced growth in comparison to those without, with herbicide improving shrub species richness in the study plots where it was used as a treatment. No significant interactions were found between competition and herbivory (Midoko-Iponga, 2004; Midoko-Iponga et al. 2005).

Seeds of many renosterveld species are produced and dispersed into agricultural land but most often do not establish with both grazing and competition from agricultural grasses inhibiting establishment. Often unpalatable Asteraceous shrubs (*Dicerothamnus*, *Relhania* and *Oedera*) establish on old lands in dense, monospecific stands with a notable absence of geophytes and grasses (Rebelo et al. 2006). One of the most successful studies to date was undertaken by Waller (2013) who examined restoration of old lands in Peninsula Shale Renosterveld on the lower slopes of Devil's Peak in Cape Town. In line with findings from other studies undertaken in this field (e.g., Midoko-Iponga, 2004), alien grass competition was found to be a significant barrier to seedling establishment. However, there was considerably improved recruitment when seeding was paired with additional treatments such as tilling or herbicide application. The majority of harvested species showed medium to high levels of germinability and occurred in the middle or upper key restoration species index range. These results are promising in informing future landscape scale restoration strategies for Peninsula Shale Renosterveld (Waller, 2013).

In contrast to the attention received by restoration in old lands in West Coast Renosterveld, there has been no research undertaken thus far into the restoration of renosterveld degraded by mismanagement such as overgrazing. Even as early as the 1980s this was identified as a significant threat to South Coast Renosterveld (Jarmin, 1986). More research is needed to better understand community recovery in response to restoration interventions to ensure better results in terms of restoration outcomes (Waller, 2013). Given the low restoration potential of old lands highlighted by previous research, coupled with multiple failed restoration experiments, a shift in restoration strategy and priority is needed (Heelemann, 2010). There needs to be a focus on restoring and therefore improving the conservation value of what renosterveld still remains but may have been degraded by mismanagement before attempting to bring back what has been lost. This research therefore seeks to use passive restoration using fenced camps to examine the degree of recovery brought about through the exclusion of domestic livestock. In addition, restoration potential will also be investigated through the study of the soil seed bank composition, richness and diversity in response to grazing by sheep and cattle.

2.9. Investigating restoration potential: The role of soil seed banks

Soil seed banks have a key role in restoration of degraded habitats and have been the subject of a considerable number of studies in recent years (De Villers et al. 2003). The soil seed bank is defined as all seeds that survive within an ecosystem and which occur both in the soil and in the upper litter layer (Wang et al. 2013). The term 'seed bank' was first used by Harper in 1977 where the assemblage of seeds from a suite of different species was compared to a current account in a bank (Moore, 1980). The soil seed bank acts as the historical 'memory' of vegetation species composition and diversity from the site historically as well as a key determinant of vegetation composition at the site in the future (Fisher et al. 2009). The soil seed bank is an important contributor to the restoration potential of any degraded habitat, particularly when indigenous species are well represented with alien invasive plant species uncommon or absent. In systems that have become degraded, leading to loss of key species from the standing vegetation, the soil seed bank can be used to drive regeneration of vegetation at the site as part of achieving conservation goals (Haussmann et al. 2019). However, there are many studies present in the literature from various ecosystems globally that have concluded that soil seed banks contain a paucity of desired restoration target species (Abella et al. 2020). In many cases ecosystem degradation is driven beyond a threshold of potential recovery with ecosystem collapse imminent. Even if the key drivers of ecosystem degradation are arrested, sometimes the damage cannot be reversed (Tessema et al. 2012).

Composition of the soil seed bank can vary in line with ecological gradients as well as land use, recent management activities, vegetation density, soil drainage and soil texture. However, soil seed bank composition can be broadly divided into five different categories as follows: 1. Depauperate seed banks with little restoration potential. 2. Seed banks that contain predominantly ruderal species, comprising a selection of more generalist species that can colonise a variety of different sites. 3. A seed bank comprising primarily alien species. 4. Predominantly restoration target species, comprising a variety of species from mature habitats. 5. A balanced mix of 2 and 4. (Abella et al. 2020). If there is a significant number of ruderal or alien species in the soil seed bank, this means that the soil seed bank can act as a barrier to effective restoration (Haussmann et al. 2019).

However, many arid, semiarid and Mediterranean type dryland ecosystems have been shown to have large seed banks (Anderson et al. 2012). This is thought to be due to slower seed decomposition due to increased aridity as well as fire activity to stimulate seed germination (Anderson et al. 2012). Although this is likely to feed into increased restoration potential in degraded habitats within these ecosystems, however, this is a subject that has been neglected. The majority of seed bank research has been concentrated in grasslands and arable fields, with wetlands, woodlands, heathlands and dune ecosystems receiving lesser attention (De Villers et al. 2003). It is possible that the soil seed bank could play an important role in restoration of habitats such as Overberg Renosterveld where it is likely that heavy grazing has contributed to the loss of key palatable species from the standing vegetation (Chimphango et al. 2020). However, more research is needed to better understand the mechanisms involved (Tessema et al. 2012). Vegetation studies are often focused only on the standing vegetation, ignoring the potentially significant role the seed bank can play in restoring ecosystems degraded by continuous heavy grazing (De Villers et al. 2003). To date there have been no soil seed bank studies previously undertaken in Overberg Renosterveld, although some research into the soil seed bank has been done in similar vegetation such as Swartland Renosterveld (Cousins et al. 2017). If seed banks are found to store target species, they can contribute to cost-effective restoration efforts without the need for expensive seeding or planting (Shiferaw et al. 2018). Use of the emergence method has shown potential for assaying seed banks as part of further empirical research in this field (Abella et al. 2020).

2.10. The role of fencing in passive restoration

The rolling landscapes of South Africa's Overberg were once wide open, with large herds of game roaming the region historically (Skead, 2009). Fences now divide this landscape, meaning that grazing by domestic livestock now places increased pressure on the surviving renosterveld fragments (Hobbs et al. 2008). The high cost of fencing means that landowners often do not have the resources available to fence individual patches of renosterveld. As a result, such patches are often subjected to the same management interventions as the surrounding agricultural lands (Curtis, 2013a). However, there is a long history of research which explores the effects of exclusion of grazing livestock and resting of continuously grazed vegetation on vegetation recovery. The majority of such research has taken place in North America (Stohlgren et al. 1999). Some of these studies have shown the benefits of collecting long-term datasets to monitor changes to the vegetation over longer time scales. For example, the work of Courtois et al. (2004) on the Nevada plots built in 1937, monitored changes in the vegetation after 65 years of exclusion from grazing. However, responses to livestock grazing from studies such as these vary by vegetation type and landscape history and so it is important not to make broad generalisations across different vegetation types (Curtis, 2013a). This is why more research such as this is needed to better understand grazing management and the effects of livestock exclusion on ecosystem recovery in ecosystems such as South Africa's mediterranean type shrublands, particularly so in Overberg Renosterveld.

Chapter 3: Effects of livestock grazing on plant species richness and diversity in Overberg Renosterveld using Modified Whittaker Plots

3.1. Introduction

The renosterveld of South Africa's Overberg region is one of the most biodiverse mediterranean type shrublands in the world (Curtis, 2013a). Mucina and Rutherford (2006) identified a total of 119 different vegetation types within South Africa's Fynbos Biome, of which 29 were renosterveld (Curtis et al. 2013). The grey-green colour of the main dominant shrub 'Renosterbos' (*Dicerothamnus rhinocerotis*) gives the outward impression that there is little species turnover or alpha diversity (Paterson-Jones, 1998; Curtis, 2013a). However, appearances can be deceptive and renosterveld contains one third of the CFR's plant diversity (Rebelo et al. 2006; Cousins et al. 2017). It is thought that this diversity is predominantly driven by pollinator specificity (Proches et al. 2006). While renosterveld occurs most commonly on shale derived soils, it can also occur on granite, dolerite, alluvium, silcrete or limestone (Rebelo et al, 2006; Bergh et al. 2014; Curtis et al. 2013). The main land use across the Overberg region is growing cash crops such as wheat, barley, oats and canola. These are grown on a crop rotation with introduced pasture crops of lucerne that are used for grazing sheep and cattle (Kemper et al. 1999; Curtis, 2013a). The biggest ongoing threats to the remaining renosterveld vegetation patches is illegal ploughing, predominantly driven by the ever increasing size of farm machinery and overgrazing (Curtis, 2013a; Moncrieff, 2021). Conditions for farming have become ever more marginal in the Overberg, further driving farmers to illegally plough virgin renosterveld to increase their areas of productive land to increase profit and for farms to remain economically viable (Kemper et al. 1999; Curtis, 2013a).

The patches of renosterveld that survive are mainly small fragments in areas of agricultural land that are too steep or rocky to plough (Groenewald, 2014; Cousins, 2017). As a result, they are subjected to management interventions applied to the cultivated areas such as grazing (Kemper et al. 1999; Curtis, 2013a). As part of the crop rotation, lucerne pastures are planted for three to five years at a time, at which point any natural, unploughed vegetation on the farm is subjected to similar grazing intensities to the surrounding pasture crop (Curtis, 2013a; Kemper et al, 1999). This can lead to renosterveld fragments being overgrazed (Curtis, 2013a) or colonised by alien grasses and other weeds with enhanced growth due to fertiliser runoff (Milton, 2007). Unmanaged heavy grazing has been become a significant threat to biodiversity in many different ecosystems (Anderson and Hoffman, 2007; Harrington and Kathol, 2008; Rutherford and Powrie, 2010). It can cause alteration of vegetation cover, altered nutrient cycling, soil compaction, loss of biological soil crust cover and soil erosion (Anderson and Hoffman, 2007; Harrington and Kathol, 2008; Rutherford and Powrie, 2010). Trampling and nutrient changes in the soil because of livestock dung and urine may also promote invasion of the ecosystem by undesirable exotic ruderals and risk of loss of species sensitive to these changes (Perevolotski and Seligman, 1998). Overgrazing is defined as excessive use of forage from the vegetation to the point that the biotic and abiotic components of the system are so changed that they cannot recover within ecological time (years or decades) (Perevolotski and Seligman, 1998). However, light to medium grazing, when used as an effective management tool, can be of significant benefit to the system (Curtis, 2013a). Grazing can also modify competition between plant species and less aggressive species that require greater access to light, such as small annuals and geophytes, may thrive when the environment receives sufficient disturbance (Perevolotski and Seligman, 1998; Koerner et al. 2018). Overberg Renosterveld has a long history of grazing (Krug et al. 2004; Bergh et al. 2014). Despite its long grazing history, we know little about the influence of herbivory on species composition and diversity in Overberg Renosterveld. Fire and herbivory are considered to be two of the main drivers of renosterveld structure and diversity, but their relative importance remains unclear (Kraaij and Van Wilgen, 2014; Cousins et al. 2017). Aside from the work of Curtis (2013a), where findings showed subtle impacts, likely due to the limited number of plots sampled, there is no quantitative data available on the impacts of grazing in Overberg Renosterveld, and little knowledge about how grazing should be used as a management tool to best achieve conservation goals (Milton, 2007).

To add to the challenge of developing conservation appropriate management practices, we also have little concrete evidence beyond anecdotal historical descriptions as to how renosterveld vegetation has changed over the last 2000 years in response to livestock grazing (Rebelo, 1995; Rebelo et al. 2006). There is a widely held perception both here and in many other parts of the world that a long history of livestock grazing has caused a decline in vegetation from a perceived pristine pre-European state (Bradd-Witt et al. 2000). It is commonly implied in the rangeland literature that pre-European systems were more diverse, resilient and productive, despite the lack of data to quantify the extent, timing and cause of change (Bradd-Witt et al. 2000). It is therefore vital when planning best practice management interventions to have an effective ecological baseline dataset from which to work (Forbes et al. 2018). It is equally critical to acknowledge that with a long timeline of anthropogenic influence on the landscape, planning management interventions are contextualised within an understanding of historic landscape change (Forbes et al. 2018). Otherwise, it is easy to fall into the trap of 'shifting baselines syndrome' whereby each generation makes decisions based on their contemporary observed experience and data with no consideration of temporal change (Forbes et al. 2018). It is likely that with a long-term history of grazing in Overberg Renosterveld that the vegetation has become more resilient, with modified competition favouring less palatable species which become more dominant in the landscape over time (Perevolotski and Seligman, 1998; Kemper et al. 1999). Impacts of livestock grazing on species richness and cover can also differ according to livestock type, with grazing by sheep and cattle having very different impacts on the veld (Esler et al. 2014). Cattle are bulk grazers that use around 70% grass in their diets along with 20% browse and 10% herbaceous plants (Esler et al. 2014). In renosterveld rangelands perennial grasses such as *Themeda triandra* (Rooigras) and *Ehrharta calycina* Sm. (Gha Grass) are commonly selected by both sheep and cattle (Samuels et al. 2016). Often medium height grass is grazed down to form dense lawns (Esler et al. 2014). Sheep tend to have a more varied diet, although this differs according to breed. They generally consume a diet of around 70% grasses, 20% dwarf shrubs and 10% annual and perennial forbs (Esler et al. 2014).

This study aims to better understand the impact that livestock grazing by sheep and cattle has on the ecological state of renosterveld vegetation in comparison to renosterveld with no grazing taking place. It further examines how livestock grazing can be used as a management tool by those who are custodians of Overberg Renosterveld in reaching conservation goals.

3.2. Objective and Hypothesis

The objective of the study was to determine the effects of livestock grazing by cattle and sheep on renosterveld plant species richness and diversity, growth form diversity and vegetation cover.

It was hypothesised that sheep grazing will result in a greater reduction in species richness, species diversity, growth form diversity and vegetation cover than grazing by cattle.

3.3. Materials and Methods

3.3.1. Study Sites

This research was undertaken in renosterveld vegetation of South Africa's Overberg region, with study sites located between the Van De Stel's Pass south of Villiersdorp and eastwards to the Suurbraak and Heidelberg areas (Figure 1.1).

3.3.2. Research Approach

Sampling was restricted to Overberg Renosterveld on shale geology, with Rûens Silcrete Renosterveld and quartz patches which commonly occur within Eastern Rûens Shale Renosterveld excluded (Curtis et al. 2013). Thirty sites where either no grazing had taken place (3 sites) or that had been grazed by cattle (11 sites) or sheep (16 sites). Sampling in old cultivated lands was avoided. This is because ploughing of vegetation for agriculture leads to extensive disturbance which irreparably alters species and growth form richness and diversity (Pineda et al. 1981; Anderson and Hoffman, 2007).

Sampling took place during Aug-Oct of 2014-2017. At each of the thirty sites, vegetation data was collected using Modified Whittaker Plots (MWPs), defined as a series of nested subplots of ten 1 m^2, two 10 m^2 and one 100 m^2 subplots nested within a 1000 m^2 plot

(Stohlgren et al. 1995). It is widely preferred over other sampling methods due to its ability to encompass far higher levels of species richness data in the least amount of time and for the generation of species area curves (Stohlgren et al. 1998, Leis et al. 2003; Petersen, 2018). In each of the 10 m^2 subplots the plant taxa that were present were identified to family, genus or species level where this was known and their percentage cover within the plot was recorded. For each subsequent subplot and the main 1000 m^2 plot, cumulative plant species were recorded, namely additional plant taxa that had not yet been recorded in any of the previous subplots.

Each species recorded was assigned to a specific growth form and functional type category that included succulent shrubs, non-succulent shrubs, perennial grasses, annual grasses, perennial forbs, annual forbs and geophytes (Anderson and Hoffman, 2011; Petersen, 2018). Identification of species encountered was done using field guides and species descriptions in Manning and Goldblatt (2012) as well as through the collection of voucher specimens and their subsequent identification in the Bolus and Compton Herbaria in consultation with taxonomic experts.

Soil samples were collected and analysed for concentration of nutrient elements to assess whether it would be a confounding variable that may influence differences between grazing treatments. One soil sample was collected from each 1 000 m^2 plot to a depth of 10 cm using an auger or a builder's trowel and mallet after Sternberg et al. (2003). The soil samples were stored in ziplock bags and labelled. They were then air dried at the laboratory to constant weight and sieved through a 2 mm sieve to remove stones and larger particles of organic material. The samples were sent to Bemlab (Pty) Ltd. (Somerset West, South Africa) for analysis of soil texture, pH (1.0 M KCl), resistance, Bray II extractable phosphorous (P Bray II), sodium (Na), potassium (K), calcium (Ca), magnesium (mg), nitrogen (N), sulphur (S) and carbon (C). Soil texture was analysed by first undertaking chemical dispersion using sodium hexametaphosphate. The three fractions of sand were determined by sieving. Silt and clay were determined by analysing sedimentation rates at 20°C using an ASTM E100 (152H-TP) hydrometer. Bray II extractable phosphorus (P Bray II) was analysed using 95% ammonium fluoride and 32% hydrochloric acid). Sodium (Na), potassium (K), calcium (Ca), magnesium (mg) and nitrogen (N) were extracted at pH>7 using 0.2 M ammonium acetate. Sulphur (S) was extracted using concentrated phosphoric acid (at pH = 4) and organic carbon (C) was extracted using the Walkley-Black method (The non-affiliated Soil Analyses Work Committee, 1990). The extracted solutions were then analysed with a Varian ICP-OES optical emission spectrometer.

3.3.3. Data Analysis

All statistical analyses were performed using the R statistical environment (R Version 3.5.2) (R Core Team, 2019). All graphs were generated using the package ggplot2 (Wickham, 2016) unless otherwise stated. One-way ANOVAs followed by post hoc tests were used to examine for statistically significant differences between livestock grazing types. Data was cleaned and processed using tidyr (Wickham and Henry, 2018) and dplyr (Wickham et al. 2018). Additional packages used for undertaking data analyses on mean, standard error and ANOVA included stringr (Wickham, 2018), vegan (community ecology package) (Oksanen et al. 2018), lmertest (Kuznetsova et al. 2017) and psycho (Mackowski, 2018).

3.4. Results

3.4.1. Variation in Species Richness ~ Grazing Type

There was a statistically significant difference (p=0.052) in species richness between sites with no grazing and sites grazed by sheep (Figure 3.1; Table 3.1). The highest median species richness at 49 species (range 40 to 51 species) occurred at sites with no grazing while the lowest median species richness was found at sites grazed by sheep with 29 species (range 19 to 39 species). The plots grazed by cattle had a median species richness of 35 (range 26 to 53 species) which was intermediate between sites that had not been grazed and sites that had been grazed by sheep only.

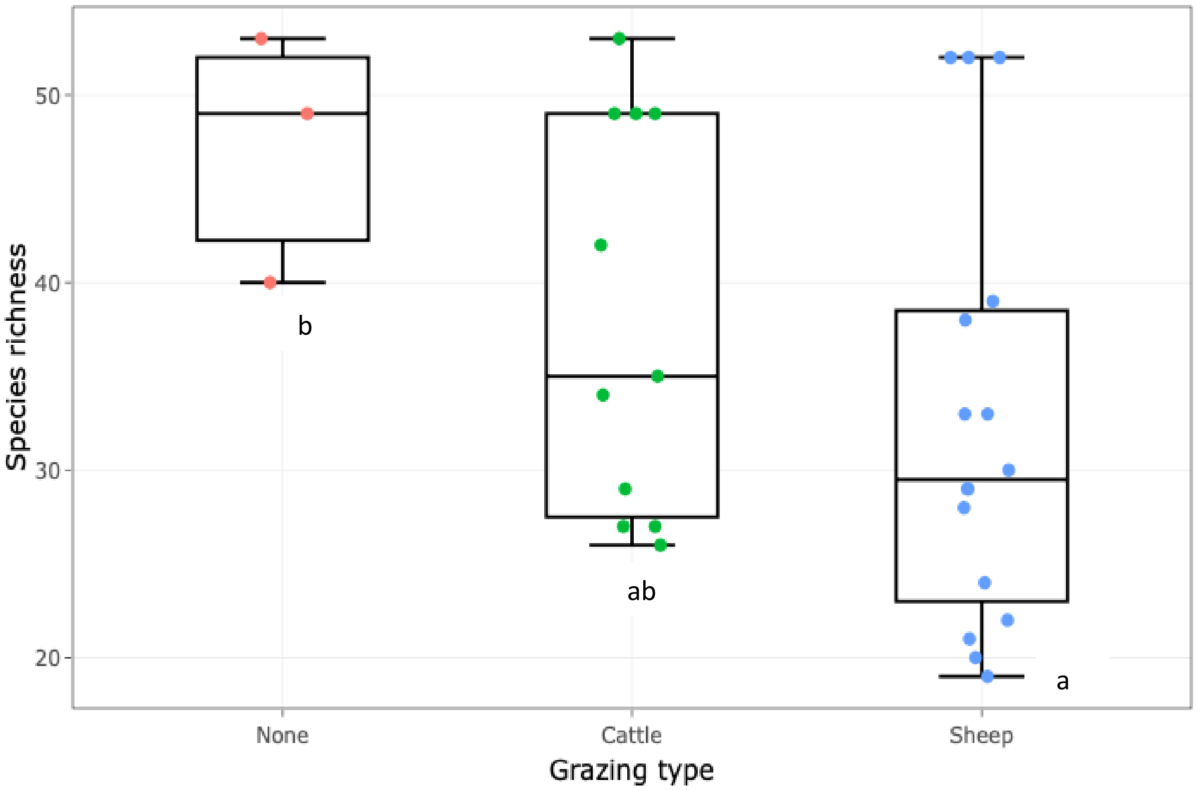

Figure 3.1: Effect of grazing type on species richness from 30 Modified Whittaker Plots in Western, Eastern and Central Shale Ruens Renosterveld with no grazing or grazing by cattle or sheep. Bars with dissimilar letters are significant at P ≤ 0.05.

Table 3.1: Results of an ANOVA comparing grazing type on species richness in 30 Modified Whittaker Plot sites which had either not been grazed (None) or had been grazed by cattle and sheep only. CL_Lower and CL_Higher = Upper and lower confidence levels of the mean.

Contrast	Difference	SE	df	t	p	CL_Lower	CL_Higher
Cattle–None	-8.0	6.7	27	-1.20	0.463	-24.51	8.51
Cattle–Sheep	8.1	4.0	27	2.02	0.127	-1.84	17.97
None–Sheep	16.1	6.5	27	2.46	0.052	-0.11	32.25

3.4.2. Species Area Curves from Modified Whittaker Plots

Species richness increased with increasing size of area from 1 m^2 to 10 m^2, 100 m^2 and 1000 m^2 (Figure 3.2). While the mean value for the intercepts for the different grazing treatments were not significantly different from each other (F = 1.437; df = 2,27; p = 0.255), the mean value for the slope of the relationship between area and species richness for each treatment were significantly different (F 3.999; df = 2,27; p = 0.030). Plots that had not been grazed (None) were significantly different (t = 2.677; df = 2,27; p = 0.032) from those that had been utilised by sheep (Appendix Table S3.2). There were no significant differences (p>0.05) in the comparison of the mean value for the slopes for sites with no grazing and cattle and for sites grazed by cattle and those grazed by sheep.

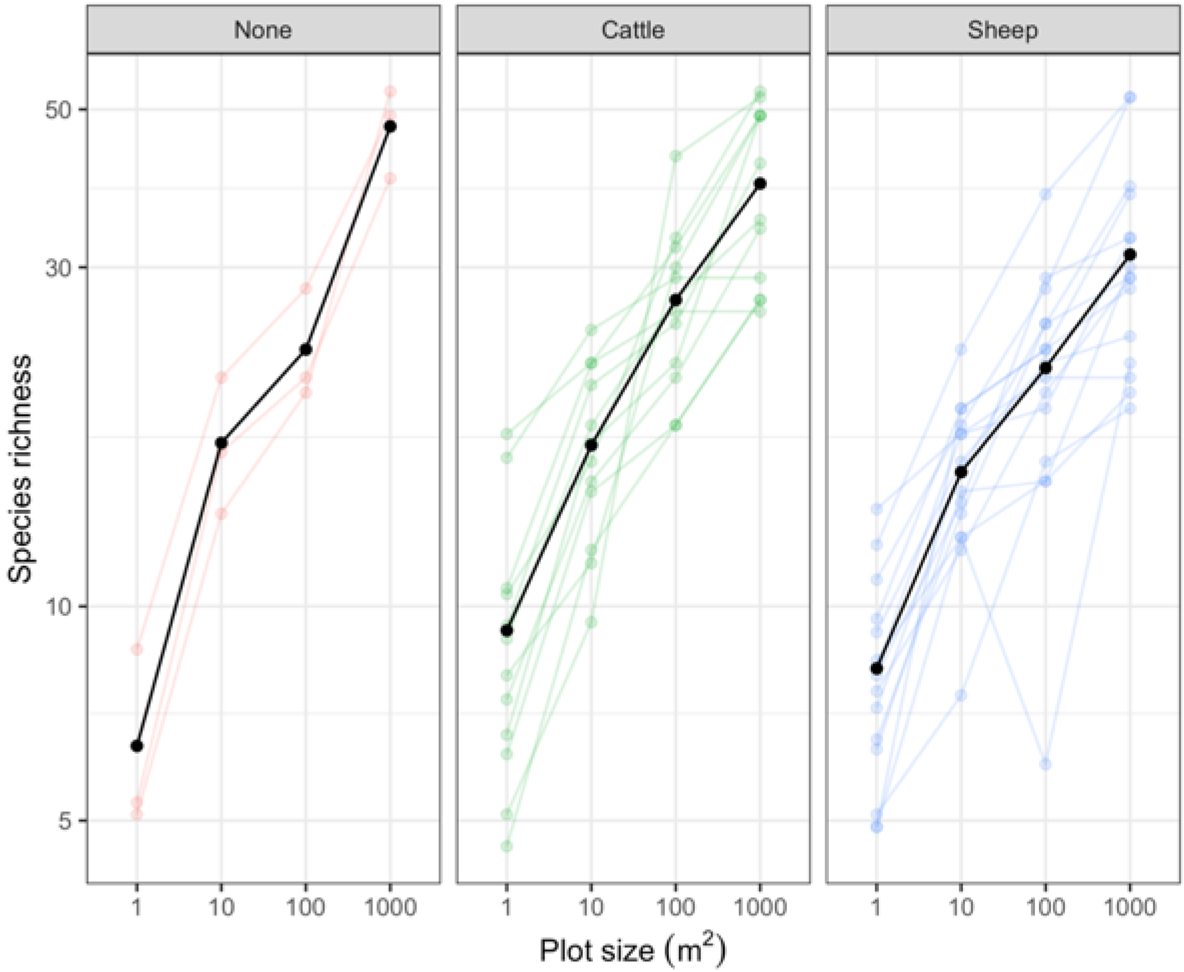

Figure 3.2: Species Area Curves from Modified Whittaker Plot vegetation data showing variation in species richness according to spatial area with grazing treatments of no grazing and grazing by cattle and sheep.

3.4.3. Variation of Species Diversity by Grazing Type

There were significant differences between the different grazing treatments using three diversity indices: Simpson, Inverse Simpson and Shannon-Wiener diversity (Table 3.2). Sites that had not been grazed (None) generally had higher values than sites that had been grazed either by cattle or sheep. However, the differences between treatments were only significant between no grazing and grazing by sheep (Appendix 3.1).

Table 3.2: Mean ± standard error values for three diversity indices from vegetation data collected from Modified Whittaker Plots with treatments of no grazing or grazed by cattle and sheep. Mean± SE with dissimilar letters indicate significant differences between grazing treatments at p<0.05.

Livestock	n	Simpson	Inverse Simpson	Shannon - Weiner
None	3	$0.98^{b} \pm 0.002$	$46.33^{b} \pm 3.283$	$3.83^{b} \pm 0.073$
Cattle	11	$0.98^{ab} \pm 0.002$	$39.08^{ab} \pm 3.019$	$3.63^{ab} \pm 0.080$
Sheep	16	$0.96^{a} \pm 0.003$	$31.13^{a} \pm 2.638$	$3.39^{a} \pm 0.081$
	df	2, 27	2, 27	2, 27
ANOVA	F	4.082	3.902	4.095
	p	**0.0283**	**0.0325**	**0.0280**

3.4.4. Variation in the richness and cover of different growth forms

The largest number of species under all grazing treatments was in the non-succulent shrub, perennial forb and geophyte growth forms (Table 3.3). The mean number of species in all other growth forms was fewer than five. There were significant differences between different grazing types ($F = 7.21$; $df = 2,7$; $p = 0.001$) only in the geophyte and non-succulent shrub growth forms. There were significantly more geophyte species in sites that had not been grazed in comparison with sites that had been grazed by cattle and sheep. Sites that had not been grazed also supported more non-succulent shrub species than sites that had been grazed by sheep, but this number was not significantly higher in sites that had been grazed by cattle.

Table 3.3. Mean± SE of species richness by growth form and grazing type in Modified Whittaker plots in Overberg Renosterveld with grazing treatments of no grazing and grazing by cattle and sheep. Mean± SE with dissimilar letters indicate significant differences between grazing treatments at p<0.05.

Growth Form	None	Cattle	Sheep
Annual Forb	0.7 ± 0.3	3.4 ± 0.9	4.5 ± 0.7
Annual Grass	0.7 ± 0.3	0.8 ± 0.3	0.9 ± 0.3
Geophyte	14.7 ± 1.9[a]	7.0 ± 0.9[b]	7.1 ± 0.9[b]
Perennial Forb	7.0 ± 1.5	17.6 ± 1.5	10.5 ± 1.4
Perennial Grass	3.0 ± 0	2.5 ± 0.3	2.3 ± 0.2
Restio	0 ± 0	2.5 ± 0.3	2.3 ± 0.2
Succulent Shrub	2.0 ± 1.2	1.7 ± 0.4	1.5 ± 0.4
Non-Succulent Shrub	18.7 ± 1.2[a]	17.6 ± 1.5[a]	10.5 ± 1.4[b]

In terms of the percentage cover of different growth forms, the vegetation plots sampled were dominated by non-succulent shrubs and perennial grasses (Table 3.4) which together accounted for 50-75% of the total cover of the vegetation in the grazing treatments. There were no significant differences in cover ($F = 0.651$; $df = 2,7$; $p = 0.523$) between the grazing treatment comparisons for any of the growth forms investigated mostly due to large variations within some plots.

Table 3.4. Mean and SE of the mean percentage cover of different growth forms in Modified Whittaker plots in Overberg Renosterveld with grazing treatments of no grazing and grazing by cattle and sheep.

Growth Form	None	Cattle	Sheep
Annual Forb	0.6 ± 0.6	0.3 ± 0.1	8.4 ± 3.7
Annual Grass	0 ± 0	1.2 ± 0.7	3.6 ± 1.7
Geophyte	4.2 ± 3.8	0.9 ± 0.3	1.4 ± 0.5
Perennial Forb	2.1 ± 0.9	2.0 ± 0.6	1.4 ± 0.5
Perennial Grass	27.6 ± 13.6	27.2 ± 4.5	15.9 ± 4.2
Restio	0 ± 0	0.3 ± 0.2	0 ± 0
Succulent Shrub	0.2 ± 0.2	2.9 ± 1.2	2.7 ± 1.8
Non-Succulent Shrub	40.8 ± 15.4	47.3 ± 5.3	35.5 ± 5.3

3.4.5. Soil Properties

Results from an analysis of soil properties across sites are shown in Figure 3.3. Results from ANOVA showed significant difference between grazing treatments only for Resistance ($F = 5.44$; $df = 2,26$; $p = 0.011$) and Mg ($F = 4.52$; $df = 2,26$; $p = 0.021$). For resistance the differences were only significant ($p<0.05$) between sites grazed by cattle and sites grazed by sheep. For Mg the differences were only significant ($p<0.05$) between sites that had not been grazed and sites that had been grazed by cattle.

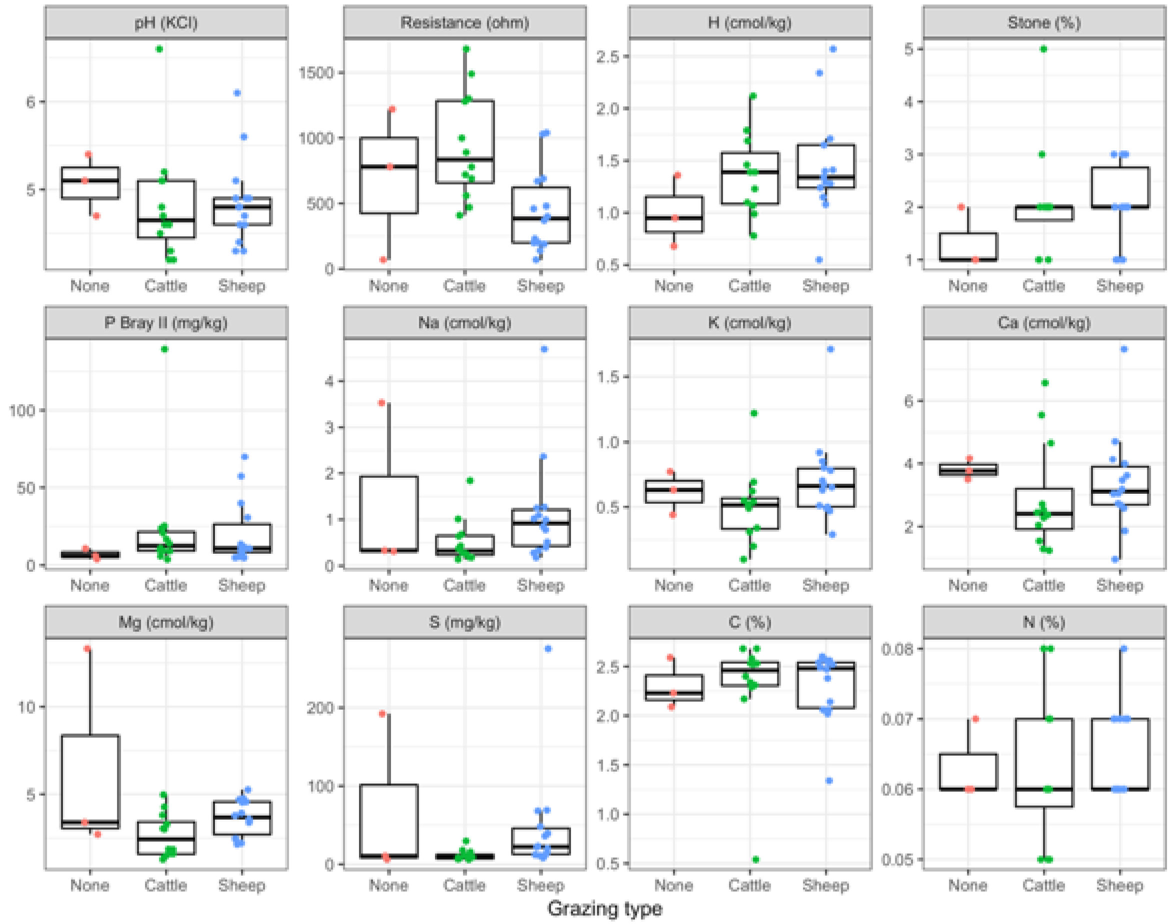

Figure 3.3: Variation in soil properties by livestock grazing type in Overberg Renosterveld with grazing treatments of no grazing and grazing by cattle and sheep.

3.5. Discussion

3.5.1. Effects of Livestock Grazing on Plant Species Richness and Diversity

The results presented agreed with the hypothesis that livestock grazing by sheep caused a reduction in species richness and diversity and vegetation cover in Overberg Renosterveld in comparison to sites where no grazing has taken place. Findings from analysis of soil samples suggest that the differences in plant species richness and diversity found are not influenced by edaphic variation across sites. Curtis (2013a) undertook similar research using grazing exclosures to try and understand the effects of livestock grazing on Overberg Renosterveld and reported that livestock grazing did not have a significant effect on plant species diversity and vegetation cover. However, palatable species such as *Themeda triandra* (Rooigras) showed a slight reduction in size and successful flowering and seed set during the study (Curtis, 2013a), thus indicating that livestock grazing had the potential to have a negative effect on the diversity of palatable plant species within the study area. It is thought that the subtle impacts recorded by Curtis (2013a) were as a result of a smaller number of sites sampled. Curtis (2013a) used six experimental sites in comparison to the thirty sites used in the current study.

Findings that grazing treatments by sheep and cattle relative to treatments of no grazing caused a reduction in species richness and species diversity were further supported by the species area curves that showed a greater increase in species richness by area at ungrazed sites in comparison with sites grazed by sheep. Similar findings were also reported in sheep-grazed, Succulent Karoo rangelands by Todd (2006) who showed that the richness of palatable species increased with distance from livestock watering points. In contrast, Verdu et al. (2000) argued that grazing by sheep and goats is beneficial to biodiversity in Mediterranean type shrublands. This case study comes from the Iberian Peninsula in Spain but is also highly applicable to Mediterranean type shrublands in other parts of the world (Verdu et al. 2000). The purported increase in species is thought to occur as a result of the modification of competition between faster growing, more vigorous species that would normally shade out slower growing and rarer species, thus reducing shade from additional cover provided by these species and allowing them to thrive (Verdu et al. 2000).

3.5.2. Effects of Livestock Grazing on Growth Form Diversity and Cover

The differences between grazing impacts from sheep and cattle are likely to be due to different grazing preferences by sheep and cattle which modify competition between plant species from different growth forms (Samuels et al. 2016). Cattle are bulk grazers with a diet consisting of 70% grasses, whereas sheep have a more varied diet with a greater focus on browsing non-succulent shrubs that can comprise as much as 20% of their diet (Esler et al. 2014; Samuels et al. 2016). Research undertaken by Samuels et al. (2016) in renosterveld vegetation in the Kamiesberg in Namaqualand found that heavy grazing by sheep leads to a reduction in the presence of overstorey shrubs. Sheep tend to be less selective in the species that they target and browse a wider range of non-succulent shrubs than cattle (Esler et al. 2014; Samuels et al. 2016). However, Samuels et al. (2016) also found that dietary selection by all livestock varied depending on the season and associated availability of annual and herbaceous perennial vegetation. During drier conditions, sheep were more likely to browse non-succulent shrubs when there were fewer other options available (Samuels et al. 2016).

In contrast to the results presented here, other research has shown that responses to livestock grazing by both cattle and sheep on geophytes is highly varied across ecosystems

(Noy-Meir and Oron, 2001). McDowell (1995) conducted one of the first studies examining the effects of livestock grazing by cattle and sheep on species richness and composition of renosterveld vegetation on the West Coast. The study revealed no difference in species richness between releves that were 'grazed' and those that were 'ungrazed' but found that grazed sites had a higher cover of geophytes, particularly those in the Iridaceae family. This is in contrast to the research findings of this study. The differences between findings from this study in Overberg Renosterveld and findings from McDowell (1995) in West Coast Renosterveld show that more research is needed encompassing a higher number of sites with no grazing to better understand the impacts of livestock grazing on geophyte cover in renosterveld vegetation at different sites across the CFR. The findings of Noy-Meir and Oron (2001) found that on average, geophyte diversity was 21% greater at sites that had been grazed than sites that had not been grazed. Research by Todd and Hoffman (1999) undertaken in Succulent Karoo vegetation in Namaqualand found that sites with increased livestock grazing in communal lands had increased species richness of geophytes than sites with a lower grazing load. Research undertaken in the New England Tablelands in Australia found no significant differences in geophyte species richness by grazing treatment (McIntyre et al. 1995). Given the reduction in geophyte species richness in response to sheep grazing revealed by this study, it is important to ensure that stocking rate does not lead to a negative impact on this growth form in Overberg Renosterveld.

3.5.3. Implications for Renosterveld Management

Early research findings have indicated that unmanaged livestock grazing may cause negative impacts in Overberg Renosterveld vegetation (Curtis, 2013a), this is in contrast to the findings from this study around cattle grazing and changes in plant species richness and vegetation cover. This demonstrates that, with care, cattle grazing likely poses a low risk of decreasing plant species richness and diversity in Overberg Renosterveld vegetation. However, this is not the case with sheep grazing which showed negative impacts on species richness of geophytes and non-succulent shrubs. Care must therefore be taken to ensure appropriate grazing intensity to sustain the geophytes in the renosterveld. Where necessary, fencing renosterveld fragments separately will ensure that the renosterveld vegetation does not become overgrazed (Esler et al. 2014).

Given that knowledge of past renosterveld vegetation structure and function is still limited (Curtis, 2013a; Rebelo, 1995; Rebelo et al. 2006), it is important to adopt a sustainable and adaptive management approach (Milton et al. 2007; Esler et al. 2014) to ensure that livestock grazing by sheep does not cause degradation to renosterveld vegetation. Damage can occur quickly and recovery can take far longer (Esler et al. 2014). Some researchers suggest that overgrazing has the potential to cause irreversible damage to renosterveld (Curtis, 2013a). It is also important to ensure that stocking levels are low enough to ensure resilience during times of drought and that livestock are removed immediately after signs of overgrazing are detected (Boucher, 1995; Curtis, 2013a; Esler et al. 2014). This is crucial to ensure that palatable plant species remain present in sufficient numbers and can reproduce (Boucher, 1995).

3.5.4. Directions for Future Research

Livestock type is a key factor influencing the impact of livestock grazing on plant species richness and diversity in Overberg Renosterveld. Further research using the same protocols to this study can be used to determine whether similar responses to livestock grazing and resting following livestock grazing are found in renosterveld in the Overberg on other geologies such as silcrete or quartz-silcrete patches (Groenewald, 2014). The grazing history of the vegetation is also important, with the lack of availability of this information for this study being a key limiting factor. This highlights the value of long-term datasets in drawing the most useful conclusions to inform management of livestock grazing. Future research should be undertaken across longer timescales to mitigate this problem, where possible using a series of controlled grazing trials where stocking density and associated grazing intensity can be monitored over time (Hester et al. 2000). One of the main challenges identified in this study concerns the scale of the Modified Whittaker Plots relative to the high species diversity of the vegetation, particularly when overgrazed. Although Modified Whittaker Plots are the most effective way of recording the maximum number of species present in an area (Stohlgren, 1995; Stohlgren et al. 1998), in sites with high alpha diversity and high species turnover, undertaking one Modified Whittaker Plot is highly time consuming. This has meant that plots were surveyed across different spring seasons because of the time constraint. Smaller plots should be used so that an increased number of replicate sites can be surveyed within the same spring season. An additional constraint was the number of sites available that had a history of no grazing, with most areas of renosterveld within areas of agricultural land that were not fenced separately and had livestock grazing adjacent pasture crops or wheat stubble. This reduced the range of statistical analyses that could be used on the dataset due to the unbalanced number of replicates between grazing treatments. To improve statistical rigour, future studies should undertake to survey a greater number of plots with a history of little to no livestock grazing to be able to better unpack the differences in vegetation between sites grazed by livestock and sites with no livestock grazing.

Chapter 4: Ecosystem resilience and restoration potential: Investigating the effects of grazing on diversity and composition of the soil seed bank in Overberg Renosterveld

4.1. Introduction

Grazing by domestic livestock, after ploughing, is considered one of the greatest threats to renosterveld vegetation in the Overberg. Environmental damage that can occur as a result of badly managed grazing practices can include changes in vegetation cover, altered nutrient cycling, soil compaction, loss of biological soil crust cover and soil erosion (Anderson and Hoffman, 2007; Harrington and Kathol, 2008; Rutherford and Powrie, 2010). This study investigated the impacts of livestock grazing on the soil seed bank in Overberg Renosterveld vegetation. It also examined the restoration potential of renosterveld from the soil seed bank following livestock grazing by cattle and sheep in comparison to sites with a treatment of no grazing for the duration of the study. Soil seed banks are an important component of ecosystem resilience and a key determinant of restoration potential in degraded ecosystems (Wang et al. 2013). In the context of community and restoration ecology, the composition of soil seed banks can be used to predict the composition of new plant recruitment and the assessment thereof is an important tool that can be used to measure ecosystem health (Wang et al. 2013).

Soil seed banks reflect the ‘memory’ of past vegetation and represent the structure of future plant populations (Fisher et al. 2009; Wang et al. 2013). Where Overberg Renosterveld vegetation has been degraded by livestock grazing, restoration potential can be assessed by examining the composition of the soil seed bank. Ecological restoration aims to recover the characteristics of an ecosystem, such as its biodiversity and function, which have been degraded or destroyed, generally as a result of human activities (Benayas and Bullock, 2012). The role of the soil seed bank in ecological restoration has been the subject of several studies. Soil seed bank health has increased importance in influencing restoration potential in areas where habitat loss has led to ecosystem fragmentation and habitat isolation which in turn makes it harder for propagules to enter the ecosystem from other vegetation fragments (Valko et al. 2011). In addition, the degree of correlation between the standing vegetation and the soil seed bank is of considerable importance in the context of ecological restoration (De Villiers et al. 2003). The duration of germinable seed survival in the soil seed bank is also crucial, with long term seed survival being very important for restoration management (Bakker et al. 1996).

Relative to many other vegetation types, there is a dearth of knowledge about seed banks in renosterveld vegetation (Cowan and Anderson, 2014). The majority of soil seed bank research globally has taken place in grasslands and arable fields, with considerably less data being available from woodlands, heathlands, dunes, deserts, alpine and wetland communities (De Villiers et al. 2003; Shi et al. 2020). Research has shown that the degree of disturbance in the ecosystem is a key determinant of the similarity or difference between plant species assemblages in the standing vegetation in comparison to the soil seed bank (De Villiers et al. 2003). Vegetation, such as renosterveld, with high levels of disturbance from key ecological drivers are likely to present with similar species composition between the seed bank and the above ground vegetation (De Villiers et al. 2003). Other ecosystem types that have experienced minimal disturbance during their landscape history have far less correspondence between the seed bank and standing vegetation (De Villers et al. 2003). However, soil seed banks have been found to have considerable spatial variation in seed density, species composition and species richness, with seed distribution being strongly influenced by vegetation heterogeneity and availability of seed traps such as vegetation and low stature obstacles such as rocks (Dreber and Esler, 2011). This is particularly likely to be the case in Overberg Renosterveld vegetation which has high microtopographic variation and a variety of different microhabitats within the main renosterveld vegetation types (Poulsen and Curtis-Scott, 2020).

Research on soil seed banks in renosterveld has predominantly focused on edge effects in renosterveld from alien grasses and restoration potential in old fields (Heelemann, 2010). The majority of research with a focus on livestock grazing and soil seed banks has taken a grazing intensity approach with minimal differentiation between different livestock types. This is despite the fact that different domestic livestock types such as sheep and cattle often target palatable plant species from different suites of functional groups (Van Oudtshoorn, 2019). Despite the likely higher restoration potential of overgrazed renosterveld in comparison to old fields, this is an area that has been neglected by researchers and there is currently almost no data available to inform management practices (Heelemann, 2010). In a calcareous grassland in Belgium, changes in species richness and diversity in the standing vegetation and the soil seed bank were studied over a period of 11 years, comparing sites grazed by cattle with mown sites and abandoned and ungrazed sites. Species richness in the cattle grazed plots remained constant throughout the duration of the study, whereas species richness in the abandoned plots declined by over 60%. This corresponded with an associated decrease in seed bank density and species richness, which was attributed to reduced availability of light for germination and associated decrease in germination opportunities (Jaquemyn et al. 2011). In the semiarid rangelands of Patagonia, little seed bank variation was observed over four years of differing livestock grazing treatments, with only a slight increase in perennial grass seeds in the germinable seed bank following a reduction in length of the grazing season during late spring (Bertiller, 1996). In contrast, in Namibia's Nama Karoo, severe grazing by a variety of livestock types was found to increase species richness of the germinable soil seed bank over time (Dreber and Esler, 2011).

Restoration interventions can greatly increase the conservation value of degraded fragments which holds significance given the paucity of remaining renosterveld and associated difficulties in meeting conservation targets (Waller, 2013). However, active restoration interventions can be incredibly costly (Waller, 2013). In consequence, it is critical that active restoration efforts are appropriately targeted within areas where there are no other alternatives for rehabilitation. However, at present in the case of overgrazed renosterveld rangelands there are no data or protocols in place to determine at what point ecosystem degradation through overgrazing moves beyond the point that recovery through passive restoration will not occur. A greater understanding of the influence of livestock grazing on soil seed bank species richness and diversity will greatly assist in determining appropriate grazing loads for Overberg Renosterveld vegetation and management interventions. It will also assist in determining the restoration potential of overgrazed renosterveld fragments to direct appropriate passive or active restoration interventions.

4.2. Objective and Hypothesis

The objective was to investigate the effect of grazing on species diversity, species richness and growth form diversity of the soil seed bank as well as to examine the relationship in Overberg Renosterveld between above ground vegetation and the soil-stored seed bank following grazing.

It was hypothesised that sheep grazing will reduce species richness, species diversity and growth form diversity in the soil seed bank in comparison with sites grazed by cattle and with a treatment of no grazing.

4.3. Materials and Methods

4.3.1. Study Site

This research took place in the Critically Endangered shale renosterveld vegetation types of South Africa's Overberg region, namely Western, Central and Eastern Rûens Shale Renosterveld (Rebelo et al. 2006) (see section 1.8 for details).

4.3.2. Research Approach

The methods used in this study are after Brown (1993), Jacquemyn et al. (2011), Sternberg et al. (2015), Sternberg et al. (2003) and Cowan and Anderson (2014). Thirty sites were selected that were either grazed by sheep (12 sites), grazed by cattle (15 sites) or had no grazing taking place when the study commenced (3 sites). Soil samples were collected during late autumn (April-May) 2017 after seed dispersal had occurred and before any germination event had taken place (Meissner and Facelli, 1999; Yagil et al. 2006). Soil samples were collected from 30 sites where above ground vegetation was assessed for species richness, species diversity and growth form diversity using Modified Whittaker Plot (MWP) (Chapter 3). From each of the thirty sites, ten soil samples were taken using an auger or a mallet and builder's trowel. Samples were taken from the top 10 cm^2 and a total of 600 g was collected per sample after Sternberg et al. (2003). Samples were air dried at 20°C for a week and then cold stored at 10°C in paper bags to ensure maximum viability of seeds until the appropriate season for sowing. Prior to sowing, all samples were sieved through a 4 mm sieve to remove any larger stones and debris before being spread on top of a 6 cm layer of commercially available potting compost in 20x15 cm seed trays. The trays were lined with shade netting to prevent soil from escaping through the drainage holes in the seed trays. Once in seed trays the soil samples were smoke treated to enhance germination (Brown, 1993; Waller, 2013). Cut branches of *Passerina corymbosa* (Thymeliaceae) (a common shrub indigenous to Overberg Renosterveld vegetation) were ignited in a metal drum and then the smoke was blown through a pipe into an attached tent (sealed with sand around the edges) which contained the seed trays (Figure 4.1). Seed trays were left in the tent for two hours to allow the chemicals from the smoke to precipitate on the soil surface.

Figure 4.1. Smoking seed trays at Kirstenbosch NBG to enhance germination.

Following the smoke treatment, the seed trays were placed in a glasshouse and germinated and grown under controlled conditions for 18 months with thermostat temperatures at 20°C and watered using aerial irrigation for ten minutes per day at 06h00 each morning (Figure 4.2). Ten control seed trays were also present containing sterilised potting compost to monitor for contaminant seedlings from the surrounding environment (Jacquemyn et al. 2011). No seedlings germinated in the control trays. When seedlings were large enough to identify differences between taxa, they were assigned a working name and grown on and monitored until large enough to identify and assign to a growth form category. The growth form categories used were Forbs, Geophytes, Annuals, Graminoids, Succulent Shrubs and

Non-Succulent Shrubs. When each plant was large enough to undertake a positive identification in the herbarium, two voucher specimens of each species were pressed. These were then identified to family, genus or species level with help of experts at the Bolus Herbarium at the University of Cape Town and the Compton Herbarium at the South African National Biodiversity Institute (SANBI). The results of the soil seed bank study were correlated with the Modified Whittaker plot data collected for Chapter 3 to examine the relationship in Overberg Renosterveld between the standing vegetation and the soil seed bank at sites across grazing treatments. Data from the Modified Whittaker plots and soil seed bank were used to record proportional distribution of each species from the standing vegetation and soil seed bank between grazing treatments. Data was also presented on presence/absence of each taxon in the soil seed bank. Family, growth form and status on the Red List of South African Plants was also recorded for each taxon.

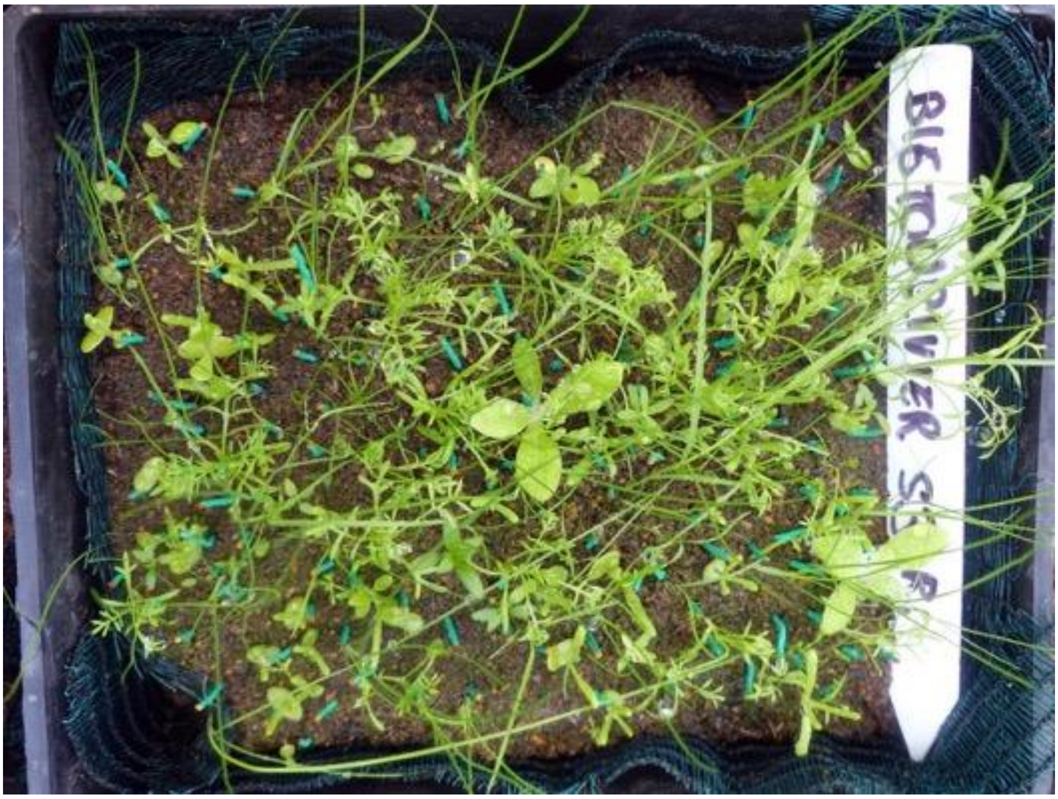

Figure 4.2. Germinated seedlings two weeks after the start of the experiment marked with cut paperclips.

4.3.3. Data Analysis

Similarity between species composition of the standing vegetation and the soil seed bank was calculated using Sorenson's index of similarity (IS_s). In this case *c* is the number of species present both in the standing vegetation and the soil seed bank, *A* is the total number of species recorded in the standing vegetation and *B* is the total number of species recorded in the soil seed bank (Muller-Dombois and Ellenberg, 1974; De Villiers et al. 2003). The equation is as follows:

$$IS_s = \frac{2c}{A + B} \times 100$$

All other statistical analyses were performed using the R statistical environment (R Version 3.5.2) (R Core Team, 2019). The R package multicomp (Hothorn et al. 2008) was used for undertaking the Kruskal Wallis rank sum tests. The Bray-Curtis distances for the non-multi-dimensional scaling (NMDS) ordination was undertaken using the R package Vegan (Oksanen et al. 2019) and the figure was generated using ggplot2 (Wickham, 2016). The hierarchical clustering dendrogram analysis was calculated using the package 'cluster' (Maechler et al. 2019) with ggplot2 (Wickham, 2016) and ggdendro (de Vries and Ripley, 2016) being used to generate the figures. The package ggplot2 was used to generate the Sorenson's index bar plot (Wickham, 2016).

4.4. Results

4.4.1. Standing Vegetation and Seed Bank Species List

A total of 322 taxa from 57 different families were recorded from the Modified Whittaker Plots and germinable soil seed bank (Appendix 3, Table S4.1). Of these, 72% were identified to species/subspecies/variant level, 11% only to genus level, 16% to family level and 1% of the taxa to only monocot/dicot level. Taxa identified to monocot/dicot, family or genus level were either too small for identification to species/subspecies/variant level or had not produced flowers/seeds to facilitate identification. This was either due to data being collected outside the flower season of these specific taxa or the seedlings not having sufficient time to reach maturity during the duration of the study. In terms of Red List status, a total of 27% of taxa could not be assigned a Red List status due to not being identified to species level. A total of 6% of taxa were not assessed on the Red List of South African Plants due to being classified as exotic naturalised weeds. A total of 1% were Red Listed as Data Deficient, 60% as Least Concern, 3% as Vulnerable, 2% as Near Threatened, 1% as Endangered and 0.3% as Critically Endangered. A total of 48% of taxa had seed present in the germinable seed bank, in comparison to 52% of taxa having no seed present across sites in the germinable seed bank (Appendix Table S4.1). All percentages were rounded to the nearest whole number if >1% and to the nearest first decimal place if <1%. A total of 48% of taxa were recorded at sites grazed by sheep, 39% were recorded at sites grazed by cattle and 13% were recorded at sites with a treatment of no grazing.

4.4.2. Seed bank species richness and diversity and # Individuals

Analysis was undertaken to determine whether soil seed bank species richness and number of individuals were statistically significantly different between grazing treatments. There were no statistically significant differences in species richness (Kruskal-Wallis chi-squared = 0.076, df = 2, $p > 0.05$), Simpsons diversity (Kruskal-Wallis chi-squared = 1.574, df = 2, $p > 0.05$) or Shannon-Weiner diversity (Kruskal-Wallis chi-squared = 0.393, df = 2, $p > 0.05$) between grazing treatments (Table 4.1). There were also no statistically significant differences between number of individuals (Kruskal-Wallis chi-squared = 0.0019817, df = 2, $p > 0.05$) between grazing types. The highest standard deviation was at sites grazed by sheep at 7.4, lower at sites grazed by cattle (7.2) and lowest at sites with a treatment of no grazing (none). The highest mean number of individuals recorded from the germinable seed bank was recorded at sites grazed by sheep (84.3). A considerably lower mean number of individuals was recorded from the germinable seed bank at sites grazed by cattle (37), with the lowest mean number of individuals recorded at sites with a treatment of no grazing (none) (33.7). The highest species diversity index measured by Shannon-Weiner diversity was also at sites grazed by cattle, but the next lowest was at sites grazed by sheep (1.970), with the lowest species diversity at sites with a treatment of no grazing (none).

Table 4.1: Mean and standard deviation of total species richness, diversity and number of individuals in the soil seed bank calculated using Simpson and Shannon-Weiner Diversity Indices for sites with grazing treatments of no grazing, grazing by cattle and grazing by sheep.

Treatment	n	# Individuals	Species Richness	Simpson	Shannon-Weiner
None	3	33.7 ± 32.49	12.0 ± 6.9	0.800 ± 0.200	1.900 ± 0.200
Cattle	11	37.0 ± 24.8	12.8 ± 7.2	0.817 ± 0.106	2.055 ± 0.603
Sheep	16	84.3 ± 152.0	13.0 ± 7.4	0.794 ± 0.126	1.970 ± 0.418

4.4.3. Seed bank growth form richness and # individuals

Mean and standard deviation of species richness of the germinable soil seed bank by growth form is shown below in Table 4.2. A Kruskal-Wallis rank sum test was used owing to the data being non-normally distributed to determine whether species richness was statistically significantly different between grazing treatments for any of the growth forms. There were no statistically significant differences in species richness (Kruskal-Wallis chi-squared = 0.075843, df = 2, $p > 0.05$) between grazing types. Analysis was undertaken to determine whether soil seed bank species richness was statistically significantly different between grazing treatments. The data analysed were non-normally distributed, so a Kruskal-Wallis rank sum test was used. There were no statistically significant differences in species richness between grazing treatments for any of the growth forms.

The highest mean species richness of 9.4 is for non-succulent shrubs at sites grazed by cattle. The lowest mean species richness for the soil seed bank is for succulent shrubs at sites grazed by cattle (1.3). At sites with a treatment of no grazing, the highest mean species richness of the soil seed bank is for non-succulent shrubs (5.7). The lowest mean species richness for this treatment is succulent shrubs. At sites grazed by cattle, the soil seed bank has the highest mean species richness for non-succulent shrubs with the lowest mean species richness for succulent shrubs. At sites grazed by sheep, the highest mean species richness is for annuals (8.8), followed by non-succulent shrubs (8.0), perennial forbs (4.9), perennial grass (2.8) and succulent shrubs (1.4) respectively.

Table 4.2: Mean and standard deviation of species richness of the soil seed bank by growth form for sites with grazing treatments of no grazing, grazing by cattle and grazing by sheep.

	None	Cattle	Sheep
Annuals	3.0 ± 1.7	4.7 ± 5.0	8.8 ± 12.8
Perennial Grass	4.3 ± 4.5	3.1 ± 2.9	2.8 ± 3.0
Perennial Forbs	4.0 ± 6.9	2.9 ± 2.6	4.9 ± 4.6
Succulent Shrubs	2.3 ± 0.6	1.3 ± 1.9	1.4 ± 2.1
Non-Succulent Shrubs	5.7 ± 5.5	9.4 ± 7.5	8.0 ± 7.3

Mean and standard deviation of the number of individuals in the germinable seed bank at sites with a treatment of no grazing, grazing by cattle and grazing by sheep is shown below in Table 4.3. The highest mean number of individuals in the soil seed bank was recorded for annuals at sites grazed by sheep. Sites with a treatment of no grazing had the highest mean number of individuals for perennial grasses, followed by non-succulent shrubs (9.3), perennial forbs (7.3), annuals (3.7) and succulent shrubs (2.3). At sites grazed by cattle, the highest number of individuals from the germinable seed bank was recorded for non-succulent shrubs (16.6), followed by annuals (10.1), perennial grasses (4.6), perennial forbs (4.1) with succulent shrubs having the lowest mean number of individuals at 1.4. At sites grazed by sheep the highest mean number of individuals is for annuals (57.4), followed by non-succulent shrubs (12.7), perennial forbs (8.3), perennial grass (3.7) respectively with the lowest mean number of individuals being succulent shrubs (1.9).

Table 4.3: Mean and standard deviation of # individuals in the soil seed bank by growth form for sites with grazing treatments of no grazing, grazing by cattle and grazing by sheep.

	None	Cattle	Sheep
Annuals	3.7 ± 2.3	10.1 ± 17.3	57.4 ± 138.2
Perennial Grass	11.0 ± 12.1	4.6 ± 6.1	3.7 ± 4.9
Perennial Forbs	7.3 ± 11.9	4.1 ± 3.4	8.3 ± 9.7
Succulent Shrubs	2.3 ± 0.6	1.4 ± 2.1	1.9 ± 3.2
Non-Succulent Shrubs	9.3 ± 8.5	16.6 ± 14.1	12.7 ± 19.0

4.4.4. Soil seed bank NMDS Ordinations and Hierarchical Clustering

NMDS ordinations with Bray Curtis distances were calculated to show arrangement of species composition for sites grazed by cattle, sites grazed by sheep and sites with a treatment of no grazing are shown below in Figure 4.3. There are no patterns of separation of the sites based on grazing treatment.

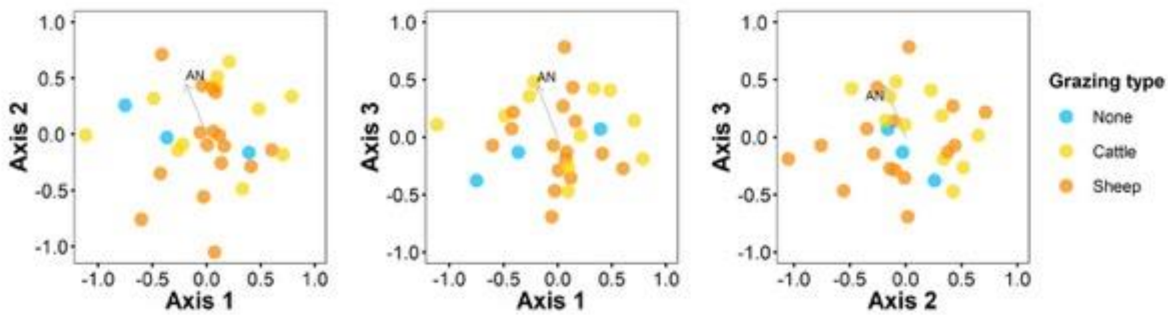

Figure 4.3: NMDS Ordinations with Bray Curtis distances between renosterveld sites with a treatment of no grazing (shown in blue), grazed by cattle (shown in yellow) or grazed by sheep (shown in orange). Only vectors with p < 0.05 are shown.

A hierarchical clustering dendrogram based on Bray Curtis distances between sites using number of individuals in the soil seed bank is shown below in Figure 4.4. No clear trends are present in the results.

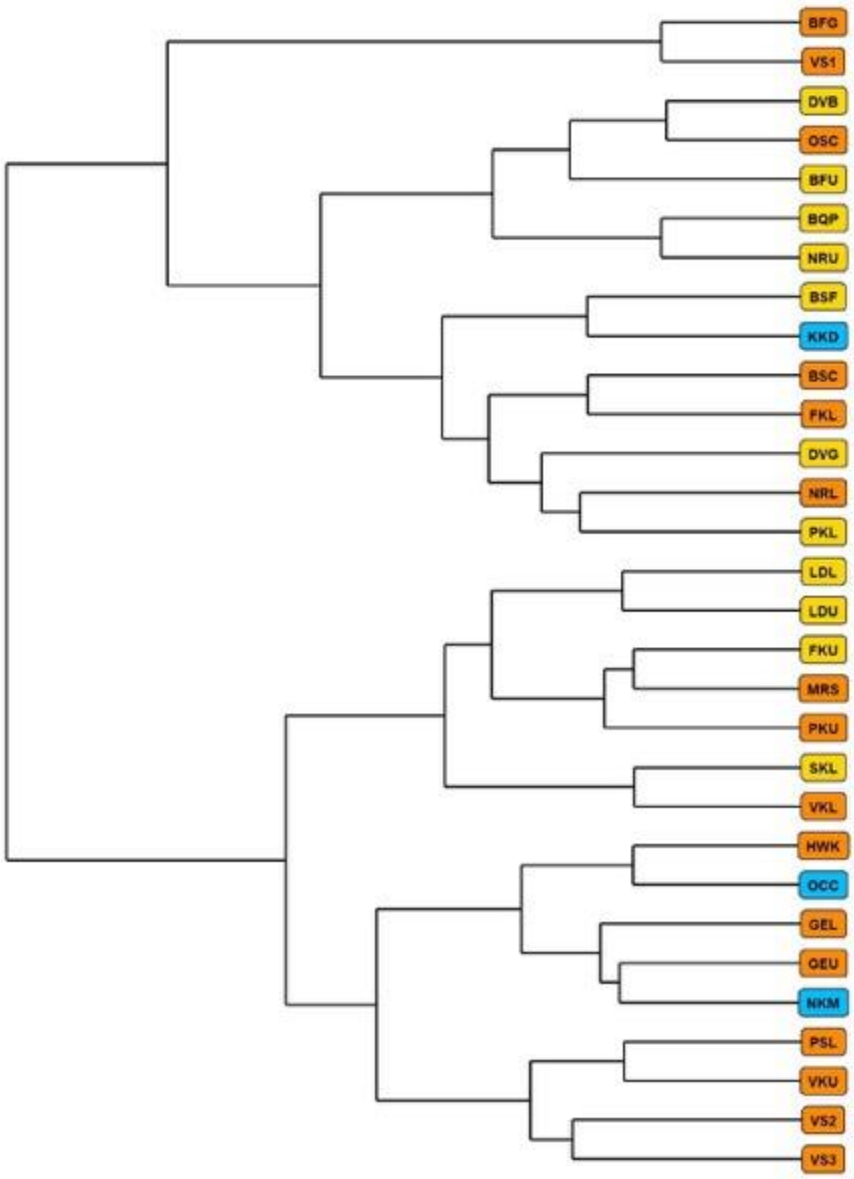

Figure 4.4: Hierarchical clustering dendrogram based on Bray Curtis distances between sites using number of individuals at renosterveld sites with a treatment of no grazing (shown in blue), grazed by cattle (yellow) or grazed by sheep (orange).

A hierarchical clustering dendrogram based in Bray Curtis distances between sites using presence only is shown in Appendix 4 (Figure S4.1). There are no clear trends present.

4.4.5. Similarity Between Standing Vegetation and Soil Seed Bank

Similarity of species composition between the above ground vegetation and the soil seed bank was calculated using Sorenson's Similarity Index (Figure 4.5). Total similarity between the standing vegetation and the soil seed bank across all sites surveyed was *IS*s = 31.41. For all species by grazing treatment, the highest similarity between the standing vegetation and the soil seed bank was cattle (30.49), followed by grazing by sheep (29.5) with the lowest similarity found at sites with a treatment of no grazing (19.04). There were no species found in both the standing vegetation and the soil seed bank for annuals at sites with a treatment of no grazing, or for geophytes at sites with a treatment of no grazing or at sites grazed by sheep. The highest similarity was for succulent shrubs at sites with a treatment of no grazing (66.7), followed by perennial grasses at sites grazed by sheep. For annuals the highest similarity was found at sites grazed by sheep (43.1), followed by sites grazed by cattle (37.8). For geophytes there was only similarity found at sites grazed by cattle, at the low level of 6.3. Perennial grasses had the highest similarity at sites grazed by sheep (53.3), followed by sites with a treatment of no grazing (46.2), with the lowest similarity found at sites grazed by cattle (40.0). Perennial forbs had a lower similarity for all grazing treatments than for perennial grasses. Perennial forbs had similar values for sites grazed by sheep and cattle, with sites with a treatment of no grazing having lower similarity at 18.2. For non-succulent shrubs the highest similarity was for sites grazed by cattle (34.41), followed by sites grazed by sheep (28.57), with the lowest similarity at sites with a treatment of no grazing (9.23).

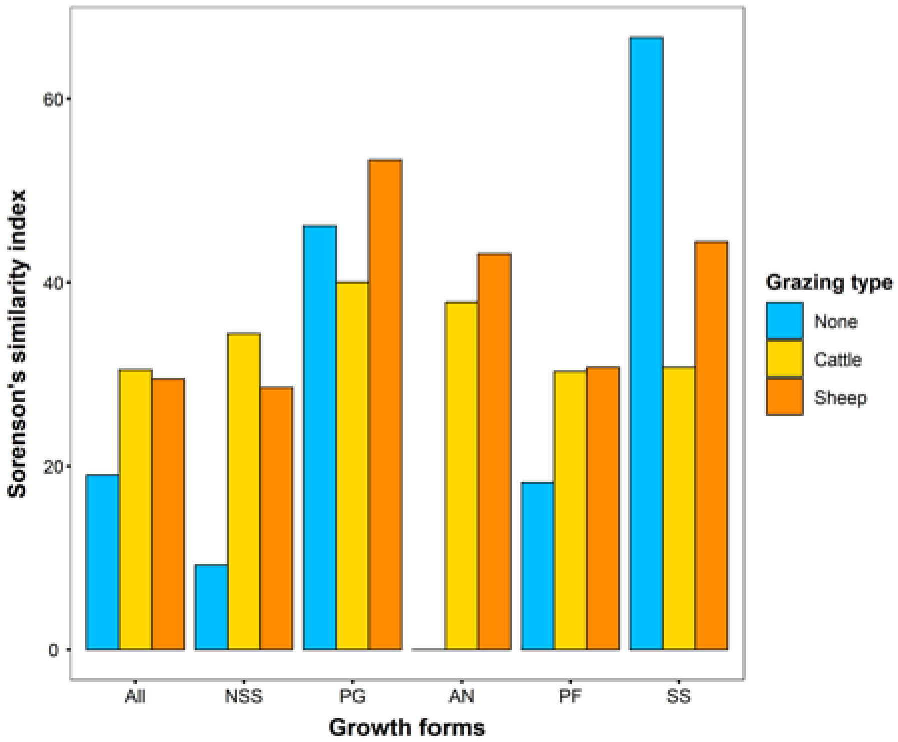

Figure 4.5: Histogram Bar plots showing similarity between the standing vegetation and soil seed bank using Sorenson's similarity index at sites with a treatment of no grazing, grazed by cattle or sheep for all species (All), non-succulent shrubs (NSS), perennial grasses (PG), annuals (AN), perennial forbs (PF) and succulent shrubs (SS).

4.5. Discussion

4.5.1. Effects of Livestock Grazing on Soil Seed Bank Richness and Diversity

This is the first study to investigate the impact of livestock grazing on the soil seed bank in Overberg Renosterveld. The taxa present in the soil seed bank represented a variety of different growth forms including palatable grasses and shrubs. The soil seed bank is well-developed and dominated by indigenous species that can be used as restoration target species with only a few alien or ruderal species present. In the context of global ecosystems, it is unusual to find vegetation with a soil seed bank of high restoration potential. A review of similarity between the standing vegetation and the soil seed bank across ecosystems undertaken by Hopfensperger (2007) found that the lowest degree of similarity was found in forest ecosystems, predominantly attributed to the low longevity of seed of most forest species. The highest Sorenson similarity values were found in grassland ecosystems (Hofensperger, 2007). This is thought to be as a result of low dispersal distances of seed from grassland species (Bossyt and Hermy, 2004; Hofensperger, 2007). However, research by Loydi et al. (2012) used NMS ordinations and showed a low degree of similarity in species composition between the standing vegetation and the soil seed bank in montane mesic grasslands in Argentina. However, the presence of a soil seed bank with high restoration potential and associated high similarity index is a more common phenomena in arid, semiarid and Mediterranean dryland ecosystems (Abella et al. 2020). For example, research by Kinloch and Friedel (2005) investigated impacts of livestock grazing on the soil seed bank in arid calcareous grazing lands in Australia's Northern Territory. They found a Sorensen's similarity index of 60% at sites with a history of several decades of heavy grazing, and a similarity index of 66% at sites with a lighter grazing load (Kinloch and Friedel, 2005).

The most commonly encountered species in the soil seed bank at both sites grazed by cattle and sheep was *Dicerothamnus rhinocerotis* (Asteraceae) (Renosterbos), a shrub that has been reported by some early authors to have increased in cover at sites with a history of continuous heavy grazing (Newton and Knight, 2004; Radloff et al. 2014). It is possible that this may represent a long history of disturbance by livestock grazing at the field sites used in the study, but more research is needed to confirm a definite link between continuous heavy livestock grazing and dominance of *Dicerothamnus rhinocerotis*. It is likely that the lack of statistically significant differences between species richness of different growth forms between grazing treatments is because Overberg Renosterveld vegetation has a long history of livestock grazing and an even longer history of being grazed by game mammals (Skead, 2009). Vegetation such as Overberg Renosterveld with a long history of grazing is likely to have become resilient to the impacts of this land use and key ecological driver (Forbes et al.2018). It is well known that livestock grazing can impact on success of seed production in palatable species, particularly palatable grasses (Tessema et al. 2012). However, sheep are more likely to target a wider range of forage species than cattle, thus increasing the impact of their grazing on the soil seed bank (Esler et al. 2014).

In Kamiesberg Granite Renosterveld, all livestock types including sheep showed a preference for grazing annuals where they are available (Samuels et al. 2016). In many ecosystems worldwide, growth of ruderal species is favoured by increased disturbance from the higher intensity grazing by sheep. For example, Teuber et al. (2013) investigated the impacts of livestock grazing in intermountain depressional prairie wetlands in British Columbia in Canada, correlating increased livestock use with increased abundance of ruderal species. However, in this study of Overberg Renosterveld, there was only one geophyte (a species of *Moraea*) that germinated across all sites. This may be because longevity of geophyte seed survival is different to the seed of other growth forms, or Overberg Renosterveld geophyte species require different growing conditions to those provided, or geophyte seeds produce fewer seeds in comparison to species from other growth forms (Abella et al. 2020).

4.7.2. Arrangement of Species Composition and Growth Form Similarity between Standing Vegetation and the Soil Seed Bank

The lack of clear patterns around the arrangement of species composition across sites is similar to the findings of other studies that have used this analysis in renosterveld vegetation. Cowan and Anderson (2014) examined the vegetation and seed bank of

Peninsula Shale Renosterveld on the lower slopes of Devil's Peak in Cape Town and also found no clear patterns evident through the use of similar analyses. Curtis (2013a) also found no clear patterns in ordinations of the arrangement of species composition when examining diversity and composition of a selection of survey plots in Overberg Renosterveld vegetation. This is attributed to extremely high beta diversity with associated high species turnover across sites (Curtis, 2013a). In contrast to findings of the current study, research by Cousins et al. (2018) in Swartland Shale Renosterveld vegetation found clearer trends with some clustering of similar results, but in this case the comparison was between the standing vegetation and the soil seed bank on north facing vs south facing slopes and smoked vs non smoked soil seed bank experiment trays. The levels of similarity found in this study is in line with findings of the only other study to date that has used the Sorenson's Similarity Index in renosterveld vegetation and seed banks by Cousins et al. (2018) which presented data showing Sorensen's similarity in a range of between 38-44% across burnt and unburnt survey plots in West Coast Renosterveld. It is possible that livestock grazing impact in the more heavily grazed plots surveyed in the current study may have lowered the similarity index overall across all plots surveyed, given that research by Kinloch and Friedel (2005) revealed lower similarity in more heavily grazed plots. This may explain the lower similarity values than those seen in the Cousins et al. (2018) study, but more research is needed to better understand the effects of various ecological drivers on similarity between the standing vegetation and the soil seed bank in Overberg Renosterveld.

In contrast to findings by Cousins et al. (2018), similar research undertaken by Heelemann et al. (2013) showed that the species found in the soil seed bank grouped separately to those in the standing vegetation. Also, in contrast to the findings presented here, research which compared standing vegetation with the composition of the soil seed bank in fynbos vegetation on the Cape Peninsula undertaken by Holmes and Cowling (1997), found low levels of similarity (*IS*s = +/- 25%) between the standing vegetation and the soil seed bank. This is however attributed to the presence of many ephemeral species only found in the soil seed bank, as well as the dominance of serotinous and sprouting species (Holmes and Cowling, 1997). Increased livestock grazing impacts have been shown to lead to lower similarity between standing vegetation and the soil seed bank in Ethiopian savanna vegetation, where sites that were heavily grazed had lower levels of similarity than sites that had been more lightly grazed (Tessema et al. 2012).

The degree of Sorensen's Similarity was varied when calculated by growth form and grazing type. However, there were a few key trends that were worthy of further discussion. Firstly, for both the non-succulent shrubs and the perennial forbs, the degree of similarity was lower for sites with a treatment of no grazing in comparison with sites grazed by cattle and sheep. It is possible that this is because the grazing livestock are contributing to dispersal of seeds of palatable plant species by grazing and then dispersing the seed as part of their dung. This may lead to some species being more consistently represented across the site in the soil seed bank than at sites with a treatment of no grazing. It is likely that historically large herds of game would have dispersed seed of palatable plants in this way in renosterveld vegetation (Shiponeni and Milton, 2006). Where large herbivorous mammals have now been lost from the landscape, it is likely that grazing cattle and sheep contribute to endozoochorous seed dispersal in renosterveld vegetation. Research in other ecosystems has shown that both cattle and sheep contribute significantly to seed dispersal in this way (Cosyns et al. 2005). Where seed dispersal agents are now missing from the ecosystem, it is highly likely that there is a bottleneck to seed dispersal through the vegetation (Pywell et al. 2002). Furthermore, there are higher similarity values for perennial grasses, annuals and succulent shrubs at sites grazed by sheep. It is possible that given that sheep are less selective about which growth forms they target, they may be acting as endozoochorous dispersal agents for these growth forms (Esler et al. 2014). Cattle do not eat all Overberg Renosterveld grasses, because the dominant *Tenaxia stricta* is wiry in texture. Field observations confirm that this species is only targeted by sheep when there is little other palatable vegetation remaining. In addition, at sites with a treatment of no grazing, there was no similarity between the standing vegetation and the soil seed bank. It is likely that this is because grazing livestock are important dispersal agents for ruderal species which make up a considerable proportion of annuals found in heavily grazed renosterveld (Shiponeni and Milton, 2006).

4.5.3. Implications for Renosterveld Management

Research findings presented here have significant implications for management of renosterveld vegetation. The unusually high species and growth form richness found in the soil seed bank in Overberg Renosterveld suggests that renosterveld vegetation has a high potential for restoration through passive restoration. The small and statistically non-significant differences in species richness between grazing treatments for any of the growth forms likely means that a long history of grazing has made renosterveld vegetation a resilient ecosystem to the deleterious effects of livestock grazing. However, these results should be interpreted by managers with caution, as vegetation from mediterranean type shrublands responds in different ways to livestock grazing often as a result of varying climate and regular drought (Hobbs et al. 2008). Furthermore, with the majority of livestock in the Overberg fenced into small areas, effects of increased grazing intensity are likely to be amplified as plant and animal resources become more closely coupled (Baker and Hoffman, 2006). It is therefore recommended that landowners adopt a strategy of adaptive management with associated close monitoring of the impacts of livestock grazing on renosterveld patches on their agricultural land (Milton, 2007).

4.5.4. Directions for Future Research

Findings from this research have demonstrated that the soil seed bank from Overberg Renosterveld is diverse, containing a significant proportion of taxa present in the standing vegetation. The potential to restore renosterveld vegetation degraded by continuous heavy grazing should therefore be an area for future research as this is the first study of its kind within this vegetation. To facilitate ecosystem restoration from seed, further research should be undertaken into seed biology and mechanisms for germination in Overberg Renosterveld, as well as the longevity of seed in the soil seed bank. This should tie in with further research into fire ecology of Overberg Renosterveld, such as documenting reseeding/resprouter post fire responses and investigating which plant taxa require smoke as a germination trigger (Cousins et al. 2018). It is possible that the experimental protocols used in this study were not suitable to trigger germination of taxa that have germination triggered by higher temperatures from fire such as members of the Fabaceae and Malvaceae families (Smykal et al. 2014). Heat shock pre-treatments of seed is a subject that has been examined in fynbos vegetation, but further research is needed to better understand this to improve germination in plant taxa found in renosterveld vegetation (Hall et al. 2017; Cousins et al. 2018; Smiedel et al. 2021). Another particular focus should be on developing germination and restoration protocols for geophytes in Overberg Renosterveld, given that they were highly underrepresented in the study and form a significant component of the diversity of this highly biodiverse mediterranean type shrubland. This is a neglected area that should also be targeted for future applied empirical research.

Chapter 5: The effect of fenced exclosures on vegetation recovery at sites grazed by cattle and sheep in Overberg Renosterveld vegetation

5.1. Introduction

The renosterveld of South Africa’s Overberg is one of the world’s most species diverse mediterranean type shrublands (Curtis et al. 2013). Historically, renosterveld would have covered as much as 30% of the Fynbos Biome (Cowan et al. 2019). However, its occurrence on fertile soils has meant that the majority of this Critically Endangered vegetation has now been ploughed up for agriculture (Heelemann et al. 2013; Topp and Loos, 2019). Renosterveld is now one of the most threatened vegetation types in South Africa (Heelemann et al. 2013) and contains the highest concentration of threatened plant species (Topp and Loos, 2019). Renosterveld fragments are often within areas of agricultural land and are often subjected to the same management interventions and impacts as the rest of the area (Curtis, 2013a). This can lead to overgrazing by agricultural livestock (Curtis, 2013a; Kemper et al. 1999), which in turn can result in habitat degradation that will take a long time or may be impossible to reverse (Curtis, 2013a; Esler et al. 2014).

To date there has only been one unpublished study undertaken by Curtis (2013a) on the effect of grazing through the use of fenced camps in Overberg Renosterveld. Findings of this study showed that livestock grazing did not have a significant effect on vegetation cover and species diversity, but palatable species such as *Themeda triandra* (Poaceae) (Rooigras) showed a slight reduction in size and successful flowering and seed set in response to livestock grazing. However, the study by Curtis (2013a) had limited replications and grazing pressure. It is therefore important that more research should be undertaken to better understand the role of both livestock grazing and cessation of livestock grazing over time on plant species richness, plant species diversity and growth form diversity in Overberg Renosterveld. This should be done with a higher number of replicates encompassing grazing by both cattle and sheep to draw the most meaningful conclusions and to better inform restoration and conservation practice.

The majority of experimental research into renosterveld restoration has largely focused on old lands which represent patches of former renosterveld vegetation that has been degraded by ploughing. Despite the low restoration potential of old lands, attempts at restoration have thus far had limited success (Heelemann et al. 2013). At present there is a lack of suitable measures for the restoration of degraded renosterveld (Topp and Loos, 2019). Given the vulnerability of the remaining Overberg Renosterveld vegetation to threats such as illegal ploughing and overgrazing, it is important that research be undertaken to better understand the role of ecological drivers such as livestock grazing on the vegetation over longer timescales. This can then feed into management planning to achieve conservation goals of best management practices. It is common knowledge that ploughing of renosterveld vegetation for agriculture alters the soil structure and destroys the majority of the soil seed bank (Saayman and Botha, 2008; Heelemann et al. 2013). Given the limited success of most restoration initiatives in renosterveld thus far (Midoko-Iponga et al. 2005), undertaking pilot studies to test restoration protocols prior to launching large scale restoration initiatives has been strongly recommended (Heelemann et al. 2013).

One of the key questions for which answers are needed around grazing in Overberg Renosterveld is whether renosterveld that has been subjected to continuous heavy grazing represents an ecological threshold being crossed. An ecological threshold is described as an abrupt change in ecological properties within the system. In the context of rangeland management, this implies that natural recovery from this form of ecosystem degradation is difficult or impossible without additional restoration interventions (Bestelmeyer, 2006). This leads to the question of what interventions are necessary to restore renosterveld that has been degraded by continuous heavy grazing. Is it possible to reverse the process of ecological degradation through the process of passive restoration, defined as the process of restoring through the removal of the processes that cause degradation (McIver and Starr, 2001)? Another question is whether renosterveld that is ecologically intact and renosterveld that has been subjected to continuous heavy grazing represent two alternate stable states (Augustine et al. 1998; Swift and Hannon, 2010). The current study investigated some of these key questions concerned with the restoration of renosterveld which has been degraded by livestock grazing. This was undertaken through the use of grazing exclosures with paired inside/outside plots at sites with a treatment of no grazing, grazing by cattle and grazing by sheep. The sites were located across the Overberg region encompassing Western, Central and Eastern Rûens Shale Renosterveld. Monitoring and ecological data collection were undertaken over five years.

5.2. Objective and Hypothesis

The objective of the study was to investigate the effects of passive restoration. This was done through the use of fenced exclosures and by an assessment of the change in vegetation cover, species richness, species diversity and growth form diversity over time at sites grazed by cattle and sheep in Overberg Renosterveld.

It was hypothesised that plots with a treatment of no grazing would exhibit an increase over time in plant species richness and diversity, vegetation cover and growth form diversity in comparison with plots grazed by cattle and plots grazed by sheep.

5.3. Materials and Methods

5.3.1. Study Site

This study was undertaken in Overberg Renosterveld vegetation of the Rûens, found in South Africa's Overberg region. The study area (Figure 1.1) extended from the Van De Stel's Pass north of Botrivier eastwards to the Heidelberg District and Suurbraak area.

5.3.2. Research Approach

Twenty field sites were selected from across the area. The sites were distributed across Western (5), Central (4) and Eastern (11) Rûens Shale Renosterveld vegetation types (Rebelo et al. 2006). Sites were selected that had either not been grazed or had been grazed by either sheep or cattle. Sampling was restricted to the shale geology renosterveld types only. Quartz patches within Eastern Rûens Shale Renosterveld and the vegetation type Rûens Silcrete Renosterveld were excluded. This was to minimise effects of soil variation on the results. Sampling on old cultivated lands was also avoided.

At each of the twenty sites, two paired adjacent 10x10 m plots were marked out and the corners delimited with metal droppers. GPS coordinates were taken to ensure easy relocation of plots for future data collection. During the first spring season of the study (August-September 2014), a baseline dataset was collected for all paired plots. This encompassed data on plant species richness and percentage cover, with each species recorded being assigned a growth form. The growth forms included succulent shrubs, non-succulent shrubs, perennial grasses, annual grasses, perennial forbs, annual forbs and geophytes (Anderson and Hoffman, 2011). Following the initial baseline data collection, 11×11 m fenced grazing exclosures were erected around one plot from each set of paired plots at each site. This left a 1 m buffer zone around each 10×10 m plot within the fence to ensure that livestock could not reach far enough to graze the plots from which all grazing was excluded. Each fenced exclosure had a gate in the corner for access. Following the construction of the fenced exclosures, in each spring season (August to September 2015-2018), a repeat follow up dataset using the same protocols was collected from each set of paired plots where one plot was inside the fenced exclosure (a grazing treatment of no grazing (none) and the other adjacent outside the fence. Throughout the remainder of the manuscript, the notation T0 will be used to refer to the dataset collected in spring (August to September) 2014, T1 for spring 2015 data, T2 for spring 2016, T3 for spring 2017 and T4 for spring 2018.

Four soil samples were collected from the plot inside the fenced exclosure and four from the plot outside the fenced camp. These soil samples were collected to a depth of 10 cm using a soil auger or builders' trowel after Sternberg et al. (2003). The soil samples were stored in ziplock bags and labelled, air dried in the laboratory to constant weight, sieved through a 2 mm sieve to remove stones and large particles of organic material. The samples were sent to Bemlab (Pty) Ltd. (Somerset West, South Africa) for analysis of soil texture, pH, resistance, Bray II extractable phosphorous (P Bray II), sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), sulphur (S), carbon (C), and nitrogen following the protocol by the non-affiliated Soil Analyses Work Committee, (1990). Soil texture was analysed first through chemical dispersion using sodium hexametaphosphate. The three fractions of sand were then determined by sieving. Silt and clay were then analysed by examining sedimentation rates at 20°C with a ASTM E100 (152H-TP) hydrometer. Bray II extractable phosphorus (P Bray II) was analysed using 95% ammonium fluoride and 32% hydrochloric acid. Sodium (Na), potassium (K), calcium (Ca), magnesium (mg) and nitrogen (N) were extracted at pH>7 using 0.2 M ammonium acetate. Sulphur (S) was extracted using concentrated phosphoric acid (at pH = 4) and organic carbon (C) was extracted using the Walkley-Black method (The non-affiliated Soil Analyses Work Committee, 1990). The extracted solutions were then analysed with a Varian ICP-OES optical emission spectrometer.

Soil samples were collected and analysed for concentration of nutrient elements to assess whether the soil environment itself could be a confounding variable that may influence differences between grazing treatments.

5.3.3. Data Analysis

All statistical analyses were performed using the R statistical environment (R Version 3.5.2) (R Core Team, 2019). All plots were generated using the package ggplot2 (Wickham, 2016) unless otherwise stated. The package lme4 (Bates et al. 2015) was used for constructing all Linear Mixed Models in this chapter. The R package multicomp (Hothorn et al. 2008) was used for undertaking the Tukey HSD post hoc tests. Species richness and diversity using the Simpson and Shannon-Wiener diversity indices were calculated in Excel. A Linear Mixed Model followed by Tukey HSD post hoc tests was run to determine whether species richness and diversity values were statistically significantly different between different time steps and grazing treatments. A Linear Mixed Effects Model was also used to determine whether there was a significant effect of time and grazing treatment on total percentage vegetation cover and relative growth form cover for the two dominant growth forms: Perennial grasses and non-succulent shrubs. Site was used as a random effect in the model. Models were validated by checking the dispersion of residuals and testing whether they follow a normal distribution. For the soil dataset, a Linear Mixed Effects Model, followed by Tukey post hoc HSD tests was used to determine whether soil properties differ statistically between grazing treatments and sites.

5.4. Results

5.4.1. Species Richness and Diversity

Results from a Linear Mixed Model (Table 5.1) showed that richness values between the grazing treatments were significantly different ($p<0.01$) at all time steps during the study. The significant differences were primarily as a result of the comparisons between sites with a treatment of no grazing and sites grazed by sheep (Appendix 1, Table S3.1). Simpson's

Diversity Index values were not significantly different between grazing treatments from time steps T0 to T2 but were significant ($p<0.05$) for T3 and T4. However, Shannon-Wiener Diversity Index values were not significantly different between grazing treatments at T0 but were significant at T1 and T2 at the $p<0.01$ level, as well as time steps T3 and T4 at the $p<0.001$ level.

The highest species richness was recorded at sites with a treatment of no grazing. These were plots within the fenced exclosures after the T0 baseline datasets were collected. Mean species richness in these plots increased from 20.6 species to 25.4 species (Table 5.1) between T0 and T4. There was a decrease in species richness during T3, likely as a result of drought during that year (2017). Plots that were grazed by cattle had a lower species richness than plots with a treatment of no grazing but had a higher species richness than plots that were grazed by sheep. Mean species richness in plots grazed by cattle started at 17.1 species at T0 and fluctuated slightly during the study duration, decreasing slightly to a mean value of 15.8 species at the end of the study. Sites grazed by sheep had the lowest minimum mean values, fluctuating between 12.6 species and 17 species during the study. However, the maximum mean value of 17 species was higher than the maximum mean value at sites grazed by cattle which reached a value of 15.8 species.

Mean species diversity values followed similar trends throughout the duration of the study, both with the Simpsons and Shannon-Wiener diversity indices (Table 5.1). Mean species diversity with the Simpsons Diversity Index increased in plots with a treatment of no grazing from 0.688 to 0.746, while the Shannon-Wiener Diversity Index increased from 1.767 to 1.946 at the end of the study. Species diversity in plots grazed by sheep increased during the study both with the Simpsons and Shannon-Wiener diversity indices. The Simpsons Diversity Index for sites grazed by sheep increased from 0.598 to 0.702 and for the Shannon-Wiener Diversity Index increased from 1.469 to 1.673.

Table 5.1: Mean and standard deviation values of variation in species richness and diversity (Simpson's and Shannon-Wiener diversity indices) by grazing treatment and over time. F-values are shown with level of significance (* = < 0.05, ** = <0.01, * = <0.001). Dissimilar superscripts denote significant differences between grazing treatment and time step at p<0.05.**

	T0	T1	T2	T3	T4
Richness					
None	20.6 ± 8.3^{a}	22.0 ± 8.1^{a}	22.0 ± 8.3^{a}	19.6 ± 6.7^{a}	25.4 ± 9.8^{a}
Cattle	17.1 ± 5.2^{ab}	16.4 ± 5.2^{b}	15.6 ± 4.4^{ab}	15.0 ± 4.6^{ab}	15.8 ± 4.5^{ab}
Sheep	15.4 ± 4.9^{b}	14.6 ± 6.1^{b}	14.7 ± 8.2^{b}	12.6 ± 5.7^{b}	16.1 ± 7.1^{b}
F-Statistic	6.79**	12.8***	18.1***	19.2***	18.8***
Simpsons					
None	0.69 ± 0.18	0.71 ± 0.18	0.71 ± 0.16	0.73 ± 0.16^{a}	0.75 ± 0.14^{a}
Cattle	0.62 ± 0.24	0.64 ± 0.21	0.63 ± 0.20	0.64 ± 0.20^{b}	0.64 ± 0.20^{b}
Sheep	0.60 ± 0.20	0.61 ± 0.18	0.61 ± 0.21	0.69 ± 0.14^{ab}	0.70 ± 0.14^{ab}
F-Statistic	1.56ns	2.12ns	3.30ns	4.28*	4.31*
Shannon-Wiener					
None	1.77 ± 0.55	1.82 ± 0.54^{a}	1.81 ± 0.52^{a}	1.82 ± 0.48^{a}	1.95 ± 0.47^{a}
Cattle	1.56 ± 0.60	1.54 ± 0.54^{ab}	1.52 ± 0.52^{b}	1.53 ± 0.50^{b}	1.54 ± 0.48^{b}
Sheep	1.47 ± 0.56	1.46 ± 0.48^{b}	1.44 ± 0.60^{b}	1.58 ± 0.43^{b}	1.67 ± 0.46^{b}
F-Statistic	2.61ns	5.71**	8.00*	13.60***	14.1***

5.4.2. Changes in total vegetation cover by treatment over time

The changes in vegetation cover over time under the grazing treatments of no grazing (none), cattle and sheep are shown in Figure 5.1. There was a steady increase in vegetation cover over time for sites that were not grazed. Mean vegetation cover for these sites increased from 71% at T0 to 120% at T4. This increase was significant at the p<0.001 level. There was also a slight increase in total vegetation cover for sites grazed by cattle, with an increase in mean total vegetation cover from 64% at T0 to 89% at T4. However, this increase was not statistically significant. Sites grazed by sheep had a decrease (p<0.01) in mean total vegetation cover over time from T0 to T3 from 75% to 50%.

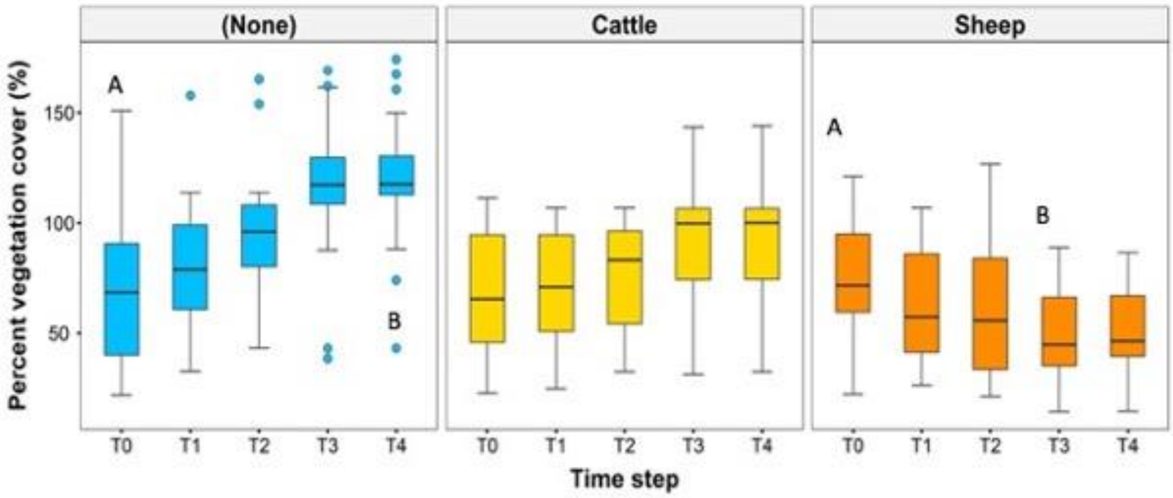

Figure 5.1: Box and whisker plots showing variation and change in percentage vegetation cover over time at sites with treatments of no grazing, grazing by cattle and grazing by sheep. Dissimilar letters denote significant differences between values at p<0.05.

Results from a Linear Mixed Model showed highly significant effects at the <0.001 level for time step (n=4, F=6.4219, p<0.001), grazing treatment (F=30.5394, p<0.001) and the interaction between time step and grazing treatment (F=5.6962, p<0.001) on total percentage vegetation cover. Results from a Linear Mixed Model followed by post hoc tests (Appendix 1, Table S1.2) which examined the significance of the change in total vegetation cover with grazing treatment between time steps revealed that between sites with a treatment of no grazing and sites grazed by cattle, there was a non-significant result between T0 and T3. However, there was a significant (p<0.05) difference in vegetation cover between no grazing and cattle grazed at T4. The differences in vegetation cover between sheep grazed and plots with a treatment of no grazing were not significantly different at T0 and T1 but were significantly different (p<0.001) for the time steps T2 to T4. Vegetation cover was not significantly different between cattle and sheep grazed plots for T0 and T2 but were significantly different for T3 (p<0.01) and T4 (p<0.05).

T0

	None	Cattle	Sheep
None		ns	ns
Cattle	ns		ns
Sheep	ns	ns	

T1

	None	Cattle	Sheep
None		ns	ns
Cattle	ns		ns
Sheep	ns	ns	

T2

	None	Cattle	Sheep
None		ns	***
Cattle	ns		ns
Sheep	***	ns	

T3

	None	Cattle	Sheep
None		ns	***
Cattle	ns		**
Sheep	***	**	

T4

	None	Cattle	Sheep
None		*	***
Cattle	*		*
Sheep	***	*	

Figure 5.2: Results of post-hoc Tukey HSD tests to determine the differences in total vegetation cover between treatments at each time step (Significance: * p<0.05, ** p<0.01, *** p<0.001).

5.4.3. Variation in cover of different growth forms over time

The changes in the cover of different growth forms over time under the grazing treatments of no grazing, grazed by cattle and grazed by sheep are shown in Figure 5.3. The cover of annuals, geophytes and perennial forbs was low in all plots under all treatments and was never more than 6% (Figure 5.3 A,B,C). Mean cover of perennial grasses increased over time in plots that were not grazed from 18% at T0 to 26% at T4. Sites grazed by cattle showed little difference in the cover of perennial grasses over time. However, sites grazed by sheep exhibited a decrease in mean perennial grass cover over time from 22% at T0 to 12% at T4 (Figure 5.3. D). Cover of succulent shrubs was highest in plots that were not grazed (Fig 5.3. E). Results from a Generalised Linear Mixed Effects Model examined the significance of the change in cover of the two dominant growth forms (perennial grass and non-succulent shrubs) with grazing treatment over time. The model revealed no significant difference for perennial grass cover for time step ($F_{4,2}$=0.0252, p>0.05) but grazing type was highly significantly different (n=3, F=17.5334, p<0.001). The interaction between time step and grazing type was also significant (F=3.0775, p<0.01). Non-succulent shrub cover was highly significantly different for both time step (n=4, F=9.5013, p<0.001) and grazing type (n=3, F=15.9275, p<0.001) (Fig 5.3F). The interaction between time step and grazing type was not significant (F=1.9209, p>0.05) for non-succulent shrubs (Figure 5.3. F).

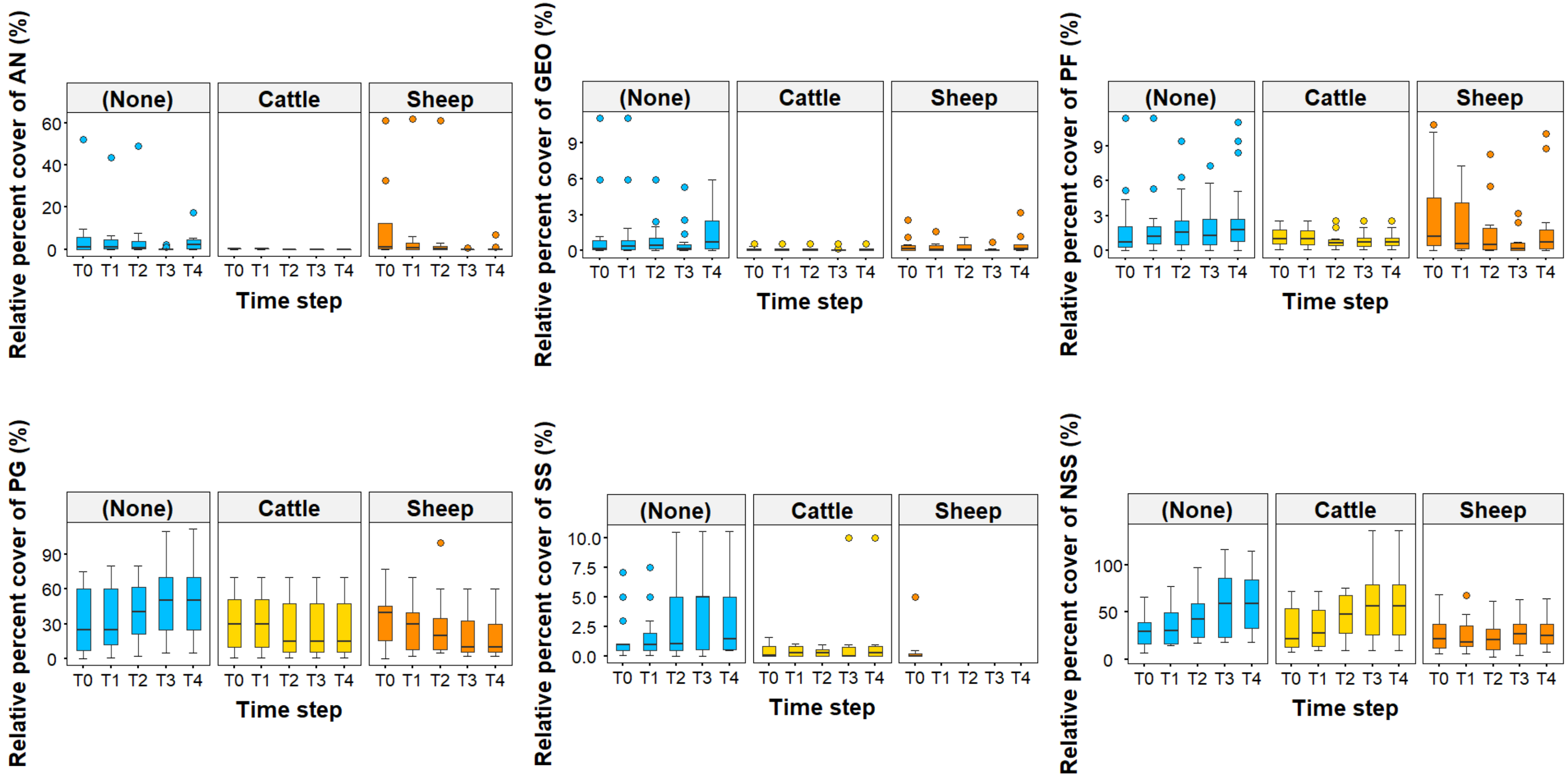

Figure 5.3: Box and whisker plots showing variation and change over time of vegetation cover by growth form with different grazing treatments of no grazing, grazing by cattle and grazing by sheep (AN = Annuals, GEO = Geophytes, PF = Perennial Forbs, PG = Perennial Grasses, SS = Succulent Shrubs, NSS = Non-Succulent Shrubs).

5.4.4. Soil Properties

Results of a Linear Mixed Model found no significant differences in all soil chemical properties apart from pH. Results for pH were found to be significant at the <0.01 level ($F=5.592$, $p=0.005$, Table 5.2). Plots that were not grazed had a higher pH (4.75) than plots grazed by cattle (4.56). Plots grazed by sheep had a higher pH (4.68) than plots grazed by cattle and plots with a treatment of no grazing.

Table 5.2: Table showing mean and standard deviation values of variation in soil properties across sites from soil samples collected from inside and outside twenty grazing exclosure plots. Dissimilar superscripts denote significant differences between values at p<0.05.

Variable	Units	None	Cattle	Sheep
pH (KCl)		4.75 ± 0.33^{a}	4.56 ± 0.41^{b}	4.68 ± 0.35^{c}
N	mg/g	0.86 ± 0.62	0.85 ± 0.58	0.78 ± 0.58
P Bray II	mg/kg	9.83 ± 4.44	10.01 ± 3.30	8.04 ± 3.53
Na	cmol/kg	0.58 ± 0.44	0.54 ± 0.33	0.69 ± 0.62
K	cmol/kg	0.34 ± 0.21	0.34 ± 0.25	0.32 ± 0.15
Ca	cmol/kg	3.66 ± 1.63	3.34 ± 1.19	3.15 ± 1.25
Mg	cmol/kg	3.34 ± 0.81	3.41 ± 0.88	3.37 ± 1.03
T-Value	-	9.19 ± 2.15	9.27 ± 1.99	9.14 ± 2.25
C	%	2.40 ± 0.36	2.46 ± 0.53	2.35 ± 0.32

5.5. Discussion

5.5.1. Effects of Fenced Exclosures Over Time on Plant Species Richness and Diversity

The results presented in this study are highly promising, suggesting that resting renosterveld (passive restoration) following continuous grazing is a potentially effective tool in improving species richness and diversity at sites that have previously shown reduced richness and diversity after livestock grazing. Renosterveld vegetation in good condition with optimum levels of species richness and diversity is more resilient to other threats such as climate change. These results also show that time is a significant factor with regards to cessation of livestock grazing allowing recovery of renosterveld after continuous heavy grazing. The more time that the renosterveld vegetation is allowed to rest, the better the natural succession and recovery as indicated by increasing species richness and diversity across field sites.

However, given that there have been few exclosure-based studies undertaken in Overberg Renosterveld to date, it is important to build upon these findings with additional research. To date, most research which uses exclosures to investigate the effect of grazing has been undertaken in North America (Stohlgren et al. 1999). Most previous studies were also carried out at a small scale, using small plots and only a few replicates as a result of the cost of building grazing exclosures (Frank et al. 1995; Lavado et al. 1996). In contrast, the current study reported on here has used a greater number of paired plots than many earlier studies across multiple grazing treatments and vegetation types, suggesting that more meaningful conclusions can be drawn around vegetation change over time in response to livestock grazing. However, a key exception to this is the work of Courtois et al. (2004), which examined vegetation changes between inside and outside a set of grazing exclosures known as the Nevada Plots, which were built in 1937 following the Taylor Grazing Act to monitor the long-term effects of livestock grazing on the Nevada rangelands. Sixteen plots remained intact and were used in the study after being fenced for 65 years prior to the study being undertaken. The research findings by Courtois et al. (2004) revealed no difference in species richness between grazed and ungrazed plots which contrasts with the findings of the current study. Furthermore, plant community diversity was also found to be similar at 11 of the 16 sites surveyed during the study (Courtois et al. 2004). In contrast to the findings of Courtois et al. (2004), research using grazing exclosures undertaken by Bradd-Witt et al. (2011) in the mulga (*Acacia aneura*) lands bioregion in southwest Queensland, Australia reported significant ecological changes over time, including higher species richness inside the grazing exclosures. In line with the results of the current study, the use of fenced exclosures as a tool for passive restoration is effective for driving an increase in species richness over time. However, Stohlgren et al. (1999) used multi-scaled plots to examine vascular plant species richness and diversity inside and adjacent to fenced exclosures in grassland vegetation in the Rocky Mountains. This research revealed that at the 1 m^2 scale there was significantly lower species richness in sites inside grazing exclosures in comparison to adjacent sites outside, but in contrast at the 1000 m^2 scale there were no significant differences in species richness and diversity between grazed and ungrazed plots (Stohlgren et al. 1999). This serves to highlight the importance of care in interpreting the findings from small exclosures and extrapolating to the landscape scale (Stohlgren et al. 1999). Virtually all these grazing exclosure studies also use sites that have been grazed both by different domestic livestock as well as by grazing wildlife such as elk and kangaroo. These datasets were also not analysed separately to allow for the examination of the impact of different domestic livestock breeds in comparison to grazing by local wildlife.

The work of the Arid Karoo Desert Stock Rate Trial (Van Der Merwe et al. 2018) further reveals that stocking levels of livestock are a key factor in vegetation change over time as a result of livestock grazing. This work is one of few studies that have been undertaken in

South Africa to examine the effects of livestock grazing on vegetation over longer timescales. In addition, unlike many other South African studies that rely on fenceline contrasts, the Arid Karoo Desert Stock Rate Trial used fenced paddocks that are considerably larger than those used in the majority of studies cited in this research. The Arid Karoo Desert Stock Rate Trial took place at the Carnarvon Experimental Station and was initiated in 1988 (Van Der Merwe et al. 2018). Four stocking rate treatments were assigned randomly to multiple blocks grazed by sheep and the study took place over 27 years. Data was collected on vegetation composition, abundance of different plant functional groups, abundance of dominant species as well as plant diversity (Van Der Merwe et al. 2018). Stocking rate and time was found to have significant effects on species composition. Greater stocking rates resulted in lower total plant cover, palatable and non-palatable shrub cover, perennial grass cover and annual herbaceous species (Van Der Merwe et al. 2018). Findings by Van Der Merwe et al (2018) highlight the importance of long-term studies in revealing important biodiversity changes in response to grazing that may only be visible over longer timescales. This is especially important in arid and semi-arid or Mediterranean ecosystems where changes may take considerable time to manifest themselves.

The work of Rutherford and Powrie (2013) takes a slightly different approach to the current study and the others presented in this section and examines the effects of grazing on vegetation by grazing intensity. The study was undertaken with the aim of better informing the compilation of the South African National Biodiversity Assessments (Rutherford and Powrie, 2013). However, it only encompassed what the authors describe as South Africa's 'rangeland biomes', including the Succulent Karoo, Nama-Karoo, grassland and thicket but excluded renosterveld vegetation despite its use for and exposure to livestock grazing in several parts of the country. There was also no temporal element or distinction by livestock type in the work of Rutherford and Powrie (2013) and their findings, therefore, only represent a 'snapshot' of general grazing impacts rather than analysing vegetation change by livestock type. The findings of Rutherford and Powrie (2013) showed highly varied responses of species richness and diversity in response to livestock grazing and were thought to be driven as much by climatic variability as by differing grazing intensities. The work of Van Der Merwe et al. (2018) and Rutherford and Powrie (2013) highlight the importance of examining climatic variability, stocking rates and grazing intensity when considering studies that build on the findings of the current report. With most studies on livestock grazing in South Africa having been undertaken in more arid systems, more research is urgently needed to better understand the impacts of livestock grazing by cattle and sheep on grazing in renosterveld vegetation.

5.5.2. Effects of Fenced Exclosures Over Time on Growth Form Diversity and Cover

Results presented as part of this study are in agreement with results by Bradd-Witt et al. (2011). As was the case with species richness and diversity, results from other grazing exclosure studies provided highly varied results both similar and in contrast to the findings reported in the current study. Hoffman et al. (2009) investigated the effect of grazing exclosures on the vegetation of Sanbona Wildlife Reserve in the Little Karoo over a similar timescale as the current study. The focus of their study was on the impact of indigenous grazing ungulates rather than domestic livestock. Their findings revealed no significant differences between fenced and open plots both for species richness and for vegetation cover (Hoffman et al. 2009). These findings were attributed to low stocking levels of game throughout the duration of the study, supporting the assertion that light to moderate grazing has little impact on vegetation cover and growth form diversity in comparison to heavy grazing as seen at several sites used in this research (Anderson and Hoffman, 2007).

Several authors have commented that grazing by sheep has a greater impact on the veld than grazing by cattle (Landsberg et al. 2003; Esler et al. 2014; Van Oudtshoorn, 2019). Sheep are mixed feeders with digestive systems that allow them to digest grasses as well as other vegetation and they are thus able to consume a wide variety of different forage species in comparison to cattle. This therefore means that their grazing has a greater impact on vegetation cover and growth form diversity in the veld (Esler et al. 2014). It is also important to note that species selection changes when sheep are present at higher stocking rates, as they tend to be less selective of grasses and will graze plants across a wider variety of different growth forms (Samuels et al. 2016). Grazing by sheep is also highly damaging to natural rangelands because of their ability to graze very close to ground level, thus damaging the growing points of plants (Van Oudtshoorn, 2019). Landsberg et al. (2003) examined vegetation change along gradients with increasing distance from watering points in an Australian chenopod shrubland and *Acacia* woodland. Bare ground decreased and grass cover increased with increasing distance from water points (Landsberg et al. 2003). Similar findings were also reported by Todd (2006) in Nama-Karoo vegetation. These findings are in line with findings from the current study that showed that areas with no grazing had higher vegetation cover and increased perennial grass cover than sites grazed by sheep.

5.5.3. Implications for Renosterveld Management

Results from this study have highlighted the benefits and importance of resting renosterveld vegetation as a key component of livestock grazing management, with findings showing significant increase of species richness, diversity and vegetation cover over time. Overgrazing is often a function of time rather than stocking levels (Van Oudtshoorn, 2019). It is crucial to allow renosterveld vegetation to rest by completely removing grazing livestock during the growing season to allow forage plants to flower and set seed (Curtis, 2013a; Van Oudtshoorn, 2019). The findings of the current study also highlight the benefit of fencing renosterveld separately from grain or pasture fields to reduce the likelihood of overgrazing. Passive restoration through fencing has the potential to be used in many cases to increase species richness, diversity and vegetation cover over time at sites where overgrazing has become a problem (Bradd-Witt et al. 2011).

5.5.4. Directions for Future Research

Results from this study have demonstrated that livestock grazing by cattle and particularly by sheep has a significant impact on renosterveld vegetation over time. Furthermore, this research has revealed that passive restoration through use of fenced camps is an effective way of increasing species richness, species diversity and vegetation cover in Overberg Renosterveld over time. However, this study only had a duration of five years which is acknowledged to be a short time in the context of vegetation change in response to both grazing and cessation of grazing (Fleischner, 1994). Further research using similar research protocols or controlled grazing trials over longer timescales is important to attain a fuller picture of the effects of livestock grazing and resting in this vegetation. Furthermore, the impacts of livestock grazing can vary according to scale and the use of only 10x10 m plots within 11x11 m exclosures was a limitation in this study as other reports using grazing exclosures have found different results using larger exclosures (Courtois et al. 2004). Future research could use larger exclosures and a multiscale plot treatment design such as that undertaken by Stohlgren et al. (1999). In addition, livestock grazing is inherently variable, and fenced exclosures can often attract grazing livestock, thus increasing grazing intensity immediately adjacent to the exclosure fences (Landsberg et al. 2003). It is therefore recommended when undertaking similar studies to this one in the future to add to the methodology a third survey plot at each site, at a randomly selected location away from the exclosure to mitigate this problem (Stohlgren et al. 1999).

Chapter 6: Synthesis: Implications for Renosterveld Restoration, Management and Conservation and Directions for Further Research

6.1. Research Synthesis

Results shed light on several key components of the rangeland and restoration ecology of Overberg Renosterveld. This study set out to better understand the impacts of livestock grazing by cattle and sheep on plant species richness, plant species diversity and growth form diversity in Western, Central and Eastern Rûens Shale Renosterveld. Results from this component of the study have created a baseline of data on the impacts of grazing livestock on Overberg Renosterveld that can be built on by future research. Sites with a treatment of no grazing were found to have higher species richness and vegetation cover of non-succulent shrubs, annual forbs and perennial forbs than at sites grazed by sheep. Findings indicate that livestock grazing by sheep leads to a significant reduction in species richness and diversity in comparison with sites with a treatment of no grazing for the duration of the study. In contrast to a treatment of sheep grazing, livestock grazing by cattle was found to have no significant differences on plant species richness, plant species diversity and growth form richness and diversity in comparison with sites with a treatment of no grazing. It has been acknowledged by several authors that livestock grazing by sheep has a greater negative impact on the veld than grazing by cattle (Landsberg et al. 2003; Van Oudtshoorn, 2019). Several possible reasons have been proposed to account for this difference in grazing impacts. As mixed feeders, sheep can consume and digest a much wider variety of forage, including both leaves and grasses, in comparison to cattle (Esler et al. 2014). In addition, when sheep are present at higher stocking rates, which is commonplace in Overberg Renosterveld rangelands, they tend to be less likely to target grasses and instead will graze across a significant variety of different growth forms (Samuels et al. 2016). In contrast, cattle tend to be bulk grazers, with their diets comprising around 70% grasses, 10% herbaceous species and only 20% browse (Esler et al. 2014). They therefore have a less significant impact on the shrubby component of the vegetation (Esler et al. 2014).

Analysis of soil samples found no significant differences in soil nutrient elements with the exception of soil resistance, thus indicating that edaphic variation did not contribute significantly to explaining the differences in vegetation documented between different sites and grazing treatments. Furthermore, results from the Modified Whittaker plots also showed that sites grazed by sheep had significantly lower species richness of geophytes than at sites with a treatment of no grazing. Livestock grazing, therefore, appears to have a negative impact on geophyte species richness in renosterveld vegetation. Given the significant geophyte species diversity in Overberg Renosterveld and that many of these species are of conservation concern, it is important that caution be exercised with sheep stocking levels and that this be considered with regards to management interventions around conservation of the renosterveld fragments that still survive.

Results are also presented from the current study on the germinable seed bank of Overberg Renosterveld. This study examined the composition of the germinable seed bank in Western, Central and Eastern Rûens Shale Renosterveld, as well as examining the effects of livestock grazing by cattle and sheep in comparison with a treatment of no grazing on the soil seed bank. There has been limited research undertaken on livestock grazing in renosterveld vegetation in the past, with the majority of these studies focused only on the impacts of grazing on the standing vegetation, for example Curtis (2013a). Previous published renosterveld soil seed bank studies have focused on Swartland Renosterveld vegetation types. The first was undertaken by Heelemann et al. (2013), with further research done by Cousins et al. (2018), focusing on abandoned agricultural fields and fire ecology respectively. Results of the current study have revealed that Western, Central and Eastern Rûens Shale Renosterveld have well-developed seed banks, comprising seeds of predominantly indigenous species from a variety of different growth forms. Presence of alien species in the soil seed bank was limited in comparison with the indigenous seed bank component. It is uncommon at the global scale for vegetation to have such a well-developed seed bank comprising a suite of species that can be selected to be restoration target species for Overberg Renosterveld vegetation. However, this is something that is more common in dryland arid, semiarid and Mediterranean type vegetation types (Abella et al. 2020). There were no statistically significant differences found in the species richness of the soil seed bank between different grazing treatments. Therefore, although livestock grazing by sheep has been shown to have a significant impact on above-ground species richness and diversity, these impacts have not been shown to be reflected in the soil seed bank by the current study. These findings hold significance for the restoration potential of renosterveld vegetation that has been degraded by continuous heavy grazing. However, there was a limited presence of geophytes found in the germinable seed bank, with only one geophyte individual (*Moraea* spp.) germinating during the glasshouse experiment. This is particularly significant in combination with the key finding that livestock grazing by sheep has a significant effect on geophyte species richness and diversity. Further research is therefore strongly recommended into conservation management and informing the development of restoration protocols for geophytes both in Overberg Renosterveld and other Mediterranean type ecosystems worldwide.

Results from the fenced grazing exclosures in comparison to plots with grazing treatments of grazing by sheep or grazing by cattle showed that sites with a treatment of no grazing for the duration of the four year study exhibited an increase in plant species richness, plant species diversity and vegetation cover over time. The cessation of grazing has allowed natural succession to take place with fast growing pioneer species such as renosterbos (*Dicerothamnus rhinocerotis*) being the most dominant. Furthermore, the accompanying increase in richness and diversity over time suggests that it is unlikely at these sites that grazing has led to ecological thresholds being exceeded and formation of an alternate stable state. Results for the plots outside the grazing exclosures showed lower species richness and diversity for sites grazed by cattle with the lowest values at sites grazed by sheep. These findings show great promise for the use of passive restoration through resting renosterveld from livestock grazing for vegetation recovery. Findings from the plots outside the exclosures further support the results from the Modified Whittaker plots in demonstrating that grazing by sheep negatively impacts on species richness and diversity in Overberg Renosterveld. Furthermore, these results also demonstrate that time is a significant factor in vegetation recovery following removal of grazing livestock. The more time that renosterveld vegetation is given to rest from grazing livestock, the better the results in terms of recovery of plant species richness and diversity. The findings from this research have filled important research gaps across several different fields of renosterveld ecology. To better inform management, conservation and restoration of Overberg Renosterveld, further research is needed to build upon the findings reported in the current study, as well as exploring areas highlighted in the following sections of the thesis.

6.2. Implications for the Conservation and Management of Overberg Renosterveld

Research findings from the current study have shown for the first time that livestock grazing, particularly grazing by sheep, reduces plant species richness, species diversity and vegetation cover in Overberg Renosterveld vegetation. This has significant implications for conservation and management given that the Conservation of Agricultural Resources Act (CARA) (Act 43 of 1983) states that farmers should not exceed the grazing capacity of their veld and should restore any area of veld on their land that is degraded or denuded (Landcare South Africa and National Department of Agriculture, 2002). For the first time, there are now empirical insights, a deeper understanding and increased knowledge that provide the opportunity to act on these progressive guidelines and policies. It is crucial when using renosterveld for livestock grazing, and particularly if grazing sheep, that landowners should err on the side of caution with regards to stocking levels. Any renosterveld being used for grazing should be subjected to regular ongoing monitoring for signs of overgrazing to safeguard veld condition. Findings lend credence to the conservative guidelines in 'Fynbos: Ecology and Management' (Esler et al. 2014) where landowners are advised to maintain at least 50% of palatable renosterveld shrub foliage after livestock grazing. When this threshold has been exceeded, livestock should be removed from the renosterveld to allow resting for a period of at least three months (Esler et al. 2014). This is best achieved by fencing renosterveld fragments separately from camps where livestock are grazed. This allows renosterveld fragments to be managed separately from the neighbouring agricultural land. When combined with care around stocking levels, this can ensure that renosterveld fragments do not become overgrazed (Curtis, 2013a; Esler et al. 2014). However, when livestock are contained within smaller units, there is a far closer coupling between animal and plant resources (Baker and Hoffman, 2006) and overgrazing is far more likely to occur (Hobbs et al. 2008). This makes monitoring renosterveld for signs of overgrazing far more important.

Results from Chapter 3 showed a significantly lower geophyte species richness at sites grazed by sheep in comparison with sites with a treatment of no grazing. This further supports prior recommendations around seasonality of livestock grazing in renosterveld. Grazing renosterveld during winter and spring should be avoided as it is the main growing and flowering season for the majority of geophytes (Curtis, 2013a; Esler et al. 2014). Livestock should also be kept out of the renosterveld for a month or two after the majority of geophytes have finished flowering to allow for seed set and dispersal. While this low geophyte presence in the seed bank is reported in other mediterranean type ecosystems, this is also a key finding of conservation concern. Landowners should be cautious with stocking levels and monitor their renosterveld for signs of overgrazing (Esler et al. 2014).

Findings provide a baseline dataset on the impacts of livestock grazing on Overberg Renosterveld, as well as contributing to an understanding of how livestock grazing impacts on renosterveld vegetation over time. However, there is still little known about this and many other key components of renosterveld ecology and management. Mediterranean climates tend to naturally experience considerable fluctuations over time in average annual temperature and rainfall. These fluctuations are projected to become more significant in the face of changing climate. In the face of this in mediterranean type ecosystems it is crucial to act with caution when implementing any new management interventions, using regular monitoring and associated adaptive management protocols to reduce the likelihood of contributing to any form of ecosystem degradation (Tompkins and Adger, 2004). There is also still little known about the historical vegetation structure and function of Overberg Renosterveld (Rebelo, 1995; Rebelo et al. 2006). It is, therefore, important that managers adopt an adaptive management approach, which reviews management and conservation interventions in line with monitoring data (Milton et al. 2007). Improved assistance to landowners should be offered by allocating additional resources to extension work undertaken by conservation practitioners. This should build on existing programmes such as the Overberg Renosterveld Conservation Trust's conservation easements or CapeNature's stewardship programme. This will help to support landowners, who are the custodians of this highly biodiversity and critically endangered vegetation, with the appropriate conservation and management advice.

6.3. Implications for Restoration of Overberg Renosterveld

Findings from this research have revealed that Overberg Renosterveld has a highly species rich seedbank comprising a variety of different growth forms that could be potential restoration target species. Furthermore, the soil seed bank of the sites sampled were low in ruderal or alien species, further increasing restoration potential of such sites. While passive restoration holds considerable potential for renosterveld, findings have also shown that some key vegetation components such as geophytes would need to be reintroduced or bulked up using active restoration. Findings from the current study also hold significance for the restoration of Overberg Renosterveld. With just 5% of the former extent of Overberg Renosterveld vegetation remaining, most has been transformed for agriculture. The fertile soils of the Overberg are considered prime quality, high value agricultural land. This combined with the low restoration potential of ploughed renosterveld means that once renosterveld has been transformed for agriculture, it is difficult to restore it to the same levels of ecosystem biodiversity and complexity (Heelemann, 2010). Some of the main structural species such as renosterbos (*Dicerothamnus rhinocerotis*) may return or survive one round of ploughing, but old lands where renosterveld has been ploughed should not be considered priorities for restoration interventions.

Although sites that have been ploughed have been shown through soil seed bank research to have low restoration potential (Heelemann, 2010), sites that have been denuded or degraded by continuous heavy grazing are still likely to have intact seed banks. The current study has shown that the use of fenced grazing exclosures to allow for the resting of renosterveld vegetation shows significant increases in plant species richness and diversity as well as vegetation cover over time. This indicates that at sites that show high restoration potential with a diversity of species in the soil seed bank still surviving, it is possible to restore some degree of species richness, species diversity and vegetation cover through use of passive restoration techniques. This is a more cost-effective form of undertaking ecological restoration in Overberg Renosterveld and avoids the need for costly seedling and planting as part of restoration protocols (Abella et al. 2020). Of the 5% of Overberg Renosterveld vegetation that remains, much of the vegetation has been degraded to some degree through continuous heavy grazing. Overgrazing could further reduce the extent of renosterveld without interventions to arrest what may be irreversible ecological degradation (Esler et al. 2014). Before attempts are made to restore old lands with lower restoration potential, renosterveld degraded by continuous heavy grazing should receive support for passive restoration through fencing off patches and resting from grazing livestock.

6.4. Implications for mediterranean type shrublands at the global scale

Mediterranean climate regions contribute to global plant diversity through comprising 20% of the world's plant species on only 5% of the land area worldwide. Some mediterranean type shrublands have biodiversity that is comparable to that of species rich tropical rainforests (Teste et al. 2017). Many mediterranean climate shrublands worldwide are used as rangelands for grazing domestic livestock (Platou and Tueller, 1988). Results from this study are, therefore, of relevance to ecologists and land managers working in other mediterranean type shrublands in other parts of the world. Findings highlight that in these highly biodiverse systems, it is important for management decisions around livestock grazing to be supported by applied ecological research to better understand differing impacts of grazing by different livestock types, as well as that of stocking rates, grazing loads and grazing intensity. Where there is limited understanding supported by ecological research around stocking rates and impacts of grazing then landowners and managers should proceed with caution when making decisions around livestock grazing for agriculture as well as conservation-focused land management. When livestock is grazed within these vegetation types, regular monitoring should be undertaken for signs of overgrazing (Esler et al. 2014). Adaptive management systems should be used to ensure that livestock grazing is in line with best management practices to achieve conservation goals and maintenance of ecosystem services (Milton et al. 2007).

6.5. Key Conclusions

I. In Overberg Renosterveld vegetation grazed by sheep, plant species richness and plant species diversity were found to be lower than at sites with a treatment of no grazing.

II. Sites with a treatment of no grazing for the duration of the study were found to have higher species richness and vegetation cover of non-succulent shrubs, annual forbs and perennial forbs than at sites that were grazed by sheep.

III. Sites grazed by sheep had significantly lower species richness of geophytes than sites with a treatment of no grazing for the duration of the study.

IV. Sites grazed by cattle showed no significant differences in plant species richness, plant species diversity and growth form diversity in comparison to sites with a treatment of no grazing for the duration of the study.

V. The soil seed bank of the Overberg Renosterveld sites surveyed during the study was found to have high species richness, species diversity and growth form diversity, with 48% of species recorded in the standing vegetation also present in the soil seed bank.

VI. These findings indicate that Overberg Renosterveld that has been degraded by continuous heavy grazing by domestic livestock has significant potential for passive restoration interventions to take place.

VII. This is further supported by findings from the fenced exclosures component of the study, which documented a significant increase in plant species richness, plant species diversity and vegetation cover over time for the duration of the study in comparison to sites with a treatment of grazing by cattle or grazing by sheep.

6.6. Directions for Future Research

Findings from the research presented here have offered a significant contribution to a growing body of literature around the conservation and management of Overberg Renosterveld, particularly at the interface of rangeland and restoration ecology. This research has also revealed several key areas where future applied research, to inform renosterveld conservation and management, should be focused. This study has documented empirically for the first time some of the impacts of livestock grazing on Overberg Renosterveld. However, this research has focused on comparing the effects of livestock grazing by cattle and sheep. Future research should focus on the effects of grazing intensity on the vegetation through the compilation of detailed records of the grazing history of the vegetation for the duration of the study, including stocking levels, encompassing all times of year that the field sites are grazed. Future studies should also investigate how the impacts of livestock grazing vary by season and over time. All of these factors could be investigated further through the use of controlled grazing trials (Hester et al. 2000) This would help to shed additional light on the carrying capacity of Overberg Renosterveld, so that more effective guidelines on stocking levels can be provided to landowners, based on empirical research. A multidisciplinary approach using similar protocols to this study but with longer term monitoring could be used to examine changes in impacts of livestock grazing in the context of global change and increasing frequency and intensity of drought events. Further research through the use of interviews and questionnaires could also consider attitudes of landowners towards renosterveld conservation, particularly around willingness of uptake of renosterveld conservation initiatives such as the Overberg Renosterveld Conservation Trust's conservation easement programme.

This research also presented findings of the first soil seed bank study to be undertaken in Overberg Renosterveld. Results found that the germinable soil seed bank comprised a variety of different growth forms, with geophytes highly underrepresented. This could be because the growing conditions for the species in this guild were not adequately met to trigger germination, or that the cold storage of the soil prior to autumn sowing and smoking exceeded the longevity of the seed (Abella et al. 2020). This would, however, require further empirical testing to examine the importance of pre-treatments in germination of renosterveld taxa using similar protocols to Hall et al. (2017). Furthermore, geophyte species richness was found to be lower at sites grazed by sheep in comparison with sites with a treatment of no grazing. Other research examining soil seed banks in mediterranean type shrublands in other parts of the world have also found very low numbers of geophytes present in the germinable seed bank (Valbuena and Trabaud, 2001), so this is not a unique phenomenon observed as part of the current study. Despite the diversity of geophytes in many mediterranean type ecosystems, there is currently little applied research published in the peer reviewed literature with a focus on developing improved restoration protocols for this guild that are specifically for mediterranean type shrubland ecosystems. Other research (e.g., Herault et al. 2005; Abella et al. 2020) indicates that different growing requirements and different protocols for restoration are needed in comparison to those used for restoring non-geophytic plant species. Further research is also needed to develop effective protocols for the restoration of geophyte species where drivers have caused them to be lost from the system. This should comprise studies of seed biology, length of viability and the development of protocols for cultivation and introduction into vegetation as part of restoration work.

The results from the fenced exclosures have shown significant promise for the use of resting from livestock grazing as a tool for passive restoration. Based on the promising findings here, to improve statistical rigour and to attain an understanding of vegetation responses to resting from livestock grazing at different scales, larger plots should be used for future grazing exclosure studies whenever resources permit. This will allow for several replicate plots to be placed within each exclosure to better understand ecological patterns and processes (Lavado et al. 1996; Stohlgren et al. 1999). The results of the grazing exclosure component of the research allowed for greater temporal depth when exploring the impacts of livestock grazing in Overberg Renosterveld. Even so, this research was only undertaken over five years, with a first year collection of baseline data and four annual follow up data collections. While this undoubtedly reveals more than a dataset with no temporal component, the full extent of the effects of grazing are sometimes only evident over decades, such as is shown in the analysis of van der Merwe et al. (2018) of a 27-year data set in the semi-arid Karoo. Further studies should therefore include longer term monitoring of larger permanent plots to better understand vegetation changes and the impacts from livestock grazing over longer timescales, to inform conservation management of this critically endangered vegetation.

References

Abella, S.R. Hodel, J.L. Schetter, T.A. (2020) 'Unusually high-quality seed banks in a Midwestern U.S. Oak Savanna region: Variation with land use history, habitat restoration and soil properties', *Restoration Ecology* (Volume 28): pp. 1100-1112.

Adamson, R.S. Salter, T.M. (1950) *Flora of the Cape Peninsula*, Juta and Co, Cape Town and Johannesburg, South Africa.

Alhamad, M.N. (2006) 'Ecological and species diversity of arid Mediterranean grazing land vegetation', *Journal of Arid Environments* (Volume 66): pp. 698-715.

Alley, N.F. (1977) 'Age and origin of laterite and silcrete duricrusts and their relationship to episodic tectonism in the mid-north of South Australia, *Journal of the Geological Society of Australia* (Volume 24): pp. 107-116.

Alkemade, R. Reid, R.S. van den Berg, M. Leeuw, J. Jeuken, M (2013) 'Assessing the impacts of livestock production on biodiversity in rangeland ecosystems', *Proceedings of the National Academy of Science of the United States of America* (Volume 110): pp. 20900-20905.

Andersen, T. Carstensen, J. Hernandez-Garcia, E. Duarte, C.M. (2008) 'Ecological thresholds and regime shifts: Approaches to identification', *Trends in Ecology and Evolution* (Volume 24): pp. 49-57.

Anderson, P.M.L and Hoffman, M.T (2007) 'The impacts of sustained heavy grazing on plant diversity and composition in lowland and upland habitats across the Kamiesberg Mountain Range in the Succulent Karoo, South Africa', *Journal of Arid Environments* (Volume 70): pp. 686-700.

Anderson, P.M.L and Hoffman, M.T (2011) 'Grazing response in the vegetation communities of the Kamiesberg, South Africa: Adopting a plant functional type approach', *Journal of Arid Environments* (Volume 75): pp. 255-264.

Anderson, T.M. Schultz, M. Risch, A.C. (2012) 'Seed germination cues and the importance of the soil seed bank across an environmental gradient in the Serengeti', *Oikos* (Volume 121): pp. 306-312.

Augustine, D.J. Frelich, L.E. Jordan, P.A. (1998) 'Evidence for two alternate stable states in an ungulate grazing system', *Ecological Applications* (Volume 8): pp. 1260-1269.

Baker, L.E. Hoffman, M.T. (2006) 'Managing variability: Herding strategies in communal rangelands of semiarid Namaqualand, South Africa', *Human Ecology* (Volume 34): pp. 765-784.

Bakker, J.P. Poschlod, P. Strykstra, R.J. Bekker, R.M. Thompson, K (1996a) 'Seed banks and seed dispersal: Important topics in restoration ecology', *Acta Botanica Neederlandica* (Volume 45): pp. 461-490.

Bakker, J.P. Bakker, E.S. Rosen, E. Verweij, G.L Bekker, R.M (1996b) 'Soil seed bank composition along a gradient from dry alvar grassland to *Juniperus* shrubland', *Journal of Vegetation Science* (Volume 7): pp. 165-176.

Barbaro, L. Dutoit, T. Cozic, P. (2001) 'A six year experimental restoration of biodiversity by shrub clearing and grazing in calcareous grasslands in the French Prealps', *Biodiversity and Conservation* (Volume 10): pp. 119-135.

Bashan, D. Bar-Massada, A. (2017) 'Regeneration dynamics of woody vegetation in a Mediterranean landscape under different disturbance-based management treatments', *Applied Vegetation Science* (Volume 20): pp. 106-114.

Bates, D. Maechler, M. Bolker, B. Walker, S. (2015) 'Fitting linear mixed-effects models using lme4', *Journal of Statistical Software* (Volume 67): pp. 1-48.

Benayas, J.M.R and Bullock, J.M (2012) 'Restoration of biodiversity and ecosystem services on agricultural land', *Ecosystems* (Volume 15): pp. 883-899.

Bergh, N.G. Verboom, G.A. Rouget, M. Cowling, R.M. (2014) 'Vegetation types of the Greater Cape Floristic Region' in Allsop, N. Colville, J.F. Verboom, G.A (Eds) *Fynbos: Ecology, Evolution and Conservation of a Megadiverse Region*, Oxford University Press, UK.

Bertiller, M.B. (1996) 'Grazing effects on sustainable semiarid rangelands in Patagonia: The state and dynamics of the soil seed bank', *Environmental Management* (Volume 20): pp. 123-132.

Bestelmeyer, B.T (2006) 'Threshold concepts and their use in rangeland management and their restoration: The good, the bad and the insidious', *Restoration Ecology* (Volume 14): pp. 325-329.

Bestelmeyer, B.T. Burkett, L.M. Lister, L. Brown, J.R. Schooley, R.L. (2019) ‘Collaborative approaches to strengthen the role of science in rangeland conservation’, *Rangelands* (Volume 41): pp. 218-226.

Beukes, P.C. (1987) ‘Responses of grey rhebuck and bontebok to controlled fire in coastal renosterveld’, *South African Journal of Wildlife Research* (Volume 17): pp. 103-108).

Beymer-Farris, B.A. Bassett, T.J. (2012) ‘The REDD menace: Resurgent protectionism in Tanzania’s mangrove forests’, *Global Environmental Change* (Volume 22): pp. 332-341.

Boshoff, A.F. Kerley, L.J. (2001) ‘Potential distributions of the medium to large-sized mammals in the Cape Floristic Region, based on historical accounts and habitat requirements’, *African Zoology* (Volume 36): pp. 245-273.

Bossuyt, B. Butaye, J. Honnay, O. (2006) ‘Soil seed bank composition of open and overgrown calcareous grassland soils - A case study from Southern Belgium’, *Journal of Environmental Management* (Volume 79): pp. 364-371.

Boucher, C. (1983) ‘Floristic and structural features of the coastal foreland vegetation south of the Berg River, Western Cape Province, South Africa’, *Bothalia* (Volume 14): pp. 669-674.

Boucher, C (1995) ‘Overview on management of renosterveld’ in Low, A.B. and Jones, F.E. (eds) *The Sustainable use and management of renosterveld remnants in the Cape Floristic Region*, Proceedings of a Symposium, FCC Report 1995/4, Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, South Africa.

Bradd-Witt, G. Berghammer, L.J. Beeton, R.J.S. Moll, E.J. (2000) ‘Retrospective monitoring of rangeland vegetation change: Ecohistory from deposits of sheep dung associated with shearing sheds’, *Austral Ecology* (Volume 25): pp. 260-267

Brand, T.S. (2000) ‘Grazing behaviour and diet selection by Dorper sheep’, *Small Ruminant Research* (Volume 36): pp. 147-158.

Briske, D.D. Illius, A.W. Anderies, J.M. (2017) ‘Nonequilibrium theory and Resilience Theory’ in Briske, D.D. (Ed) *Rangeland Systems: Processes, Management and Challenges,* Springer Series on Environmental Management.

Brown, N.A.C (1993) ‘Promotion of germination of Fynbos seeds by plant-derived smoke’, *New Phytologist* (Volume 123): pp. 575-583.

Brown, N.A.C. Botha, P.A. (2004) 'Smoke seed germination studies and a guide to seed propagation of plants from the major families of the Cape Floristic Region, South Africa', *South African Journal of Botany* (Volume 70): pp. 559-581.

Bunbury, C.J.F. (1842) 'Botanical excursions in South Africa', *London Journal of Botany* (Volume 2): pp. 15-41.

Butt, B. Turner, M.D. (2012) 'Clarifying competition: The case of wildlife and pastoral livestock in east Africa', *Pastoralism: Research, Policy and Practice* (Volume 2): pp.1-15.

Callmander, M.W. Schatz, G.E. Lowry, P.P. (2005) 'IUCN Red List Assessment and the Global Strategy for Plant Conservation: Taxonomists must act now', *Taxon* (Volume 54): pp. 1047-1050.

Cameron, D.R. Marty, J. Holland, R.F. (2014) 'Whither the rangeland? Protection and conversion in California's rangeland ecosystems', *PLOS One* (Volume 9): pp.1-12.

Chandra, R. Knight, R. Omlin, C.W. (2009) 'Renosterveld conservation in South Africa: A case study for handling uncertainty in knowledge-based neural networks for environmental management', *Journal of Environmental Informatics* (Volume 13): pp. 56-65.

Chimphango, S.B.M. Gallant, L.H. Poulsen, Z.C. Samuels, M.I. Hattas, D. Curtis, O.E. Muasya, A.M. Cupido, C. Boatright, J.S. Howieson, J. (2020) 'Native legume species as potential fodder crops in the mediterranean renosterveld shrubland, South Africa', *Journal of Arid Environments* (Volume 173): pp. 1-9.

Cosyns, E. Claerbout, S. Lamoot, I. Hoffman, M. (2005) 'Endozoochorous seed dispersal by cattle and horse in a spatially heterogenous landscape', *Plant Ecology* (Volume 178): pp. 149-162.

Courtois, D.R. Perryman, B.L. Hussein, S.H (2004) 'Vegetation change after 65 years of grazing and grazing exclusion', *Journal of Rangeland Management* (Volume 57): pp. 574-582.

Cousins, S.R. (2017) 'Renosterveld remnants of the Swartland: A rainbow of colour and rarity veiled in black', *Veld & Flora* (December 2017): pp. 158-162

Cousins, S.R. Witkowski, E.T.F. Esler, K.J. (2018) 'Influence of fire on critically endangered Swartland Shale Renosterveld in the Cape Floristic Region', *Applied Vegetation Science* (Volume 21): pp. 144-155.

Cowan, O.S. (2019) *The Functional Ecology of Eastern Rûens Shale Renosterveld*, Unpublished PhD Thesis, University of Cape Town, South Africa.

Cowan, O.S. Anderson, P.M.L. (2014) 'The Peninsula Shale Renosterveld of Devil's Peak, Western Cape: A study into the vegetation and seedbank with a view towards potential restoration', *South African Journal of Botany* (Volume 95): pp. 135-145.

Cowan, O.S. Anderson, P.M.L. (2019) 'Three complete plant pollinator networks along a secondary successional gradient in critically endangered renosterveld, South Africa', *Journal of Pollination Ecology* (Volume 25): pp. 24-35.

Cowling, R.M. (1984) 'A syntaxonomic and synecological study in the Humansdorp region of the Fynbos Biome', *Bothalia* (Volume 15): pp. 175-227.

Cowling, R.M. Pierce, S.M. Moll, E.J. (1986) 'Conservation and utilisation of South Coast Renosterveld, an endangered South African vegetation type', *Biological Conservation* (Volume 37): pp. 363-377.

Cowling, R.M. Proches, S. Partridge, T.C (2009) 'Explaining the uniqueness of the Cape Flora: Incorporating geomorphic evolution as a factor for explaining its diversification', *Molecular Phylogenetics and Evolution* (Volume 51): pp. 64-74.

Curtis, O.E (2009) 'The race to save the renosterveld', *Farmers Weekly* (October 2009): pp. 48-51.

Curtis, O.E (2013a) *Management of Critically Endangered Renosterveld Fragments in the Overberg, South Africa*, Unpublished PhD Thesis, University of Cape Town.

Curtis, O.E (2013b) 'The remains of the Renosterveld', *Farmers Weekly* (July 2013): pp. 48-49.

Curtis, O.E and Bond, W (2013) 'What is Renosterveld? Grassy shrubland or shrubby grassland? Have recent analyses of soil carbon isotopes provided an answer?' *Veld and Flora* (Volume 99): pp. 180-183.

Curtis, O.E. Stirton, C.H Muasya, A.M (2013) 'A conservation and floristic assessment of poorly known species rich quartz-silcrete outcrops within Rûens Shale Renosterveld', *South African Journal of Botany* (Volume 87): pp. 99 – 111.

Curtis-Scott, O.E. Goulding, M. Helme, N. McMaster, R. Privett, S. Stirton, C. (2020) *Field Guide to Renosterveld of the Overberg,* Struik Nature, Cape Town, South Africa.

Deacon, H.J (1992) ‘Human settlement’ in Cowling, R.M. (Ed) *The Ecology of Fynbos: Nutrients, Fire and Diversity*, Oxford University Press, Cape Town, South Africa (pp. 260-270).

De Villers, A.J. Van Rooyen, M.W. Theron, G.K. (2003) ‘Similarity between the soil seed bank and the standing vegetation in the strandveld Succulent Karoo, South Africa’, *Land Degradation & Development* (Volume 14): pp. 527-540.

De Vries, A. Ripley, B.D. (2016) ggdendro: Create Dendrograms and Tree Diagrams Using ‘ggplot2’, R Package Version 0.1-20. https://CRAN.R-project.org/package=ggdendro

Donaldson, J. Nanni, I. Zachariades, C. Kemper, J. (2002) ‘Effects of habitat fragmentation on pollinator diversity and plant reproductive success in renosterveld shrublands of South Africa’, *Conservation Biology* (Volume 16): pp. 1267-1276.

Dougill, A.J. Fraser, E.D.G. Reed, M.S. (2010) ‘Anticipating vulnerability to climate change in dryland pastoral systems: Using Dynamic Systems Models for the Kalahari’, *Ecology and Society* (Volume 15): 17.

Dreber, N. Esler, K.J. (2011) ‘Spatio-temporal variation in soil seed banks under contrasting grazing regimes following low and high seasonal rainfall in arid Namibia’, *Journal of Arid Environments* (Volume 75): pp. 174-184.

Eggleton, P. (2020) ‘The State of the World’s Insects’, *Annual Review of Environment and Resources* (Volume 45): pp. 61-82.

Esler, K.J. Pierce, S.M. de Villiers, C. (2014) *Fynbos: Ecology and Management*, Briza Publications, Pretoria, South Africa.

Espinosa, R. Tago, D. Treich, N. (2020) ‘Infectious diseases and meat production’, *Environmental and Resource Economics* (Volume 76): pp. 1019-1044.

Fisher, J. Loneragan, W. Dixon, K Veneklaas, E (2009) ‘Soil seed bank compositional change constrains biodiversity in an invaded species rich woodland’, *Biological Conservation* (Volume 142): pp. 256-269.

Forbes, C.J. *Benchmarks for the future: Long term vegetation change derived from palaeoecological techniques in West Coast Renosterveld, South Africa*, Unpublished Masters Thesis, University of Cape Town, South Africa.

Forbes C.J. Gillson L. Hoffman M.T. (2018) ‘Shifting baselines in a changing world: restoration targets in endangered heathlands of the Cape Floristic Region, South Africa’, *Anthropocene* (Volume 22): 81-93.

Fornoni, J. (2011) ‘Ecological and evolutionary implications of plant tolerance to herbivory’, *Functional Ecology* (Volume 25): pp. 299-407.

Forsyth, G.G. and Van Wilgen, B.W. (2008) ‘The recent fire history of the Table Mountain National Park and implications for fire management’, *Koedoe* (Volume 50): pp. 3-9.

Fleischner, T (1994) ‘Ecological costs of livestock grazing in western North America’, *Conservation Biology* (Volume 8): pp. 629-644.

Frank, A.B. Tanaka, D.L. Hofmann, L. Follett, R.F. (1995) ‘Soil carbon and nitrogen of the northern Great Plains grasslands as influenced by long term grazing’, *Journal of Range Management* (Volume 48): pp. 470-474.

Freese, C.H. Fuhlendorf, S.D. Kunkel, K. (2014) ‘A management framework for the transition from livestock production towards biodiversity conservation on Great Plains rangelands’, *Ecological Restoration* (Volume 32): pp. 358-368.

Fuhlendorf, S.D. Engle, D.M Kerby, J Hamilton, R (2008) ‘Pyric herbivory: Rewilding landscapes through the recoupling of fire and grazing’, *Conservation Biology* (Volume 23): pp. 588-598.

Fuhlendorf, S.D. Engle, D.M. Elmore, R.D. Limb, R.F. Bidwell, T.G. (2012) ‘Conservation of pattern and process: Developing an alternative paradigm of rangeland management’, *Rangeland Ecology and Management* (Volume 65): pp. 579-589.

Fuhlendorf, S.D. Fynn, R.D.S. McGranahan, D.A. Twidwell, D. (2017) ‘Heterogeneity as the basis for rangeland management’ in Briske, D.D. (Ed) *Rangeland Systems: Processes, Management and Challenges*, Springer Series on Environmental Management.

Gao, J. Carmel, Y. (2020) ‘A global meta-analysis of grazing effects on plant richness’, *Agriculture, Ecosystems and Environment* (Volume 302): pp. 1-10.

Gillson, L. Hoffman, M.T. (2007) ‘Rangeland ecology in a changing world’, *Science* (Volume 315): pp. 53-54.

Godde, C.M. Boone, R.B. Ash, A.J. Waha, K. Sloat, L.L. Thornton, P.K. Herrero, M. (2020) ‘Global rangeland production systems and livelihoods at threat under climate change and variability’, *Environmental Research Letters* (Volume 15): pp. 1-15.

Gonthier, D.J. Ennis, K.J. Farinas, S. Hsieh, H.Y. Iverson, A.L. Batary, P. Rudolphi, J. Tscharntke, T. Cardinale, B.J. Perfecto, I (2014) ‘Biodiversity conservation in agriculture requires a multi-scale approach’, *Proceedings of the Royal Society B* (Volume 281): 20141358.

Grice, A.C. Hodgekinson, K.C. (2002) ‘Challenges for rangeland people’, in Grice, A.C. Hodgekinson, K.C. (Eds) *Global Rangelands: Progress and Prospects*, Cabi Publishing, Oxon, UK. pp. 1-10.

Groenewald, J.P. (2014) *Biodiversity and Ecology of Critically Endangered Rûens Silcrete Renosterveld in the Buffeljagsrivier area, Swellendam*, Unpublished Masters thesis, Stellenbosch University, South Africa.

Groffman, P.M. Baron, J.S. Blett, T. Gold, A.J. Goodman, I. Gunderson, L.H. Levinson, B.M. Palmer, M.A. Paerl, H.W. Peterson, G.D. Poff, N.L. Rejeski, D.W. Reynolds, J.F. Turner, M.G. Weathers, K.C. Wiens, J. (2006) ‘Ecological thresholds: The key to environmental management or an important concept with no practical application?’, *Ecosystems* (Volume 9): pp. 1-13.

Haarmeyer, D.H. Schmeidel, U. Dengler, J Bösing, B.M (2010) ‘How does grazing intensity affect different vegetation types in arid Succulent Karoo, South Africa? Implications for conservation management’, *Biological Conservation* (Volume 143): pp. 588-596.

Haddad, C.R and Dippenaar-Schoeman, A.S. (2009) ‘A checklist of the non-acarine Arachnids (Chelicerata: Arachnida) of the De Hoop Nature Reserve, Western Cape Province, South Africa’, *Koedoe* (Volume 51): pp. 17-25.

Hall, S.A. Newton, R.J. Holmes, P.M. Gaertner, M. Esler, K.J. (2017) ‘Heat and smoke pre-treatment of seeds to improve restoration of an endangered Mediterranean climate vegetation type’, *Austral Ecology* (Volume 42): pp. 354-366.

Hamilton III, W.J. Buskirk, R.E. Buskirk, W.H. (1978) ‘Omnivory and utilisation of food resources by Chacma baboons, Papio ursinus’, *The American Naturalist* (Volume 112): pp. 911-924.

Harrington, J.A and Kathol, E (2008) ‘Responses of shrub midstorey and herbaceous layers to managed grazing and fire in a North American savanna (Oak Woodland) and prairie landscape’ *Restoration Ecology* (Volume 17): pp. 234-244.

Hasanpouri, R. Sepehry, A. Barani, H. (2020) ‘Effects of rangeland conversion to dryland farming on soil chemical properties (Case Study: Kian rangelands, Lorestan, Iran’, *Journal of Rangeland Science* (Volume 10): pp. 49-55.

Haussmann, N.S. Delport, C. Kakembo, V. Mashiane, K.K. le Roux, P.C. (2019) ‘Restoration potential of invaded abandoned agricultural fields: What does the seed bank tell us?’, *Restoration Ecology* (Volume 27): pp. 813-820.

Heelemann, S (2010) *Renosterveld Remnants – Current Ecological Situation and Restoration Perspectives*, Unpublished PhD Thesis, University of Regensburg, Germany.

Heelemann, S. Krug, C.B. Esler, K.J. Reisch, C. Poschlod, P. (2013) ‘Soil seed banks of remnant and degraded Swartland Shale Renosterveld’, *Applied Vegetation Science* (Volume 16): pp. 585-597.

Hendley, Q.B. (1983) ‘Palaeontology and palaeoecology of the Fynbos region: An introduction’, in Deacon, H.J. Hendley, Q.B. Lambrechts, J.J.N. (Eds) *Fynbos Palaeoecology: A Preliminary Synthesis*, pp. 87-99, CSIR, Pretoria, South Africa.

Herault, B. Honnay, O. Thoen, T. ‘Evaluation of the ecological restoration potential of plant communities in Norway spruce plantations using a life-trait based approach’, *Journal of Applied Ecology* (Volume 42): pp. 536-545.

Hester, A.J. Edenius, L. Buttenschon, R.M. Kuiters, A.T. (2000) ‘Interactions between forests and herbivores: The role of controlled grazing experiments’, *Forestry* (Volume 73): pp. 381-391.

Heywood, P (2013) ‘The quagga and science: What does the future hold for this extinct zebra?’, *Perspectives in Biology and Medicine* (Volume 56): pp. 53-64.

Hobbs, R.J. Arico, J. Aronson, J.S. Baron, P. Bridgewater, V.A. Cramer, P.R. Epstein, J.J. Ewel, C.A. Klink, A.E. Lugo, D. Norton, D. Ojima, D. Richardson, D.M. Sanderson, E.W. Valladares, F. Vila, M. Zamora, Z. (2006) ‘Novel ecosystems: theoretical and management aspects of the new ecological world order’, *Global Ecology and Biogeography* (Volume 15): pp. 1-7.

Hobbs, N.T. Galvin, K.A. Stokes, C.J. Lackett, J.M. Ash, A.J. Boone, R.B. Reid, R.B. Thornton, P.K. (2008) 'Fragmentation of rangelands: Implications for humans, animals and landscapes', *Global Environmental Change* (Volume 18): pp. 776-785.

Hoffman, M.T. Madden, C.F. Erasmus, K. Saayman, K. Botha, J.C. (2009) 'The impact of indigenous ungulate herbivory over five years (2004-2008) on the vegetation of the Little Karoo, South Africa', *African Journal of Range and Forage Science* (Volume 26): pp. 169-179.

Holechek, J.L. Geli, H.M.E. Cibils, A.F. Sawalhah, M.N. (2020) 'Climate change, rangelands and sustainability of ranching in the western United States', *Sustainability* (Volume 12): pp. 1-24.

Holmes, P.M. Cowling, R.M. (1997) 'Diversity, composition and guild structure relationships between soil stored seed banks and mature vegetation in alien plant invaded South African fynbos shrublands', *Plant Ecology* (Volume 133): pp. 107-122.

Holmes, P.M and Richardson, D.M (2002) 'Protocols for restoration based on recruitment dynamics, community structure and ecosystem function: Perspectives from South African Fynbos', *Restoration Ecology* (Volume 7): pp. 215-230.

Holmes, P.M. Esler, K.J. Gaertner, M. Geerts, S. Hall, S.A. Nsikani, M.M. (2020) 'Biological invasions and ecological restoration in South Africa', in Van Wilgen, B.W. Measey, J. Richardson, D.M. Wilson, J.R. Zengeya, T.A. (Eds) *Biological Invasions in South Africa*, Invading Nature, Spring Series in Invasion Ecology (Volume 14), Springer, Cham.

Hopfensperger, K.N. (2007) 'A review of similarity between seed bank and standing vegetation across ecosystems', *Oikos* (Volume 116): pp. 1438-1448.

Hothorn, T. Bretz, F. Westfall, P. (2008) 'Simultaneous inference in General Parametric Models', *Biometrical Journal* (Volume 50): pp. 346-363.

Hrusker, T. Huntsigner, L. Brunson, M. Wenjun, L. Marshall, N. Oviedo, J.L. Whitcomb, L. (2017) 'Rangelands as social-ecological systems', in Briske, D.D. (Ed) *Rangeland Systems: Processes, Management and Challenges*, Springer Series on Environmental Management.

Humphreys, A.J.B. (1981) 'Before Van Riebeeck: Some thoughts on the later Prehistoric inhabitants of the Western Cape', *Kronos* (Volume 4): pp. 1-9.

Huntsinger, L. Oviedo, J.L. (2014) 'Ecosystem services are socio-economic services in a traditional pastoral system: The case of California's Mediterranean rangelands', *Ecology and Society* (Volume 19): pp. 8.

IPBES (2019): Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Díaz, J. Settele, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany.

Jacquemyn, H. Van Mechelen, C. Brys, R. Honnay, O (2011) 'Management effects on the vegetation and soil seed bank of calcareous grasslands', *Biological Conservation* (Volume 144): pp. 416-422.

Jarmin, M.L. (1986) *Conservation priorities in the lowland regions of the Fynbos Biome*, South African National Scientific Programmes Report 87.

Joubert, C. (1991) *History and description of contemporary vegetation of Signal Hill, Cape Town*. Unpublished Masters thesis, University of Cape Town, South Africa.

Kemper, J. Cowling, R.M. Richardson, D.M. (1999) 'Fragmentation of South African renosterveld shrublands: Effects on plant community structure and conservation implications', *Biological Conservation* (Volume 90): pp. 103-111.

Kinloch, J.E and Friedel, M.H (2005) 'Soil seed reserves in arid grazing lands of central Australia Part 1: Seed bank and vegetation dynamics', *Journal of Arid Environments* (Volume 60): pp. 133-161.

Kinucan, R.J and Smiens, F.E (1992) 'Soil seed bank of a semiarid Texas grassland under three long term (36 year) grazing regimes', *American Midland Naturalist* (Volume 128): pp. 11-21.

Koerner, S.E. Smith, M.D. Burkepile, D.E. Hanan, M.P. Avolio, M.P. Collins, S.L. Knapp, A.K. Lemoine, N.P. Forrestel, E.J. Eby, S. Thompson, D.L. Aguado-Santacruz, G.A. Anderson, J.P. Anderson, T.M. Angassa, A. Bagchi, S. Bakker, E.S. Bastin, G. Baur, L.E. Beard, K.H. Beever, E.A. Bohlen, P.J. Boughton, E.H. Canestro, D. Cesa, A. Chaneton, E. Cheng, J. D'Antonio, C.M. Deleglise, C. Dembele, F. Dorrough, J. Eldridge, D.J. Fernandez-Going, B. Fernandez-Lugo, S. Fraser, L.H. Freedman, B. Garcia-Salgado, G. Goheen, J.R. Guo, L. Husheer, S. Karembe, M. Knops, J.M.H. Kraaij, T. Kulmatiski, A. Kytoviita, M.M. Lezama, F. Loucougaray, G. Loydi, A. Milchunas, D.G. Milton, S.J. Morgan, J.W. Moxham, C. Nehring, K.C. Olff, H. Palmer, T.M. Rebollo, S. Riginos, C. Risch, N.L. Rueda, M. Sankaran, M. Sasaki, T. Schoenecker, K.A. Schultz, N.L. Schwabe, A. Siebert, F. Smit, C. Stahlheber, K.A. Storm, C. Strong, D.J. Su, J.

Tiruvaimozhi, Y.V. Tyler, C. Val, J. Vandegehuchte, M.L. Veblen, K.E. Vermeire, L.T. Ward, D. Wu, J. Young, T.P. Yu, Q. Zelikova, T.J. (2018) 'Change in dominance determines herbivore effects on plant biodiversity', *Nature Ecology & Evolution* (Volume 2): pp. 1925-1932.

Kosmas, C. Detsis, V. Karamesouti, M. Kounalaki, K. Vassiliou, P. Salvati, L. (2015) 'Exploring long term impact of grazing management on land degradation in the socio-ecological system of Asteroussia Mountains, Greece', *Land* (Volume 4): pp. 541-559.

Kraaij, T. Novellie, P.A. (2010) 'Habitat selection by large herbivores in relation to fire at the Bontebok National Park (1974-2009): The effects of management changes', *African Journal of Range and Forage Science* (Volume 27): pp. 1-27.

Kraaij, T. (2010) 'Changing the fire management regime in the renosterveld and lowland fynbos of the Bontebok National Park', *South African Journal of Botany* (Volume 76): pp. 550-557.

Kraaij, T and Van Wilgen, B.W.(2014) 'Drivers, ecology and management of fire in Fynbos', Allsopp, N. Colville, J.F. Verboom, A.G. (Eds) *Fynbos: Ecology, Evolution and Conservation in a Megadiverse Region* (pp. 47-72), Oxford University Press, Oxford, UK.

Krug, C (2004) *Practical Guidelines for the Restoration of Renosterveld*, University of Stellenbosch, South Africa.

Krug, R.M. Krug, C.B. Midoko Iponga, D. Walton, B. Milton, S.J. (2004) 'Reconstructing West Coast Renosterveld: Past and present ecological processes in a Mediterranean shrubland of South Africa', Proceedings of the 10th MEDECOS Conference, Rhodes, Greece.

Kuznetsova, A. Brockhoff, P.B. Christensen, R.H.B (2017). "lmerTest Package: Tests in Linear Mixed Effects Models." *Journal of Statistical Software*, (Volume 82): pp. 1–26.

Landcare South Africa. National Department of Agriculture (2002) *CARA Legislation Made Easy: The Conservation of Natural Resources Act, 1983 (Act No. 43 of 1983) (CARA)*, National Department of Agriculture, Pretoria, South Africa.

Landsberg, J. James, C.D. Morton, S.R. Mueller, W.J. Stol, J. (2003) 'Abundance and composition of plant species along grazing gradients in Australian rangelands', *Journal of Applied Ecology* (Volume 40): pp. 1008-1024.

Lavado, R.S. Sierra, J.O. Hashimoto, P.N. (1996) 'Impacts of grazing on soil nutrients in a Pampean grassland', *Journal of Range Management* (Volume 49): pp. 452-457.

Leis, S.A. Engle, D.M.L. Jeffrey, S.F. Kretzer, J. (2003) ‘Comparison of vegetation sampling procedures in a disturbed mix grass prairie’ *Proceedings of the Oklahoma Academy of Science* (Volume 83): pp. 7-15.

Leopold, A.S. (1949) *A sandy county almanac with sketches here and there*, Oxford University Press, New York, USA.

Liao, C. Agrawal, A. Clark, P.E. Levin, S.A. Rubenstein, D.I. (2020) ‘Landscape sustainability in the drylands: Mobility, rangelands and livelihoods’, *Landscape Ecology* (Volume 35): pp. 2433-2447.

Le Roux, G.H. (1988) ‘Die gebruik van die Kaapse fynbos vir weiding en die probleme wat dit skep’, *Suid Afrikaanse Bosbouttydskrif* (Volume 114): pp. 51-54.

Lombard, A.T. Cowling, R.M. Pressey, R.L. Mustart, P.J. (1997) ‘Reserve selection in a species rich and fragmented landscape on the Agulhas Plain, South Africa’, *Conservation Biology* (Volume 11): pp. 1101-1116.

Low, A.B and Jones F.E (Eds.) *The sustainable use and management of renosterveld fragments in the Cape Floristic Region*, Proceedings of a Symposium, FCC Report 1995/4, Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town.

Louhaichi, M. Salkini, A.K. Petersen, S.L. (2009) ‘Effect of small ruminant grazing on the plant community characteristics of semiarid Mediterranean ecosystems’, *International Journal of Agriculture and Biology* (Volume 11): pp. 681-689.

Loydi, A. Zalba, S.M. Distel, R.A. (2012) ‘Viable seed banks under grazing and exclosure conditions in montane mesic grasslands of Argentina’, *Acta Oecologica* (Volume 43): pp. 8-15.

Maechler, M. Rousseeuw, P. Struyf, A. Hubert, M. Hornik, K. (2019) *cluster: Cluster Analysis Basics and Extensions,* R Package Version 2.1.0.

Magee, A.R. Curtis, O.E. Van Wyk, B.E. (2016) ‘A refined circumspection of *Notobubon striatum* and the resurrection of *Dregea collina* Ecklon & Zeyer (Apiaceae, Apioideae)’, *Phytotaxa* (Volume 266): pp. 27-32.

Makowski, D. (2018). The Psycho Package: An Efficient and Publishing-Oriented Workflow for Psychological Science. Journal of Open Source Software, 3(22), 470. Available from https://github.com/neuropsychology/psycho.R

Manning, J. and Goldblatt, P (2012) *Plants of the Greater Cape Floristic Region 1: The Core Cape Flora*, *Strelitzia* 29, South African National Biodiversity Institute, Pretoria.

McDowell, C.R. (1995) 'Grazing and Renosterveld Management', in Low, A.B. and Jones, F.E. (Eds) *The Sustainable use and management of renosterveld remnants in the Cape Floristic Region*, Proceedings of a Symposium, FCC Report 1995/4, Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, South Africa.

McGranahan, D.A. Kirkman, K.P. (2013) 'Multifunctional rangeland in southern Africa: Managing for production, conservation and resilience with fire and grazing', *Land* (Volume 2): pp. 176-193.

McIntyre, S. Lavorel, S. Tremont, R.M. (1995) 'Plant life history attributes: Their relationship to disturbance response in herbaceous vegetation', *Journal of Ecology* (Volume 83): pp. 31-44.

McIver, J. Starr, L (2001) 'Restoration of degraded lands in the interior Columbia River basin: Passive vs. active approaches', *Forest Ecology and Management* (Volume 153): pp. 15-28.

McKeon, G.M. Stone, G.S. Syktus, J.I. Carter, J.O. Flood, N.R. Ahrens, D.G. Bruget, D.N. Chilcott, C.R. Cobon, D.H. Cowley, R.A. Crimp, S.J. Fraser, G.W. Howden, S.M. Johnston, P.W. Ryan, J.G. Stokes, C.J. Day, K.A. (2009) 'Climate change impacts on northern Australian rangeland livestock carrying capacity: A review of issues', *The Rangeland Journal* (Volume 31): pp. 1-29.

Meadows, M.E. (2003) 'Soil erosion in the Swartland, Western Cape Province, South Africa: Implications of past and present policy and practice', *Environmental Science and Policy* (Volume 6): pp. 17-28.

Meissner, R.A and Facelli, J.M (1999) 'Effects of sheep exclusion on the soil seed bank and annual vegetation in chenopod shrublands of South Australia', *Journal of Arid Environments* (Volume 42): pp. 117-128.

Menke, J. Bradford, G.E. (1992) 'Rangelands' *Agriculture, Ecosystems and Environment* (Volume 42): pp. 141-163.

Merdas, S. Kouba, Y. Mostephaoui, T. Farhi, Y. Chenchouni, H. (2021) 'Livestock grazing induced large scale biotic homogenization in arid Mediterranean steppe rangelands', *Land Degradation Development* (Volume 2021): pp.1-9.

Midoko-Iponga, D. (2004) *Renosterveld restoration: The role of competition, herbivory and other disturbances*, Unpublished Masters Thesis, Stellenbosch University, South Africa.

Midoko-Iponga, D. Krug, C.B. Milton, S.J (2005) 'Competition and herbivory influence growth and survival of shrubs on old fields: Implications for restoration of renosterveld shrublands', *Journal of Vegetation Science* (Volume 16): pp. 685-692.

Milton, S.J. (2007) Rapid Assessment Method for Renosterveld. Renosterveld Assessment Workshop, unpublished report.

Molteno, S. Smith, G.F. (2019) 'A significant westerly extension of the natural distribution range of *Tulisa minor* (Aiton) GideonF.Sm & Molteno (Asphodelaceae: Alooidae) in the Overberg, Western Cape Province, South Africa', *Bradleya* (Volume 37): pp. 201-211.

Moncrieff, G.R. (2021) 'Locating and dating land cover change events in the Renosterveld, a Critically Endangered shrubland ecosystem', *Remote Sensing* (Volume 13): pp. 1-9.

Mongomery, R.A. Borona. K. Kasozi, H. Mudumba, T. Ogada, M. (2020) 'Positioning human heritage at the centre of conservation practice', *Conservation Biology* (Volume 34): pp. 1122-1130.

Moore, P.D (1980) 'Soil seed banks', *Nature* (Volume 284): pp. 123-124.

Mueller-Dombois, D. Ellenberg, H. (1974) *Aims and Methods of Vegetation Ecology*, Wiley: New York, New York, USA.

Myers, N. Mittermeier, R.A. Mittermeier, C.G. Da Fonseca, A.B. Kent, J. (2000) 'Biodiversity hotspots for conservation priorities', *Nature* (Volume 403): pp. 853-858.

Newton, I.P. Knight, R. (2004) 'The matrix: The evolutionary development of coastal renosterveld', *Veld and Flora*, December 2004 Issue, pp. 162-164.

Newton, I.P. Knight, R. (2005) 'The use of a 60 year series of aerial photographs to assess local agricultural transformations of West Coast Renosterveld, an endangered South African vegetation type', *South African Geographical Journal* (Volume 87): pp. 18-27.

Newton, I.P. Knight, R. Cowling, R.M. (2005) 'The use of Landsat imagery for the identification of the remaining West Coast Renosterveld fragments, Western Cape Province, South Africa', *South African Journal of Botany* (Volume 71): pp. 67-75.

Newton, I.P and Knight, R.S (2010) 'How homogenous is West Coast Renosterveld? Implications for conservation', *Bothalia* (Volume 40): pp. 219 – 226.

Nhamo, L. Matchaya, G. Mabhaudi, T. Nhlengethwa, S. Nhemachena, C. Mpandeli, S. (2019) ‘Cereal production trends under climate change: Impacts and adaptation strategies in Southern Africa’, *Agriculture* (Volume 9): pp. 1-16.

Noy-Meir, I and Oron, T (2001) ‘Effects of grazing on geophytes in Mediterranean vegetation’, *Journal of Vegetation Science* (Volume 12): pp. 749-760.

Oksanen, F.J. Blanchet, G. Friendly, M. Kindt, R. Legendre, P. McGlinn, D. Minchin, P.R. O'Hara, R.B. Simpson, G.L. Solymos, P Henry H. Stevens, M. Szoecs, E. Wagner, H (2018) vegan: Community Ecology Package. R package version 2.5-3. https://CRAN.R-project.org/package=vegan

Papanastasis, V.P. (2009) ‘Restoration of degraded lands through grazing management: Can it work?’, *Restoration Ecology* (Volume 17): pp. 441-445.

Paterson-Jones, C. (1998) ‘Renosterveld floral treasures: Headed for extinction?’, *Africa – Environment and Wildlife* (Volume 6): pp. 26-32.

Perevolotsky, A and Seligman, N.G (1998) ‘The role of grazing in Mediterranean Rangeland Ecosystems’, *Bioscience* (Volume 48): pp. 1007-1017.

Perevolotsky, A. (2005) ‘Livestock grazing and biodiversity conservation in Mediterranean environments: The Israeli experience’ in Biala, L. Morand-Fehr, P. (Eds) *Sustainable Grazing, Nutritional Utilisation and Quality of Sheep and Goat Products*, CIHEAM, pp. 51-56.

Petersen, H. (2018) ‘Patterns of plant species richness and diversity across two habitat types in the Upper Karoo, South Africa’, Unpublished Masters thesis, University of Cape Town, South Africa.

Picker, M.D. Hoffman, M.T. Leverton, B. (2007) ‘Density of *Microhodotermes viator* (Hodotermitidae) mounds in southern Africa in relation to rainfall and vegetative productivity gradients’, *Journal of Zoology* (Volume 271): pp. 37-44.

Pillay, K.R. Wilson, A.L. Ramesh, T. Downs, C.T. (2015) ‘Digestive parameters and energy assimilation of Cape porcupine on economically important crops’, *African Zoology* (Volume 50): pp. 321-326.

Pineda, F.D. Nicholas, J.P. Pou, A. Galiano, E.F. (1981) ‘Ecological succession in oligotrophic pastures of central Spain’, *Vegetatio* (Volume 44): pp. 1-1.

Platou, K.A. Tueller, P.T. (1988) ‘The ecology of shrubland/woodland for range use’, In Tueller, P.T (Ed) *Vegetation Science Applications for Rangeland Analysis and Management*, Handbook of Vegetation Science Volume 14, Springer, Dordrecht, Germany.

Poulsen, Z.C. Curtis-Scott, O.E. (2020) ‘Life in miniature’, *Veld & Flora* (June 2020): pp. 26-31.

Pratt, D.J. Greenway, P. Gwynne, D. (1966) ‘A classification of East African Rangeland, with an Appendix on Terminology’, *Journal of Applied Ecology* (Volume 3): pp. 369-382.

Pressey, R.L. Cowling, R.M. Rouget, M. (2003) ‘Formulation of conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa’, *Biological Conservation* (Volume 112): pp. 99-127.

Proches, S. Cowling, R.M. Goldblatt, P. Manning, J.C. Snijman, D (2006) ‘An overview of the Cape geophytes’, *Biological Journal of the Linnaean Society* (Volume 87): pp. 27-43.

R Core Team (2019) R: A language and environment for statistical computing. http://www.r-project.org.

Pywell, R.F. Bullock, J.M. Hopkins, A. Walker, K.J. Sparks, T.H. Burkes, M.J.W. Peel, S. (2002) ‘Restoration of species rich grassland on arable land: Assessing the limiting processes using a multi-site experiment’, *Journal of Applied Ecology* (Volume 39): pp. 294-309.

Radloff, F.G.T. (2008) *The Ecology of Large Herbivores native to the coastal lowlands of the Fynbos Biome in the Western Cape, South Africa*, Unpublished PhD thesis, Stellenbosch University, South Africa.

Radloff, F.G.T. Mucina, L. Snyman, D (2014) ‘The impact of native large herbivores and fire on the vegetation dynamics in the Cape renosterveld shrublands of South Africa: Insights from a six year field experiment’, *Applied Vegetation Science*: pp. 1-14.

Raitt, G.R. (2005) *Themeda triandra Renosterveld in the Heidelberg District*, Unpublished Masters Thesis, Stellenbosch University, South Africa.

Rebelo, A.G. (1992) ‘Preservation of biotic diversity’, in Cowling, R.M. (Ed) *The Ecology of Fynbos: Nutrients, Fire and Diversity*, Oxford University Press, Cape Town, South Africa, pp. 309-344.

Rebelo, A.G. (1995) ‘Renosterveld: Conservation and Research’, in Low, A.B and Jones F.E (Eds.) *The sustainable use and management of renosterveld fragments in the Cape Floristic Region*, Proceedings of a Symposium, FCC Report 1995/4, Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town.

Rebelo, A.G. Boucher, C. Helme, N. Mucina, L. Rutherford, M (2006) ‘Fynbos Biome’ in Mucina, L. Rutherford, M (Eds) *The Vegetation of South Africa, Lesotho and Swaziland*, Strelitzia 19, South African National Biodiversity Institute, Pretoria.

Rebelo, A.G. Holmes, P.M. Dorse, C. Wood, J. (2010) ‘Impacts of urbanisation in a biodiversity hotspot: Conservation challenges in metropolitan Cape Town’, *South African Journal of Botany* (Volume 77): pp. 20-35.

Reid, R.S. Fernandez-Giminez, M.E. Wilmer, H. Pickering, T. Kassam, K.S. Yasin, A. Porensky, L.M. Derner, J.D. Nkedianye, D. Jamsranjav, C. Jamiyyansharav, K. Ulambayar, T. Oteros-Rotaz, E. Ravera, F. Bulbulbshoev, U. Kaziev, D.S. Knapp, C.N. (2021) ‘Using research to support transformative impacts on complex “wicked problems” with pastoral peoples in rangelands’, *Frontiers in Sustainable Food Systems* (Volume 4): pp. 1-24.

Richard, C.D. Jaymee, M. Holland, R.F. (2014) ‘Whither the rangeland? Protection and conversion in California’s rangeland ecosystems’, *PloS ONE* (Volume 9): pp. 1-12.

Rookmaaker, L.C (1992) ‘Additions and revisions to the list of specimens of the extinct Blue Antelope (*Hippotragus leucophaeus*)’, *Annals of the South African Museum* (Volume 102): pp. 131-141.

Ruwanza, S. (2017) ‘Towards an integrated ecological approach for abandoned agricultural fields in renosterveld, South Africa’, *South African Journal of Science* (Volume 113): pp. 1-4.

Rutherford, M.C and Powrie, L.W (2010) ‘Severely degraded rangeland: Implications for plant diversity from a case study in the Succulent Karoo, South Africa’, *Journal of Arid Environments* (Volume 74): pp. 692-701.

Rutherford, M.C. Powrie, L.W. (2013) ‘Impacts of heavy grazing on plant species richness: A comparison across rangeland biomes of South Africa’, *South African Journal of Botany* (Volume 87): pp. 146-156.

Saayman, N. Botha, J.C. (2008) ‘Does the soil seed bank of veld dominated by *Pteronia paniculata* change as a result of brushcutting?’ *Grassroots: Newsletter of the Grassland Society of Southern Africa* (Volume 8): pp. 32-35.

Samuels, I. Cupido, C. Swarts, M.B. Palmer, M.R. Paulse, J.W (2016) ‘Feeding ecology of four livestock species under different management in a semi-arid pastoral system in South Africa’, *Journal of Rangeland and Forage Science* (Volume 33): pp. 1-9.

Sayre, N.F. McAllister, R.R.J. Bestelmeyer, B.T. Moritz, M. Turner, M.D. (2013) 'Earth Stewardship of rangelands: Coping with ecological, economic and political marginality', *Frontiers in Ecology and the Environment* (Volume 11): pp. 348-354.

Schmiedel, U. Mucina, L. (2006) 'Vegetation of quartz fields in the Little Karoo, Tanqua Karoo and eastern Overberg (Western Cape Province, South Africa)', *Phytocoenologia* (Volume 36): pp. 1-44.

Schmiedel, U. Siemen, S.E. Dludu, M.N. Oldeland, J. (2021) 'Germination success of habitat specialists from the Succulent Karoo and Renosterveld on different soil types', *South African Journal of Botany* (Volume 137): pp. 320-330.

Scogings, P. de Bruyn, T. Vetter, S. (1999) 'Grazing into the future: Policy making for South African communal rangelands', *Development of Southern Africa* (Volume 16): pp. 403-414.

Segan, D.B. Murray, K.A. Watson, J.E.M. (2016) 'A global assessment of current and future biodiversity vulnerability to habitat loss-climate change interactions', *Global Ecology and Conservation* (Volume 5): pp. 12-21.

Shaw, J.M. Jenkins, A.R. Ryan, P.G. and Smallie, J.J (2010) 'A preliminary survey of avian mortality on power lines in the Overberg, South Africa', *Ostrich* (Volume 81): pp. 109-113.

Shi, Z. Zhang, J. Wei, H. (2020) 'Research progress on soil seed bank: A bibliometrics analysis', *Sustainability* (Volume 12): pp. 1-17.

Shiferaw, W. Demissew, S. Bekele, T. (2018) 'Ecology of soil seed banks: Implications for conservation and restoration of natural vegetation: A review', *International Journal of Biodiversity and Conservation* (Volume 10): pp. 380-393.

Shiponeni, N.N. Milton, S.J. (2006) 'Seed dispersal in the dung of large herbivores: Implications for restoration of renosterveld shrubland old fields', *Biodiversity & Conservation* (Volume 15): pp. 3161-3175.

Silva, V. Catry, F.X. Fernandes, P.M. Rego, F.C. Paes, P. Nunes, L. Caperta, A.D. Sergio, C. Bugalho, M.N. (2019) 'Effects of grazing on plant composition, conservation status and ecosystem services of Natura 2000 shrub-grassland habitat types', *Biodiversity and Conservation* (Volume 28): pp. 1205-1224.

Skead, C.J. (2009) *Historical plant incidence in southern Africa*, Strelitzia 24, South African National Biodiversity Institute, Pretoria, South Africa.

Skead, C.J. (2011) *Historical incidence of the larger land mammals in the broader Western and Northern Cape*, Second Edition (Eds: Boshoff, A.F. Kerley, G.I.H. Lloyd, P.H), Port Elizabeth: Centre for African Conservation Ecology, Nelson Mandela Metropolitan University.

Skowno, A.L. Poole, C.J. Raimondo, C.D. Sink, K.J. Van Deventer, H. Van Niekerk, L. Harris, L.R. Smith-Adao, L.B. Tolley, K.A. Zengeya, T.A. Foden, W.B. Midgley. G.F. Driver, A. (2019) *National Biodiversity Assessment 2018: The status of South Africa's ecosystems and biodiversity*, Synthesis Report, South African National Biodiversity Institute, an entity of the Department of Environment, Forestry and Fisheries, Pretoria.

Smykal, P. Vernoud, P. Blair, M.W. Soukup, A.Thompson, R.D. (2014) 'The role of the testa during development and in establishment of dormancy of the legume seed', *Frontiers in Plant Science* (Volume 5): pp. 1-19.

Society for Ecological Restoration International Science and Policy Working Group (2004) *The SER International Primer on Ecological Restoration*, www.ser.org and Tucson: Society of Ecological Restoration International.

Sternberg, M. Gutman, M. Perevolotski, A. Ungar, E.D. Kigel, J. (2000) 'Vegetation response to grazing management in a Mediterranean herbaceous community: A functional group response', *Journal of Applied Ecology* (Volume 37): pp. 224-237.

Sternberg, M. Gutman, M. Perevolotsky, A. Kigel, J (2003) 'Effects of grazing on soil seed bank dynamics: An approach with functional groups', *Journal of Vegetation Science* (Volume 14): pp. 375-386.

Sternberg, M. Golodets, C. Gutman, M. Perevolotsky, A. Ungar, E.D. Kigel, J Henkin, J (2015) 'Testing the limits of resistance: A 19 year study of Mediterranean grassland response to grazing regimes', *Global Change Biology* (): 1-12.

Steinaker, D. Twomlow, S. (2012) 'Revolutionary land use change in the 21st Century: Is (Rangeland) science relevant?', *Rangeland Ecology & Management* (Volume 65): pp. 590-598.

Steynor, R.C. Hewitson, B.C. Tadross, M.A. (2009) 'Projected future runoff of the Breede River under climate change', *Water SA* (Volume 35): pp. 433-440.

Stohlgren, T.J. Falkner, M.B. Schell, L.G (1995) 'A modified Whittaker nested sampling method', *Vegetatio* (Volume 117): pp. 113-121.

Stohlgren, T.J. Bull, K.A. Otsuki, Y (1998) 'Comparison of rangeland vegetation sampling techniques in the Central Grasslands', *Journal of Rangeland Management* (Volume 51): pp. 164-172.

Stohlgren, T.J. Schell, L.D. Van Den Heuvel, B (1999) 'How grazing and soil quality affect native and exotic plant diversity in rocky mountain grasslands', *Ecological Applications* (Volume 9): pp. 45-64.

Suding, K.N. Gross, K.L. Houseman, G.R. (2004) 'Alternate states and positive feedback in restoration ecology', *Trends in Ecology and Evolution* (Volume 19): pp. 46-53.

Suding, K.N. Hobbs, R.J. (2009) 'Threshold models in restoration and conservation: A developing framework', *Trends in Ecology and Evolution* (Volume 24): pp. 271-279.

Swift, T.L. Hannon, S.J (2010) 'Critical thresholds associated with habitat loss: A review of the concepts, evidence and applications', *Biological Reviews* (Volume 85): pp. 35-53.

Szejner, P. Soumaya, B. Ehleringer, J.R. Monson, R.K. (2020) 'Recent increases in drought frequency cause observed multi-year drought legacies in the tree rings of semi-arid forests', *Oecologia* (Volume 192): pp. 1-12.

Tardella, F.M. Bricca, A. Goia, I.G. Catorci, A. (2020) 'How mowing restores montane Mediterranean grasslands following cessation of traditional livestock grazing', *Agriculture, Ecosystems and Environment* (Volume 295): pp. 1-12.

Tassone, E.E. Miles, L.S. Dyer, R.J. Rosenberg, M.S. Cowling, R.M. Verrelli, B.C. (2021) 'Evolutionary stability, landscape heterogeneity and human land usage shape population genetic connectivity in the Cape Floristic Region Biodiversity hotspot', *Evolutionary Applications* (Volume 14): pp. 1109-1123.

Tessema, Z.K. de Boer, W. Baars, R.M.T. Prins, H.H.T. (2012) 'Influence of grazing on soil seed banks determines restoration potential of aboveground vegetation in a semiarid savanna of Ethiopia', *Biotropica* (Volume 44): pp. 211-219.

Teste, F.P. Kardol, P. Turner, B.L. Wardle, D.A. Zemunik, G. Renton, M. Laliberte, E. (2017) 'Plant-soil feedback and the maintenance of diversity in Mediterranean climate shrublands', *Science* (Volume 355): pp. 173-176.

Teuber, L.M. Holzel, N. Fraser, L.H. (2013) 'Livestock grazing in intermountain depressional wetlands: Effects on plant strategies, soil characteristics and biomass', *Agriculture, Ecosystems and Environment* (Volume 175): pp. 21-28.

The Non-Affiliated Soil Analysis Work Committee (1990) *Handbook of Standard Soil Testing Methods for Advisory Purposes*, Soil Science Association of South Africa, Sunnyside, Pretoria, South Africa.

Thuiller, W. Midgley, G.F. Rouget, M. Cowling, R.M. (2006) 'Predicting patterns of plant species richness in Megadiverse South Africa', *Ecography* (Volume 29): pp. 733-744.

Todd, S.W. and Hoffman, M.T (1999) 'A fenceline contrast reveals effects of heavy grazing on plant diversity and community composition in Namaqualand, South Africa', *Plant Ecology* (Volume 142): pp. 169-178.

Todd, S.W. (2006) 'Gradients in vegetation cover, structure and species richness of Nama-Karoo shrublands in relation to distance from livestock watering points', *Journal of Applied Ecology* (Volume 43): pp. 293-204.

Tompkins, E.L. Adger, W.N. (2004) 'Does adaptive management of natural resources enhance resilience to climate change?', *Ecology and Society* (Volume 9): 10.

Topp, E.N. and Loos, J. (2019) 'Fragmented landscape, fragmented knowledge: A synthesis of renosterveld ecology and conservation', *Environmental Conservation* (Volume 46): pp. 171-179.

Topp, E.N. Tscharntke, T. Loos, J. (2021) 'Fire and landscape context shape plant and butterfly diversity in a South African shrubland', *Diversity and Distributions* (Volume 00): pp. 1-15.

Valbuena, L. Trabaud, L. (2001) 'Contribution of the soil seed bank to post fire recovery of a heathland', *Plant Ecology* (Volume 152): pp. 175-183.

Valko, O. Torok, P. Tothmeresz. Matus, G (2011) 'Restoration potential in seed banks of acidic fen and dry mesophilous meadows: Can restoration be based on local seed banks?', *Restoration Ecology* (Volume 19): pp. 9-15.

Van Der Merwe, H. Van Rooyen, M.W. (2011) 'Vegetation trends following fire in the Roggeveld, Mountain Renosterveld, South Africa', *South African Journal of Botany* (Volume 77): pp. 127-136.

Van Der Merwe, H. Du Toit, J.C.O. Van Den Berg, L. O' Connor, T.G. (2018) 'Impact of sheep grazing intensity on vegetation in the Arid Karoo Stock Rate Trial after 27 years, Carnarvon, South Africa', *Journal of Arid Environments* (Volume 155): pp. 36-45.

Van Oudtshoorn, F (2019) *Veld Management: Principles and Practices*, Briza Publications, Pretoria, South Africa.

Van Wilgen, B.W. Carruthers, J. Cowling, R.M. Elser, K.J. Forsyth, A.T. Gaertner, M. Hoffman, M.T. Kruger, F.J. Midgley, G.F. Palmer, G. Pence, G.Q.K. Raimondo, D.C. Richardson, D.M. Van Wilgen, N.J. Wilson, J.R.U. (2016) 'Ecological research and conservation management in the Cape Floristic Region between 1945 and 2015: History, current understanding and future challenges', *Transactions of the Royal Society of South Africa* (Volume 71): pp. 207-303.

Verdu, J.R. Crespo, M.B. Galante, E (2000) 'Conservation strategy of a nature reserve in Mediterranean ecosystems: The effects of protection from grazing on biodiversity', *Biodiversity & Conservation* (Volume 9): pp. 1707-1721.

Virgilio, A. Lambertucci, S.A. Morales, J.M. (2019) 'Sustainable grazing management in rangelands: Over a century searching for a silver bullet', *Agriculture, Ecosystems and Environment* (Volume 283): pp. 1-11.

Voeten, M.M. Prins, H.H.T. (1999) 'Resource partitioning between sympatric wild and domestic herbivores in the Taragire region of Tanzania', *Oecologica* (Volume 120): pp. 287-294.

Von Hase, A. Rouget, M. Maze, K. Helme, N (2003) *A fine scale conservation plan for Cape Lowlands Renosterveld: Technical Report*, Report CCU2/03, Botanical Society of South Africa, Kirstenbosch.

Waller, P.A (2013) *Towards ecological restoration strategies for Peninsula Shale Renosterveld: Testing the effects of disturbance-intervention treatments on seed germination on Devil's Peak, Cape Town',* Unpublished Masters Thesis, University of Cape Town, South Africa.

Wang, Y. Jiang, D. Toshio, O. Zhou, Q (2013) 'Recent advances in soil seed bank research', *Contemporary Problems of Ecology* (Volume 6): pp. 520-524.

Wang, H. Grant, W.E. Teague, R. (2020) 'Modelling rangelands as spatially explicit complex adaptive systems', *Journal of Environmental Management* (Volume 269): pp. 1-8.

Wessels, K.J. Prince, S.D. Carroll, M. Malherbe, J. (2007) 'Relevance of rangeland degradation in semiarid north-eastern South Africa to the nonequilibrium theory', *Ecological Applications* (Volume 17): pp. 815-827.

Western, D. Tyrrell, P. Brehony, P. Russell, S. Western, G. Kamanga, J. (2020) 'Conservation from the inside-out: Winning space and a place for wildlife in working landscapes', *People and Nature* (Volume 2): pp. 279-291.

Wickham, H (2016) ggplot2: Elegant Graphics for data analysis, Springer-Verlag, New York 2016. ISBN 978-3-319-24277-4. http://ggplot2.org.

Wickham, H (2018). stringr: Simple, Consistent Wrappers for Common String Operations. R package version 1.3.1.https://CRAN.R-project.org/package=stringr.

Wickham, H and Henry, L (2018) tidyr: Easily Tidy Data with "spread ()" and "gather ()" Functions. R package version 0.8.1. https://CRAN.R-project.org/package=tidyr.

Wickham, H. Romaine, R. Henry, L. Muller, K (2018) dplyr: A grammar of data manipulation. R package version 0.7.6. https://CRAN. R-project.org/packa.

Willis, K.J. (Ed) (2017) *State of the World's Plants 2017*, Report, Royal Botanic Gardens Kew.

Winter, S.J. Prozesky, H. Esler, K.J. (2007) 'A case study of landowner attitudes and behaviour towards the conservation of renosterveld, a critically endangered vegetation type in the Cape Floral Kingdom, South Africa', *Environmental Management* (Volume 40): pp. 46-61.

Yagil, O. Perevolotsky, A and Kigel, J (2006) 'Similarity between seed bank and vegetation in a semi-arid annual plant community: the role of productivity and grazing', *Journal of Vegetation Science* (Volume 17): pp. 29-36.

Yan, J. Yingying, W.U. Zhang, Y. (2011) 'Adaptation strategies for pasture degradation: Gap between government and local nomads in the eastern Tibetan Plateau', *Journal of Geographic Science* (Volume 21): pp. 1112-1122.

Zhigila, D.A. Verboom, A.G. Muasya, A.M. (2020) 'An infrageneric classification of *Thesium* (Santalaceae) based on molecular phylogenetic data', *Taxon* (Volume 69): pp. 100-123.

Appendices

Appendix 1

Table S3.1: Table showing results of one-way ANOVAs of diversity indices (Shannon-Wiener, Simpson and Inverse Simpson) from vegetation data collected from Modified Whittaker Plots with treatments of no grazing or grazed by cattle and sheep at differing grazing intensities.

Simpson

Contrast	Difference	SE	df	t	p	Cl_Lower	Cl_Higher
Cattle – None	-0.006	0.006	27	-0.967	0.603	-0.0201	0.009
Cattle – Sheep	0.008	0.003	27	2.212	0.0871	-0.001	0.016
None – Sheep	0.013	0.006	27	2.342	0.0669	-0.001	0.0275

Inverse Simpson

Contrast	Difference	SE	df	t	p	Cl_Lower	Cl_Higher
Cattle – None	-7.25	6.489	27	-1.117	0.5120	-23.341	8.841
Cattle – Sheep	7.95	3.894	27	2.042	0.122	-1.705	17.605
None – Sheep	15.2	6.359	27	2.390	0.0604	-0.566	30.966

Shannon – Wiener

Contrast	Difference	SE	df	t	p	Cl_Lower	Cl_Higher
Cattle – None	-0.199	0.186	27	-1.071	0.539	-0.661	0.262
Cattle – Sheep	0.239	0.112	27	2.145	0.099	-0.037	0.517
None – Sheep	0.439	0.182	27	2.407	0.058	-0.013	0.892

Appendix 2

Table S3.2: Results of ANOVAs on species area regression parameters showing variation in species richness by livestock type and spatial area from Modified Whittaker Plots in Overberg Renosterveld with grazing treatments of no grazing and grazing by cattle and sheep.

Contrast	Difference	SE	df	t	p	Cl_lower	Cl_higher
None – Cattle	0.011	0.006	27	1.660	0.239	-0.005	0.026
None – Sheep	0.016	0.006	27	2.677	0.032	0.001	0.031
Cattle – Sheep	0.006	0.004	27	1.605	0.261	-0.003	0.015

Appendix 3

Table S4.1: Species list from renosterveld vegetation in Modified Whittaker Plots from sites with no grazing (3), grazed by cattle (11) and sheep (16) and the soil seed bank, including information on plant growth form, family, red list status, proportion in plots with a treatment of no grazing, grazed by cattle or sheep and presence in seedbank study. Abbreviations used: GF = Growth form; CG = Renosterveld grazed by cattle; SG = Renosterveld grazed by sheep; NG = Renosterveld with a treatment of no grazing during the study; A = Annual; NSS = Non Succulent Shrub; G=Geophyte; PF = Perennial Forb; PG = Perennial Grass; SS = Succulent Shrub.

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Acanthaceae	*Barleria pungens* L.f.	NSS	Least Concern	0	0	33	×
Aizoaceae	*Delosperma neethingiae* (L. Bolus) Schwantes.	SS	Data Deficient	0	8	0	×
Aizoaceae	*Drosanthemum ambiguum* L.Bolus	SS	Least Concern	0	8	0	✓
Aizoaceae	*Drosanthemum asperulum* (Salm-Dyck) Schwantes	SS	Least Concern	13	25	33	✓
Aizoaceae	*Drosanthemum calycinum* (Haw.) Schwantes	SS	Near Threatened	0	8	0	✓
Aizoaceae	*Drosanthemum hispidum* (L.) Schwantes	SS	Vulnerable	33	42	33	✓
Aizoaceae	*Drosanthemum hispifolium* (Haw.) Schwantes	SS	Vulnerable	0	0	0	✓
Aizoaceae	*Drosanthemum parvifolium* (Haw.) Schwantes.	SS	Least Concern	7	0	33	✓
Aizoaceae	*Drosanthemum striatum* (Haw.) Schwantes	SS	Vulnerable	0	0	0	✓
Aizoaceae	*Erepsia villiersii* L. Bolus	SS	Critically Endangered	13	0	0	×
Aizoaceae	*Galenia africana* L.	NSS	Least Concern	7	25	0	✓
Aizoaceae	*Galenia secunda* (L.f.) Sond.	NSS	Least Concern	0	25	0	×
Aizoaceae	*Glottiphyllum longum* (Haw.) N.E.Br.	PF	Least Concern	0	25	0	×
Aizoaceae	*Lampranthus* 1	SS	N/A	0	0	0	✓
Aizoaceae	*Mestoklema tuberosum* (L.) N.E.Br. ex Glen	SS	Least Concern	0	8	0	×
Aizoaceae	*Ruschia tenella* (Haw.) Schwantes	SS	Least Concern	7	8	0	✓
Aizoaceae	*Ruschia vaginata* (Haw.) Schwantes	SS	Least Concern	7	0	0	×
Alliaceae	*Tulbaghia capensis* L.	G	Least Concern	7	0	0	×
Amaranthaceae	Amaranthaceae 1	A	N/A	0	0	0	✓
Amaranthaceae	Amaranthaceae 2	A	N/A	0	0	0	✓

Table continued on next page

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Amaranthaceae	Amaranthaceae 3	NSS	N/A	0	0	0	✓
Amaranthaceae	Amaranthaceae 4	NSS	N/A	0	0	0	✓
Amaranthaceae	*Atriplex semibaccata* R.Br.	NSS	Least Concern	0	17	0	✓
Amaryllidaceae	*Boophone disticha* L.f.Herb.	G	Least Concern	0	17	67	×
Anacardiaceae	*Searsia laevigata* (L.) F.A.Barkley	NSS	Least Concern	40	33	33	×
Anacardiaceae	*Searsia lucida* (L.) F.A.Barkley	NSS	Least Concern	7	0	0	×
Anacardiaceae	*Searsia rosmarinifolia* (Vahl) F.A.Barkley	NSS	Least Concern	7	8	0	×
Apiaceae	Apiaceae 1	A	N/A	0	0	0	✓
Apiaceae	*Arctopus echinatus* L.	G	Least Concern	13	8	33	×
Apiaceae	*Centella* 1	NSS	N/A	0	0	0	✓
Apocynaceae	*Gomphocarpus cancellatus* (Burm.f.) Bruyns	NSS	Least Concern	7	0	0	×
Apocynaceae	*Microloma sagittatum* (L.) R.Br.	NSS	Least Concern	7	17	67	×
Apocynaceae	*Microloma tenuifolium* (L.) K.Schum	NSS	Least Concern	20	0	33	×
Asparagaceae	*Asparagus capensis* L. var. *capensis*	NSS	Least Concern	67	100	100	×
Asphodelaceae	*Aloe ferox* Mill.	SS	Least Concern	0	8	0	×
Asphodelaceae	*Bulbine mesembryanthoides* Haw. subsp. *mesembryanthoides*	PF	Least Concern	0	0	33	×
Asphodelaceae	*Haworthia mutica* Haw.	PF	Data Deficient	0	17	0	×
Asphodelaceae	*Trachyandra ciliata* (L.f) Kunth	G	Least Concern	0	0	0	×
Asteraceae	*Arctotheca calendula* (L.) Levyns	PF	Least Concern	0	0	0	✓
Asteraceae	*Arctotheca populifoliia* (P.J. Bergius) Norl.	PF	Least Concern	0	0	0	✓
Asteraceae	*Arctotheca prostrata* (Salis.) Britten	PF	Least Concern	0	0	0	✓
Asteraceae	*Arctotis acaulis* L.	PF	Least Concern	33	50	67	✓
Asteraceae	*Arctotis angustifolia* Eckl. & Zeyh.	PF	Least Concern	0	0	0	✓
Asteraceae	Asteraceae 1	A	N/A	0	0	0	✓
Asteraceae	Asteraceae 10	A	N/A	0	0	0	✓

Table continued on next page

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Asteraceae	Asteraceae 11	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 12	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 13	A	N/A	0	0	0	✓
Asteraceae	Asteraceae 14	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 15	A	N/A	0	0	0	✓
Asteraceae	Asteraceae 2	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 3	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 4	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 5	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 6	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 7	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 8	NSS	N/A	0	0	0	✓
Asteraceae	Asteraceae 9	NSS	N/A	0	0	0	✓
Asteraceae	*Athanasia* 1	NSS	N/A	0	0	0	✓
Asteraceae	*Athanasia dentata* (L.) L.	NSS	Least Concern	0	0	0	✓
Asteraceae	*Athanasia juncea* (DC.) D.Dietr	NSS	Least Concern	7	8	33	✓
Asteraceae	*Athanasia trifurcata* (L.) L.	NSS	Least Concern	13	17	0	✓
Asteraceae	*Berkheya armata* (Vahl) Druce	PF	Least Concern	27	25	33	×
Asteraceae	*Berkheya rigida* (Thunb.) Erwart, Jean White & B.Rees	A	Least Concern	27	58	0	✓
Asteraceae	*Chrysocoma ciliata* L.	NSS	Least Concern	33	50	67	×
Asteraceae	*Cotula* 1	A	N/A	0	0	0	✓
Asteraceae	*Cotula* 2	A	N/A	0	0	0	✓
Asteraceae	*Cotula ceniifolia* DC.	PF	Least Concern	0	0	33	✓
Asteraceae	*Cotula turbinata* L.	A	Least Concern	0	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Asteraceae	*Cotula zeyheri* Fenzl	A	Least Concern	20	92	0	✓
Asteraceae	*Curio crassulifolius* (D.C.) P.V.Heath	SS	Data Deficient	7	8	33	×
Asteraceae	*Cymbopappus adenosolen* (Harv.) B.Nord.	NSS	Least Concern	0	8	0	✓
Asteraceae	*Dimorphotheca nudicaulis* (L.) DC. var. *nudicaulis*	A	Least Concern	0	8	0	×
Asteraceae	*Dimorphotheca pluvialis* (L.)	A	Least Concern	0	8	0	×
Asteraceae	*Dicerothamnus rhinocerotis* (L.f.) Less.	NSS	Least Concern	73	100	100	✓
Asteraceae	*Erigeron bonariensis* L.	A	Exotic naturalised weed	0	0	0	✓
Asteraceae	*Eriocephalus africanus* L. var. *africanus*	NSS	Least Concern	27	25	0	×
Asteraceae	*Euryops abrotanifolius* (L.) DC.	NSS	Least Concern	13	0	0	✓
Asteraceae	*Felicia flifolia* (Vent.) Burtt Davy subsp. *filifolia*	NSS	Least Concern	13	8	0	×
Asteraceae	*Felicia minima* (Hutch.) Grau	A	Least Concern	27	25	0	✓
Asteraceae	*Gazania cilliaris* DC.	PF	Least Concern	0	0	33	×
Asteraceae	*Gazania krebsiana* Less.	PF	Least Concern	33	67	33	✓
Asteraceae	*Gorteria personata* L. subsp. *personata*	A	Least Concern	0	8	0	×
Asteraceae	*Helichrysum* 1	NSS	N/A	0	0	0	✓
Asteraceae	*Helichrysum* 2	NSS	N/A	0	0	0	✓
Asteraceae	*Helichrysum* 3	NSS	N/A	0	0	0	✓
Asteraceae	*Helichrysum asperum* (Thunb.) Hilliard & B.L.Burtt	NSS	Least Concern	0	25	0	✓
Asteraceae	*Helichrysum cymosum* (L.) D.Don	NSS	Least Concern	0	8	0	✓
Asteraceae	*Helichrysum helianthemifolium* (L.) D.Don	NSS	Least Concern	0	8	0	✓
Asteraceae	*Helichrysum petiolare* Hilliard & B.L.Burtt	NSS	Least Concern	40	50	33	✓
Asteraceae	*Helichrysum rosum* (P.JBergius) Less. var. *rosum*	NSS	Least Concern	40	42	33	✓
Asteraceae	*Hypochaeris radicata* L.	A	Exotic naturalised weed	0	8	0	✓
Asteraceae	*Leysera gnaphalodes* (L.) L.	NSS	Least Concern	0	8	0	×

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Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Asteraceae	*Macledium spinosum* (L.) S.Ortiz	NSS	Least Concern	0	17	0	×
Asteraceae	*Metalasia acuta* P.O.Karis	NSS	Least Concern	27	8	0	✓
Asteraceae	*Oedera genistifolia* (L.) Anderb. & K.Bremer	NSS	Least Concern	7	17	33	×
Asteraceae	*Oedera squarrosa* (L.) Anderb & K.Bremer	NSS	Least Concern	47	33	33	✓
Asteraceae	*Oedera uniflora* (L.f) Ander. & K.Bremer	NSS	Least Concern	33	25	0	✓
Asteraceae	*Oncosiphon piluliferum* (L.f) Kallersjo	NSS	Least Concern	0	0	0	✓
Asteraceae	*Osteospermum calendulaceum* L.f.	A	Least Concern	0	0	0	✓
Asteraceae	*Osteospermum moniliferum* L. subsp. *moniliferum*	NSS	Least Concern	0	0	33	×
Asteraceae	*Osteospermum tomentosum* (L.f.) Nori	PF	Least Concern	20	50	0	×
Asteraceae	*Pentzia incana* (Thunb.) Kuntze	NSS	Least Concern	0	8	0	×
Asteraceae	*Printzia polifolia* (L.) Hutch.	NSS	Least Concern	13	8	33	×
Asteraceae	*Pteronia incana* (Burm.) DC.	NSS	Least Concern	13	17	0	×
Asteraceae	*Relhania garnottii* (Less.) K.Bremer	NSS	Vulnerable	27	17	0	×
Asteraceae	*Senecio* 1	NSS	N/A	0	0	0	✓
Asteraceae	*Senecio laevigatus* Thunb. var. *laevigatus*	NSS	Least Concern	0	8	0	✓
Asteraceae	*Senecio pterophorus* DC.	NSS	Least Concern	0	0	0	✓
Asteraceae	*Senecio purpureus* L.	NSS	Least Concern	0	0	33	×
Asteraceae	*Senecio speciosus* Willd.	NSS	Least Concern	0	0	33	×
Asteraceae	*Sonchus oleraceus* (L.) L.	A	Exotic naturalised weed	33	25	0	✓
Asteraceae	*Ursinia anthemoides* (L.) Poir. subsp. *anthemoides*	NSS	Least Concern	0	8	0	✓
Asteraceae	*Ursinia discolor*	NSS	Least Concern	13	8	33	✓
Asteraceae	*Ursinia nana* DC. subsp. *nana*	NSS	Least Concern	27	58	0	×

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Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Boraginaceae	*Lobostemon echioides* Lehm.	NSS	Least Concern	0	8	0	×
Boraginaceae	*Lobostemon fruticosus* (L.) H.Buek	NSS	Least Concern	7	0	0	×
Brassicaceae	*Heliophila* 1	A	N/A	7	25	0	✓
Brassicaceae	*Heliophila* 2	A	N/A	0	0	0	✓
Brassicaceae	*Lepedium africanum* (Burm.f.) DC. subsp. *africanum*	NSS	Least Concern	0	25	0	×
Brassicaceae	*Raphanus raphanistrum* L.	A	Exotic Naturalised Weed	20	25	0	×
Campanulaceae	*Prismatocarpus pedunculatus* (P.J. Berguis)	NSS	Least Concern	0	8	0	✓
Campanulaceae	*Wahlenbergia thunbergii* (Schult) B. Nord. var. *thunbergii*	NSS	Least Concern	33	25	0	✓
Caryophyllaceae	Caryophyllaceae 1	A	N/A	0	0	0	✓
Caryophyllaceae	*Dianthus caespitosus* Thunb. *caespitosus*	NSS	Least Concern	0	17	0	×
Caryophyllaceae	*Silene gallica* L.	A	Exotic naturalised weed	20	33	0	✓
Colchicaceae	*Wurmbea marginata* (Desr) B.Nord.	G	Least Concern	27	0	0	×
Convulvulaceae	*Falkia repens* Thunb.	PF	Least Concern	13	42	0	✓
Crassulaceae	*Adromischus triflorus* (L.f) A.Berger	SS	Least Concern	7	17	0	×
Crassulaceae	*Crassula tetragona* L. subsp. *tetragona*	PF	Least Concern	20	8	0	✓
Crassulaceae	*Tylecodon* 1	SS	N/A	0	0	33	×
Cyperaceae	*Ficinia* 1	PF	N/A	0	0	0	✓
Cyperaceae	*Ficinia filiformis* (Lam.) Schrad.	PF	Least Concern	0	0	0	✓
Cyperaceae	*Ficinia gracilis* Schrad.	PF	Least Concern	20	67	33	×
Cyperaceae	*Ficinia indica* (Lam.) Pfeiff.	PF	Least Concern	0	0	0	✓
Cyperaceae	*Ficinia nigrescens* (Schrad.) J.Raynal	PF	Least Concern	27	0	67	×
Cyperaceae	*Ficinia overbergensis* Muasya & C.H.Stirt.	PF	Near Threatened	13	8	33	×
Cyperaceae	*Ficinia tristachya* (Rottb.) Nees	PF	Least Concern	0	0	0	✓
Cyperaceae	*Isolepis marginata* (Thunb.) A.Dietr.	PF	Least Concern	0	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Juncaceae	*Juncus oxycarpus* E. Mey. ex Kunth	PF	Least Concern	0	0	0	✓
Cyperaceae	*Pycreus polystachyos* (Rottb.) P.Beauv. var. *polystachyos*	PF	Least Concern	0	8	0	✓
Dipsacaceae	*Scabiosa africana* L.	NSS	Least Concern	7	0	0	×
Droseraceae	*Drosera cistiflora* L.	PF	Least Concern	7	8	0	×
Droseraceae	*Drosera trinervia* Spreng.	PF	Least Concern	0	0	33	×
Ericaceae	*Erica* 1	NSS	N/A	13	8	0	✓
Euphorbiaceae	*Clutia govaertsii* Radcl. Sm.	NSS	Least Concern	0	8	0	×
Euphorbiaceae	*Clutia tomentosa* L.	NSS	Least Concern	20	8	100	×
Euphorbiaceae	*Euphorbia peplus* L.	A	Exotic naturalised weed	0	0	0	✓
Euphorbiaceae	*Euphorbia tuberosa* L.	PF	Least Concern	13	50	33	×
Fabacaeae	*Trifolium uniflorum* L.	A	Exotic naturalised weed	0	0	0	×
Fabaceae	*Argyrolobium pachyphyllum* Schltr.	NSS	Endangered	7	0	0	×
Fabaceae	*Aspalathus alpestris* (Benth.) R.Dahlgren	NSS	Least Concern	40	17	100	×
Fabaceae	*Aspalathus lebeckioides* R.Dahlgren	NSS	Vulnerable	0	0	33	×
Fabaceae	*Aspalathus nigra* L.	NSS	Least Concern	47	42	67	×
Fabaceae	*Aspalathus spinosa* L. subsp. *spinosa*	NSS	Least Concern	13	8	67	×
Fabaceae	*Aspalathus steudeliana* Brongn.	NSS	Vulnerable	27	25	33	×
Fabaceae	*Aspalathus submissa* R.Dahlgren	NSS	Least Concern	7	17	0	×
Fabaceae	*Indogofera heterophylla* Thunb.	PF	Least Concern	67	100	100	×
Fabaceae	*Lessertia capensis* (P.J.Bergius) Druce	NSS	Least Concern	0	8	0	×
Fabaceae	*Lessertia frutescens* (L.) Goldblatt & J.C.Manning subsp. frutescens	NSS	Least Concern	0	25	33	×
Fabaceae	*Medicago polymorpha* L.	A	Exotic naturalised weed	13	50	0	✓
Fabaceae	*Medicago sativa* L.	A	Exotic naturalised weed	7	0	0	×
Juncaceae	*Juncus oxycarpus* E..Mey. ex Kunth	PF	Least Concern	0	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Fabaceae	*Otholobium candicans* (Eckl. & Zeyh.) C.H.Stirt.	NSS	Least Concern	7	0	0	×
Fabaceae	*Polhillia pallens* C.H.Stirt.	NSS	Vulnerable	7	0	0	×
Fabaceae	*Trifolium burchellianum* Ser.	A	Exotic naturalised weed	7	8	33	✓
Fabaceae	*Vicia sativa* L. subsp. *sativa*	A	Exotic naturalised weed	0	8	0	×
Gentianaceae	Gentianaceae 1	A	N/A	0	0	0	✓
Gentianaceae	Gentianaceae 2	A	N/A	0	0	0	✓
Gentianaceae	Gentianaceae 3	A	N/A	0	0	0	✓
Gentianaceae	Gentianaceae 4	A	N/A	0	0	0	✓
Gentianaceae	*Sebaea aurea* (L.f.) Roem, & Schult.	G	Least Concern	7	25	67	×
Geraniaceae	*Pelargonium* 1	NSS	N/A	0	0	0	✓
Geraniaceae	*Pelargonium* 2	NSS	N/A	7	0	33	✓
Geraniaceae	*Pelargonium lobatum* (Burm.f.) L'Her	G	Least Concern	7	17	67	×
Haemadoraceae	*Wachendorfia paniculata* Burn.	G	Least Concern	0	8	0	×
Hyacinthaceae	*Albuca setosa* Jacq.	G	Least Concern	20	0	33	×
Hyacinthaceae	*Drimia capensis* (Burm.f.) Wijnands	G	Least Concern	27	25	0	×
Hyacinthaceae	*Lachenalia fistulosa* Baker	G	Vulnerable	0	8	0	×
Hyacinthaceae	*Lachenalia lutea* G.D.Duncan	G	Least Concern	7	0	0	×
Hyacinthaceae	*Lachenalia unifolia* Jacq.	G	Least Concern	0	17	0	×
Hyacinthaceae	*Ledebouria revoluta* (L.f.) Jessop	G	Least Concern	7	50	33	×
Hyacinthaceae	*Ornithogalum dubium* Houtt.	G	Least Concern	13	17	33	×
Hypoxidaceae	*Empodium plicatum* (Thunb.) Gardside	G	Least Concern	7	0	0	×

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Iridaceae	*Aristea africana* (L.) Hoffmanns.	G	Least Concern	0	8	0	×
Iridaceae	*Babiana* 1	G	N/A	13	0	0	×
Iridaceae	*Babiana* 2	G	N/A	0	8	0	×
Iridaceae	*Babiana ambigua* (Roem. & Schult.) G.J.Lewis	G	Least Concern	0	8	0	×
Iridaceae	*Babiana patula* N.E.Br.	G	Least Concern	0	17	0	×
Iridaceae	*Freesia caryophyllacea* (Burm.f.) N.E. Br.	G	Near Threatened	33	67	33	×
Iridaceae	*Freesia refracta* (Jacq.) Klatt	G	Least Concern	0	8	0	×
Iridaceae	*Geissorhiza ovata* (Burm.f.) Asch & Graebn.	G	Least Concern	0	8	33	×
Iridaceae	*Gladiolus graciis* Jacq.	G	Least Concern	13	8	0	×
Iridaceae	*Gladiolus lilliaceus* Houtt.	G	Least Concern	0	0	33	×
Iridaceae	*Hesperantha falcata* (L.f.) Ker Gawl.	G	Least Concern	0	8	67	×
Iridaceae	Iridaceae 1	G	N/A	0	0	0	✓
Iridaceae	*Ixia micrandra* Baker	G	Near Threatened	13	8	0	×
Iridaceae	*Lapeirousia pyramidalis* (Lam.) Goldblatt subsp. *pyramidalis*	G	Least Concern	7	17	0	×
Iridaceae	*Micranthus tubulosus* (Burm.) N.E.Br.	G	Least Concern	0	8	0	×
Iridaceae	*Moraea comptonii* (L. Bolus) Goldblatt	G	Endangered	0	8	0	×
Iridaceae	*Moraea debilis* Goldblatt	G	Endangered	7	8	0	×
Iridaceae	*Moraea gawleri* Spreng.	G	Least Concern	7	8	0	×
Iridaceae	*Moraea miniata* Andrews	G	Least Concern	7	25	0	×
Iridaceae	*Moraea tripetala* (L.f.) Ker Gawl. subsp. *tripetala*	G	Least Concern	0	0	33	×
Iridaceae	*Romulea rosea* (L.) Eckl. var. *rosea*	G	Least Concern	0	17	0	×

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Iridaceae	*Tritoniopsis antholyza* (Poir.) Goldblatt	G	Least Concern	0	8	0	×
Iridaceae	*Watsonia aletroides* (Burm.f) Ker Gawl.	G	Near Threatened	7	17	33	×
Lamiaceae	Lamiaceae 1	PF	N/A	0	0	0	✓
Lamiaceae	Lamiaceae 2	NSS	N/A	0	0	0	✓
Lamiaceae	*Stachys aethiopica* L.	NSS	Least Concern	0	0	33	×
Linaceae	*Linum africanum* L.	NSS	Least Concern	0	0	0	✓
Lobeliaceae	*Cyphia digitata* (Thunb.) Willd. subsp. *digitata*	PF	Least Concern	7	0	0	×
Lobeliaceae	*Cyphia longipetala* C. Presl.	PF	Least Concern	0	8	33	×
Lobeliaceae	*Cyphia phyteuma* (L.) Willd. var. *phyteuma*	PF	Least Concern	0	8	0	×
Lobeliaceae	*Lobelia* 1	PF	N/A	0	0	0	✓
Lobeliaceae	*Lobelia erinus* L.	PF	Least Concern	20	17	33	✓
Malvaceae	*Hermannia alnifolia* L.	NSS	Least Concern	0	0	33	×
Malvaceae	*Hermannia althaeifolia* L.	NSS	Least Concern	0	0	0	✓
Malvaceae	*Hermannia confusa* T.M.Salter	NSS	Least Concern	7	8	0	×
Malvaceae	*Hermannia cuneifolia* Jacq. var. *cuneifolia*	NSS	Least Concern	0	0	33	×
Malvaceae	*Hermannia diversistipula* C.Presl ex Harv. var. *diversistipula*	NSS	Least Concern	0	17	0	×
Malvaceae	*Hermannia flammea* Jacq.	NSS	Least Concern	7	0	0	×
Malvaceae	*Hermannia flammula* Harv.	NSS	Least Concern	33	25	67	✓
Malvaceae	*Hermannia hyssopifolia* L.	NSS	Least Concern	0	8	0	×
Malvaceae	*Hermannia lavandulifolia* L.	NSS	Vulnerable	0	8	33	×
Malvaceae	*Hermannia saccifera* (Turcz.) K.Schum.	NSS	Least Concern	0	17	0	×

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Malvaceae	*Hermannia ternifolia* C.Presl. ex Harv.	NSS	Least Concern	7	8	0	×
Malvaceae	*Hermannia trifoliata* L.	NSS	Least Concern	13	8	33	×
Malvaceae	*Hibiscus aethiopicus* L. var. *aethiopicus*	PF	Least Concern	7	0	0	×
Malvaceae	*Hibiscus pusillus* Thunb.	PF	Least Concern	47	50	100	×
Molluginaceae	Molluginaceae 1	A	N/A	0	0	0	✓
Montinaceae	*Montinia caryophyllacea* Thunb.	NSS	Least Concern	7	0	33	×
N/A	Dicot 1	NSS	N/A	0	0	0	✓
N/A	Dicot 2	A	N/A	0	0	0	✓
N/A	Dicot 3	NSS	N/A	0	0	0	✓
N/A	Dicot 4	NSS	N/A	0	0	0	✓
Orchidaceae	*Disa bracteata* Sw.	PF	Least Concern	7	0	33	×
Orchidaceae	*Disperis bolusiana* Schltr. ex Bolus subsp. *bolusiana*	PF	Least Concern	0	0	33	×
Orchidaceae	*Disperis villosa* (L.f.) Sw.	PF	Least Concern	0	8	0	×
Orchidaceae	*Pterygodium catholicum* (L.) Sw.	PF	Least Concern	0	0	33	×
Orchidaceae	*Pterygodium orobanchoides* (L.f.) Schltr.	PF	Least Concern	0	8	0	×
Orchidaceae	*Pterygodium orobanchoides* (L.f.) Schltr.	PF	Least Concern	0	0	0	×
Orchidaceae	*Satyrium erectum* Sw.	PF	Least Concern	7	8	0	×
Orobanchaceae	*Hyobanche sanguinea* L.	PF	Least Concern	27	8	33	×
Oxalidaceae	*Oxalis 1*	G	N/A	27	58	67	×
Oxalidaceae	*Oxalis 2*	G	N/A	33	33	100	×
Oxalidaceae	*Oxalis 3*	G	N/A	20	42	100	×
Oxalidaceae	*Oxalis 4*	G	N/A	7	25	0	×
Oxalidaceae	*Oxalis* 5	G	N/A	7	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Oxalidaceae	*Oxalis hirsuta* Sond.	G	Data Deficient	0	8	33	×
Oxalidaceae	*Oxalis multicaulis* Eckl. & Zeyh.	G	Least Concern	0	0	33	×
Oxalidaceae	*Oxalis obtusa* Jacq.	G	Least Concern	0	8	0	×
Oxalidaceae	*Oxalis pers-caprae* L. var. *pers-caprae*	G	Least Concern	0	8	0	×
Oxalidaceae	*Oxalis polyphylla* Jacq. var. *polyphylla*	G	Least Concern	0	0	33	×
Oxalidaceae	*Oxalis purpurea* L.	G	Least Concern	13	33	67	×
Oxalidaceae	*Oxalis truncatula* Jacq.	G	Least Concern	7	0	33	×
Plantaginaceae	*Plantago lanceolata* L.	A	Exotic naturalised weed	13	42	0	✓
Poaceae	*Briza maxima* L.	A	Exotic naturalised weed	0	17	67	×
Poaceae	*Briza minor* L.	A	Exotic naturalised weed	0	25	0	✓
Poaceae	*Chloris gayana* Kunth.	PG	Exotic naturalised weed	0	8	0	×
Poaceae	*Cymbopogon marginatus* (Steud.) Stapf. ex Burtt Davy	PG	Least Concern	13	42	67	×
Poaceae	*Cynodon dactylon* (L.) Pers.	PG	Least Concern	0	8	0	✓
Poaceae	*Digitaria sanguinalis* (L.) Scop.	A	Exotic naturalised weed	13	8	0	×
Poaceae	*Ehrarta calycina* Sm.	PG	Least Concern	27	92	67	✓
Poaceae	*Eragrostis curvula* (Schrad.) Nees	PG	Least Concern	7	8	0	×
Poaceae	*Hyparrhenia hirta* (L.) Stapf	PG	Least Concern	7	0	0	✓
Poaceae	*Lolium multiflorum* Lam.	A	Exotic naturalised weed	27	50	0	✓
Poaceae	*Pentameris eriostoma* (Nees) Steud.	PG	Least Concern	33	17	0	×
Poaceae	*Poa annua* L.	A	Exotic naturalised weed	20	25	0	✓
Poaceae	Poaceae 1	PG	N/A	0	0	0	✓
Poaceae	Poaceae 10	A	N/A	0	0	0	✓
Poaceae	Poaceae 11	PG	N/A	0	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Poaceae	Poaceae 2	PG	N/A	0	0	0	✓
Poaceae	Poaceae 3	PG	N/A	0	0	0	✓
Poaceae	Poaceae 4	A	N/A	0	0	0	✓
Poaceae	Poaceae 5	PG	N/A	0	0	0	✓
Poaceae	Poaceae 6	A	N/A	0	0	0	✓
Poaceae	Poaceae 7	A	N/A	0	0	0	✓
Poaceae	Poaceae 8	A	N/A	0	0	0	✓
Poaceae	Poaceae 9	PG	N/A	0	0	0	✓
Poaceae	*Tenaxia stricta* (Schrad.) N.P.Barker & H.P.Linder	PG	Least Concern	0	8	0	✓
Poaceae	*Themeda triandra* Forssk.	PG	Least Concern	53	100	100	✓
Polygalaceae	*Polygala garcinii* DC.	NSS	Least Concern	27	42	33	✓
Polygalaceae	Polygalaceae 1	NSS	N/A	0	0	0	✓
Primulaceae	*Anagallis arvensis* L.	A	Exotic naturalised weed	33	67	33	✓
Resedaceae	*Oligomeris* 1	A	N/A	0	0	0	✓
Restionaceae	Restionaceae 1	PF	N/A	13	0	0	×
Restionaceae	Restionaceae 2	PF	N/A	7	8	0	×
Restionaceae	Restionaceae 3	PF	N/A	0	0	0	✓
Rosaceae	*Cliffortia filicaulis* Schltdl. var. *filicaulis*	NSS	Least Concern	0	8	0	×
Rosaceae	*Cliffortia ruscifolia* L.	NSS	Least Concern	0	8	0	×
Rubiaceae	*Anthospermum* 1	NSS	N/A	0	0	0	✓
Rubiaceae	*Anthospermum* 2	NSS	N/A	0	0	0	✓
Rubiaceae	*Anthospermum aethiopicum* L.	NSS	Least Concern	20	0	0	✓

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Rubiaceae	*Anthospermum paniculatum* Cruse	NSS	Least Concern	13	8	0	×
Rubiaceae	*Anthospermum prostratum* Sond.	PF	Least Concern	60	83	100	✓
Rubiaceae	*Gallium* 1	NSS	N/A	0	0	0	✓
Ruscaceae	*Eriospermum* 1	G	N/A	40	33	100	×
Ruscaceae	*Eriospermum* 2	G	N/A	0	33	0	×
Rutaceae	*Diosma* 1	NSS	N/A	7	0	0	×
Santacaeae	*Thesium* 2	PF	N/A	33	25	33	×
Santalaceae	*Thesium* 3	PF	N/A	0	8	33	×
Santalaceae	*Thesium* 4	PF	N/A	13	0	0	×
Santalaceae	*Thesium* 5	PF	N/A	7	0	0	×
Scrophulariaceae	*Chaenostoma calcifilum* (Hilliard) Kornhall	NSS	Least Concern	0	0	0	✓
Scrophulariaceae	*Chaenostoma hispidum* (Thunb.) Benth.	NSS	Least Concern	47	25	33	✓
Scrophulariaceae	*Freylinia undulata* (L.f.) Benth.	NSS	Least Concern	13	8	0	×
Scrophulariaceae	*Oftia africana* (L.)	NSS	Least Concern	7	0	0	×
Scrophulariaceae	Scrophulariaceae 1	NSS	N/A	0	0	0	✓
Scrophulariaceae	Scrophulariaceae 2	NSS	N/A	0	0	0	✓
Scrophulariaceae	*Selago corymbosa* L.	NSS	Least Concern	33	8	0	✓
Scrophulariaceae	*Selago fruticosa* L.	NSS	Least Concern	13	25	33	✓
Solanaceae	*Lyceum afrum* L.	NSS	Least Concern	0	8	33	×
Solanaceae	Solanaceae 1	NSS	N/A	0	0	0	✓
Tecophilaeaceae	*Cyanella hyacynthoides* L.	G	Least Concern	7	8	0	×
Tecophilaeaceae	*Cyanella lutea* L.f. subsp. *lutea*	G	Least Concern	20	58	33	×
Thymeliaceae	*Gnidia laxa* (L.f) Gilg	NSS	Least Concern	40	50	67	×

Table continued on next page.

Table 1: (Continued)

Family	Species Name	GF	Status	NG	CG	SG	Seedbank
Thymeliaceae	*Gnidia scabra* Thunb.	NSS	Least Concern	0	8	0	×
Thymeliaceae	*Gnidia sericea* L. var. *sericea*	NSS	Least Concern	7	0	0	×
Thymeliaceae	Thymeliaceae 1	NSS	N/A	0	8	0	✓
Thymeliaceae	Thymeliaceae 2	NSS	N/A	0	0	0	✓
Thymeliaceae	Thymeliaceae 3	NSS	N/A	0	0	0	✓

Appendix 4

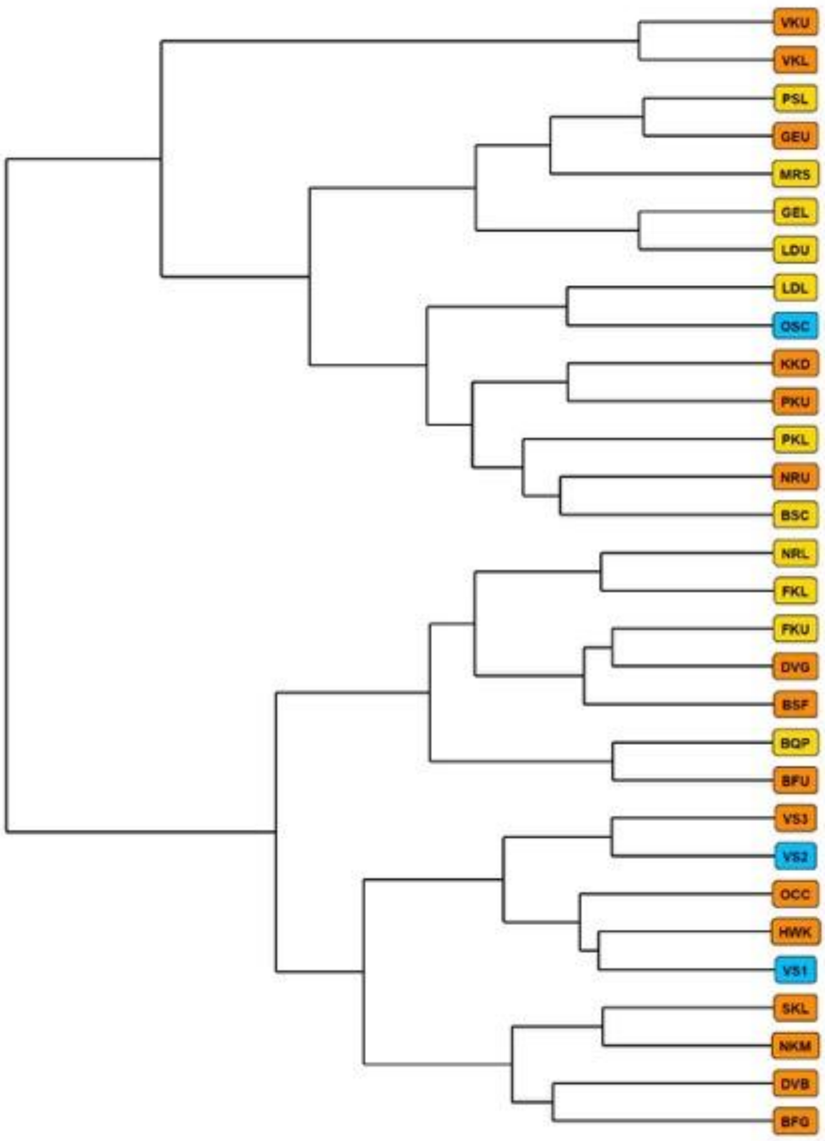

Figure S4.1: Hierarchical clustering dendrogram based on Bray Curtis distances between sites using presence only at renosterveld sites with a treatment of no grazing (blue), grazed by cattle (yellow) or grazed by sheep (orange).

Appendix 5

Table S5.1: Linear Mixed Model on relative growth form cover within each grazing treatment followed by post hoc tests to determine at which time step the differences lie.

	NONE-CATTLE			NONE-SHEEP			CATTLE-SHEEP		
	Z-value	**adj. p-value**	**Significance**	**Z-value**	**adj. p-value**	**Significance**	**Z-value**	**adj. p-value**	**Significance**
Annuals									
T0	-0.183	0.981	n.s.	1.208	0.432	n.s.	0.801	0.689	n.s.
T1	-0.212	0.974	n.s.	0.552	0.838	n.s.	0.474	0.877	n.s.
T2	-0.293	0.951	n.s.	0.066	0.997	n.s.	0.286	0.953	n.s.
T3	-1.411	0.327	n.s.	-1.263	0.408	n.s.	0.506	0.865	n.s.
T4	-1.328	0.374	n.s.	-1.313	0.382	n.s.	0.377	0.924	n.s.
Geophytes									
T0	-1.337	0.373	n.s.	-0.938	0.615	n.s.	0.344	0.936	n.s.
T1	-1.532	0.274	n.s.	-1.273	0.409	n.s.	0.223	0.973	n.s.
T2	-2.183	0.0725	n.s.	-1.853	0.1495	n.s.	0.264	0.9618	n.s.
T3	-1.668	0.214	n.s.	-1.613	0.237	n.s.	0.045	0.999	n.s.
T4	-2.609	0.02385	*	-3.016	0.00676	**	-0.312	0.94635	n.s.
Perennial Forbs									
T0	-0.123	0.9913	n.s.	2.241	0.0612	n.s.	1.670	0.2071	n.s.
T1	-0.483	0.875	n.s.	-0.338	0.936	n.s.	0.118	0.992	n.s.
T2	-1.464	0.297	n.s.	-1.711	0.193	n.s.	-0.128	0.991	n.s.
T3	-1.596	0.24078	n.s.	-3.431	0.00148	**	-1.324	0.37421	n.s.
T4	-1.247	0.4119	n.s.	-3.372	0.0021	**	-1.451	0.3025	n.s.
Perennial Grasses									
T0	-0.307	0.948	n.s.	0.858	0.661	n.s.	0.891	0.640	n.s.
T1	-0.695	0.761	n.s.	-0.686	0.766	n.s.	0.028	1.000	n.s.
T2	-1.661	0.217	n.s.	-1.531	0.273	n.s.	0.152	0.987	n.s.

T3	-2.358	0.04715	*	-3.488	0.00126	**	-0.829	0.68170	n.s.
T4	-2.349	0.04839	*	-3.494	0.00145	**	-0.840	0.67424	n.s.
Succulent Shrubs									
T0	-1.488	0.293	n.s.	-0.989	0.580	n.s.	0.465	0.886	n.s.
T1	-1.692	0.0906	n.s.	-	-	-	-	-	-
T2	-2.01	0.0445	*	-	-	-	-	-	-
T3	-1.713	0.0867	n.s.	-	-	-	-	-	-
T4	-1.419	0.156	n.s.	-	-	-	-	-	-
Non-Succulent Shrubs									
T0	-0.322	0.943	n.s.	-0.372	0.924	n.s.	-0.027	1.000	n.s.
T1	-0.511	0.863	n.s.	-1.518	0.276	n.s.	-0.754	0.726	n.s.
T2	-0.351	0.93244	n.s.	-3.166	0.00413	**	-2.093	0.08824	n.s.
T3	-0.267	0.96038	n.s.	-3.506	0.00119	**	-2.424	0.03935	*
T4	-0.306	0.9486	n.s.	-3.765	<0.001	***	-2.596	0.0247	*

Appendix 6

Table S5.2: Tables showing results from post hoc Tukey HSD test on total vegetation cover within each time step followed by post-hoc tests to determine the differences between treatments (Significance: * p<0.05, ** p<0.01, *** p<0.001).

NONE – CATTLE

Time Step	Z-value	adj. p-value	Significance
T0	-0.891	0.638	n.s.
T1	-1.520	0.273	n.s.
T2	-2.137	0.081	n.s.
T3	-2.230	0.06527	n.s.
T4	-2.650	0.0216	*

NONE – SHEEP

Time Step	Z-value	adj. p-value	Significance
T0	0.992	0.573	n.s.
T1	-1.871	0.142	n.s.
T2	-3.707	<0.001	***
T3	-5.860	<0.001	***
T4	-6.161	<0.001	***

CATTLE – SHEEP

Time Step	Z-value	adj. p-value	Significance
T0	1.420	0.322	n.s.
T1	-0.210	0.975	n.s.
T2	-1.191	0.454	n.s.
T3	-2.926	0.00935	**
T4	-2.888	0.0107	*