The effect of rehabilitation on ecosystem services in the semi-arid Succulent Karoo lowlands of the Little Karoo, South Africa

Petra de Abreu

DST/NRF Centre of Excellence at the
Percy FitzPatrick Institute of African Ornithology
University of Cape Town, Rondebosch,
South Africa 7701

March 2011

Dissertation presented in partial fulfilment of the requirements for the degree of Master of Science in the Department of Zoology, Faculty of Science, University of Cape Town.

Supervisors:
Prof. Sue Milton, Renu-Karoo Veld Restoration
Prof. Timm Hoffman, Plant Conservation Unit (UCT)
Dr. David Le Maitre, Council for Scientific and Industrial Research (CSIR)

Declaration

"I know the meaning of plagiarism and declare that all of the work in the dissertation, save for that which is properly acknowledged, is my own"

Petra de Abreu   March 2011

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Abstract

Ecosystem services ('the benefits that humans obtain from ecosystems') represent a significant contribution to sustaining human well-being. Conservation efforts worldwide are being increasingly focussed outside formally protected areas in order to ensure the protection of biodiversity and ecosystem services. Highlighting ecosystem services attained through the rehabilitation of degraded lands has shown the potential to motivate for conservation on private lands. Conservationists, landowners and society are faced with a fundamental challenge of how to achieve a balance between the utilization of natural capital for improved economic opportunity and the conservation of natural capital for ecosystem service provisioning, both of which are necessary for human well-being.

The South African Ostrich Business Chamber through their Long-term Biodiversity Management Strategy is working towards achieving this balance. Degradation from the ostrich industry has been most severe within the Succulent Karoo lowlands in the Little Karoo. Large tracts of land particularly in the eastern Little Karoo have been seriously degraded because of poor ostrich farming practises over the last 150 years. In 2009, the South African Ostrich Business Chamber motivated for an industry switch to environmentally friendly farming systems as well as resting and actively rehabilitating degraded lands in order to ensure the long-term sustainability of the ostrich industry, conserve biodiversity and improve productivity. Without active intervention, severely degraded rangelands may not return to functional states within timescales practical for landowners. While there is potential for landowners to benefit from rehabilitating degraded rangelands, this option is more likely to be realised if the benefits of rehabilitation are investigated and made known. The central aim of this study was to investigate the short-term (*c.* one year to one-and-a-half years) potential and relative cost of active rehabilitation for increasing ecosystem services on degraded ostrich rangeland.

The study firstly evaluated the short-term (one-year) effect of four common restoration techniques (ripping, micro-catchments, sowing seed and mulching) on vegetation recovery, grazing capacity and plant diversity on a severely degraded ostrich farm. In the short term (one-year), ripping (R $0.13/m^2$) was the least costly method although it was also the least effective. In terms of costs, this was followed by sowing seed (R $0.37/m^2$), micro-catchments (R $1.54/m^2$) and mulching (R $2.47/m^2$). Micro-catchments were the most effective treatment and less costly then mulching. Micro-catchments increased palatable plant species richness and plant density; seed treatments increased palatable species richness, and mulching increased plant diversity (Shannon diversity index). Treatments, however had no effect on perennial vegetation cover or grazing capacity (LSU/ha).

Secondly, the study tested the short-term (18 months) effect of rehabilitation using a combination of treatments on landscape functioning and ecosystem services like water flow regulation, erosion control, forage production, nutrient cycling and plant diversity on four disused and degraded ostrich farms. The Landscape Functional Analysis Indices of water infiltration, nutrient cycling and soil stability (Tongway & Hindley 2004) in conjunction with empirical/quantitative measurements were used to evaluate the performance of active rehabilitation as a management technique for enhancing ecosystem service provision. Furthermore, the study assessed the ability and usefulness of the Landscape Function Analysis technique for quantifying differences in water and organic matter retention within the Succulent Karoo vegetation communities. Within 18 months, rehabilitation enhanced water flow regulation as indicated by improved water infiltration rates (ml/s). It also increased erosion control as indicated by an increase in percentage cover of resource sink zones that capture nutrients, soil and water and are indicative of improved erosion control. Rehabilitation was however not sufficient for increasing nutrient cycling (soil organic carbon and nitrogen), forage production (LSU/ha) and plant diversity (species richness) ecosystem services. Empirical evidence was more useful in determining the effect of rehabilitation on landscape functioning and ecosystem

services than the Landscape Functional Analysis Indices, which were not consistent with empirical results and were relatively complex to interpret. The LFA landscape organisation index (total patch length/length of transect) was, however, useful in evaluating the effect of rehabilitation on erosion control and correlated well with empirical measurements of water infiltration.

This study has shown that using active rehabilitation in combination with improved management strategies (e.g. rest from grazing) may induce recovery of certain ecosystem services within practically relevant time scales. However, these benefits might not be tangible for landowners or society as of yet, due to the small-scale nature of the rehabilitation application along with the relatively high, short-term associated costs. To alleviate financial constraints on farmers and in order to socially and financially boost depressed local communities, the focus for now should be on sourcing funds from government and private sectors for rehabilitation implementation. It is likely that a Payment for Ecosystem Services scheme similar to the 'working for wetlands/water' community programmes funded through poverty alleviation money could be most successful in the Little Karoo. Alternatively, landowners can use resources available on-site to reduce costs. Further research is needed to evaluate the long-term effects of rehabilitation on ecosystem services.

Chapter 1: General Introduction

1.1 Contextualisation

The existing protected area network covers about 12% of the Earth's surface (Jenkins & Joppa 2009). Even if these areas were to be protected from adverse anthropogenic influences, including the effect of climate change this would not be enough to ensure the future of living plant and animal populations (Secretariat of the Convention on Biological Diversity 2010a). As a result, conservation efforts are being increasingly focussed within private and communal landscapes worldwide (Gallo *et al.* 2009; Plieninger & Gaertner 2011; Secretariat of the Convention on Biological Diversity 2010b). Promoting conservation outside protected areas and in particular on privately owned land is seen as a major challenge. This is especially true for diverse biodiversity hotspots such as the Succulent Karoo biome in South Africa (O'Farrell 2005). Ecosystems not formally protected have been transformed, or degraded by human use and in most cases are still in use for economic production (Farley & Costanza 2010; Plieninger & Gaertner 2011). Functioning ecosystems provide services essential to human survival and well-being such as climate regulation, erosion control, aesthetic beauty and protection from storms and floods (MA 2005; Costanza *et al.* 1997; Daily 1997). However, degradation results in the loss of ecosystem function and services which leads directly to losses of soil organic carbon, nutrients, biodiversity, soil water storage and water regulation (Gisladottir & Stocking 2005; Brauman *et al.* 2007; Plieninger & Gaertner 2011).

Ecological restoration which has been defined by the Society for Ecological Restoration as "assisting the recovery of degraded or damaged lands" (SER 2004), offers an opportunity to recover ecosystem services and benefits as well as the intrinsic value of degraded lands (Aronson *et al.* 2010a). There are two approaches for restoration. Restoration in its strictest sense seeks a direct and full return of a landscape's full indigenous historic ecosystem state and includes the return of all goods, services and species. As restoration in its strictest sense is seldom possible, a broader sense of restoration, known as rehabilitation is generally used. Rehabilitation seeks to halt degradation and to redirect a disturbed ecosystem in a trajectory resembling its pre-disturbance state, while positively effecting landscape functioning and ecosystem services (Aronson *et al.* 1993). Large costs make rehabilitation efforts unfeasible for most private landowners (Reyers *et al.* 2009; Farley *et al.* 2010; Ribaudo *et al.* 2010). The concepts of natural capital, ecosystem services and Payment for Ecosystem Services (PES) have shown the potential that rehabilitation interventions can play in mainstreaming such practices outside of protected areas (Daily 1997; Jenkins 2003; Reyers *et al.* 2010). PES defined broadly is the practise of offering incentives to owners of tracts of land in exchange for improvements in land management

for enhancing the production of ecological services (Daniels *et al.* 2010; Kemkes *et al.* 2010; Ribaudo *et al.* 2010). Ecosystem services, and the natural capital assets that produce them, represent a significant contribution to sustaining human well-being (Farley & Costanza 2010). However, a fundamental question faced by conservationists, landowners and society is how to strike a balance between the overutilization of natural capital for improved economic opportunities and the conservation of natural capital to provide ecosystem services, both of which are essential to human well-being (Carpenter *et al.* 2009).

1.2 Background to the study

The semi-arid Succulent Karoo Biome of southern Africa is one of the world's centres of endemism and biodiversity (Mittermeier *et al.* 2005). It is one of the most diverse biomes in the country and covers a wide geographic range extending from southern Namibia in the north into the intermontane valleys of the Little Karoo in the south. Only 5.8% of the biome falls within formally protected areas (Mucina *et al.* 2006). The majority of the area is used for livestock farming with sheep, goats, cattle, donkeys and ostriches (O'Farrell *et al.* 2008). Inappropriate management and overstocking have led to significant rangeland degradation and a subsequent decline in rangeland quality (Dean & Milton 2003; Thompson *et al.* 2009). Degradation has been most severe within the Succulent Karoo lowlands in the Little Karoo (Le Maitre *et al.* 2009). Large tracts of land particularly in the eastern Little Karoo have been seriously degraded due to the pressure from the ostrich industry over the last 150 years (Hoffman & Ashwell 2001; Keay-Bright & Boardman 2006; Rouget *et al.* 2006; Herling *et al.* 2009; Thompson *et al.* 2009). Overstocking ostriches for breeding purposes in free ranging paddocks has caused a severe decline in biodiversity, nutrient cycling, forage production, water flow regulation, tourism and erosion control (Reyers *et al.* 2009). Declines in regulating (e.g. water regulation) and supporting (e.g. nutrient cycling) ecosystem services (Reyers *et al.* 2009), together with biodiversity losses (Rouget *et al.* 2006) have the potential to decrease the long-term productivity and resilience of the Little Karoo and increase its vulnerability to floods and droughts (Reyers *et al.* 2009).

The South African Ostrich Business Chamber (SAOBC) identified the inadequate management of biodiversity and ecosystem services as a major threat to their long-term sustainability (SAOBC 2009). It is believed that an industry switch to a more environmentally friendly pen-breeding farming system and rehabilitating degraded lands can decrease the industry's vulnerability to market shifts, disease and floods (Murray 2007; O'Farrell *et al.* 2008; Reyers *et al.* 2009; SAOBC 2009). Environmental pressure groups in countries that import goods from the Succulent Karoo also appear

to be driving ecological rehabilitation (Murray 2007; Le Maitre *et al.* 2009). Therefore, one important way to mainstream rehabilitation practices in the area is to identify the benefits and costs associated with these practises (Herling *et al.* 2009; Aronson *et al.* 2010b). However, before the costs and benefits can be demonstrated to farmers and the populace, the ecological and hydrological effects of rehabilitation need to be established in order to ascertain the most successful and feasible approaches to rehabilitation. There is potential for landowners to benefit from the sustainable use of their land (O'Farrell *et al.* 2008; Egoh *et al.* 2010). However, without incentives or payment structures rehabilitation is improbable due to the large costs involved (Herling *et al.* 2009). Payments for Ecosystem Services could be an option for the Little Karoo and research into the impacts and benefits of rehabilitation could supply the information from which relevant support from private and government sources can be garnered (Herling *et al.* 2009; Reyers *et al.* 2009; O' Farrell *et al.* 2010).

Linking ecosystem services to beneficiaries of ecosystem rehabilitation, and demonstrating values to society, has only recently emerged in the mainstream literature (Goldstein *et al.* 2008; Galatowitsch 2009; Rey Benayas *et al.* 2009; Aronson *et al.* 2010a). Most research has been at broad scales with concentrated efforts in forest, riverine and marine ecosystems (Alexander *et al.* 1997; Daily *et al.* 1997; Jansson *et al.* 1999; Arrow *et al.* 2000; Falkowski *et al.* 2000; Kaplowitz 2001; Wu *et al.* 2002; Phat *et al.* 2004; Dahdouh-guebas *et al.* 2005; Danielsen *et al.* 2005; MA 2005; Calder & Aylward 2006; Bradshaw *et al.* 2007; Sodhi *et al.* 2007; Daniels *et al.* 2010; Wendland *et al.* 2010). There has only been limited work in arid and semi-arid regions (O'Farrell 2005; Mills & Cowling 2006; Aronson *et al.* 2010a; Mills & Cowling 2010). Broad scale studies have raised international awareness of the benefits of rehabilitation on ecosystem services and biodiversity. However, rehabilitation benefits should be evaluated at fine-scale to encourage conservation and sustainable use on privately owned lands such as the degraded ostrich rangeland in the Little Karoo (O'Farrell 2005; O'Farrell *et al.* 2010). If the benefits derived from rehabilitation and appropriate management of land can be demonstrated at a farm-scale, this would act as motivation for conservation (Edwards & Abivardi 1998; Kemper *et al.* 1999; O'Farrell *et al.* 2009).

A switch to pen-breeding ostrich farming and rehabilitation could present one way to balance the overutilization of the regions natural resources and the conservation of the Little Karoo's threatened natural capital (SAOBC 2009). However, no experimental rehabilitation trials have been conducted on ostrich farmlands. Although some rehabilitation studies have been conducted in denuded areas in parts of the Karoo (Van der Merwe & Kellner 1999; Beukes & Cowling 2003; Anderson *et al.* 2004; Visser *et al.* 2004; Van den berg & Kellner 2005; Simons & Allsopp 2007; Schmiedel *et al.* 2010), approaches differ with soil type, rainfall distribution as well as disturbance. Currently rehabilitation

within degraded rangelands in the Succulent Karoo biome is still in its pioneer phase (Schmiedel *et al.* 2010). Baseline and follow up monitoring of rehabilitation is generally poor (Herling *et al.* 2009) and existing large-scale rehabilitation interventions are rarely monitored (Wagner *et al.* 2008; Deri *et al.* 2009).

1.3 Study aims and objectives

This study investigates the short-term effects of rehabilitating degraded ostrich farmland for the improvement of landscape functioning, ecosystem services and biodiversity. The main aim is to determine the short-term ecological and hydrological effects of rehabilitation on degraded ostrich farmland within the Little Karoo Succulent Karoo lowlands. The key question addressed is; does rehabilitating ostrich farmland increase the provisioning of ecosystem services such as grazing capacity, rainwater infiltration, nutrient cycling, erosion control and plant diversity in the short-term (i.e. one year to 18 months)? It is hypothesised that restoring degraded ostrich farmland using soil disturbance, woodchip mulching and sowing palatable species improves land productivity, grazing capacity, rainfall infiltration, nutrient cycling, erosion control and plant diversity even over a relatively short time period.

The study has two main emphases. Firstly the study tests different types of rehabilitation methods such as ripping the soil, creating micro-catchments, mulching and sowing seed for increasing natural vegetation and grazing services in one disused and degraded ostrich camp in the Oudtshoorn area in order to provide information to improve the efficiency of future rehabilitation practises. Secondly, field-level sampling is used to estimate the effects of rehabilitation on five ecosystem services; water flow regulation, biodiversity, forage production, nutrient cycling and erosion control on four disused and degraded ostrich camps. Both nominal/qualitative and empirical/quantitative methods are used to assess the success of rehabilitation practises. Nominal indicators of ecosystem function from the Tongway and Hindley (2004) Landscape Function Analysis (LFA) along with empirical measurements are used to evaluate the functional status of damaged and rehabilitated ostrich rangelands. The relationship between the nominal and empirical data is assessed in order to evaluate the capabilities of nominal methods like LFA within the Succulent Karoo semi-arid environment. Links between ecosystem services provided by rehabilitated ostrich rangeland to beneficiaries are discussed.

This study has four main objectives. These are:
(1) To evaluate the relative success of three different rehabilitation methods for returning indigenous vegetation and grazing services to a severely degraded ostrich camp.

(2) To determine the effects of rehabilitation on landscape functioning and ecosystems services on four disused and degraded ostrich camps.

(3) To assess the ability and usefulness of the Landscape Function Analysis technique for quantifying differences in water and organic matter retention within the Succulent Karoo lowland vegetation communities of the Little Karoo.

(4) To discuss the implications of the results for management and rehabilitation of semi-arid degraded Succulent Karoo vegetation in the Little Karoo.

1.4 Significance of study

This research is critical for land management decisions in the Little Karoo, and adds to the findings of ecological research within other parts of the Succulent Karoo Biome (Schmiedel *et al.* 2010). Information on the success of rehabilitation treatments is required by local land users and conservationists and contributes to the literature on ecological rehabilitation in South African rangelands. This study provides information for those farmers and conservationists who practice and support rehabilitation. It also provides a crucial baseline survey for rehabilitation efforts within the Little Karoo. The datum will aid towards developing models for estimating ecosystem services generated from rehabilitated ostrich farms over the short-term. It also addresses the gaps in the literature with regard to the lack of rehabilitation and ecosystem services studies within dryland ecosystems and investigates rehabilitation, biodiversity, ecosystem function and services.

This study forms part of the South African Water Research Commission's larger Restoring Natural Capital project entitled "*The Impact of Re-establishing Indigenous Plants and Restoring the Natural Landscape on Sustainable Rural Employment and Land Productivity through Payment for Environmental Services*". This latter project is a meta-analysis of the ecological, hydrological and economic impacts and benefits that rehabilitation can hold for society throughout South African ecosystems ranging from Fynbos to Succulent Karoo as well as Savanna areas and grasslands in the Drakensberg foothills. More locally, this study will be used by economists and social scientists to determine how rehabilitation can enhance the social acceptability of ostrich farming in the Little Karoo and will contribute to the long-term biodiversity and ostrich management strategy.

1.5 Thesis structure

This thesis has been divided into four chapters in order to describe the effect of rehabilitation on ecosystem services. Chapter (1) contextualizes the study and provides the general background, study aims and objectives. It also highlights the importance of the study. Chapter (2) provides a literature

review, which focuses on ecosystem services, land degradation and rehabilitation and provides an in-depth look at the background of the ostrich industry and the natural environment and history of the Little Karoo. Two data chapters follow which address the key aims and objectives of the study.

Chapter (3) assesses different rehabilitation methods for restoring degraded ostrich rangelands within a practical time-scale (one-year). It investigates the impact of different rehabilitation methods for increasing species richness, plant density, plant diversity (Shannon diversity index), vegetation cover and grazing services (Large Stock Unit/ha) on a severely degraded ostrich farm in the Little Karoo. The consequences of these findings for rehabilitation practises that aim to maintain both the forage production services and species richness levels of the area are described.

Chapter (4) investigates the effect of rehabilitation on landscape functioning and ecosystem services using nominal Landscape Functional Analyses Indices and empirical measurements on four farms of differing degradation levels. Critical factors such as chemical soil properties (total nitrogen (%) and organic carbon (%), LFA nutrient cycling index (%)), soil hydrology (infiltration rates ml/s, LFA infiltration index (%)), grazing capacity (LSU/ha), plant diversity (species richness) and erosion control (vegetation cover (%), resource sink zones (%), LFA soil stability index (%)) are used to assess the success of rehabilitation efforts. Relationships between the nominal and empirical measurements are also examined in order to assess the usefulness of nominal methods for monitoring rehabilitation within these semi-arid rangelands.

The final chapter (5) provides a brief synthesis, highlighting the key findings of the study. It also includes discussions on conservation and Payment for Ecosystem Services in the Little Karoo and identifies future research priorities. The implications for rehabilitation practises in the Succulent Karoo vegetation within the Little Karoo are discussed highlighting the short-term nature of these results and the importance of follow up measurements.

Chapter 2: Literature study

2.1 Introduction

In order to place my research within the context of the scientific literature, I have presented a literature study focussing on the topics of ecosystem services, land degradation and rehabilitation. This literature study provides a closer look at the topics of natural capital, ecosystem services, land degradation, restoring natural capital, rehabilitation techniques applied in rangelands worldwide focussing on Africa and the United States, arid and semi-arid rangelands, Payments for Ecosystem Services and techniques for measuring rehabilitation success. I present a closer look at the Succulent Karoo biome and in particular the portion of the Succulent Karoo biome, which falls within the Little Karoo region in South Africa. I focus on declines in ecosystem services as a result of unsuitable farming practises utilised by the ostrich industry as well as opportunities within the region for balancing ostrich production with biodiversity and ecosystem service conservation.

2.2 Natural capital and ecosystem services

The terms 'natural capital' and 'ecosystem services' have become widely adopted among scientific and policy communities (Carpenter *et al.* 2009; Reyers *et al.* 2010). These concepts were explicitly recognised in the literature during the 1970s and 1980s (Mooney & Ehrlich 1997; Reyers *et al.* 2010). However, it was only after the publications of Daily's (1997) Nature's Services, Costanza *et al.*'s (1997) paper on the value of the world's ecosystem services and natural capital in the journal Nature, and following the introduction of the United Nations Millennium Ecosystem Assessment in 2000 (MA 2005) that these concepts were mainstreamed within the sciences (Reyers *et al.* 2010).

Natural capital is one of five principal forms of capital, the others being human, social, physical and financial capital (MA 2005). It is an economic metaphor for the limited reserve of physical (air, soil, water) and biological (living organisms) natural resources found on Earth and unlike other forms of capital there are no adequate substitutes for it (Costanza & Daily 1992; Janzen 1998; Aronson *et al.* 2010b). The Millennium Ecosystem Assessment (MA) recognised four main types of natural capital. These are non-renewable (e.g. coal), replenishable (e.g. fertile soils), cultivated or production (e.g. farms and crops) and renewable (e.g. biodiversity) forms of natural capital. The latter three are all described as providing flows of ecosystem services that are essential for human life and economic production (MA 2005; Aronson *et al.* 2010b).

Fisher *et al.* (2008) provide an overview of definitions for ecosystem services, which range from the Millennium Ecosystem Assessment's "the benefits people obtain from ecosystems", to their own "the aspects of ecosystems utilized (actively or passively) to produce human well-being" (Farley & Costanza 2010). The MA's definition is most frequently used in the literature. The MA distinguishes between provisioning services, such as food and water; regulating services, such as regulation of floods, drought, and disease; supporting services, such as soil formation and nutrient cycling; and cultural services, such as recreational, spiritual, and other non-material benefits (MA 2005). Most definitions emphasise that services are processes or functions of value to humans (Farley *et al.* 2010). This emphasis (Farley & Costanza 2010) has led some critics to reject the entire concept of ecosystem services (McCauley 2006) because they incorrectly interpret it as meaning the transformation of ecosystem goods and services into commodities (Farley & Costanza 2010). Others state that thinking of ecosystems as stocks that provide flows of benefits will result in narrow market-based solutions to complex ecological problems (Liu *et al.* 2008; Farley & Costanza 2010; Norgaard 2010).

Recently Farley and Costanza (2010) and Farley *et al.* (2010) proposed an additional definition, which is less broad and therefore easier to understand. They distinguish between ecosystems as either funds or stocks. Ecosystem goods are described as stock-flow resources and ecosystem services as fund-services provided by nature. An ecosystem fund is a particular organization of structural components such as water, minerals, soil, plants animals and so on, that generates a flux of services such as climate and water regulation. These fund services are not provided at a given rate over time and the ecosystem fund is not physically transformed into the services it provides. In contrast, the structural components of ecosystems serve as stock-flow resources. This type of flow is a physical flow of raw materials and stored energy from nature, which is then transformed into economic products and returned as waste back to the natural environment. Such stock-flow resources can be used up at any given rate and are physically transformed into products (e.g. forests into houses). Stock-flow resources can be stored when inflows exceed outflows (known as stockpiling) whereas fund services cannot be stored for future use (Farley *et al.* 2010; Farley & Costanza 2010).

One of the most vital and immediate services of ecosystems is the provisioning and regulation of water resources. Brauman *et al.* (2007) classified hydrologic services into five broad categories; improvement of extractive water supply, improvement of in-stream water supply, water damage mitigation, provision of water related cultural services, and water-associated supporting services. Soil is one of the most critical elements of natural capital. It is also, however the most unappreciated and therefore one of the most over-utilised and degraded (Myers 1993; Yeld 1993; Pimentel *et al.*

1995). According to Daily *et al.* (1997), soil provides six major ecosystem services, which are, moderating the hydrologic cycle, physically supporting plants, retention and delivery of nutrients to plants, disposal of wastes and dead organic matter, renewal of soil fertility and regulation of major element cycles.

It is almost impossible to list all the ecosystem services and natural products that people directly utilise. However, ecosystems with intact groundcover and root systems are generally very effective at improving water quality, increasing recharge and reducing runoff by increasing the rate at which water moves into the subsurface of soils (Brauman *et al.* 2007). Infiltration is the process by which surface water becomes groundwater. The positive impact of vegetation cover on runoff water retention is extensively discussed in the literature. Vegetation reduces the rate of runoff and enhances water infiltration while canopy cover also directly intercepts raindrops, allowing water to gradually descend into the soil rather than hit soil surfaces directly. In this way, interception of rainwater by plants reduces the magnitude of soil erosion and decreases the risk and magnitude of floods (Castillo *et al.* 1997; Descroix *et al.* 2001; Descheemaeker *et al.* 2006; Bradshaw *et al.* 2007; Bartley *et al.* 2010a; Bartley *et al.* 2010b). Vegetation also tempers water flow velocity (Bochet *et al.* 1998; Descheemaeker *et al.* 2006). Vegetation positively influences soil properties through the incorporation of organic matter and therefore indirectly also increases infiltration rate (Bochet *et al.* 1998; Brauman *et al.* 2007).

In summary vegetation and soils, provide important ecosystem services because they remove pollutants from overland flow and from ground water by reducing water speed and enhancing infiltration, by physically trapping water and sediments, by stabilizing eroding banks and by absorbing water and nutrients from the root zone (Brauman *et al.* 2007).

2.3 Biodiversity and ecosystem services

The role of biodiversity in providing ecosystem services is actively debated in ecology. Many authors treat biodiversity and ecosystem services separately. They consider ecosystem services to be defined by their link to human values and to particular beneficiaries (Egoh *et al.* 2010). Some authors state that the diversity of functional groups is as important as species diversity, if not more so (Kremen 2005), and in most cases suggest that a few dominant species seem to play the major role in ecosystem service provision (Hooper *et al.* 2005). There is, however, abundant evidence which suggests that biodiversity enhances both the productivity and stability of ecosystems, maintains resilience in the presence of shocks and ensures the continued provision of ecosystem services over time (Tilman & Downing 1994; Tilman *et al.* 1996; Gowdy 1997; Naeem & Li 1997; Tilman 1997;

Chapin *et al.*, 1998; Naeem 1998; Ricketts *et al.* 2004; Hooper *et al.* 2005; Worm *et al.* 2006; Hobbs *et al.* 2007). Although dominant species are likely to be important for ecosystem function (Raffaelli 2004; Smith *et al.* 2004), it has been shown that even rare species can have a role (Lyons & Schwartz 2001).

Despite the number of theories concerning the importance of species diversity in ecosystem functioning (Ehrlich & Ehrlich 1981; Walker 1992; Johnson *et al.* 1996; Schwartz *et al.* 2000) most authors suggest that when biodiversity is retained, ecosystem processes are also more likely to be retained (Simberloff 1999). Many ecosystem services are products of complex ecosystem processes and are delivered through a variety of landscape settings (Farley *et al.* 2010; Ribaudo *et al.* 2010). The more complex an ecosystem is, the more biodiversity will increase ecosystem function, as more species are needed to exploit the many combinations of environmental variables (Tilman 1997). Ecosystems that are more diverse are also likely to be more stable (Hooper *et al.* 2005; Worm *et al.* 2006). Farley and Costanza (2010) suggest further that in the face of ecosystem complexity humans do not know enough to make decisions about whether biodiversity is crucial for ecosystem functioning or even which parts of biodiversity are vital. Therefore, rather than separating biodiversity from ecosystem services, biodiversity and other ecosystem services should be combined into a bundle of loosely defined services in order to maximise social benefits (Farley & Costanza 2010; Farley *et al.* 2010; Kemkes *et al.* 2010; Wainger *et al.* 2010; Wendland *et al.* 2010).

When considering the Millennium Ecosystem Assessment's full definition of ecosystem services (including provisioning, supporting, regulating and cultural) (MA 2005) as well as the new proposed definition by Farley and Costanza (2010), there is little doubt that biodiversity is a critical component of ecosystem funds and generates a variety of services.

2.4 Land degradation and declines in ecosystem services

Land degradation is defined by the UN Environment Programme as "a long-term loss of ecosystem function and services, caused by disturbances from which the system cannot recover unaided" (Dent 2007). It is one of the most fundamental and persistent environmental challenges which leads directly to losses of soil organic carbon, nutrients, soil water storage and regulation and biodiversity. Land degradation also indirectly causes habitat loss as lands lose the ability to support vegetation biomass. In addition, sensitive and vulnerable species are lost (Gisladottir & Stocking 2005; Plieninger & Gaertner 2011). Human's utilisation of provisioning (stock-flow resources) ecosystem services has increased extensively at the expense of regulating, supporting and cultural ecosystem services (fund services) (MA 2005; Carpenter *et al.* 2009). Current large-scale land degradation and

biodiversity losses through habitat fragmentation as well as altered biogeochemical cycles, climate and hydrology result from trends of lop-sided or unbalanced use between provisioning and regulating, supporting and cultural ecosystem services (Carpenter *et al.* 2009; Reyers *et al.* 2009). The current decline in regulating services such as climate and water regulation foreshadows future declines in other provisioning services due to positive feedback loops (MA 2005; Carpenter *et al.* 2009). One of the best-documented causes of environmental degradation with positive feedback loops is rangeland overgrazing (Aronson *et al.* 2010a).

Rangelands can be regarded as land carrying natural or semi-natural vegetation, which provides a habitat for herds of wild ungulates or domestic livestock (Pratt *et al.* 1966). Rangelands have primarily native vegetation as opposed to cultivated plants. They are managed with extensive practises of livestock grazing and prescribed fire rather than intensive agricultural practises such as irrigation. In their natural state, they consist of a vegetation mixture of grasses and woody plants, in various proportions. Types of rangelands include pure grasslands such as prairies, desert grasslands, shrublands, woodlands, savannas, chaparrals, steppes and tundras (Walker 1993). Worldwide, plant species composition has changed because of inappropriate land utilization by domestic livestock (Watkinson & Ormerod 2001; Tietjen & Jeltsch 2007; Todd & Hoffman 2009). Overgrazing has been shown to decrease palatable perennial plants in favour of less palatable, undesirable vegetation (Milton 1992a; Milton 1995a, 1995b; Perrings & Walker 1997; Todd & Hoffman 1999; Van der Westhuizen *et al.* 1999; Anderies *et al.* 2002; Snyman 2004; Cingolani *et al.* 2005; Anderson & Hoffman 2007; Tietjen & Jeltsch 2007; Todd & Hoffman 2009). This shift in plant species composition is accompanied by reductions in primary productivity and grazing capacity (Illius & O'Connor 1999; Anderson & Hoffman 2007; O'Farrell *et al.* 2010). Globally, rangelands are severely degraded requiring extensive land use change and/or management in order to restore stocks of natural capital (Lund 2007; Aronson *et al.* 2010b).

Most workers agree that overgrazing by livestock has played a major role in rangeland degradation (Fleischner 1994; Archer *et al.* 1995; Daily 1995; Aronson *et al.* 2010b; O'Farrell *et al.* 2010). However, the main cause of degradation has been debated for decades as climate, grazing and fire regime have all been shown to play a role (Ellis & Swift 1988; Vetter 2005; Vetter 2009). Semi-arid ecosystems typically show abrupt, discontinuous and irreversible transitions between discrete intact or degraded states. Successful reproduction (i.e. establishment) and mortality in plants depend on particular conditions (Wiegand *et al.* 1995). The typical Clementsian range succession model has been criticized especially in arid and semi-arid zones because of its inability to deal with the vegetation changes observed (Noy-Meir 1973; Wiegand *et al.* 1995; Wiegand & Milton 1996). These

complicated dynamics within semi-arid and arid rangelands are often described using state and transition models, characterized by threshold dynamics and alternate stable states (Ellis & Swift 1988; Westoby *et al.* 1989; Milton & Hoffman 1994). Transitions between alternative stable states may be triggered in two main ways. The first occurs via altered biotic interactions (e.g. grazing and competitive dynamics) which provide sufficient perturbation to force the state to cross a threshold. Alternatively, changes to the abiotic conditions (rainfall and soil nutrients) of a site may lower the threshold (Vetter 2009).

2.5 Ecosystem services in arid and semi-arid rangelands

The supply of ecosystem services in arid and semi-arid regions have received considerably less attention in comparison to forests and aquatic ecosystems (rivers, wetlands, marine and coastal systems) (Aronson *et al.* 2010a). River catchments, watersheds and fresh water ecosystems have received considerable research attention in a variety of countries (Alexander *et al.* 1997; Daily *et al.* 1997; Jansson *et al.* 1999; Falkowski *et al.* 2000; Kaplowitz 2001; Wu *et al.* 2002; Phat *et al.* 2004; Dahdouh-guebas *et al.* 2005; Danielsen *et al.* 2005; MA 2005; Calder & Aylward 2006; Bradshaw *et al.* 2007; Brauman *et al.* 2007; Sodhi *et al.* 2007; Daniels *et al.* 2010; Wendland *et al.* 2010). Forest ecosystems have been well-studied in terms of the role they play in watersheds of river systems and the role they play in climate regulation. Bradshaw *et al.* (2007) estimated that a 10% decrease in natural forest area would lead to a flood frequency increase between 4% and 28% and to a 4–8% increase in total flood duration at the country scale. The value of mangroves, seagrass beds and coral reefs has also received attention by researchers (Ewel *et al.* 1998; Arrow *et al.* 2000; Kaplowitz 2001). Overall, the bulk of research concerning ecosystem effects on water supply and water hazard mitigation comes from research undertaken in temperate ecosystems followed by tropical ecosystems. However, researchers are increasingly seeking to evaluate ecosystems services and in particular hydrologic response in arid and semi-arid ecosystems (MA 2005; Brauman *et al.* 2007).

Semi-arid areas cover, 40% of the world's land surface and are home to 38% of the global population (Aronson *et al.* 2010a). These 'drylands' are characterized by a mean annual rainfall of 25 to 500 mm and are areas with rainfall lower than the evaporative demand (Tietjen & Jeltsch 2007). Drylands fulfil an array of functions such as water and nutrient cycling, biomass production, biodiversity conservation as well as socio-economic services by providing the basis for sustainable rural livelihoods (Vohland & Barry 2009). Livestock production is an important land use in arid and semi-arid areas and according to Puigdefábregas (1998), drylands support about 50% of the world's livestock.

The degradation of drylands negatively affects *c.* one billion humans on 35–40 million km^2 of land (Goodrich *et al.* 2000). Degradation amplifies water stress in an environment where water is the limiting factor for plant growth even under undisturbed conditions (Noy-mier 1973). While water shortages and desertification affect all dryland areas, developing countries are particularly vulnerable to the economic and social costs associated with the decline of agricultural and natural ecosystem productivity (Goodrich *et al.* 2000). Stabilization of the agricultural landscape by restoring degraded natural grazing lands is essential for managers aiming to sustain human well-being within drylands (Aronson *et al.* 2010b).

2.6 Rehabilitating degraded rangelands
(Restoration, rehabilitation and restoring natural capital)

Rehabilitation is being increasingly implemented throughout the world, to reverse environmental degradation caused by human activities (Rey Benayas *et al.* 2009). The Convention on Biological Diversity and the Millennium Ecosystem Assessment support rehabilitation action through global policies (MA 2005; Secretariat of the Convention on Biological Diversity 2010b). Ecological restoration is defined by the Society for Ecological Restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). Many restoration actions are undertaken with the aim of increasing biodiversity. However, another important direct benefit of restoration is the socio-economic impacts of ecological restoration as well as long-term conservation of natural resources. This is generally recognised as "Restoring Natural Capital" (RNC). RNC refers to all investments in replenishable, renewable and cultivated natural capital stocks in order to improve the functioning of both human and natural managed ecosystems (Aronson *et al.* 2010b). The term restoration in its strictest sense is defined as a direct and full return of a landscape's full indigenous historic ecosystem state. Rehabilitation, in contrast, seeks to halt degradation and redirect a disturbed ecosystem back towards pre-disturbed functional state (Aronson *et al.* 1993; Aronson *et al.* 2010a, 2010b). The definition of restoration has been repeated in the literature countless of times and while some argue that preciseness of the definition is essential (Aronson & Le Floc'h 1996; Higgs 1997) others feel that, the clear enunciation of restoration goals is more important (Hobbs & Harris 2001).

Many authors demonstrate the need to factor in the degree of rangeland degradation when planning and assessing the viability for rehabilitation (Milton *et al.* 1994a; Maestre & Cortina 2004; Maestre *et al.* 2006). This is because the relationship between ecosystem structure, functioning, and restorability (i.e., the difficulty to bring a degraded ecosystem to a desired target state, or the effort

needed to do so) is not well understood (Cortina *et al.* 2006; Maestre *et al.* 2006; Mendez *et al.* 2008). Most conceptual frameworks suggest that the probability of reversing grazing-induced changes may be inversely related to the amount of disturbance involved in degrading the land in the first place (Milton *et al.* 1994a). One influential model proposed by Bradshaw (1984) assumes that increases in structure (any description of community composition, and the way organisms are organized) parallel the recovery of ecosystem functioning in a linear way.

Other models, which describe the recovery of ecosystem function after disturbance, recognize that the steps in this trajectory may not be of equal magnitude or importance and that some of them can hardly be reversed spontaneously (Hobbs & Norton 1996). Many of the conceptual models developed imply that the lack of certain ecosystem components and/or functions may limit rehabilitation efforts (Cortina *et al.* 2006). For example, Milton *et al.* (1994a) anticipated that for every step descended along a degradation gradient, rehabilitation becomes more costly in terms of loss of secondary productivity and expenditure. For ecosystems in a highly degraded state, it is suggested that rehabilitation efforts should focus on the recovery of ecosystem structure by increasing the number of resource-traps. A technique employed in Australia to achieve this is to insert brush piles parallel to land contours (Ludwig & Tongway 1996). Such artificially created patches are thought to act as sinks for resources (soil, water, and nutrients) and seeds and thereby providing favourable micro-environments for the recovery of vascular plants and biological soil crusts (Aguiar & Sala 1999; Bowker 2007). Once this intervention has reduced some of the degradation processes, the introduction of seeds of indigenous palatable plants should be the next step. This introduction aids the recovery of nutrient cycling in the long term, increases ecosystem resilience, and provides suitable habitats for further plant and animal colonization (Mendez *et al.* 2008).

In the context of state and transition models, severely degraded rangelands can be viewed as lands that have undergone dramatic state shifts or threshold transitions (Milton & Dean 1995; Bestelmeyer 2006). For the rehabilitation of these lands, passive methods (e.g., the removal of livestock) are usually insufficient, because the degraded system is relatively stable in its undesirable state (Seymour *et al.* 2010). Slow recovery and irreversibility translate into long-term losses of ecosystem services and persistent problems for farmers and managers (Le Maitre *et al.* 2007; O'Farrell *et al.* 2008). Several studies from South Africa and the United States suggest that without active intervention, vegetation regeneration does not occur on timescales that are practical for land managers (Milton & Dean 1995; O'Connor & Roux 1995; Wiegand & Milton 1996; Valone *et al.* 2002; Valone & Sauter 2005; Sasaki *et al.* 2009; Miehe *et al.* 2010; Seymour *et al.* 2010).

Restoring severely degraded lands over management timeframes requires complex interventions such as introducing transplants and sowing seeds, (Anderson *et al.* 2004; Padilla & Pugnaire 2006; King & Stanton 2008), the application of brushpacks (Ludwig & Tongway 1996; Simons & Allsopp 2007), or organic mulches (Beukes & Cowling 2003; Visser *et al.* 2004; Van den berg & Kellner 2005). Another approach is the digging of micro-catchments (rainwater harvesting) to capture runoff (Critchley *et al.* 1994; Li *et al.* 2005; Bainbridge 2007; Simons & Allsopp 2007; Hanke *et al.* 2011) or using alternative soil disturbances like ripping and ploughing to break open compacted soils (Snyman 2003; Kinyua *et al.* 2009). Active rehabilitation techniques mainly aim to improve soil water status by increasing infiltration rates or decreasing evaporative losses (Thurow 2000).

In some areas, particularly United States rangelands, improvements have been in use for nearly a century (Branson *et al.* 1966). Technologies such as contour furrowing, pitting, and contour terracing were implemented in the 1930s, followed by the alternatives of level bench terracing, gully plugs, ripping, and chiselling in 1970s. In general, the goals of reducing erosion and runoff were achieved. Benefits included reduced soil salinity (Branson *et al.* 1966; Shanan *et al.* 1970), increased plant biomass (Branson *et al.* 1966; Fisser *et al.* 1974; Wight *et al.* 1978; Suleman *et al.* 1995), altered species composition to increase forage species (Wight *et al.* 1978), and the re-establishment of woody species (Shanan *et al.* 1970). Branson *et al.* (1966) and Wight *et al.* (1978) recommended using soil disturbance techniques for harvesting runoff water for fine-to-medium-textured soil as the poor structure and crusted surfaces result in low infiltration rates. In the United States, these soil disturbance methods were shown to be most effective for arid and semi-arid lands with low infiltration rates and sparse vegetation (Wentz 2004).

The creation of micro-catchments for soil and water conservation have also had a long tradition in African, middle-eastern and Indian countries (Vohland & Barry 2009). They are actively promoted by NGOs as well as natural agricultural extension services and government agencies. There is a variety of micro-catchment systems and although these systems are referred to by different names, they fulfil similar roles. The main difference between the different systems is the size of the pits. In semi-arid zones of western Africa, pitting takes place in the form of semi-circular bunds, half moons or Demi lunes, Zai pit systems and Tassa pits. In East Africa Chololo pits and Ngoro pits of the Matengo people have also been shown to be effective (Vohland & Barry 2009).

Most micro-catchment structures have been installed for crop plants, and much less frequently for rangeland improvement or ecosystem conservation (Vohland & Barry 2009). They have been shown to modify water flow in the landscape mainly by enhancing water infiltration at the plot-scale

(Zougmore´ *et al.* 2003; Wakindikiand & Ben-Hur 2004). However, high variability of rainfall means that there is a high degree of uncertainty when using these techniques in other dryland areas (Vohland & Barry 2009). For example, Simons and Allsopp (2007) and Hanke *et al.* (2011) both showed that micro-catchments in the semi-arid winter rainfall region in South Africa did not affect soil water content shortly after a rainfall event. The intensity of the typical winter rainfall event was apparently too low for the development of runoff. In addition, Hanke *et al.* (2011) showed that soil water storage was not improved. A study in China further indicated that rainfall events with intensities lower than 3 mm/h and sizes smaller than 6 mm did not generate runoff pooling in micro-catchments (Li *et al.* 2005). In addition, Simons and Allsopp (2007) found that micro-catchments did not improve natural recruitment or the establishment of perennial seedlings. This was attributed to the removal of fertile topsoil when digging the pits. However, in Niger micro-catchments (Zai pits) have been shown to act as sediment traps and therefore enhance nutrient availability (Vohland & Barry 2009). In addition, biomass production has been improved by applying mulch or another form of organic matter into the micro-catchment before planting or sowing seeds (Zougmore´ *et al.* 2003). In the semi-arid rangelands of South Africa, however, Simons and Allsopp (2007) found that mulching did not improve the performance of micro-catchments. Another negative consequence that has been highlighted by several authors is that under poor drainage conditions micro-catchments might lead to water logging (Simons & Allsopp 2007; Vohland & Barry 2009) and some authors warn that the inappropriate use of micro-catchments might lead to severe side effects as shown by erosive events in Kenya (Ngigi 2003). In South African rangelands Van der Merwe and Kellner (1999) and Snyman (2003) found that using different types of dyker ploughs which make shallow micro-catchments scattered across the soil surface was very successful at capturing runoff.

There is insufficient literature on the impact of using micro-catchments on biodiversity conservation in agricultural landscapes. In addition, the effects of micro-catchments on landscape functions are still poorly understood (Vohland & Barry 2009). Another approach, showing improved infiltration on a field scale, is conservation tillage known as ripping (Visser *et al.* 2004; Van den berg & Kellner 2005; Kinyua *et al.* 2009). In the Nama-Karoo region of South Africa, Visser *et al.* (2004) and Van den berg and Kellner (2005) found that a combination of ripping, sowing seed and brushpacking was the most successful treatment at increasing the establishment of sown seed and the abundance of naturally occurring plants. Ripping alone was found to have large increases in plant cover and biomass in Kenyan rangelands (Kinyua *et al.* 2009). Micro-catchments have been found to be more effective for clayey soils (between 1% and 20% clay) and ripping more suitable for sandy soils

(<20% clay) (Van der Merwe & Kellner 1999; Bainbridge 2007). This is because the effectiveness of micro-catchments decline as they fill in with blown or washed soils and debris (Bainbridge 2007).

Most studies have shown that sowing seeds without an additional soil treatment is ineffective and authors agree that seeding alone in semi-arid environments is not successful. Selective clearing of unpalatable plants is recommended in order to reduce competition from established plants. However, this has been shown to have negative effects and to have no influence on seedling establishment (Saayman *et al.* 2009). Planting functional plants (i.e. plants with a functional role such as facilitation of other plants or reduction of soil erosion) (Anderson *et al.* 2004) and the application of mulch (Beukes & Cowling 2003) have been shown to improve soil properties and to offer considerable potential for enhancing vegetation re-establishment. Brushpacking and wood-chip mulching, which simulates the protective effect of natural plant cover, have been successfully applied in semi-arid environments throughout the world (Ludwig & Tongway 1996). Brushpacks and mulch protect the soil against erosion and solar radiation thereby improving soil moisture conditions. They may also generate fertile islands by trapping soil particles and organic material (Ludwig & Tongway 1996).

2.7 Payments for Ecosystem Services (PES) and government policy tools

Ecological rehabilitation has direct and obvious benefits for society like watershed protection, waste treatment and carbon sequestration (Rey Benayas *et al.* 2009). In the short-term, however, rehabilitation can also lead to perceivable benefits to society by improving the supply and quality of ecosystem services such as increased productivity of farmland (Geerken & Ilaiwi 2004; Aronson *et al.* 2010a), reduced soil erosion and greater protection against floods and offshore storms (Clewell & Aronson 2006, Aronson *et al.* 2007; Le Maitre *et al.* 2007; O'Farrell *et al.* 2007; O'Farrell *et al.* 2008; O'Farrell *et al.* 2009; Reyers *et al.* 2009; Aronson *et al.* 2010a). Rehabilitation actions cannot be implemented without incurring direct and indirect costs and transaction costs for environmental problems are typically enormous (Bromley 1991; Farley & Costanza 2010; Muradian *et al.* 2010). Costs can include amongst others opportunity costs, treatment costs, damage costs and acquisition costs (Naidoo *et al.* 2006). Most case studies show that restoring degraded ecosystems is not profitable for the private landowner, as yields in the short-term are not high enough to justify the investments in labour and materials, although the long-term gross margins are generally higher (Vohland & Barry 2009). Because of this, financial incentives are needed for ecological rehabilitation to be widely implemented (Daniels *et al.* 2010; Farley & Costanza 2010).

Payments for Ecosystem Services (PES) have become an increasingly popular approach to dealing with environmental problems around the world (Farley & Costanza 2010). In addition, it is an increasingly mainstream tool for influencing land-use on private land. PES focuses on increasing private investment in ecosystem services through creating markets for non-commodity ecosystem services (Daniels *et al.* 2010). Successful approaches include emissions trading, mitigation banking, and eco-labelling (Ribaudo *et al.* 2010). Others like water quality trading, wetland mitigation, carbon cap-and-trade, over-the-counter carbon and fee hunting have also shown potential (Ribaudo *et al.* 2010). Most PES schemes actually pay for land uses associated with generating an ecosystem service (Farley & Costanza 2010).

PES should be developed in order to help pay for, and reward, ecological rehabilitation (Aronson *et al.* 2007; Aronson *et al.* 2010a). However, ignorance concerning how ecosystems generate services and how human activities affect them is a challenge for any PES scheme (Farley *et al.* 2010). Ecosystem services are mostly non-rival and/or non-excludable. Non-rival means that the consumption by one person does not reduce another person's consumption and non-excludable means that no one can be excluded from benefiting from the provision of services (Ribaudo *et al.* 2010). These characteristics generally prevent the development of a market, primarily because ownership cannot be defined or enforced and it is difficult to ration an ecosystem service by pricing (Ribaudo *et al.* 2010). Ecosystem services are nearly always characteristics of social goods. Therefore, it is difficult to decide what activities a PES scheme should pay for and the level of services provided by those activities (Farley *et al.* 2010; Ribaudo *et al.* 2010). Ecosystem services are generally produced as joint products (referred to as bundles) from intact ecosystems (Farley & Costanza 2010). Certain authors suggest that a PES scheme should focus on bundles of loosely-defined services instead of on a defined single service (Rørstad *et al.* 2007; Farley & Costanza 2010). It is believed that this will maximise the benefits to society while preventing perverse incentives such as monoclonal *Eucalyptus* plantations. Although these *Eucalyptus* plantations increase the ecosystem service of carbon sequestration, they have the potential to degrade the services offered by biodiversity, water provisioning and nutrient cycling (Farley & Costanza 2010).

Governments have a range of approaches for increasing the production of ecosystem services on farms (Ribaudo *et al.* 2010). Salzman (2005) identifies five types of policy tools; prescription, penalty, property rights, persuasion and payment (PES). If a payment option is available, (e.g. tax expenditure, grant allocation or direct payment) landowners can voluntarily supply ecosystem services on their property while being compensated. For example, forty-six farmers are reported to have been compensated for protecting a watershed in Bolivia's Los Negros Valley. This program

bundled the services of habitat protection and hydrological flow for beneficiaries, including the US Fish and Wildlife Service and the Los Negros municipal government on behalf of local irrigators (Asquith *et al.* 2008; Kemkes *et al.* 2010). In South Africa, the PES program 'Working for Water" restores hydrological services from riparian zones and attempts to address local poverty issues while doing so (Turpie *et al.* 2008). Local municipalities contract government funded programs creating employment for individuals (Turpie *et al.* 2008; Kemkes *et al.* 2010). Public education, in an effort to change landowner behaviour, is another option. However, communicating public information across policy and social institutions has low levels of coerciveness when opportunity costs are high and implementation efforts are costly. In these cases, information alone is generally not enough to change landowner's behaviours (Kemkes *et al.* 2010).

One of the biggest issues facing landowners wishing to participate in markets for ecosystem services is uncertainty about the environmental performance of management practices like rehabilitation (Ribaudo *et al.* 2010). The uncertainty about the quality or quantity of ecosystem services a farm can produce make it difficult for a producer to decide on the long-term economic benefits in, for example, investing in a mitigation bank or making wildlife habitat improvements for agri-tourism (Ribaudo *et al.* 2010). There is a definitive need for research on the effectiveness of different conservation practises particularly on a local-scale (O'Farrell 2005; O'Farrell *et al.* 2010; Ribaudo *et al.* 2010). Agricultural landscapes are inherent with ecological, economic and social complexities. This necessitates significantly greater planning to ensure that the expenditure of money within these programmes is cost-effective, provides the greatest rehabilitative benefits and has the least impact on farm profitability (Crossman & Bryan 2009; Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). The quantification of natural capital and the benefits provided by rehabilitation action is necessary to mainstream rehabilitation as a tool for increasing ecosystem services (Crossman & Bryan *et al.* 2009; O'Farrell *et al.* 2009).

2.8 Measuring rehabilitation success

The most commonly used measures to monitor the change in habitats following rehabilitation are species richness and diversity (Perner & Malt 2003; Piper *et al.* 2007; Deri *et al.* 2009). However, species richness can be misleading because it does not always correlate with conservation objectives or ecological function and its related services (Borrwall & Ebenman 2008). More recently, scientists have focused on the ecological functioning of an ecosystem. Both nominal and empirical approaches are being used for quantifying rehabilitation success (Holm *et al.* 2002; Maestre & Cortina 2004, Maestre *et al.* 2006). The nominal approaches do not directly measure soil properties. They are

based on categorical variables and are subjectively assessed according to a scoring system. Empirical approaches, directly measure continuous variables such as soil carbon and nitrogen, landscape heterogeneity, erosion, infiltration and water holding capacities of soils (Holm *et al.* 2002). Although nominal methods are often employed due to their affordability, especially in arid and semi-arid ecosystems, they can compromise accuracy (Whitford 2002; Maestre & Cortina 2004).

The Landscape Function Analysis (LFA), developed in Australia (Ludwig & Tongway 2000; Tongway & Hindley 2004), is essentially a nominal technique to determine broad biogeochemical processes occurring at the soil surface. The framework builds on earlier work by Noy-Meir (1973) and Westoby *et al.* (1989) and provides powerful insight into the dynamics of semi-arid rangelands (Lechmere–Oertel *et al.* 2005). The LFA framework predicts that functional landscapes have a high degree of spatial organization at a patch-scale. Perennial vegetation patches are associated with resource-rich soils and are surrounded by infertile bare patches (Ludwig & Tongway 1995; Tongway & Hindley 2004; Lechmere–Oertel *et al.* 2005). The framework suggests that healthy landscapes conserve resources and have processes that enrich fertile patches. In contrast, dysfunctional or 'leaky' landscapes have lost their spatial organization of vegetated fertile patches and resource reserves of nutrients and organic matter (Lechmere-Oertel *et al.* 2005). Conceptually, landscapes are positioned along a continuum. Fully-functional landscapes at one end of the continuum are highly patchy and, ideally, capture all resources allowing little runoff and sediment loss. Dysfunctional landscapes at the other end do not trap or retain any resources and all runoff, sediments, and litter are lost (Ludwig *et al.* 2000; Tongway & Hindley 2004).

The LFA Soil Surface Assessments (SSA) evaluate eleven features, which are compiled into three soil quality indices: soil stability, infiltration and nutrient cycling. The soil surface assessment indices have been verified against established scientific measurements at numerous sites in both rangelands and minesites (Tongway & Hindley 2004). Laboratory-measured aggregate stability has correlated well with the field-assessed soil stability index (Chaney & Swift 1984). Linear relationships have also been found between infiltration measured by a disk permeater and the LFA infiltration index (Perroux & White 1988) while linear relationships have been established between soil respiration (biological activity in the soil) and the LFA nutrient cycling index (Tongway & Hindley 2004). Several correlations have been recorded between the size of the nutrient pool (including nitrogen and organic carbon percentage) with the nutrient cycling index (Gianello & Bremner 1986; Tongway & Hindley 2004). Many international authors accept these measurements as providing reliable and useful information that has scientific backing (Cortina & Maestre 2004; Maestre *et al.* 2006). However, according to some authors LFA does have constraints as a

management technique (Palmer *et al.* 2001; Petersen *et al.* 2004a; O'Farrell & Donaldson unpublished). It is a subjective method in the sense that it relies to some extent on the practitioners' skills and experience in assessing the eleven soil surface indicators. In addition, although it is less time-consuming than many empirical techniques it still involves many person-hours to collect field data (Furniss 2009).

Only a few studies using Landscape Function Analysis have been carried out in South Africa. Palmer *et al.* (2001), working in the Peddie district of the Eastern Cape, found that the soil surface assessment technique was ineffective, describing it as insufficiently sensitive to measure shifts in the rangeland which had been detected in both vegetation field surveys, and through remote sensing methods. Petersen *et al.* (2004a) noted that the LFA technique was unable to draw a distinction between communal and commercial grazing strategies in the Succulent Karoo Biome. In contrast, O'Farrell & Donaldson (unpublished) found that LFA was able to detect differences between grazing systems within the Nama-Karoo. However, these differences were not verified when using rainfall simulation experiments. Given the popularity and success of these methods in other semi-arid ecosystems around the globe (Maestre & Cortina 2004; Maestre *et al.* 2006) as well as the scarcity of robust assessments of the technique in South African semi-arid rangelands, LFA could still have potential for monitoring the impacts of degradation or rehabilitation in semi-arid South African ecosystems. However, currently empirical methods are favoured over these more subjective techniques.

One indicator of rangeland soil health, which is measured using the LFA technique, is the cover of biological soil crusts within an area (Tongway & Hindley 2004). Biological soil crusts (biocrusts) are a soil surface community made up of cyanobacteria, lichens, and moss. They are globally widespread and are an integral part of the soil system in arid and semi-arid regions where they stabilize soil surfaces, aid in vascular plant establishment, and are significant sources of ecosystem nitrogen and carbon (Grote *et al.* 2010). An increasing number of studies suggest that biocrusts provide important ecosystem functions (Belnap 2006; Bowker 2007; Grote *et al.* 2010). South Africa's biocrusts are among the most diverse worldwide mainly because of the unusual high cyanobacterial species richness comprising 58 species in 21 genera. The Succulent Karoo contains 49 of these species (Büdel *et al.* 2009). Despite thousands of increasingly international studies documenting the important roles of these organisms, biocrusts are rarely discussed in the rehabilitation literature (Bowker 2007).

2.9 The semi-arid Succulent Karoo biome in South Africa

The Succulent Karoo biome (83,000 km^2) of South Africa is one of only two semi-arid global biodiversity hotspots (Mittermeier *et al.* 2005). Like most semi-arid parts of the world, the Succulent Karoo is home to some of the most vulnerable people and places in the country. Local communities have utilized ecosystem goods and services, particularly grazing services from the land for *c.* 2000 years (James *et al.* 2005; O'Farrell *et al.* 2010). For example, ninety percent of the biome is used for grazing livestock including sheep, goats, cattle, donkeys and ostrich (Mucina *et al.* 2006; O'Farrell *et al.* 2009). The unsustainable farming practises of sedentary grazing, overstocking and ploughing of alluvial areas for subsistence crops have, however, been widespread across large areas (Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). Intensive grazing and trampling has resulted in decreased land productivity and biodiversity (Esler *et al.* 2006; Mucina *et al.* 2006; Rouget *et al.* 2006; Reyers *et al.* 2009). This has severely compromised the ability of the land to provide a flux of ecosystem services (Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). Reductions in grazing services increase livestock mortalities during drought periods and generally lead to poorer quality livestock. This decreases the production of meat and milk, which has serious repercussions for the local populace (Anderson & Hoffman 2007; O' Farrell *et al.* 2010).

The impact of unsustainable farming practises on plant community composition has received considerable research attention within the Succulent Karoo biome (Cowling *et al.* 1999; Todd & Hoffman 1999; Anderson & Hoffman 2007; Cousins *et al.* 2007; Desmet 2007; Hoffman & Rohde 2007; Hoffman *et al.* 2007; O'Farrell *et al.* 2010; Schmiedel *et al.* 2010). Severe degradation is typically characterised by a loss of perennial plant cover, which increases the amount and connectivity of bare ground (Le Maitre *et al.* 2007; Schmiedel *et al.* 2010; Hanke *et al.* 2011). Under heavy grazing the diverse dwarf shrub communities tend to be replaced by few often-unpalatable species. This has been shown particularly in the valley bottoms and low-lying areas (Todd & Hoffman 1999; Anderson & Hoffman 2007; Todd & Hoffman 2009). The change in species composition is usually attributed to selective grazing which can reduce the reproductive output of these preferred forage species (Milton 1992a; Milton 1995a, 1995b; Riginos & Hoffman 2003) and favours the establishment of toxic and unpalatable plants (Noy Meir *et al.* 1989; Milton & Dean 1990). Anderson and Hoffman (2007) found that sustained heavy grazing results in a reduction in leaf succulent and woody plant cover and increases in dwarf shrub cover. Intensive grazing has also been found to alter vegetation composition towards more ephemeral communities (Todd & Hoffman 1999; Anderson & Hoffman 2007; Todd & Hoffman 2009).

Annual plants die during dry times and higher temperatures beat down onto exposed soils accelerating soil water loss (Snyman & du Preez 2005). The reduction in soil moisture recharge further reduces plant productivity and increases the impacts of overgrazing on the already stressed vegetation. Soil crusting and compaction, declines in water infiltration and increased runoff further erode, and aridify areas (Mills & Fey 2004; Snyman & du Preez 2005; Le Maitre *et al.* 2007; Du Toit *et al.* 2009). Most areas, which were originally covered in biocrusts, are replaced with mineral crusts (Mills & Fey 2003). Denuded, compacted and smooth surfaces develop which accelerate water runoff as well as plant nutrients and seed loss from the landscape (Snyman & Van Rensburg 1986; Le Maitre *et al.* 2007). Higher temperatures of exposed soils, together with wind moving over these bare areas cause dust storms, which damage and smother vegetation (Le Maitre *et al.* 2009). All these processes tend to be self-reinforcing leading to positive feedback loops within the landscape (Le Maitre *et al.* 2007).

2.10 Degradation in the Little Karoo succulent karoo lowlands

The Little Karoo region (19,730km2) of the Succulent Karoo biome has some of the worst levels of documented degradation (Le Maitre *et al.* 2009). The Little Karoo is a semi-arid, intermontane basin where vegetation associated with the Succulent Karoo biome as well as two other biomes (Fynbos, and Subtropical Thicket biomes) interconnect (Vlok *et al.* 2005; Thompson *et al.* 2009). All three biomes are recognized biodiversity hotspots, the Succulent Karoo, Maputaland- Pondoland-Albany (Subtropical Thicket), the Cape Floristic Region (Fynbos), and contain high plant diversity and endemism (Mittermeier *et al.* 2005). Rangelands in the Little Karoo have provided forage for large numbers of livestock since European settlement of the area in the 1730s (Thompson *et al.* 2009). Intensive grazing practises have left the Little Karoo with widespread degradation (Hoffman & Ashwell 2001). Approximately 15% of the region is severely degraded and 53% is moderately degraded. Ten percent of the region is cultivated leaving only 22% as largely natural vegetation (Thompson *et al.* 2009). River floodplains have been severely affected, with 51% of all lower river reaches being severely degraded and only 11% in good condition (Vlok *et al.* 2005; Le Maitre *et al.* 2007; Smith-Adoa *et al.* 2010). Land cover changes have had substantial impacts on the biodiversity and ecosystem services, including water supply, erosion, and flood control (Rouget *et al.* 2006; Gallo *et al.* 2009; Reyers *et al.* 2009; Egoh *et al.* 2010). Rouget *et al.* (2006) and Gallo *et al.* (2009) demonstrated that degradation has resulted in a 35% decline in biodiversity conditions. Furthermore, 20% of the Little Karoo is recognized as threatened ecosystems. Reyers *et al.* (2009) highlight declines of 18% to 44% in ecosystem service levels for grazing, carbon sequestration, erosion control, and water provision. Most authors agree that declines in fund (regulating and supporting)

services, together with the documented biodiversity losses have the potential to decrease the region's long-term productivity and resilience and hence lead to an increase in the region's vulnerability (Reyers *et al.* 2009). Overgrazing is also believed to be associated with salinization, soil loss, sedimentation, declines in water quality, and reductions in nitrogen input, which further undermines the current and future productivity of the system (Le Maitre *et al.* 2009; Reyers *et al.* 2009).

2.11 Overstocking ostrich in the Little Karoo

Evidence from historical records indicates that large areas of the Little Karoo were probably degraded via heavy overstocking of cattle, horses, donkeys, sheep, goats and ostriches by the close of the 19th century (Dean & Milton 2003). However, over the last 150 years the ostrich industry has been identified as the major agent of habitat degradation (Burman 1981; Cupido 2005; O'Farrell *et al.* 2008; Le Maitre *et al.* 2009). The Oudtshoorn Basin (10,163 km2) is located in the eastern region of the Little Karoo (Beinart 2003; Cupido 2005). The ostrich industry started at Oudtshoorn in 1864 however, collapsed during the First World War (Beinart 2003). A spectacular recovery provided the foundation for growth of the ostrich industry within the eastern Little Karoo, which is now the core ostrich production region in South Africa (SAOBC 2009). The deregulation of the industry in 1996, as well as the growing demand for relatively lean ostrich meat, resulted in the area experiencing its greatest increase in ostrich numbers during the latter part of the 20th century (O'Farrell *et al.* 2008; Le Maitre *et al.* 2009; Reyers *et al.* 2009). Ostrich numbers reached a record 300,000 in the 1990s, more than three times greater than the peak numbers recorded during the ostrich boom of the early 1900s (Bonora 2006; Thompson *et al.* 2009). Census estimates for 2002 included 250 000 bird sales and 150 000 birds on farms in the Little Karoo. This is more than five times the total potential capacity of 27 000 large stock units in the Little Karoo (one ostrich = 0.35 LSU) and typifies current agricultural practices in the region (O' Farrell *et al.* 2008).

Ostriches have a significant impact on rangeland vegetation because they strip off leaves or pull out plants rather than bite off foliage. In addition, trampling and territorial displays lead to soil compaction, the removal of the biological soil crust, and the formation of pathways that channel surface water (Milton *et al.* 1994b; Cupido 2005). The Ostrich Industry engages in various methods of ostrich breeding. The 'tropparing' method also known as 'flock breeding' is the most preferred with approximately 60% of farmers in the Little Karoo utilising this method (Cupido 2005). Flock breeding is, however, also the most environmentally destructive method of ostrich farming and has significant impacts on vegetation cover and plant composition. Ostrich camps averaging 300 ha in size are used for breeding ostriches at unnaturally high densities of *c.* 1.5 ha/ostrich compared to the

Department of Agriculture's recommended stocking rate of 22 ha/ostrich. Three separate camps are generally used. This allows for a rest period of two seasons, following the utilization of a camp for one complete breeding season (May/June – January) (Cupido 2005; Murray 2007; Herling *et al.* 2009). Unlike goat, sheep or cattle farming, flock breeding with ostriches does not offer any incentive for maintaining vegetation in good condition. Unused veld or veld in good condition only provides up to 25% of the nutrition for the ostriches in the camp for the initial first 3 months, after which the vegetation in the camp becomes too degraded to provide additional forage. Therefore, ostriches receive full rations of supplementary feed, predominantly lucerne throughout the breeding season, which sustains the current high stocking rates (Cupido 2005; Murray 2007). Ostrich farmers are thus only dependent on the these natural ecosystems for space, an ecosystem service defined as a habitat function by de Groot *et al.* (2002), and are not dependent on the natural vegetation for any forage production services or related goods and services (Herling *et al.* 2009).

Poor ostrich grazing practises have been concentrated within the Succulent Karoo plant communities of the Little Karoo, leaving less than 2% of the area in good condition. This is to be compared with 12% of the thicket, 68% of the renosterveld, and 85% of the fynbos, which is well preserved mainly because it provides poor grazing for livestock (Le Maitre *et al.* 2007). Approximately 27% of the Succulent Karoo vegetation in the Little Karoo has been assessed as severely degraded, 61% moderately degraded and 9% cultivated (Le Maitre *et al.* 2009). Le Maitre *et al.* (2009) calculated, according to these levels of degradation, that potential stocking levels have fallen from 5 087 LSU under pristine conditions to 3 328 LSU under current conditions. This represents a 35% reduction in carrying capacity. Assuming a LSU value of R1 800 for this region, this translates into a R3.1 million decline in annual grazing values over the past 100 or so years (Le Maitre *et al.* 2009). Within the Oudtshoorn basin, degradation is most severe in bottomland and pediment Succulent Karoo habitats known as Gannaveld and Apronveld, where less than 2% of each habitat is intact, and 39% and 54%, respectively, is severely degraded (Thompson *et al.* 2009). These two habitats are most accessible to livestock and when in an intact state include a high abundance of palatable plants (Vlok *et al.* 2005; Esler *et al.* 2006). The two habitats associated with upland, rocky terrain Karoo- Thicket Mosaic and Spekboom Thicket are less accessible to livestock and therefore have the highest proportions of extant intact habitat (Thompson *et al.* 2009).

2.12 Striking the balance between ostrich production and biodiversity conservation
Three internationally-funded conservation and development programs (the Cape Action Plan for People and the Environment, the Subtropical Thicket Ecosystem Program, and the Succulent Karoo

Ecosystem Plan) have identified the Little Karoo as an area of conservation importance due to its global significance, high biodiversity value and threatened state (Reyers *et al.* 2009). This has attracted local and international investment in conservation programs, including the establishment of the Gouritz Initiative and the Biodiversity Ostrich Initiative (Reyers *et al.* 2009). The Biodiversity Ostrich Initiative was formed through a partnership between the Gouritz Initiative, the Succulent Karoo Ecosystem Programme (SKEP) and the South African Ostrich Business Chamber (SAOBC) in 2007. It aims to promote the sustainability of the South African ostrich industry through the development of farm-level best practice guidelines and the provision of extension and advisory support to ostrich farmers (Le Maitre *et al.* 2009; SAOBC 2009).

The Biodiversity Unit completed a "Long Term Biodiversity Management Strategy for the South African Ostrich Industry" in 2009 (SAOBC 2009). During 2010, funds for the project dried up and the Biodiversity Unit was no longer operational. Recently, in January 2011, the SAOBC received fresh funds to implement the Biodiversity Management Strategy (S. Botha pers. comm. 2010). This strategy has as its mission to recognise and safeguard the biodiversity of the Little Karoo and support sustainable farming practises through improved veld management, rehabilitation, climate change adaptation and research. In considering various strategies to achieve this mission, the SAOBC identified a possible industry switch to pen breeding as one of the key strategic initiatives. It is not commercially viable for farmers to stock at the densities recommended by the Department of Agriculture. Therefore, rather than reducing ostrich numbers and thus income for the private landowner, the SAOBC believes that an industry-wide adoption of pen-based ostrich breeding or strictly environmentally regulated and monitored veld-based flock breeding provides a "win-win" solution for both conservationists and farmers (SAOBC 2009).

Pen breeding consists of a farmer keeping two, three or four ostriches in a fenced camp of 0.25 ha, on complete feed. Pen breeding encourages continuous genetic improvement through proper record keeping and selection and shows huge potential for increased production (Murray 2007; SAOBC 2009). Alternative strategies for maintaining biodiversity within a sustainable production system are also being researched. These include "trade-offs", whereby a farmer sacrifices portions of his land for concentrated flock breeding, thus allowing more ostriches per hectare on natural veld than the prescribed stocking rate. However, under this regime it is suggested that the farmer sign an agreement with the mandated conservation authority to improve veld management on other parts of the property. Another strategy has as its aim to increase the number of ostriches per pen and increase pen sizes while removing ostriches from the natural veld (SAOBC 2009). The rationale behind promoting all three strategies, particularly pen breeding, is to decrease the pressure on the natural

veld. If a farmer can achieve increased production from a pen breeding system, s/he will remove breeding ostriches from the natural veld and the land will be available for rehabilitation. It has also been suggested that indigenous game or cattle can be introduced and stocked at a low rate within the natural veld in order to generate alternate income (Murray 2007; SAOBC 2009; Herling *et al.* 2009). Agri-tourism is growing in the region and this could encourage farmers to make the switch (O'Farrell *et al.* 2008). However, currently most of the ostrich farms are severely degraded and need to be rested for several years to decades in order for any type of sustainable farming (Herling *et al.* 2009; Thompson *et al.* 2009). In reality, the short-term costs attached to the withdrawal of ostrich, rehabilitation of veld, and the switch to alternative livestock are very high and hold minimal appeal for farmers (Herling *et al.* 2009).

2.13 Paying for rehabilitation in the Little Karoo

The Little Karoo, like the rest of South Africa, consists of mostly privately-owned land with state land totalling <20% of the region (Gallo *et al.* 2009). Costs of rehabilitation include acquisition or opportunity and other compensatory costs (Le Maitre *et al.* 2007; Gallo *et al.* 2009; Egoh *et al.* 2010). The South African ostrich industry is one of the largest exporters of red meat in the country and ninety percent of ostrich products (meat, leather and feathers) are exported to Europe, adding R1.7 billion in revenue per annum to the gross domestic product (SAOBC 2009). The ostrich industry forms a critical part of the local economy, employing over 30% of the population and provides 20 000 jobs (Murray 2007; O'Farrell *et al.* 2008; SAOBC 2009). It is important to note that a switch to pen breeding ostrich farming should not affect the number of workers required per ostrich farm (H. Jonker pers. Comm. 2010). Therefore, as there is an alternative option for increased and continued ostrich farming in the region the shift to pen breeding should not affect the local economy. In fact, rehabilitation of degraded lands could provide further job opportunities as well as openings for entrepreneurship in the region. This is especially in the form of indigenous nurseries as well as employment opportunities for seed collectors and rehabilitation teams. It is evident that the ostrich industry in partnership with the local community and conservationists are trying to conserve the Little Karoo's threatened natural capital. However, without incentives or payment structures it is unlikely that the farmers will be able to make the switch to pen breeding in addition to rehabilitating degraded lands in the short-term. This is due to the lack of knowledge, high start-up costs for farmers, discount rates and enormous costs of rehabilitation programs (Wiegand *et al.* 1995, Le Maitre *et al.* 2007; Herling *et al.* 2009; Le Maitre *et al.* 2009).

The question of who pays for the rehabilitation or the protection of ecosystem services versus who benefits from this protection or rehabilitation is complicated and can be problematic. Farmers in the Little Karoo produce a wide variety of ecosystem services that are valued by society. These include regulation functions, habitat functions, provisioning functions, and information functions (de Groot *et al.* 2002; Reyers *et al.* 2009; Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). Some outputs, such as meat, leather and feathers, are sold in well-established markets (SAOBC 2009). Many, however, like the production of forage for domestic livestock, erosion control, freshwater flow regulation, carbon storage, biodiversity, cultural information and tourism do not have established markets. Without well-defined markets for these ecosystem services, landowners are not rewarded financially for supplying them. If a portion of the value that society places on ecosystem services could be captured by farmers, they would be more likely to keep their land in a natural state (Ribaudo *et al.* 2010). Payment for Ecosystem services deserves investigation in the Little Karoo (Le Maitre *et al.* 2007; Herling *et al.* 2009). Establishing a PES could highlight the private benefits of conserving ecosystems and therefore encourage landowners to increase their efforts in switching to an alternative ostrich farming system and resting or rehabilitating other parts of their land. Public-funded poverty relief programs that clear invasive alien plants and restore hydrological function set up as Payment for Ecosystem Services projects have shown potential in South Africa (Turpie *et al.* 2008) and could be an option for the Little Karoo (Reyers *et al.* 2009).

The following four ecosystem services have been identified by the Gouritz Initiative forum as being important in the Little Karoo, the production of forage for domestic livestock, erosion control, freshwater flow regulation, and tourism (Reyers *et al.* 2009). Although carbon storage was not identified by the GI Forum, it has shown potential for rehabilitation activities in the region (Mills & Cowling 2006, 2010). Tourism is seen as a significant economic growth opportunity because it creates employment. Recent tourism surveys in the Little Karoo region found that the unspoilt nature of the region was a major draw-card for tourists. When tourists were asked, "What did you enjoy most about the area?" the most frequent response was "scenery" followed by "nature" (Le Maitre *et al.* 2009).

2.14 Rehabilitation in the Little Karoo

There has been no research describing the success of active rehabilitation inventions on degraded ostrich rangelands in the Little Karoo. Although various rehabilitation methods have been described for practise (Coetzee 2005), none of these techniques have been scientifically evaluated and the results published.

Over the last decade, the BIOTA (BIOdiversity Monitoring Transect Analysis) research project in Africa conducted four major rehabilitation trials in the Succulent Karoo biome (Simons & Allsopp 2007; Schmiedel *et al.* 2010; Hanke *et al.* 2011). The evaluation of these rehabilitation experiments has opened new insights into the treatment effects on soil properties and vegetation. Studies occurred in three major ecological regions of the Succulent Karoo biome - Namaqualand, Richtersveld and the Knersvlakte. The treatments used in these rehabilitation projects included a combination of the following techniques: livestock exclusion, brushpacking, dung mulching, quartz stones, micro-catchments and the use of functional plants. Overall results indicate that annuals respond more strongly to treatments such as livestock exclusion, dung mulching, scattering of quartz stones and brushpacking than perennial plants. Only two of the rehabilitation treatments increased perennial vegetation cover (scattering quartz stones and transplants of adult plants). Chemical soil properties were increased by all treatments, which were associated with the introduction of organic matter. This was usually reflected by an increase of carbon in the soil. Storage of soil water was improved by planting, brushpacks and dung (Simons & Allsopp 2007; Schmiedel *et al.* 2010; Hanke *et al.* 2011). For micro-catchments, no effect on soil water was detected (Hanke *et al.* 2011). All treatments that shaded the soil surface and protected the soil surface from wind (i.e. transplants, brushpacks, and dung) had a positive effect on soil water storage (Schmiedel *et al.* 2010).

Rehabilitation of degraded rangelands in the Succulent Karoo Biome is considered to be in its pioneer phase (Schmiedel *et al.* 2010). Apart from the BIOTA studies, only two other types of active rangeland rehabilitation treatments have been conducted in the Succulent Karoo. These consisted of planting adult plants (Anderson *et al.* 2004) and application of mulch and gypsum (Beukes & Cowling 2003). In addition, experiences from mine spoil rehabilitation projects (Blignaut & Milton 2005; Carrick & Kruger 2007) have shown that floristic features like the occurrence of succulence and the high drought tolerance of many seedlings present good opportunities for the success of species re-introductions. The Succulent Karoo biome is an extraordinarily heterogeneous biome where the various habitats are driven by a high diversity of topography, soil types, rainfall patterns, floristic inventories and historical and recent land-use. Results from rehabilitation experiments conducted on different soil types and within different rainfall zones may not apply to the Succulent Karoo vegetation within the Little Karoo region as habitats have been shown to respond differently to rehabilitation measures (Schmiedel *et al.* 2010). Most of the research on rehabilitation in the Succulent Karoo has been concentrated within the typical uni-modal winter rainfall region of the Succulent Karoo whereas ostrich farming is concentrated within the eastern Little Karoo where rain

may fall at any time of the year and the most predictably wet seasons are spring (September-October) and autumn (April-May) (Mucina *et al.* 2006).

2.15 Conclusion

The Little Karoo presents an opportunity for assessing the costs and benefits of reversing the impacts of land degradation in an arid socio-ecological system (O'Farrell *et al.* 2008). The effect of inappropriate land-use practices on biodiversity and ecosystem services has been profound. A study identifying the benefits of rehabilitation for improving ecosystems services at farm level has the possibility of highlighting these benefits to farmers (O'Farrell *et al.* 2007; 2009). This could incentivize farmers to rehabilitate their land (Hobbs & Harris 2001; Herling *et al.* 2009). The benefits of rehabilitation need to be quantified in order to motivate for fiscal and regulatory methods such as incentives, payments, punitive actions, and statutory requirements (Le Maitre *et al.* 2009). The broad range of basic ecological research as well as the various rehabilitation trials conducted over the last decade within the winter rainfall regions of the Succulent Karoo have contributed significantly to an understanding of these ecosystems (Schmiedel *et al.* 2010). However, many important aspects are still not clearly understood. Research identifying the most affordable, practical, sustainable and socially-acceptable rehabilitation techniques is needed in the area. This would improve the rehabilitation potential of the ostrich farms within the Little Karoo (Le Maitre *et al.* 2007; Herling *et al.* 2009). The socio-economic benefits of ecological rehabilitation are critical for their societal and political appreciation and support. However, they are insufficiently quantified in scientific studies (Aronson *et al.* 2010a).

Chapter 3:

The effects of seed, mulch, ripping and micro-catchment rehabilitation treatments on vegetation recovery, plant diversity and grazing services in the Succulent Karoo lowlands of the Little Karoo.

3.1 Abstract

Heavy grazing alters vegetation composition and decreases primary productivity, especially palatable plant species. This reduces the ability of rangelands to render vital ecosystem services such as forage production for livestock. Active rehabilitation is required to reverse the degradation process and increase grazing capacity on a time-scale relevant to landowners. This study investigated whether four common restoration techniques (ripping, micro-catchments, sowing seed and mulching) could generate short-term (one-year after rehabilitation) improvements on degraded ostrich rangeland in the Little Karoo. Micro-catchments increased palatable plant species richness and plant density; seed treatments increased palatable species richness, and mulching increased plant diversity (Shannon diversity index). Although direct benefits of rehabilitation were highlighted in this study, none of the treatments proved cost-effective, as the return on investments in the short-term is not likely to be more than the rehabilitation costs. Ripping (R $0.13/m^2$) was the least costly however least effective method in the short-term. In terms of cost, this was followed by sowing seed (R $0.37/m^2$), micro-catchments (R $1.54/m^2$) and mulching (R $2.47/m^2$). Micro-catchments were the most effective and less costly then mulching. Increases in plant density were mainly of newly established palatable seedlings and young plants. Seedling growth was hindered by grazing springbok (*Antidorcas marsupialis*). Therefore, despite increases in palatable plant density, treatments had no positive effects on vegetation cover and did not contribute to increasing grazing services. Mulching actually reduced grazing capacity by creating a barrier for seed germination as well as physically damaging less palatable and unpalatable herbaceous and succulent seedlings. Decreases in abundant less palatable and unpalatable plants allowed for greater evenness within mulched plant communities and this increased Shannon plant diversity (H). Decreases in less- and unpalatable community dominance may benefit palatable species in the long-term due to reduced competition within mulched areas. Government subsidisation for rehabilitation programmes in the Little Karoo should be investigated as rehabilitation treatments are labour intensive and costly. Alternatively, landowners can use resources available on site to reduce costs. Further long-term monitoring of this site is crucial for the effects of rehabilitation to be adequately quantified.

3.2 Introduction

Degradation of semi-arid rangelands poses a serious problem worldwide and their rehabilitation has become an important field of research (Schmiedel *et al.* 2010). Restoration intends to restore an ecosystem to its former indigenous historic state. Rehabilitation on the other hand seeks to halt degradation and redirect a disturbed ecosystem back to a pre-disturbed functional state (Aronson *et*

al. 1993; SER 2004). Arid and semi-arid rangelands in South Africa have undergone wide-scale and dramatic changes attributed primarily to overgrazing (Yeaton & Esler 1990; Dean & MacDonald 1994; Milton & Dean 1995; Allsopp 1999; Todd & Hoffmann 1999; Snyman & du Preez 2005). This has resulted in large-scale degradation and subsequent biodiversity loss (Milton & Dean 1995; Todd & Hoffmann 1999; Hoffman & Ashwell 2001; Rouget *et al.* 2006).

The semi-arid Succulent Karoo biome is one of the world's centres of biodiversity and endemism. It is also one of only two semi-arid global biodiversity hotspots (Mittermeier *et al.* 2005). Degradation due to unsustainable farming practises has been widespread (Anderson & Hoffman 2007; Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). Ninety percent of the biome is used for grazing livestock such as sheep, goat, cattle and ostrich. Only 6% is formally conserved (Mucina *et al.* 2006; O'Farrell *et al.* 2009). Several studies have shown that high stocking rates transform the Succulent Karoo lowlands from a diverse mix of palatable perennial shrubs to a relatively uniform mix of unpalatable perennial shrubs and annual herbaceous species. Annuals increase in density and cover in good rainfall years, however seeds remain dormant under dry conditions (Todd & Hoffman 1999; Riginos & Hoffman 2003; Anderson & Hoffman 2007; Todd & Hoffman 2009). Where perennial vegetation has been reduced, denuded patches dominate during the dry period and are susceptible to soil erosion and compaction (Snyman & du Preez 2005; Le Maitre *et al.* 2007; Schmiedel *et al.* 2010; Hanke *et al.* 2011).

Degradation in the Succulent Karoo biome has been most severe within the intermontane Succulent Karoo lowlands of the Little Karoo (Le Maitre *et al.* 2009; O'Farrell *et al.* 2010). The Little Karoo is the major ostrich production region in South Africa (Murray 2007). Ostriches are kept at unnaturally high densities (1.5 ha/ostrich versus the recommended 22 ha/ostrich) in breeding camps *c.* 300 ha in size (Cupido 2005; Herling *et al.* 2009). Ostriches strip plant branches, often pull out whole plants, as well as damage vegetation and biocrust through their trampling activity (Milton *et al.* 1994b; Cupido 2005). Trampling, in turn compacts the soil and reduces water infiltration (Cupido 2005). The reduction in cover of forage plants through poor grazing management decreases the forage production potential of the land, causes erosion and reduces tourism value (Reyers *et al.* 2009). Indications are that poor biodiversity management could result in trade restrictions for the ostrich industry because of potential buyers demanding 'ethical trade' and/or clean production. As a result the South African Ostrich Industry has funded a Biodiversity Unit tasked with the implementation of a Long Term Biodiversity Management Strategy. Improved land management and active rehabilitation has been identified as a way to ensure sustainability for the ostrich industry and continued international market access (SAOBC 2009).

Active rehabilitation aims to reduce the loss of topsoil, improve vegetation cover, particularly palatable plants, and aims to enhance ecosystem resilience by increasing plant diversity (Van der Merwe & Kellner 1999; Beukes & Cowling 2003; Snyman 2003; Visser *et al.* 2004; Van den Berg & Kellner 2005). Soil interventions commonly used to break up compacted soil include ripping and micro-catchments (Snyman 2003; Visser *et al.* 2004; Van den Berg & Kellner 2005; Vohland & Barry 2009; Hanke *et al.* 2011). Brushpacking, mulch and gypsum have been used to stabilise the soil and promote vegetative growth (Beukes & Cowling 2003; Schmiedel *et al.* 2010). The success of ripping, seeding and soil amendments (fertilization, mulching, brushpacking) have been evaluated on private ranches in Kenya (Kinyua *et al.* 2009), semi-arid rangelands in the Nama-Karoo Biome (South Africa) (Visser *et al.* 2004; Van den Berg & Kellner 2005) and in western United States (Sheley *et al.* 2006; Wentz 2004). Micro-catchments have been used extensively in African and Indian rangelands, however, mainly for crop plants and not for rangeland improvements or biodiversity conservation (Vohland & Barry 2009). Three studies in South Africa have included ripping and micro-catchments (Van der Merwe and Kellner 1999; Snyman 1999). Only one study has focused on using ripping, micro-catchments and sowing seed (Snyman 2003). In the winter rainfall Succulent Karoo rangelands, studies have focused on seeding, mulching, gypsum (Beukes & Cowling 2003), planting plants that function as seed and silt traps (functional plants) (Anderson *et al.* 2004; Schmiedel *et al.* 2010), brushpacking, micro-catchments (Schmiedel *et al.* 2010; Hanke *et al.* 2011), dung treatments and quartz stones (Schmiedel *et al.* 2010).

Vegetation recovery of severely degraded areas by means of natural succession is very slow. Studies from South Africa and the United States suggest that without active intervention, vegetation regeneration does not occur on practical time-scales (Milton & Dean 1995; O'Connor & Roux 1995; Wiegand & Milton 1996; Valone *et al.* 2002; Valone & Sauter 2005; Sasaki *et al.* 2009; Miehe *et al.* 2010; Seymour *et al.* 2010). However, short-term costs along with the low likelihood of success make rehabilitation efforts by private landowners undesirable (Esler & Kellner 2001; Herling *et al.* 2009). Successful funding mechanisms in South Africa are Payments for Ecosystem Services (PES) (Blignaut *et al.* 2008) such as the Working for Water and the Landcare programmes (Turpie *et al.* 2008). PES is an option for the Little Karoo (Herling *et al.* 2009). However, research on the effects of rehabilitation treatments is required before reasonable support for PES can be garnered. There is little information on soil preparation and seeding methods for vegetation re-establishment in ostrich camps. Although some experimental rehabilitation trials have been conducted in parts of the Karoo (Beukes & Cowling 2003; Visser *et al.* 2004; Van den Berg & Kellner 2005; Carrick & Kruger 2007; Simons & Allsopp 2007; Schmiedel *et al.* 2010), approaches differ with soil type and rainfall

distribution. This study was designed to test the effects of seed, mulch, ripping and micro-catchment rehabilitation treatments on palatable and less palatable plant recovery. Treatments were applied to a severely degraded ostrich farm to test whether improving infiltration through soil disturbance, conserving moisture through mulching, and adding species through sowing seed, leads to an increase in vegetation recovery and forage value. A major goal was to identify effective and cost-efficient rehabilitation methods for short-term improvements in plant species richness, plant density, vegetation cover, plant diversity and grazing services.

3.3 Methods

3.3.1 Study Site

The study was carried out on a severely degraded ostrich farm (Morestêr farm, 33º35'50.1" S, 22º02'36.7" E) in the Succulent Karoo lowlands of the Oudtshoorn basin (10,163 km^2) in the Little Karoo (19,730 km^2). The Little Karoo is located between the coastal Langeberg and Outeniqua mountains and the inland Anysberg, Swartberg and Antoniesberg mountain ranges. It extends from the town of Montagu in the west to Uniondale in the east, and encompasses four local municipal areas, Ladismith, Calitzdorp, Oudtshoorn and Uniondale (O'Farrell *et al.* 2008) (Fig. 3.1).
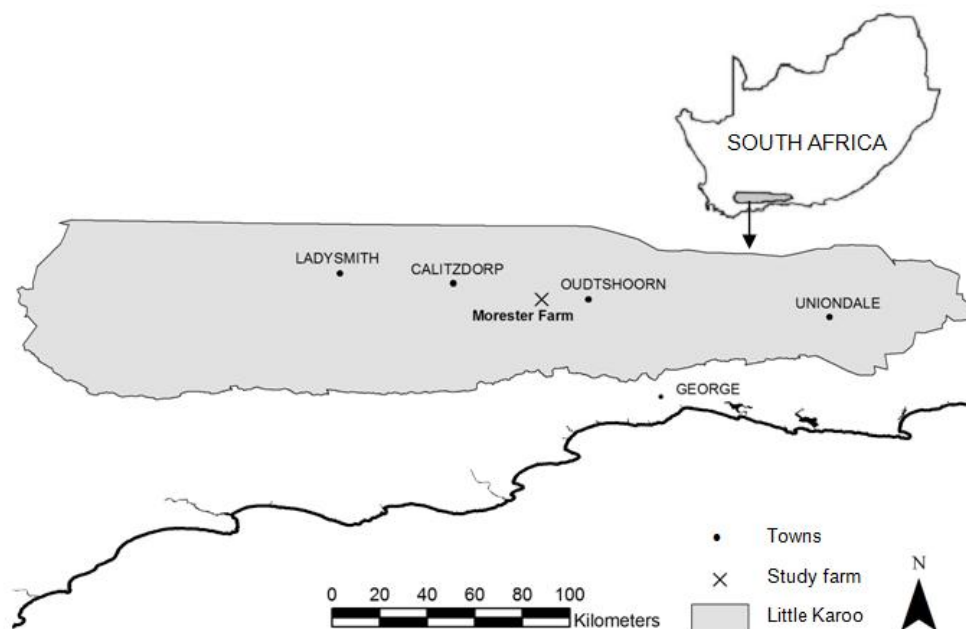

Figure 3.1 Map showing the study farm, Morestêr and the major district towns within the Little Karoo (shaded) in South Africa (Vlok *et al.* 2005).

Vlok *et al.* (2005) found that only 17.4% of the Little Karoo is made up of Succulent Karoo vegetation types, with sub-tropical thicket and Fynbos accounting for more than 60%. Succulent

Karoo habitats of the Little Karoo occur in low-lying areas, where soils are nutrient rich and rainfall is less than 350mm per annum (Vlok *et al.* 2005).

The Oudtshoorn basin falls between two mountain ranges, the Swartberg mountain range in the north and the Outeniqua mountain range in the South and is situated approximately between the towns of Calitzdorp in the west and Oudtshoorn in the east (Thompson *et al.* 2009) (Fig. 3.2). Since 1865, the Oudtshoorn basin has been at the centre of the Little Karoo's ostrich industry (Beinart 2003; Cupido 2005). It is now considered the most important ostrich production region producing more than 70% of South Africa's ostrich products (Cupido 2005, Murray 2007).

The climate is warm-temperate and sub-mediterranean. Mean annual rainfall ranges from 150 to 250 mm, increasing to 400 mm on the lower slopes of the mountains, and up to 1000 mm on the highest, coast-facing peaks (Thompson *et al.* 2009). Rain may fall at any time of the year, however the most predictably wet seasons are spring (September–October) and autumn (April–May). Summers are hot (daily maxima up to 40 °C are frequent) and dry (Thompson *et al.* 2009).
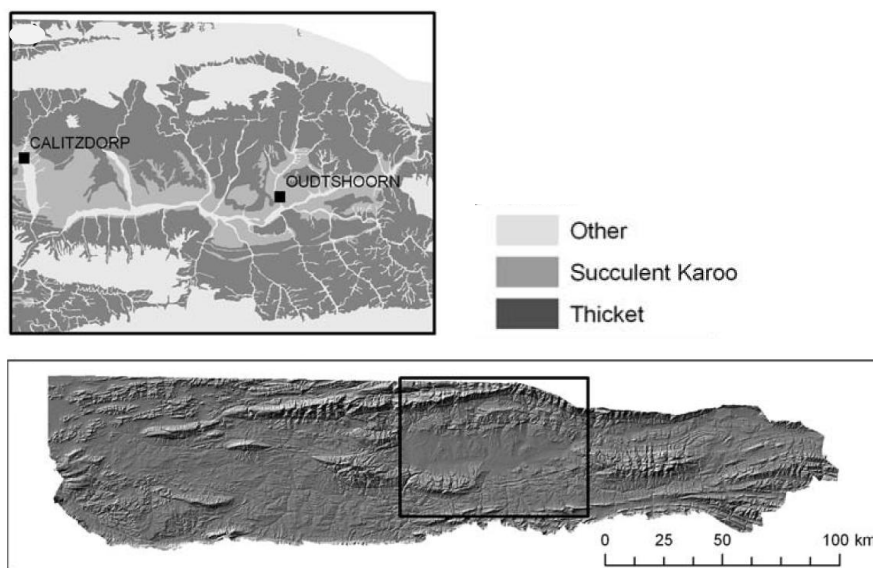
Figure 3.2 Location of the Oudtshoorn basin between Calitzdorp and Oudtshoorn within the Little Karoo, showing the topography of the area and the Succulent Karoo lowlands (Thompson *et al.* 2009).

The low-lying parts of the basin are dominated by the dwarf, succulent shrublands, mainly Gannaveld and Apronveld vegetation types, associated with the Succulent Karoo biome. Lower slopes are covered in dense thicket associated with the sub-tropical thicket biome; the upper slopes consist of fire-prone shrublands and heathlands of the fynbos biome (Low & Rebelo 1996; Vlok *et al.* 2005; Thompson *et al.* 2009). Gannaveld and Apronveld have been severely degraded by overstocking ostrich (Cupido 2005; Thompson *et al.* 2009). The recommended grazing capacity is 60 ha/Large Stock Unit (LSU) (22ha/ostrich), however farmers overstock at *c.* 1.5 ha/ostrich (Cupido

2005; Herling *et al.* 2009). According to Thompson *et al.*'s (2009) degradation map, less than 2% of these vegetation types are in relatively natural condition. 39% of Gannaveld and 54% Apronveld have been severely degraded with 59% and 44% being moderately degraded (Thompson *et al.* 2009).

Fieldwork on Morestêr farm was conducted within a historically intensively used ostrich camp (247 ha). The camp is dominated by Gannaveld vegetation, an open semi-succulent (<0.5 m) shrubland comprising *Salsola*, *Lycium* and *Tripteris* species, occurring on bottomlands with deep loamy sands characterised by a high salt content (Vlok *et al.* 2005). It falls within an area of severely degraded Gannaveld (Thompson *et al.* 2009) where rehabilitation has been proposed (Vlok *et al.* 2005). Since the 1940s, this camp has been stocked every two to three years with ±80 ostriches (3 ha/ostrich). In other words, ostriches have been on the veld for ±23 breeding seasons with ±20 years of resting dispersed between stocking years. Ostriches were removed from the camp in 2007 (J. Ernst pers. comm. 2010). In 2008, the camp was classified as severely degraded (S. Milton and K. Coetzee *in litt.* 2008). About 60% of the area lacked vegetation and large patches of soil were exposed and compacted. Palatable plant species and biological soil crust was lost except for under a few shrub clumps. Although the shrubland had been severely degraded in the most intensively used parts of the camp it still included a number of disturbance-adapted species such as *Ruschia spinosa*, *Drosanthemum lique*, *Malephora lutea*, *Augea capensis*, *Leipoldtia schultzii*, *Delosperma spp.*, *Mesembryanthemum junceum*, *Drosanthemum hispidum*, *Pteronia pallens*, *Galenia pubescens* and *Rosenia humilis* (K. Coetzee and S. Milton *in litt.* 2008).

3.3.2 Experimental layout

A factorial (crossed) plot design, incorporating ripping, micro-catchments, seed and mulch rehabilitation treatments was set out in January 2009. The design consisted of three 45 m by 45 m study blocks which were marked out in close proximity (±20 m) (Fig. 3.3). In February 2009, the three study blocks were rehabilitated according to the crossed plot design. Micro-catchments, ripping and untreated control plots were randomized within nine 15 m by 15 m plots within each 45 m by 45 m study block. Each 15 m by 15 m soil treated or control plot was further sub-divided into a non-randomised design consisting of four 7.5 m by 7.5 m subplots (SP 1-4) (Fig. 3.4). Mulch was applied to two 7.5 m by 7.5 m subplots within all plots and study blocks. Seed of palatable and less palatable species were sown into one of the mulched subplots and one of the unmulched plots within all plots and study blocks. Soil and mulch treatments were applied in the first week of February 2009 and seed was sown on the 19th of February 2009.

There was no disturbance from ostriches within the camp during this study; however, eleven springbok (*Antidorcas marsupialis*) were kept on the farm until the end of 2009. In addition, steenbok (*Raphicerus campestris*) and other small herbivores were not excluded from browsing the rehabilitation site.
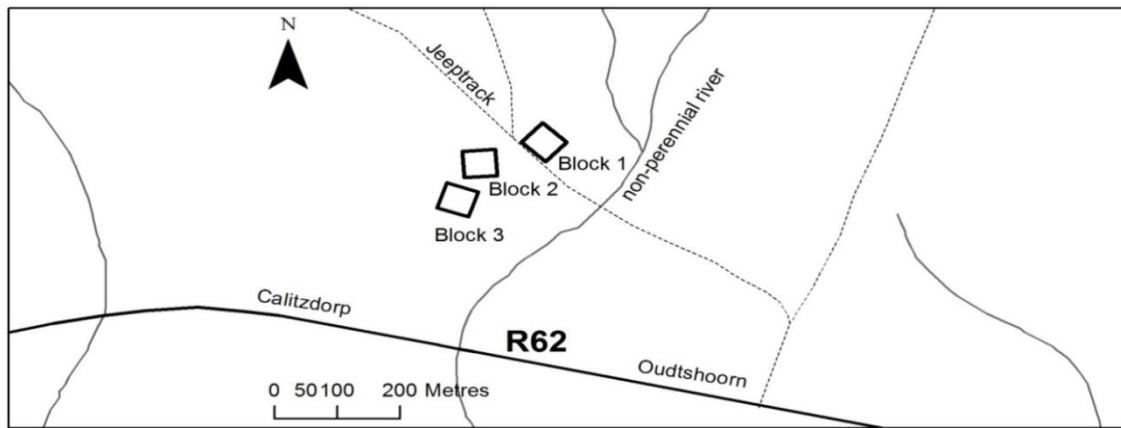

Figure 3.3 Three 45 m x 45 m rehabilitation study blocks on Morestêr ostrich farm – a severely degraded farm in the Little Karoo.
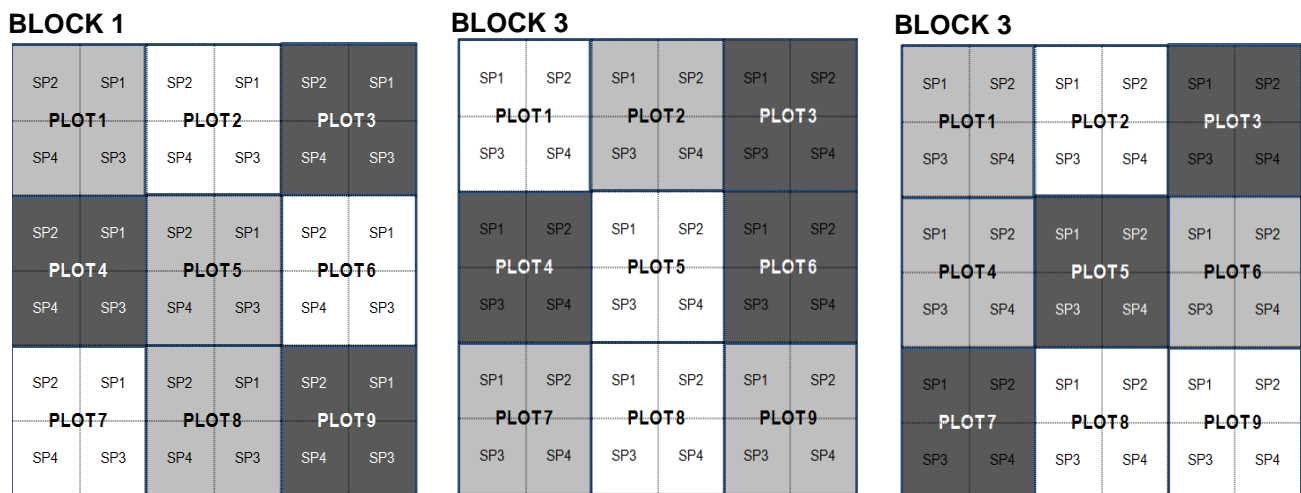
Figure 3.4 Randomized soil treatments (dark grey = ripped, light grey = micro-catchments, white = control (untreated)) at the study site within each 45 m by 45 m study block; overlaid by a crossed plot non-randomised design of additional treatments (SP1 = Soil treatment/control; SP2 = Soil treatment/control + seed; SP3 = Soil treatment/control + mulch; SP4 = Soil treatment/control + seed + mulch).

3.3.4 Rehabilitation Treatments

Micro-catchment and ripping soil treatments

Micro-catchments (i.e. holes *c*. 600 mm in depth and *c*. 500–900 mm in diameter) were made in the hard soil by hand, using a soil pick. Loosened material was shovelled into a low berm (micro-catchment wall) on the downhill side of the micro-catchment. Micro-catchments were spaced about 2 m apart and rows were spaced 1 m apart. Each micro-catchment was filled with a layer of wood-chip mulch (60 mm deep). Ripping the soil involved using a ripper that ripped long furrows to a depth of

100 mm using a single-tine ripper drawn by a tractor. Rip lines were cut along the contour (Coetzee 2005; W. Stroebel. pers. comm. 2010).

Mulch treatments

A thick layer (*c.* 100 mm deep) of coarse wood-chip mulch was applied to one-half of each 15 m by 15 m plot. Mulch consisted of a rough consistency of gum and pine wood-chips approximately 100–150 mm in length. These were placed to form a dense mat on the soil surface (Coetzee 2005; W. Stroebel. pers. comm. 2010).

Seed treatments

The seed mix consisted of two highly palatable, two palatable and two less-palatable indigenous plant species. One grass, two succulent and three shrub species were sown at the site. A total of 2.2 kg of seed was sown throughout the 54 subplots (3,037.5 m^2) with a mean of 36-filled seeds/m^2 (Table 3.1) (S. Milton. *in litt*. 2009).

Table 3.1 Species and quantities of seed sown at Morestêr farm in the Little Karoo (S. Milton. *in litt.* 2009).

Seed mix			Quantities	
Growth form	Species	Kg seeded	Total no. of seed sown	seed/m^2
Grass	*Fingerhuthia africana*	0.2	54,560	17.96
Succulent	*Ruschia spinosa*	0.1	40,000	13.17
Succulent	*Drosanthemum hispidum*	0.2	280,000	92.18
Shrub	*Tripteris sinuata*	0.4	24,880	8.19
Shrub	*Tetragonia fruticosa*	0.3	8,520	2.80
Shrub	*Lessertia annularis*	1	144,800	47.67
	Total (mean/m^2)	2.2	552,760	36

3.3.5 Field measurements

Before rehabilitation was applied in February 2009, baseline vegetation surveys were conducted within the three study blocks. Vegetation surveys measured plant density, species richness and vegetation cover. In January 2010, these surveys were repeated in order to determine the effect of rehabilitation on the change in plant density, species richness, vegetation cover, plant diversity and grazing capacity.

Species richness and plant density

All plant species were identified and individuals per species were counted within 5 m by 5 m quadrats within each subplot (1-4) for the three-study blocks. This resulted in 108 quadrats being surveyed in January 2009 and resurveyed in January 2010.

Shannon Plant Diversity

The Shannon index of diversity was calculated from the species richness and density data. The index is defined as:

$$H = -\sum pi \ln pi.$$

The term *pi* is the proportion of a particular species in a sample, which is multiplied by the natural logarithm of itself. *H* is derived by summing the product for all species in the sample.

Vegetation cover

The line intercept method was used to measure vegetation percentage cover at the site. Two transects of 7.5 m were used per subplot. Transects were 3 m apart and set down slope. Cover in meters was summed and divided by the total transect length to calculate percentage cover.

Current Grazing capacity

Vegetation cover data were used to determine the current grazing capacity (ha/LSU) of the site and was calculated according to the grazing index method (Du Toit 1995, Du Toit *et al.* 1995; Du Toit 2000; Du Toit 2002; Esler *et al.* 2006). Current grazing capacity was calculated as, (Benchmark value (500) / veld condition index) x (7.14). The cover of each individual species recorded was multiplied by its specific objective grazing index value, e.g. *Pentzia incana* (5.6% * 2.88 (OGIV)), in order to calculate the veld condition index (16.13). All these individual veld condition indices were summed, to give the total veld condition index for the site. This veld condition index was then compared to the known standard benchmark index (500) (Du Toit 2002; Esler *et al.* 2006) by dividing the benchmark index by the site veld condition index in order to calculate the grazing capacity of the veld in hectares per Small Stock Unit - ha/SSU (Merino sheep). This was then multiplied by a conversion factor of 7.14 to convert the grazing capacity into hectares per Large Stock Unit – ha/LSU (Du Toit 1995, Du Toit *et al.* 1995; Du Toit 2000; Du Toit 2002; Esler *et al.* 2006). This was converted to LSU per hectare for analyses and is presented as LSU/1000 hectares due to very small stocking rates observed. Objectively estimated grazing indices for plant species (OGIV) were obtained from the literature (Du Toit 1995, Du Toit *et al.* 1995; Du Toit 2000; Du Toit 2002; Esler *et al.* 2006). When objective grazing indices were not available for a particular species its subjective index value was used (Du Toit *et al.* 1995).

3.3.6 Statistical analyses

To determine overall changes in vegetation, plant diversity and grazing capacity at the study site the non-parametric Wilcoxon signed-rank test for paired samples was used to compare variables in 2009 to 2010. Once overall changes for the study site had been evaluated, the 2009 measurements for each

variable were subtracted from the 2010 measurements in order to represent the change in plant density, species richness, vegetation cover, plant diversity and grazing capacity after rehabilitation in order to perform statistical analyses.

A multifactor linear model (factorial ANOVA) incorporating four categorical factors was used in order to test the effect of different rehabilitation treatments on the changes observed. Soil, mulch and seed treatment were included as fixed factors and study block was included as a random factor. Study block had no significant effect on response variables and it was removed from the model (Table 3.2).

Table 3.2 The results from the crossed factorial linear model, which tested for random effects of study block on response variables (df=2, n=36).

Response variable	F - test statistic	*p-values*
Perennial Plant species richness/m^2		
Palatable plant species richness	0.39	0.72
Less palatable plant species richness	2.26	0.76
Total	2.87	0.57
Annual species richness/m^2		
Total	3.53	0.14
Perennial plant density/m^2		
Palatable plant density	3.03	0.86
Less palatable plant density	1.41	0.36
Total plant density	1.5	0.38
Annual plant density/25m^2		
Total	0.15	0.86
Shannon diversity index (H)		
Perennial Plants	1.47	0.32
Perennial and annual plants	0.58	0.59
Perennial vegetation cover (%)		
Palatable vegetation cover	1.24	0.46
Less palatable vegetation cover	0.07	0.93
Total	0.01	0.98
Annual vegetation cover (%)		
Total	0.39	0.71
Grazing capacity (LSU/ha)		
Perennial plants	0.012	0.98
Perennial and annual plants	0.009	0.92

Therefore, all factors were fixed and analysed using fixed effects linear models (three Factor ANOVA), also termed Model 1 analyses of variance. The model consisted of a factorial (crossed) design with three factors (soil, mulch and seed treatments). The effect of soil treatment (three levels, micro-catchments, ripping and no treatment), sowing seed (two levels, with and without) and mulching (two levels, with and without) were evaluated for the change in annual and perennial species richness, plant density, plant diversity, vegetation cover and grazing capacity. Pairwise

comparisons were used to compare treatments and controls (for the soil treatment factor) when main effects or interactions were significant.

The three factor ANOVA was used to measure the main effect of factors and the interaction between factors. The main effect of each factor is the effect of each factor independent of other factors. The interaction between factors is a measure of how the effects of one factor depend on the level of one or more of the additional factors. The absence of an interaction means that the combined effect of two or more factors is predictable by just adding their individual effects together. The presence of an interaction indicates a synergistic or antagonistic effect of two factors. All possible combinations of the three factors were used i.e. complete factorials (Quinn & Keough 2002).

Three general Null hypotheses were tested. The first three were tests of main effects and the second three were tests of interaction. The null hypotheses were as follows:

Main effects:
No difference between mean species richness, density, diversity and vegetation cover in each soil treatment, pooling all levels of mulched and seed factors.
No difference between mean species richness, density, diversity and vegetation cover in mulched or unmulched plots, pooling all levels of soil treatments and seed factors.
No difference between mean species richness, density, diversity and vegetation cover in each seeded or unseeded plots, pooling all levels of soil treatments and mulched factors.

Interaction effects:
No interaction between soil treatments and mulch i.e. the effect of soil treatment on the response variable is independent of mulching and vica-versa.
No interaction between soil treatment and seed i.e. the effect of soil treatment on the response variable is independent of seed and vica-versa.
No interaction between mulch and seed i.e. the effect of mulch on the response variable is independent of seed and vica-versa.
No interaction between soil treatment, mulch and seed i.e. the effect of soil treatment on the response variable is independent of mulching and seeding and vica-versa.

Plants were initially grouped into four subjective palatability groups, highly palatable, palatable, less palatable and unpalatable (Van Breda *et al.* 1990; S. Milton. pers. comm. 2010; J. Vlok pers. comm. 2010; A. Vlok pers. comm. 2010.). However, for analyses, highly palatable and palatable species were grouped together as palatable and less-palatable and unpalatable species were grouped together

as less palatable. Therefore, response variables, perennial species richness, plant density and vegetation cover were analysed separately for two plant groups, palatable and less palatable.

Marginal means (mean for the levels of one factor pooling over the levels of the second and third factor) are reported for main effects and cell means (means of the observations within each treatment combination) are reported for any interaction effects.

Contingency tables were used to determine statistically significant associations between the changes in plant density and cover of certain life forms and rehabilitation treatments. Rank based non-parametric tests were used to assess the effects of soil, seed and mulch treatments on the density and cover of seedling species sown at the study site. The Man-Whitney-U test was used to assess changes between mulch and seed plots versus unmulched or no seed plots. The Kruskal-Wallis test was used determine effects of micro-catchments and ripping. Means are reported for descriptive purposes although these tests are based on ranked data (Quinn & Keough 2002). All analyses were conducted with the IBM SPSS Statistics 19 (SPSS Inc., Chicago, IL, U.S.A.).

Assumptions:

The assumptions of normality and homogeneity of variances for the error terms from the model and the observations were checked using residual plots, examining studentized residuals versus predicted values. Levenes formal tests of homogeneity were also used, however using residual plots are considered a more informative and accurate way to assure that the data meets the assumptions of the tests (Quinn & Keough 2002) (Appendix 3.1). Quinn & Keough (2002) recommend using diagnostic checks such as residual plots over the more formal Levenes tests. Variables with skewed distributions were transformed in order to improve normality and homogeneity. Variables that were transformed using Log_{10} transformations were palatable species richness, annual species richness, palatable plant density, less-palatable plant density, annual plant density and total perennial plant density. Variables that were arcsine and square root transformed were palatable, less palatable and annual vegetation cover and grazing capacities (Quinn & Keough 2002). Parametric tests were used for Shannon diversity indices. Taylor (1978) pointed out that if the Shannon index is calculated for a number of samples the indices will be normally distributed. This property makes it possible to use parametric statistics to compare sets of samples for which the diversity index has been calculated (Sokal & Rohlf 1981; Magurran 1991).

3.4 Results

3.4.1 Overall study site changes (one-year after rehabilitation)

Perennial plant species richness and plant density increased almost two-fold and more than two-fold respectively. This was mainly due to increases in palatable species richness and both palatable and less palatable plant density. Similarly, annual plant species richness and plant density mostly doubled from 2009 to 2010 (Table 3.3). In January 2009, 43 plant species (38 perennial) were recorded on the site, 11 palatable and 27 less palatable. By comparison, 85 (76 perennial) species were recorded at the site in 2010, 29 palatable and 58 less palatable. This is indicative of the overall mean two-fold increase in species richness/$25m^2$. In contrast to the significant increases in species richness and plant density, total perennial vegetation cover remained similar. Palatable vegetation cover increased significantly over the year however only by 1%. Annual vegetation also showed an increase of 1%. There were significant increases in plant diversity, whereas increases in grazing services were not significant for the study site (Table 3.3).

Table 3.3 Overall study farm changes in species richness, plant density, vegetation cover, plant diversity and grazing capacity, one year after rehabilitation (January 2009 to January 2010) (n=108).

Variable	2009 Mean (SE)	2010 Mean (SE)	Diff.	Z-stat	P-value
Perennial plant species richness/25m^2					
Palatable	1.0 (0.2)	3.4 (0.3)	2.4	-7.71	<0.001
Less palatable	4.8 (0.3)	6.4 (0.4)	1.6	-4.90	<0.001
Total	5.8 (0.5)	9.8 (0.6)	4.0	-7.04	<0.001
Annual plant species richness/25m^2					
Total	1.0 (0.1)	2.7 (0.2)	1.7	-8.19	<0.001
Perennial plant density/25m^2					
Palatable	2.1 (0.5)	12.9 (1.9)	10.8	-8.17	<0.001
Less palatable	32.3 (2.9)	101.6 (15.4)	69.3	-5.98	<0.001
Total	34.4 (2.9)	114.5 (16.1)	80.1	-6.68	<0.001
Annual plant density/25m^2					
Total	8.2 (2.4)	20.7 (3.3)	12.5	-5.77	<0.001
Shannon Diversity Index (H)					
Perennial plants	1.0 (0.1)	1.5 (0.1)	0.5	-6.93	<0.001
Perennial & annual plants	1.2 (0.1)	1.7 (0.1)	0.5	-7.28	<0.001
Perennial vegetation cover (%)					
Palatable	2.0 (0.7)	3.0 (0.8)	0.9	-3.96	<0.001
Less palatable	17.0 (2.6)	15.5 (2.1)	-1.6	-0.91	0.36
Total	19.0 (2.9)	18.4 (2.4)	-0.6	-0.32	0.75
Annual vegetation cover (%)					
Total	0.1 (0.1)	1.7 (0.4)	1.6	-6.89	<0.001
Grazing capacity (LSU/1000ha)					
Perennial plants	8.4 (1.2)	8.9 (1.2)	0.6	-0.72	0.47
Perennial & annual plants	8.4 (1.2)	9.5 (1.2)	1.1	-1.72	0.09

There were no significant associations between life form and change in density (χ^2=14.15, df=10, p>005) or cover (χ^2=2.15, df=10, p>005) between 2009 and 2010. Trends showed that increases in perennial palatable plant density were mainly in the form of highly palatable woody shrubs such as

Tripteris sinuata, palatable perennial forbs such as *Chaenostoma subnudum* and palatable perennial grasses e.g. *Fingerhuthia africana*. Increases in less palatable perennial plant density were in the form of less palatable dwarf shrub succulents, mainly *Malephora lutea*, unpalatable dwarf succulents such as *Crassula expansa* and unpalatable dwarf woody shrubs mainly *Galenia pubescens (*Table 3.4). Increases in annual plant density consisted of *Chenopodium mucronatum* a palatable species. Perennial less-palatable dwarf shrub succulents such as *Malephora lutea* also showed increases in cover over the year; however, others showed declines such as *Leipoldtia schultzii. Chenopodium mucronatum* was the species that increased the most in vegetation cover, with increases of 1.2%.

Table 3.4 Overall study farm plant density and vegetation cover in 2009 (before rehabilitation) compared to 2010 (after rehabilitation) for different life forms.

	Plant density/25m^2			Vegetation cover (%)		
Life form	Mean 2009 (SE)	Mean 2010 (SE)	Diff.	Mean 2009 (SE)	Mean 2010 (SE)	Diff.
Annuals	8.24 (3.4)	20.06 (6.1)	11.82	0.11 (0.1)	1.57 (0.6)	1.46
Annual grasses	0.00	0.62 (0.3)	0.62	0.00	0.10 (0.1)	0.10
Dwarf shrub succulents	8.59 (2.6)	32.41 (10.4)	23.81	9.62 (3.9)	7.69 (3.1)	-1.93
Dwarf succulents	0.13 (0.1)	24.35 (8.6)	24.22	0.01 (0.01)	0.43 (0.2)	0.42
Dwarf woody shrubs	24.28 (3.8)	47.00 (9.4)	22.72	6.56 (1.8)	6.97(2.1)	0.41
Geophytes	0.00	0.12 (0.1)	0.12	0.00	0.01(0.01)	0.01
perennial forbs	0.04 (0.02)	2.63 (1.2)	2.59	0.01 (0.01)	0.06(0.04)	0.05
Perennial grasses	0.00	1.12 (0.4)	1.12	0.00	0.10 (0.1)	0.10
Succulent Shrubs	0.50 (0.2)	0.56 (0.2)	0.06	0.36 (0.2)	0.35 (0.2)	-0.01
Woody shrubs	0.89 (0.4)	6.32 (2.4)	5.44	2.48 (1.8)	2.82 (1.9)	0.34

3.4.2 Effect of soil, seed and mulch rehabilitation treatments on species richness, plant density, vegetation cover, grazing capacity and plant diversity

There were no significant interactions between treatments and therefore the null hypotheses of no interaction between rehabilitation treatments were accepted for all response variables. There were significant main effects of treatments on response variables and therefore the null hypotheses of no main effects of soil, seed and mulch treatments were rejected for certain response variables. This means that the combined effect of two or all three of the factors (soil, seed and mulch treatments) are predictable by adding their individual effects together i.e. there were no synergistic or antagonistic effects of two or more rehabilitation treatments. See Appendix 3.2 and 3.3 for species lists with palatability and life form groupings as well as mean changes (±SE) of plant density and vegetation cover per species per treatment.

Effects of rehabilitation treatments on species richness/25m^2 changes

Soil (*F=3.63, df=2, p=0.03*) and seed (*F=4.61, df=1, p=0.034*) treatments had significant effects on perennial palatable species richness, with significantly higher increases in species richness (3

species/25m^2) in the micro-catchment and seed treatments compared to control, rip and unseeded plots (2 species/25m^2) (Fig. 3.5). There were no effects of soil, seed or mulch treatments on the change in less palatable species richness (Fig. 3.6), overall perennial species richness (Fig. 3.7) and annual species richness (Figs 3.8). Trends for higher increases in species richness within micro-catchments were consistent for perennial plants. This was however not significant.
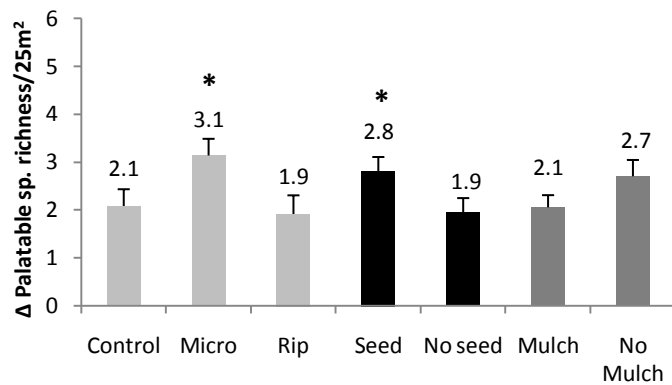

Figure 3.5 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial palatable species richness on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
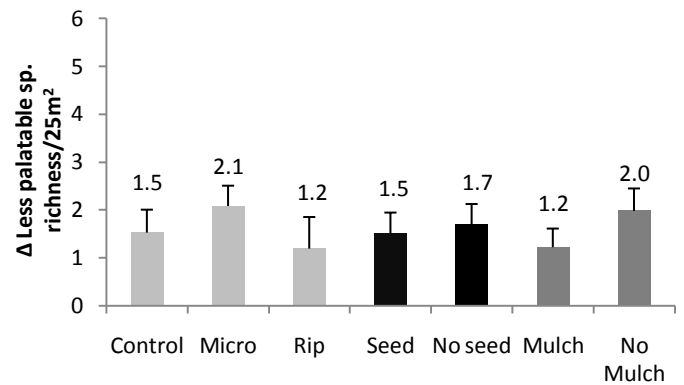

Figure 3.6 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial less palatable species richness on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), no significant differences were found.

Figure 3.7 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial species richness on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), no significant differences were found.

Figure 3.8 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in annual species richness on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), no significant differences were found.
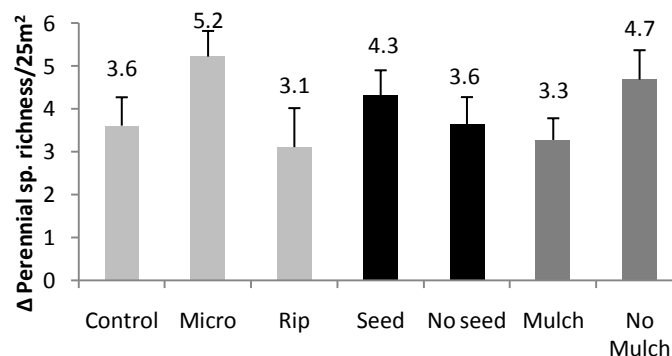
Effects of rehabilitation treatments on plant density/25m^2 changes

Soil treatments (*F=4.85, df=2, p=0.01*) had a significant effect on the change in palatable perennial plant density, with significantly higher increases within micro-catchment treatments (16 plants/m^2) than control and rip plots (8 plants/25m^2) (Fig 3.9). Mulch treatments had a significant negative effect on the change in less palatable plant density (*F=57.32, df=1, p<0.001*) (Fig. 3.10), total

perennial (*F=51.99, df=1, p<0.001*) (Fig. 3.11) and annual plant density (*F=6.74, df=1, p=0.01*) (Fig. 3.12). Mulch had no significant effect on palatable plant density with similar change occurring in mulched (10 plants/25m^2) and unmulched plots (12 plants/25m^2).
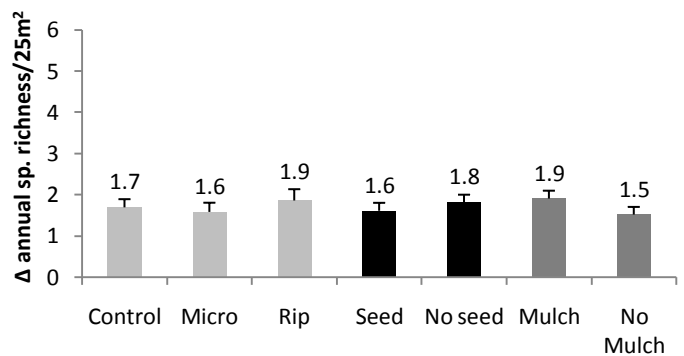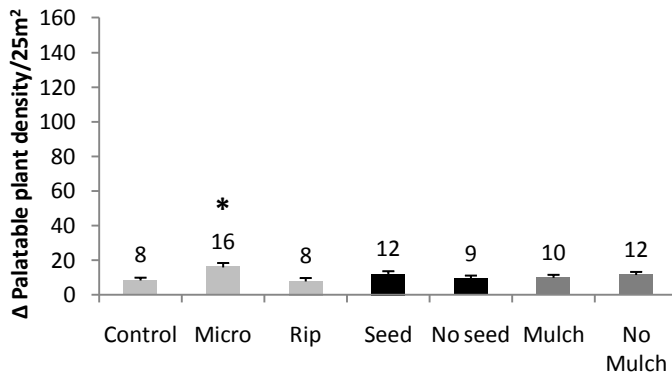

Figure 3.9 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial palatable plant density on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
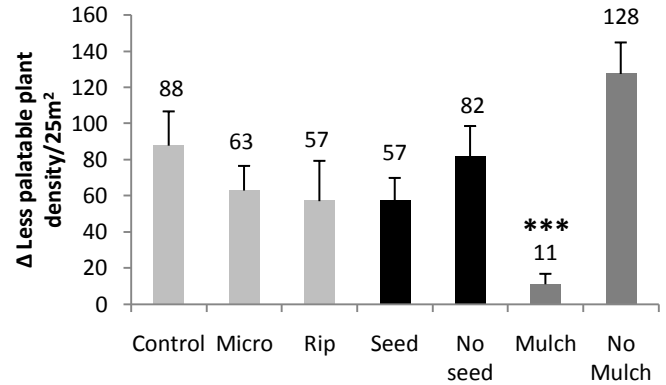
Figure 3.10 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial less palatable plant density on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
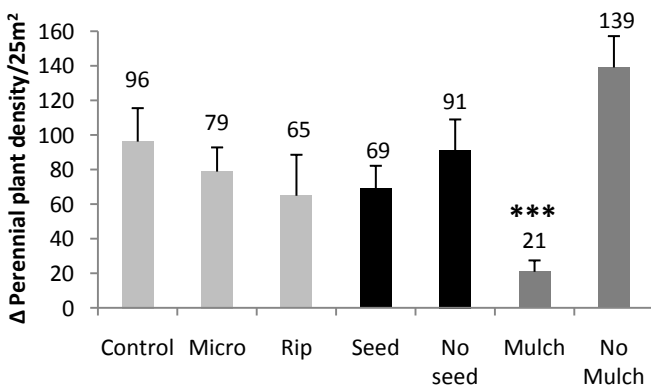
Figure 3.11 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in perennial plant density on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
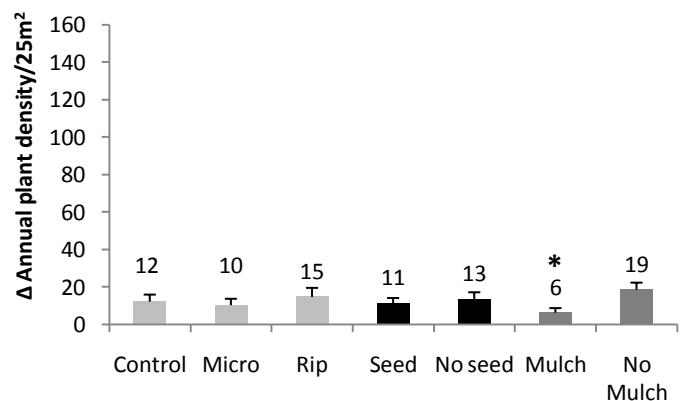
Figure 3.12 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on the change (Δ) in annual plant density on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

There was a statistically highly significant association between changes in plant life forms with soil (χ^2 =19.7, df=10, p<0.05) and mulch treatments (χ^2=49.97, df=10, p<0.01). There were however, no significant associations between plant life forms and seed treatments (χ^2=7.46, df=10, p>0.05). Palatable woody shrubs and dwarf woody shrubs like *Tripteris sinuata* and *Tetragonia fruticosa* showed the highest increases within mulched treatments. Dwarf shrub succulents, like the less-palatable *Malephora lutea*, unpalatable dwarf succulents, like *Crassula expansa* and unpalatable dwarf woody shrubs, e.g. *Galenia pubescens* showed the highest decreases (Table 3.5). Mulching also decreased palatable dwarf shrub succulents such as *Mesembryanthemum noctiflorum* and

perennial forbs like *Chaenostoma subnudum.* Mulch decreased annuals e.g. palatable *Chenopodium mucronatum* and unpalatable *Mesembryanthemum crystalinum.* The only annual species, which were not suppressed by mulch treatments, were the unpalatable *Oligocarpus calendulaceus* and less-palatable *Chenopodium murali*. Increases of palatable plant density in micro-catchment treatments were mainly highly palatable woody shrubs, palatable perennial forbs and palatable perennial grasses like *Tripteris sinuata*, *Chaenostoma subnudum* and *Fingerhuthia africana* respectively (Table 3.5).

Table 3.5 The effect of soil (micro-catchments, ripping, n=36), seed (n=54) and mulch (n=54) treatments on changes of plant density/m^2 grouped together for different life forms on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed as means (±SE).

Life form	Control	Micro	Ripping	Seed	No Seed	Mulch	No Mulch
Annuals	12.1(8.1)	9.2(6.2)	14.2(8.3)	10.8(6.1)	12.9(6.9)	5.5(5.3)	18.2(6.8)
Annual grasses	0.2(0.1)	1.2(0.6)	0.5(0.2)	0.6(0.3)	0.6(0.3)	0.8(0.3)	0.4(0.2)
Dwarf shrub succulents	26.2(12)	26.5(12.1)	18.7(9.3)	22.9(9.6)	24.8(9.8)	8.9(4.6)	38.7(12.7)
Dwarf succulents	37.2(12.9)	15.6(4.9)	19.9(12.3)	16.3(5.5)	32.2(11.1)	5.5(1.7)	42.9(11.8)
Dwarf woody shrubs	25.2(12.1)	22.3(10.2)	20.7(11)	18.8(9.1)	26.6(9.4)	-3.6(5.2)	49(10.4)
Geophytes	0.1(0.1)	0.1(0.1)	0.1(0.1)	0.1(0.01)	0.2(0.2)	0.2(0.1)	0.1(0.1)
Perennial forbs	2.4(1.5)	3.6(1.6)	1.8(0.9)	1.7(0.7)	3.5(1.4)	1.9(1)	3.3(1.1)
Perennial grasses	0.5(0.2)	1.7(0.7)	1.1(0.3)	1.9(0.4)	0.3(0.2)	1.0(0.4)	1.3(0.3)
Succulent Shrubs	0.2(0.2)	0.3(0.4)	-0.4(0.2)	0.2(0.3)	-0.1(0.2)	0.1(0.2)	0.0(0.2)
Woody shrubs	4.4(2.1)	8.9(3.7)	3.0(1.6)	7.2(2.3)	3.7(2)	7.1(2.7)	3.8(1.5)

Considering species sown at the study site, micro-catchments had significantly higher density of *Tripteris sinuatum* (*K=8.77, df=2, p=0.012*) and *Ruschia spinosa* (*K=11.86, df=2, p=0.003*) (Table 3.6). There were however no significant effects of soil treatments on *Tetragonia fruticosa, Lessertia annularis, Fingerhuthia africana* and *Drosanthemum hispidum*. Seed treatments significantly increased the density of *Tripteris sinuatum* (*U=755, p<0.001*) and *Fingerhuthia africana* (*U=649, p<0.001*) however showed no significant effects on the other four seedling species. Mulching had no significant effects on sown species. However, non-significant increases of *Tetragonia fruticosa* and *Tripteris sinuata* were observed.

Table 3.6 The effect of soil (micro-catchments, ripping, n=36), seed (n=54) and mulch (n=54) treatments on changes in sown seedling species density/25m^2 on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed as means (±SE). Significant differences are indicated with an asterisk *

Species	Control	Micro	Ripping	Seed	No Seed	Mulch	No Mulch
Tripteris sinuata	1.58(0.5)	*5.25(1.6)	2.42(0.8)	*4.89(1.1)	1.28(0.6)	4.31(1.2)	1.85(0.4)
Tetragonia fruticosa	1.83(0.6)	2.50(0.9)	0.33(0.2)	1.6(0.4)	1.51(0.6)	2.12(0.7)	0.99(0.3)
Lessertia annularis	0.00(0.1)	0.14(0.1)	0.08(0.1)	0.16(0.1)	-0.01(0.01)	0.08(0.06)	0.06(0.05)
Fingerhuthia africana	0.50(0.2)	1.67(0.6)	1.08(0.3)	*1.89(0.3)	0.28(0.2)	0.91(0.4)	1.26(0.3)
Ruschia spinosa	-0.61(0.3)	*0.89(0.4)	-0.11(0.2)	0.15(0.3)	-0.04(0.3)	-0.19(0.2)	0.30(0.3)
Drosanthemum hispidum	0.44(0.3)	0.03(0.03)	0.17(0.1)	0.09(0.05)	0.33(0.2)	0.06(0.04)	0.37(0.2)

Effects of rehabilitation treatments on vegetation cover (%) changes

Rehabilitation treatments had no significant effect on palatable perennial vegetation cover. Palatable plants increased *c.* 1% across treated and untreated plots (Fig. 3.13). Mulching significantly decreased less palatable vegetation cover ($F=13.32$, $df=1$, $p<0.001$) (Fig. 3.14) and total perennial vegetation cover ($F=13.35$, $df=1$, $p<0.001$) (Figs 3.15). Similarly, mulch had significant negative effects on annual plants, however difference in cover were only 1% between mulched and unmulched plots ($F=6.36$, $df=1$, $p=0.013$) (Fig. 3.16).

Figure 3.13 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in perennial palatable vegetation cover on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), no significant differences found.

Figure 3.14 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in perennial less palatable vegetation cover on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

Figure 3.15 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in perennial vegetation cover on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

Figure 3.16 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in annual vegetation cover on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

There were no significant associations between perennial plant life forms and soil, seed or mulch treatments. However, trends showed that mulching caused declines in the cover of dwarf shrub succulents like less palatable *Malephora lutea* and *Leipoldtia schultzii*, unpalatable dwarf woody shrubs like *Galenia pubescens* and annuals like palatable *Chenopodium mucronatum*. Most of the

plots showed decreases in dwarf shrub succulents these were mainly due to declines in *Leipoldtia schultzii* across treatments (Table 3.7).

Table 3.7 The effect of soil (micro-catchments, ripping, n=36), seed (n=54) and mulch (n=54) treatments on changes of perennial vegetation cover grouped together for different life forms on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed as means (±SE).

Total	Control	Micro	Ripping	Seed	No Seed	Mulch	No Mulch
Annuals	1.01(0.5)	1.13(0.9)	2.25(0.44)	1.41 (0.6)	1.51(0.6)	0.78(0.2)	2.14(0.3)
Annual grasses	0.13(0.1)	0.14(0.02)	0.03(0.06)	0.10(0.1)	0.11(0.1)	0.09(0.01)	0.11(0.04)
Dwarf shrub succulents	-2.66(4.1)	-3.07(1.98)	-0.05(3.16)	-2.87(2.8)	-0.98(2.8)	-4.06(0.3)	0.20(3.1)
Dwarf succulents	0.51(0.2)	0.49(0.2)	0.26(0.13)	0.35(0.2)	0.50(0.2)	0.10(0.1)	0.75(0.1)
Dwarf woody shrubs	1.83(2.8)	2.17(2.1)	-2.76(1.9)	1.48(2.1)	-0.66(1.9)	-1.68(0.1)	2.50(1.9)
Geophytes	0.00(0.01)	0.01(0.02)	0.02(0.00)	0.00(0.02)	0.02(0.00)	0.02(0.00)	0.00(0.02)
Perennial forbs	0.01(0.1)	0.13(0.03)	0.00(0.02)	0.00(0.07)	0.10(0.07)	-0.01(0.01)	0.11(0.01)
Perennial grasses	0.04(0.05)	0.08(0.15)	0.18(0.1)	0.18(0.01)	0.01(0.04)	0.13(0.1)	0.07(0.1)
Succulent Shrubs	0.03(0.2)	-0.12(0.26)	0.06(0.2)	-0.20(0.14)	0.18(0.2)	-0.16(0.01)	0.15(0.2)
Woody shrubs	0.68(1.4)	0.52(1.45)	-0.19(1.5)	0.08(0.9)	0.60(1.27)	0.55(0.05)	0.12(1.2)

Soil and mulch treatments did not significantly affect the vegetation cover of seedling species sown at the site. Seed treatments, however significantly increased the change in vegetation cover of *Tripteris sinuata* (*U=1201, p<0.03*) and *Fingerhuthia africana* (*U=1295, p=0.028*) (Table 3.8).

Table 3.8 The effect of soil (micro-catchments, ripping, n=36), seed (n=54) and mulch (n=54) treatments on changes in sown seedling species vegetation cover on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed as means (±SE). Significant differences are indicated with an asterisk *

Species	Control	Micro	Ripping	Seed	No Seed	Mulch	No Mulch
Tripteris sinuata	0.31(0.1)	0.23(0.1)	0.49(0.18)	*0.46 (0.1)	0.23 (0.1)	0.40 (0.1)	0.29 (0.1)
Tetragonia fruticosa	0.14(0.2)	0.73(0.4)	0.33(0.2)	0.56 (0.2)	0.25 (0.3)	0.33 (0.3)	0.47 (0.2)
Lessertia annularis	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fingerhuthia africana	0.04(0.03)	0.08(0.05)	0.18(0.15)	*0.18 (0.1)	0.01 (0.01)	0.13 (0.1)	0.07 (0.03)
Ruschia spinosa	-0.05(0.16)	-0.32 (0.06)	0.05(0.1)	-0.22 (0.1)	0.01 (0.1)	-0.11 (0.1)	-0.10 (0.1)
Drosanthemum hispidum	0.14(0.1)	-0.07(0.07)	0.00	0.01 (0.01)	0.04 (0.1)	-0.04 (0.5)	0.08 (0.9)

Effects of rehabilitation treatments on grazing capacity (LSU/1000ha) changes

Mulch had a significant negative effect on perennial and total grazing capacity (*F=10.49, df=1, p=0.002*; F=*10.44, df=1, p=0.002*). Mulching reduced perennial and total grazing capacity by 1LSU/1000ha compared to increases 2 and 3LSU/1000ha without mulching. Soil treatments and seed treatments had no significant effect on grazing capacities (Fig 3.17 & 3.18).

Effects of rehabilitation treatments on plant diversity (Shannon diversity index (H)) changes

Mulch had significantly higher increases in perennial plant diversity and total plant diversity (*F=6.74, df=1, p=0.01*; *F=11.92, df=1, p=0.001*). Trends showed higher increases within micro-catchment and rip treatments compared to control plots. These changes however were not significant (Fig. 3.19 & 3.20).
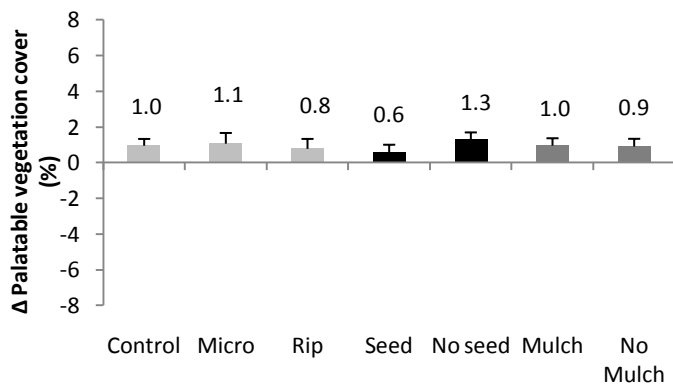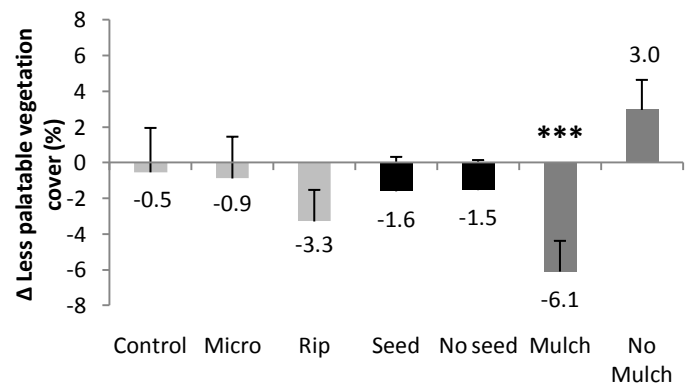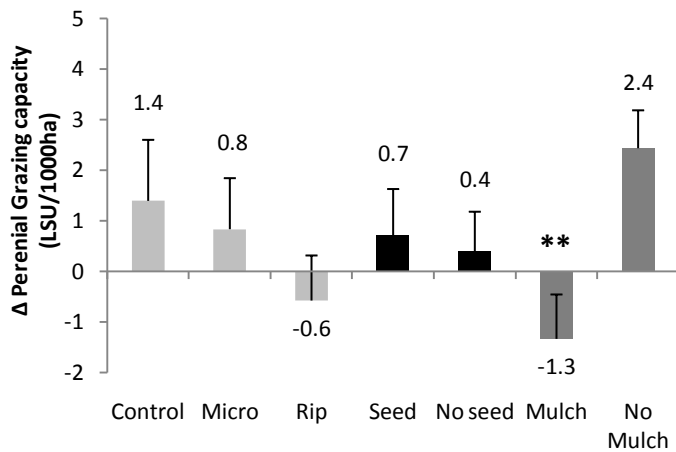
Figure 3.17 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in perennial grazing capacity (LSU/1000ha) on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

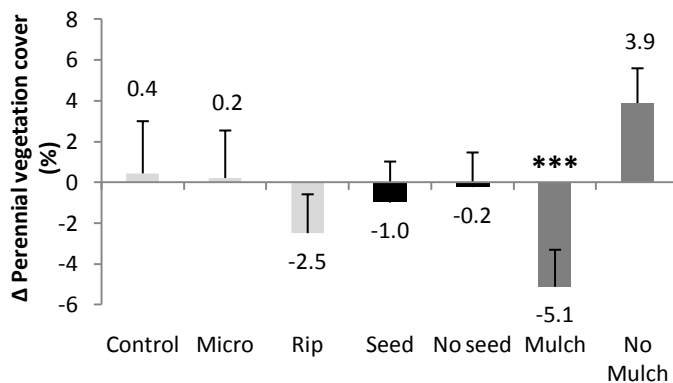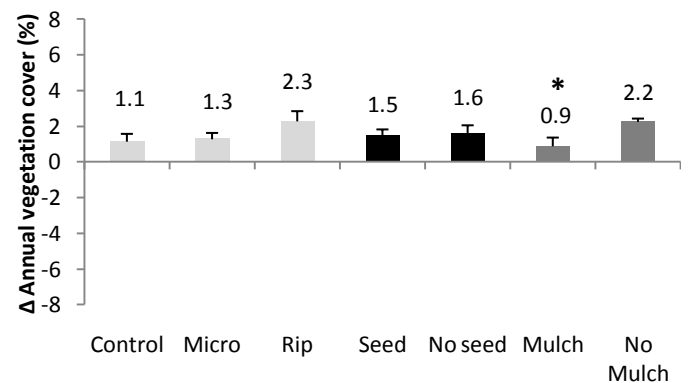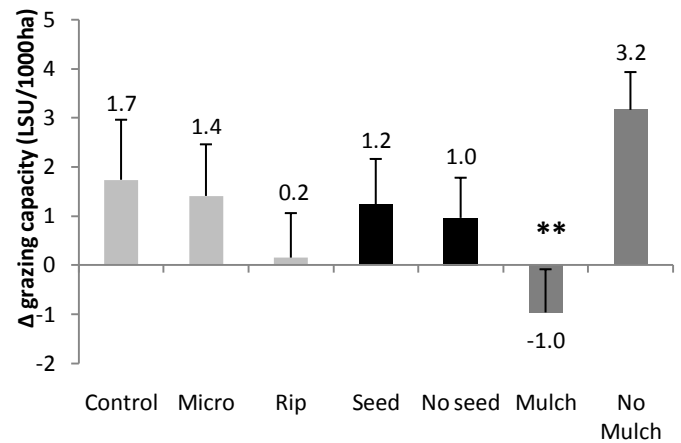**Figure 3.18** The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in total grazing capacity (LSU/1000ha), including annual and perennial plants on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
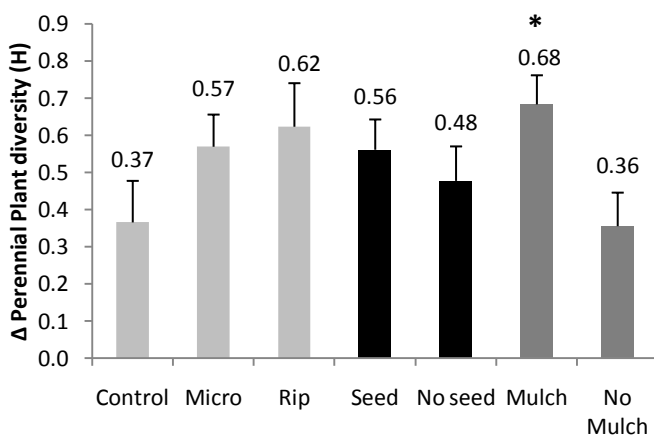
Figure 3.19 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in perennial Shannon plant diversity (H) on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).
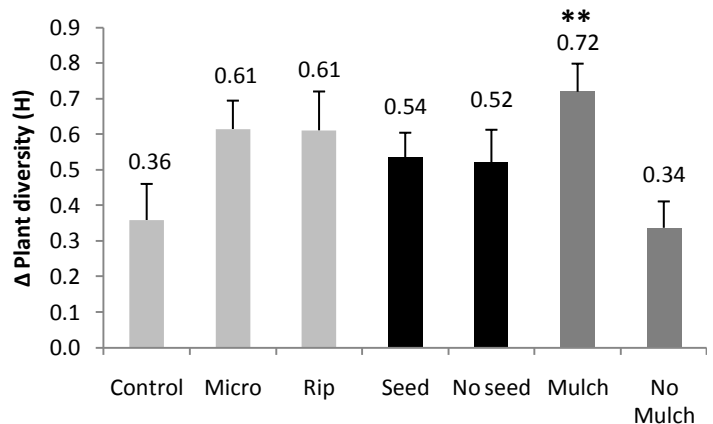
Figure 3.20 The effect of soil (grey, n=36), seed (black, n=54) and mulch (dark grey, n=54) treatments on changes (Δ) in total Shannon plant diversity (H), including annual and perennial plants on Morestêr farm – a degraded ostrich farm in the Little Karoo. Changes are expressed in means (±SE), (*p<0.05; **p<0.01; ***p<0.001).

3.4.3 Rehabilitation Treatment costs/m²

Ripping was the least expensive treatment applied at the study site (W. Stroebel. pers. comm. 2011), followed by sowing seed, micro-catchments (with mulch in the base) and then mulching (Table 3.9). The cost of seed amounted to R600.00. Labour for broadcasting seeds for the day totalled R520.00 (S. Milton *in litt.* 2009). Ripping costs include the use of a 60-kilowatt tractor with a three-tine ripper and labour. Ripping labour costs cover two personnel, one driving the tractor and the other on foot to

remove large rocks. Micro-catchment costs include labour and mulch (W. Stroebel per. Comm. 2011).

Table 3.9 Rehabilitation costs for Morestêr farm (W. Stroebel pers. comm. 2011 and S. Milton *in litt.* 2009**)**

Rehabilitation treatment	Cost/m^2	Cost/25m^2
Ripping	R 0.13	R 3.25
Seed	R 0.37	R 9.22
Micro-catchments with mulch	R 1.54	R 38.50
Mulch	R 2.47	R 61.75

3.4.4 Rainfall and temperature during the study period

Total rainfall for 2009 (259 mm) was slightly higher than the long-term average (242 mm) and considerably higher than the rainfall in 2008 (152 mm) (Fig. 3.21). The year before rehabilitation (2008) was a considerably dry year with rainfall occurring well below the long-term average. The two years prior to 2008 were relatively wet years with rainfall occurring above the long-term average particularly within 2007 (325 mm) (Fig. 3.21). The distribution of rainfall throughout the year after rehabilitation differed from the long-term average distribution with higher amounts of rain falling within February, April, June and December and lower rainfall within January, March, May, August and September. Rainfall in July, October and November were similar to the long-term distribution (Fig 3.22).
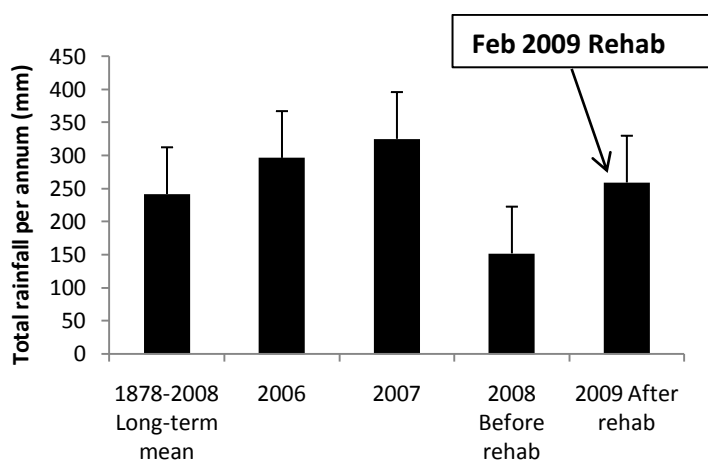
Figure 3.21 Total rainfall in 2009 after rehabilitation compared with rainfall three years prior to rehabilitation (2006-2008) and the long-term average (±SD) for the town of Oudtshoorn in the Little Karoo (South African Weather Service 2010)
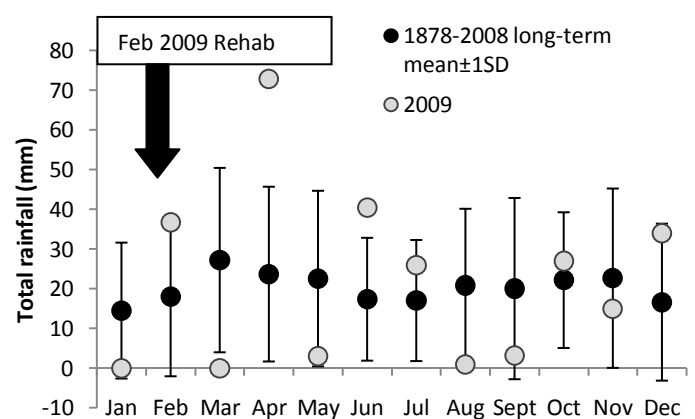
Figure 3.22 Rainfall distribution in 2009 compared to the long-term average for the town of Oudtshoorn in the Little Karoo (South African Weather Service 2010)

Minimum and maximum temperatures for the study period were similar to the long-term minimum and maximum mean temperatures (Fig 3.23).
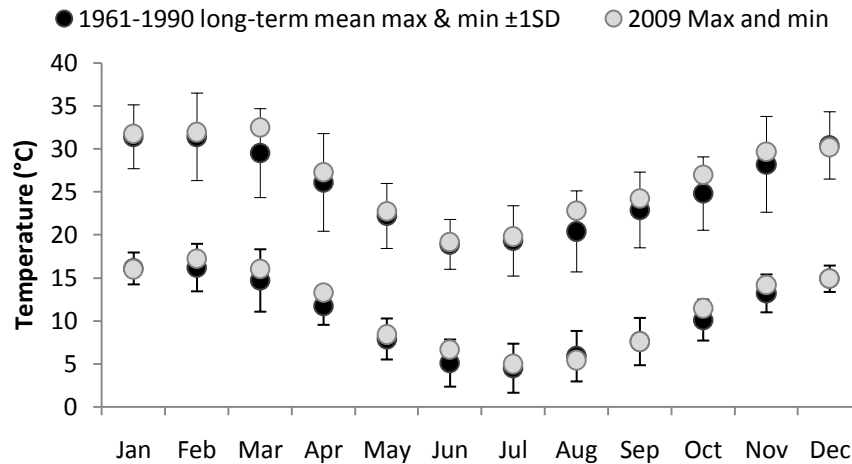
Figure 3.23 Minimum and maximum temperature in 2009 after rehabilitation compared to the long-term average(±SD) for the town of Oudtshoorn in the Little Karoo (South African Weather service 2010)

3.5 Discussion

A key goal of rangeland rehabilitation is vegetation recovery, particularly palatable plants and an increase in grazing capacity (Seymour *et al.* 2010). At Morestêr farm, micro-catchments increased palatable species richness and density, sowing seed increased palatable species richness and mulching increased plant diversity. Mulching reduced less- and unpalatable plant density and vegetation cover, resulting in lower grazing capacity. In the long-term, forage species density and cover could increase because of the reduction of competing less palatable plants. This was however not evident one-year after mulch application. Ripping treatments showed no benefits for rangeland restoration over one-year even with good rainfall. The unsuccessfulness of the ripping treatments could merely be a result of the short-term nature of the research project and benefits could only become noticeable in the long-term. Active rehabilitation studies in South African, US, African and Indian rangelands have shown similar patterns. These studies emphasize the necessity of soil treatments and the desirability of seeding palatable species for increases in plant density and species richness (Snyman 2003; Visser *et al.* 2004; Wentz 2004; Van den Berg & Kellner 2005; Sheley *et al.* 2006; Bainbridge 2007; Kinyua *et al.* 2009; Vohland & Barry 2009).

3.5.1 Overall study farm changes

Before rehabilitation (January 2009), the study farm had low perennial palatable (2 plants/25 m^2) and less palatable plant (30 plants/25 m^2) densities, species richness (6 species/$25m^2$) and vegetation cover (<20%). Grazing capacity (0.48 LSU/60ha) was below the recommended stocking rate (1LSU/60ha) for the area (Cupido 2005). One year after rehabilitation (January 2010), species richness increased almost two-fold (10 species/$25m^2$) and plant density increased three-fold (114

plants/25m^2) for both palatable (12 plants/m^2) and less palatable plants (100 plants/25m^2) across treated and untreated sites. Vegetation cover remained largely unchanged.

The availability of soil water in the cool season determines the fate of germination events in the Succulent Karoo (Esler 1993; Milton 1995a, 1995b). Increases in plant density and species richness could be attributed to the change in rainfall conditions before and after rehabilitation. It is assumed that in 2007, with the above average rainfall and the removal of ostrich disturbance, existing adult plants were able to flower, set seed and therefore contribute to the local seed bank as well as increase in biomass. Before rehabilitation in 2008, rainfall was well below the long-term average inhibiting seed germination and negatively effecting adult plant growth. After rehabilitation in 2009, rainfall was similar to the long-term average. Naturally occurring seeds within the seed bank along with the sown seedling species germinated and established as young plants. These increases in naturally occurring plant density were due to the recent years' seed production and not a long-dormant seed bank (Esler 1993; Milton 1995b). Density measurements were able to pick up these fine-scale changes. Young plants and seedlings however were not large enough to contribute to an increase in vegetation cover at the site. Eleven springbok (*Antidorcas marsupialis*) grazed on the rehabilitated site for the duration of 2009. This is an additional reason for the incongruity between increases in plant density and no changes in vegetation cover. Herbivores continuously remove new growth preventing plants from increasing in cover (Esler *et al.* 2006; Milton 1992b; Milton 1994).

3.5.2 Micro-catchments and ripping soil treatments

The short-term benefits of micro-catchments at Morestêr farm are an increase in forage species richness and density, however with no subsequent increases in vegetation cover and grazing capacities. The short-term (1-3 years) and long-term (5-10 years) effects of micro-catchment and ripping cultivation techniques on plant density and species richness have been reviewed for several semi-arid and arid areas in South Africa (Snyman 1999; Van der Merwe & Kellner 1999; Snyman 2003; Visser *et al.* 2004; Van den Berg & Kellner 2005; Simons & Allsopp 2007; Schmiedel *et al.* 2010). Most studies link increases in plant density and richness to increases in forage production and grazing capacity (Snyman 1999; Van der Merwe & Kellner 1999; Snyman 2003; Seymour *et al.* 2010). However, few studies measure the rehabilitation effects on vegetation cover (Schmiedel *et al.* 2010) and or translate this into potential grazing capacity (ha/LSU).

At Morestêr, increases of palatable plant density within micro-catchments were mainly due to increases of seedlings or young plants, contributing little to overall cover. Van der Merwe and Kellner (1999) reviewed 25 case studies in semi-arid (451-600 mm) and arid (300-450 mm) areas in

South Africa and found that the high plant density (100 plants/m^2) recorded in micro-catchment treatments even after 5 to 10 years, were mainly reflected by a higher presence of seedlings within micro-catchments versus ripped areas (49 plants/m^2). In addition, these increases were mainly due to the high proportion of annual (annual grasses) and perennial pioneer species (Van der Merwe & Kellner 1999; Van den Berg & Kellner 2005). Similarly, in Kenyan rangelands after 6 months ripping alone tripled plant density however mostly herbaceous forbs and small annual grasses. Even after ten years, increases were principally by annual plants, which have limited value for grazing in the long-term (Kinyua *et al.* 2009). Rangeland rehabilitation studies in the Succulent Karoo have found that brushpacking (*Brownanthus sp.; Salsola sp.*; *Galenia africana*), dung mulching, sowing seed and micro-catchments over a three-year period has little effect on perennial vegetation cover (Simons & Allsopp 2007; Schmiedel *et al.* 2010). Directly planting adult plants with functional roles (e.g. seed, soil and water traps) has however been found to be successful (Anderson *et al.* 2004; Schmiedel *et al.* 2010).

Successful palatable plant species within micro-catchments on Morestêr farm included both species that were sown at the site (*Tripteris sinuata, Tetragonia fruticosa, Fingerhuthia africana*), as well as naturally occurring perennial plant species like *Chaenostoma subnudum*, *Mesembryanthemum noctiflorum* and *Euphorbia burmanii*. Micro-catchments also showed trends for increased density of the less palatable pioneer species, *Malephora lutea* (19 plants/25m^2). Succulent plants were mainly observed growing on the micro-catchment walls whereas the woodier seedlings like *Tripteris sinuata* grew within the mulched base. Seedlings have been recorded to germinate on micro-catchment walls quite effectively, especially during wet years due to water logging (Bainbridge 2007). At Morestêr farm, small seeded succulents were probably able to avoid water logging this way, whereas large seeded woodier species benefitted from the increases of water harvested as runoff within the base of the micro-catchment (Snyman 2003; Bainbridge 2007, Vohland & Barry 2009).

Micro-catchments have been successfully applied for soil and water conservation throughout Africa and India (Vohland & Barry 2009). Micro-catchments have been shown to improve water infiltration, increase moisture content and reduce evaporation, facilitating plant establishment and growth (Wight & White 1974; Stern *et al.* 1992; Van der Merwe & Kellner 1999; Wentz 2004; Bainbridge 2007). In contrast, a study in the Succulent Karoo found that micro-catchments did not improve natural recruitment of perennial seedlings and resulted in only minor increases in the cover of annuals (Simons & Allsopp 2007). This was initially attributed to the removal of fertile soils when digging micro-catchments and water logging (Simons & Allsopp 2007). However, more recently it has been linked to low rainfall intensities (5.2 mm) experienced in Succulent Karoo winter rainfall

areas (Hanke *et al.* 2011). Rainfall events <5.2 mm are apparently too low for the development of runoff into micro-catchments and result in similar soil water moisture and storage within the micro-catchments compared to bare areas (Hanke *et al.* 2011). Although, rainfall intensities were too low to generate runoff, Hanke *et al.* (2011) found that infiltration rates were higher within micro-catchments. Therefore, if there were higher intensity rainfalls, runoff would accumulate in micro-catchments, infiltrating into the soil more efficiently (Hanke *et al.* 2011). At Morestêr farm, after rehabilitation, during the 2009 growing season, there were 29 days where rainfall was less than 5.2 mm, however, there were ten days where rainfall was greater than 5.2 mm, reaching greater than 10 mm and 20 mm on certain days. Micro-catchments were observed to harvest rainwater effectively from these heavier rainfall events (Snyman 2003; Van der Merwe & Kellner 2005). The soils on Morestêr farm develop impenetrable mineral crusts, which result in a high degree of compaction and decreased infiltration rates (Mills & Fey 2003; Mills & Fey 2004; Le Maitre *et al.* 2007). Although soil moisture, storage and infiltration rates were not measured, it is likely that rainwater ran off bare crusted soil surfaces and accumulated in micro-catchments (Stern *et al.* 1992; Snyman 1999; Van der Merwe & Kellner 1999; Snyman 2003; Le Maitre *et al.* 2007; Vohland & Barry 2009). This probably promoted seedling germination from the natural seedbank as well as from sown seeds and increased perennial palatable plant density within micro-catchments (Esler 1993; Visser *et al.* 2004; Vohland & Barry 2009).   Sown seedling species showing highest recruitment within micro-catchments included the woody shrub *Tripteris sinuata* and the dwarf shrub succulent *Ruschia spinosa*.

Micro-catchments have been found to be more effective for clayey soils (between 1% and 20% clay) and ripping more suitable for sandy soils (<20% clay) (Van der Merwe & Kellner 1999; Bainbridge 2007). This is because the effectiveness of micro-catchments decline as they fill in with blown or washed soils and debris (Bainbridge 2007). The soils at Morestêr farm have low clay content (<20%) and are classified as sandy with less than 20% clay (BemLab, Somerset West 2011) or sandy loam (Vlok *et al.* 2005). Within one year, ripping was not as effective as Micro-catchments at increasing palatable plant density or species richness. Field observations suggest that micro-catchments at post one-year on Morestêr farm were not completely filled with sediments. In addition, sediment and litter that washed into the micro-catchments probably increased soil fertility and aided plant establishment (Bainbridge 2007). Micro-catchments also protected seedlings from wind and sand blast (Bainbridge 2007). At Morestêr farm, it is possible that ripping was too shallow or too closely spaced and therefore filled quickly with sediment as opposed to the large micro-catchments. Ripped areas were therefore unable to protect seedlings or to be effective at catching water run-off (Van der Merwe & Kellner 1999; Snyman 2003). Depth of and distance between rips and micro-catchments are very

important because these factors influence the results of restoration to a considerable extent (Van der Merwe & Kellner 1999). However, there are still many views on these parameters (Snyman 1999; Tainton *et al.* 2000). Snyman (2003) recommended a ripping depth of 400 mm. Van der Merwe & Kellner (1999) recommended that ripping width should be at least 2 m. Some studies have shown ripping to have limited effects on plant establishment and sometimes even decreased emergence. It is believed that ripping can bury or destroy germinated pioneer species, rather than promoting emergence (Montalvo *et al.* 2002).

In the short-term, there were no effects of ripping or micro-catchments on total annual or less palatable plant density, richness or cover. An increase however in long-lived forage species richness and density in micro-catchments could be viewed as a success over an increase in total perennial pioneer and annual species.

3.5.3 Mulching
Mulching decreased less palatable plant density (by 90 plants/25 m^2) and vegetation cover (by 9%). Less- and unpalatable plant species have objective grazing indices which contribute to the calculated grazing capacity of an area (Du Toit 1995, Du Toit *et al.* 1995; Du Toit 2000; Du Toit 2002; Esler *et al.* 2006). Mulching had the greatest effect on less- and unpalatable high-density species like the dwarf shrub succulent *Malephora lutea*, dwarf woody shrub *Galenia pubescens,* the dwarf succulent *Crassula expansa.* Annuals like *Chenopodium mucronatum* were also largely affected. This decreased grazing capacity however increased the plant diversity. The Shannon diversity index takes into account how evenly the total number of individuals in a plot is apportioned between each species (known as 'equitability') (Magurran 1991). The index is increased either by having additional unique species, or by having greater species evenness. Mulched treatments had similar species richness to unmulched areas. However, the large decreases in high-density less- and unpalatable plants resulted in greater species evenness and therefore higher plant diversity (Magurran 1991).

Plants in degraded semi-arid environments are likely less palatable, long-lived, hardy perennials and/or pioneer plants whose abundance may be maintained by prolific seed production (Milton & Dean 1994; Wiegand *et al.* 1995; Wiegand & Milton 1996; Saayman *et al.* 2009). Recruitment and establishment of palatable shrubs is suppressed in the presence of competing unpalatable plants (Milton 1994; Todd 2000; Anderson & Hoffman 2007), ultimately lowering the production potential of rangelands (Dean & Macdonald 1994; Milton 1994, 1995b). Although mulching resulted in short-term grazing capacity declines, it has also potentially decreased competition from less palatable

plants. Certain palatable long-lived species like woody and dwarf woody shrubs e.g. *Tripteris sinuata* and *Tetragonia fruticosa* were able to increase in density within mulch treatments. These significant decreases in high-density plants altered the ratio of palatable to less palatable plants from 1:12 to 1:3 within mulched areas. This increased species evenness and could increase future grazing capacity due to reduced interspecific competition.

Mulch and brushpacks have been successfully applied in semi-arid environments throughout the world (Ludwig & Tongway 1996). Soil moisture, temperature, and nutrient availability have all been shown to be positively affected by mulch. Organic mulches capture sediment, litter, and seeds (Ludwig *et al.* 1994), absorb raindrop impact, reduce evaporation, insulate the soil and re-create fertile patches (Ludwig & Tongway 1996; Sharma *et al.* 1998; Beukes & Cowling 2003; Athy *et al.* 2006). Brushpacking and mulching has been shown to cause marked responses in plant development including increased growth (Haywood 1999), seedling survival (Tomlinson *et al.* 1997; Beukes & Cowling 2003; Visser *et al.* 2004) and increased crop yields (Moitra *et al.* 1996; Lal 2000). Mulching however, has also been found to have negative effects on plant density, vegetation cover and no consistent effect on soil moisture conditions (Athy *et al.* 2006; Simons & Allsopp 2007; Schmiedel *et al.* 2010).

Brushpacking (*Galenia africana*) at Paulshoek in the Succulent Karoo did not have any effect on perennial plant species richness or abundance (Simons & Allsopp 2007). Brushpacks (*Salsola sp.*) at Soebatsfontein in the Succulent Karoo actually reduced perennial species richness and cover (Schmiedel *et al.* 2010). It was assumed that adult plants and seedlings were damaged during the treatment installation, through spreading brushpacks (Schmiedel *et al.* 2010). The mulching applied at Morestêr consisted of wood chip, which was applied directly on to the ground. Large woody and succulent adult shrubs were probably not physically effected by mulching as care was taken to apply mulch around the base of these plants. It is however possible that smaller seedlings and especially dwarf succulents and dwarf succulent shrubs were physically disturbed during application procedures. Wood chip mulches, have been shown to reduce the emergence of small-seeded species (Zak *et al.* 1972; Petersen *et al.* 2004b) as well as suppress fine herbaceous biomass growth (Athy *et al.* 2006). It is possible that robust and large seeded plants like *Tripteris sinuata* were able to germinate and grow through the mulch barrier at Morestêr Farm. Small-seeded succulent species like *Malephora lutea* or less robust forbs did not have enough bulk to penetrate the above layer of mulch (Athy *et al.* 2006).

In the Ceres Succulent Karoo, Beukes and Cowling (2003) found mulched areas (Restionaceae thatching reed) to have significantly higher naturally seeded annual and perennial plant volumes. This was believed to be mainly due to improved infiltration rates (Beukes & Cowling 2003; Schmiedel *et al.* 2010). At Morestêr farm it is possible that if sparser layers of mulch were used it would have aided rather than reduced naturally seeded annuals and perennials. Studies have found that thick mulches especially in dry areas can prevent limited precipitation from reaching plant roots (Petersen *et al.* 2004b). In contrast, Athy *et al.* (2006) found mulch to have no consistent effect on soil moisture as high rainfall caused the proliferation of a fungus layer in the mulch. The impermeable fungus layer sealed moisture within the mulch and thereby desiccated the soil below (Athy *et al.* 2006). A fungus layer of this type has been observed in mulches of >150 mm. The layer of mulch at Morestêr measured at ±100 mm. The fact that certain species were able to germinate and grow through the mulch suggests that a proliferation of a fungus layer at Morestêr is unlikely.

Nitrogen is considered the most limiting nutrient for plant growth in semi-arid ecosystems (Mazzarino & Bertiller 1999; Snyman & du Preez 2005). Carbon amendments such as wood-chip mulch have been shown to reduce plant available nitrogen. The carbohydrates in the wood-chip stimulate the growth of microbes, which utilise the available nitrogen in the soil (Wilson & Gerry 1995; Blumenthal *et al.* 2003; Corbin & D' Antonio 2004). This has been shown to reduce certain plant species biomass and community dominance (Corbin & D' Antonio 2004; Wilson *et al.* 2004). At Morestêr farm, it is possible that woodchip mulch reduced the available nitrogen in the soil, which temporarily stunted plant growth and possibly caused death from malnutrition (Corbin & D' Antonio 2004; Wilson *et al.* 2004). Bulmer (2000) found that mulch decreased growth over three years. Other authors have found that the negative effect of decreased nitrogen levels because of mulching lasted for 2 years (Corbin & D' Antonio 2004; Wilson *et al.* 2004).

There are a number of plausible reasons for plant decreases within mulched areas. Mulch presented a germination barrier to herbaceous and small-seeded plants and could have physically damaged dwarf vegetation present during application. Palatable woodier species however, managed to germinate through the barrier. It is also possible that nitrogen levels were reduced which affected plant growth of annual and pioneer perennial plants. This could aid the hardier long-lived plant species, which managed to germinate through the mulch barrier. Decreased available nitrogen has been correlated with the replacement of early successional species by mid-successional species in a variety of systems (McLendon & Redente 1992; Paschke *et al.* 2000; Blumenthal *et al.* 2003).

Despite the negative effects of mulching, mulch offers protective functions, which benefit developing seedlings and established plants in the long-term (Ludwig *et al.* 1994; Milton 1995b; Beukes & Cowling 2003). Many existing succulent and non-succulent adult plants were observed to have much higher water contents within mulched plots compared to plants off mulched plots. Existing mulched areas could be populated by these plant species in the long-term (Petersen *et al.* 2004b).

3.5.4 Seed treatments

The low abundance of palatable and highly palatable species and an absence of their seed in the soil seed bank are considered among the main factors limiting recruitment and establishment (Milton 1992a, 1994, 1995b; Jones & Esler 2004). Three of the species sown (*Tripteris sinuata, Drosanthemum hispidum, Fingerhuthia africana*) were not recorded at the site in 2009. Three however, were present in low numbers (*Tetragonia fruticosa, Ruschia spinosa, and Lessertia annularis*). *Tripteris sinuata* and *Fingerhuthia africana* had significantly higher cover and density within seeded plots. Both these species however, were also recorded within unseeded areas. It is likely that seed, which was sown at the study site, was blown or washed into unseeded plots by wind and rain. Species that were recorded in 2009 probably germinated from the natural seedbank or more likely their seed were washed or blown into the rehabilitated site from adjacent areas. As ostriches concentrate mainly on flat low-lying areas, hilly areas (koppies) occurring in close proximity to the rehabilitation site probably provided a natural seed source (Milton 1995a, 1995b).

Ruschia spinosa established better within micro-catchments compared to ripped or control sites. Succulent seedlings (mainly Mesembryanthemaceae) both seeded and naturally recruited were observed to establish on micro-catchment walls instead of mulched micro-catchment floors as well as bare surfaces in between micro-catchments. Mesembryanthemaceae family have hygrochastic capsules and are water dispersed (Parolin 2001). Micro-catchment walls are relatively smooth and probably trapped small seeds of succulents. The lack of competition on the walls provided the seeds with an opportunity to germinate when moisture was sufficient. Wind dispersed perennial species, with winged seeds like *Tripteris sinuata* had significantly higher density within micro-catchments. These were mainly observed within the mulched layer on the floor of the micro-catchment.

Researchers have reported that local seed banks have contributed to vegetation rehabilitation in a range of environments (Beukes & Cowling 2003; Milton 1994). Small seeded Aizoaceae species and certain annuals and pioneers can survive more than two years in the soil seed bank (Esler 1993, Wiegand *et al.* 1995). Naturally occurring species, which have shown to be successful in terms of

plant density on Morestêr farm, include *Limeum aethiopicum, Tetragonia fruticosa, Malephora lutea, Galenia pubescens. Limeum aethiopicum* is considered a very important fodder plant for livestock and is heavily browsed. It is remarkably resilient to grazing and is able to recover even after being completely browsed down particularly after good rains and rest from grazing (Vlok & Vlok 2010). Hardy succulents e.g. *Malephora lutea* serve as pioneers, as they establish more easily in open eroded sites in comparison to shrubs (Yeaton & Esler 1990). Beukes and Cowling (2003) similarly noticed the importance of soil stored seed and naturally occurring plants such as *Limeum aethiopicum, Galenia sp.* and *Malephora sp* for the revegetation of bare areas. The potential of the surrounding, mainly non-forage (but also partly forage) community, to expand into bare areas has to be exploited. This will avoid the costs of an exogenous seed source (Milton & Dean 1994, Milton & Dean 1995, Jones & Esler 2004; Snyman 2004).

According to the stress gradient hypothesis, it is assumed that competition becomes less important and facilitation more important in drylands because abiotic conditions are harsh (Callway & Walker 1997; Maestre *et al.* 2009). However, a study by Carrick (2003) demonstrated that facilitation appears to play a less important role. Competition from annuals has been shown to increase the mortality of seedlings of perennials plants under arid conditions (van Epps & Mckell 1983). By contrast, annuals had no detectable effect on the seedlings of perennials in a clearing experiment in the southern Karoo (Milton 1995b). There were no competition and facilitation effects evident within this short-term study at Morestêr farm. It is possible that micro-catchments decreased the number of existing plants within the micro-catchment area and on the micro-catchment wall and this aided increases in palatable species richness and plant density. However, a decrease in competing less- and unpalatable plants was not significant between treated and untreated sites. Future monitoring at this site could assist in a better understanding of the interplay between competition and facilitation in Succulent Karoo rangelands.

3.5.5 Paying for rehabilitation and management implications

Farmer and rangeland managers are generally interested in low-cost methods where the best results can be obtained over a short period. This is understandable, especially in semi-arid areas where rangeland restoration contains large elements of risk because of the low probability of follow-up rain (Van der Merwe & Kellner 1999; Beukes & Cowling 2003).

Short-term rehabilitation (post one-year) of soil, seed and mulch treatments have not proved cost-effective in terms of representing a positive return on investment. This is because the treatments did not increase vegetation cover or grazing capacity and therefore, the farmer cannot sustainably stock

livestock. The short-term value of an increase in palatable species richness, plant density and plant diversity reflected mainly in the form of seedlings and young plants is hard to determine and will probably be based on personal opinion and intrinsic value rather than an actual rand value (Ribaudo *et al.* 2010; Farley & Costanza 2010). Rare long-term monitoring (10 years) has shown that initial restoration successes may be transient (Snyman 2003). Others (Kinyua *et al.* 2009), however suggest a relatively stable rehabilitation response. From a long-term perspective, rehabilitation could be viable and cost-effective. Continued monitoring of the experiments will show if perennial palatable plant density and pioneer density facilitate an increase in perennial plant cover and grazing capacity. Long-term benefits could include increased carrying capacity for livestock and game, improved veld condition, increased biodiversity and the possibility of reversing the desertification process (Visser *et al.* 2004). Benefits such as these are more tangible for private landowners.

The return interval for palatable plant recovery on rested overgrazed rangelands in the Karoo is more than two decades (Seymour *et al.* 2010). Increases in perennial palatable plant density and species richness within micro-catchments may be a satisfactory achievement over one-year. It implies an improvement of the veld as more forage plants are available for grazing. Whether these short-term effects of rehabilitation are sufficient motivation for private landowners to spend time and money on rehabilitation will depend on the individual and the intrinsic value they place on their veld. However, given the unpredictable nature of semi-arid areas and the low natural grazing capacity (60ha/LSU) of these Succulent Karoo lowlands, only inexpensive conservation measures should be justified for private landowners (Snyman 2003). In the short-term ripping treatments were the least costly (R 0.13/m^2) method however the most ineffective. Micro-catchment treatments were the most effective at increasing palatable plant density and species richness, however were more costly (R 1.54/m^2) than ripping and sowing seed (R 0.37/m^2). Micro-catchments could however provide long-term benefits for grazing capacity. Mulching (R2.47/m^2) was more costly than ripping, sowing seed and micro-catchments. Removal of herbivores from rehabilitation sites increases seedling establishment and survival (Esler *et al.* 2006). Morestêr farm was exposed to continual grazing by springbok (*Antidorcas marsupialis*) after rehabilitation. Springbok were removed at the end of 2009. This could lead to an increase in growth of established seedlings and young plants (Milton 1992b; Milton & Wiegand 2001). The potential for existing long-lived palatable and less palatable plants within the area especially on the nearby hilltops (koppies) needs to be fully investigated. Using resources adjacent to degraded sites can reduce the costs of rehabilitation treatments for landowners.

Seeding alone is not recommended (Milton & Dean 1995; Visser *et al.* 2004) as seeding was not able to increase palatable plant density. It is possible that seed should have been introduced at the farm

one to three years after the soil and mulch treatments. Many authors suggest that for ecosystems in a highly degraded state, restoration efforts should first focus on increasing the number of micro-sites for the recovery of vascular plants and biological soil crusts. Once this intervention has reduced the degradation processes, the introduction of seeds of indigenous palatable plants should be the next step (Milton *et al.* 1994a; Mendez *et al.* 2008). If funds are available for seed, large winged seed should be sown within micro-catchments. Alternatively, seed can be collected on site in areas, which still harbour palatable species. Mesembryanthemaceae seed should be avoided. Instead, succulent adult plants from areas around the rehabilitated site should be transplanted. Species that have shown potential in other parts of the Succulent Karoo include *Brownanthus sp.* and *Cephalophyllum sp.* (Anderson *et al.* 2004; Schmiedel *et al.* 2010). These functional plants although mostly not palatable act as ecosystem engineers and facilitate seedling establishment by reducing high temperatures near the soil surface as well as providing microhabitats with a higher soil nutrient content (Turner *et al.* 1966; Humberto *et al.* 1996). Caution should be taken to allow for sufficient gaps between transplanted functional plants as established perennial plants have been shown to out-compete seedlings in certain Karoo environments (Milton 1994, 1995b). Competitive interactions are caused by light deprivation and competition for water and nutrients (Yeaton & Esler 1990; Esler & Phillips 1994; Milton 1994).

Micro-catchment, seeding and mulching treatments are all labour intensive. In communities where unemployment rates are high (around 40% in the Little Karoo) labour-intensive projects are welcomed and sometimes subsidised by government (Murray 2007; Turpie *et al.* 2008). Outside funding would alleviate financial constraints faced by landowners, while job creation would act as a social and financial boost for the local community. Further longer-term studies are needed to understand the effect of rehabilitation on vegetation conditions and if necessary generate support for government subsidisation or stimulate Payment for Ecosystem Service schemes. Ecosystem services such as key hydrological services (water flow regulation and erosion control) were not measured in this study. It is possible that the rehabilitation treatments increased rainfall infiltration and reduced erosion at Morestêr Farm in the short-term. This could provide the conditions for future improvements in grazing capacities and vegetation cover. This however was not tested in this study. Given the short-term nature of this research, follow up monitoring will be imperative for understanding these short-term effects.

3.6 Conclusion

In most cases, the aim of rehabilitation is to increase biodiversity for higher resilience, increase the vegetation cover to combat erosion and to increase the production potential for higher grazing capacity (Bakker *et al.* 1996; Seymour *et al.* 2010). This study was designed to investigate the importance of ripping, micro-catchments, seeding, and mulching for the rehabilitation of severely degraded ostrich rangeland. It appears that using micro-catchments, seeding and mulching all have short-term benefits for creating opportunities for further long-term recovery. Micro-catchments increased forage species richness and density and seed treatments increased forage species richness. Mulching increased plant diversity by increasing species evenness. The effect of rehabilitation over one-year however does not provide tangible benefits for landowners in terms of increasing grazing capacity of the farm. However, increases in palatable plant density, species richness and plant diversity might provide these benefits in the long-term. Almost all species present on heavily overgrazed Succulent Karoo rangelands are less palatable. Rehabilitation treatments involving micro-catchments and seed treatments have shown the potential to enable the return of grazing capacities and system resilience. System resilience is important especially in semiarid areas where perturbations are a common phenomenon. However, follow up monitoring is recommended to evaluate whether increases in density and richness translate into increases in vegetation cover and grazing services.

Chapter 4:

The effect of rehabilitation on landscape functioning and ecosystem services in the semi-arid Succulent Karoo lowlands of the Little Karoo, South Africa

4.1 Abstract

Without active intervention, severely degraded rangelands may not return to functional states within timescales practical for landowners. While there is potential for landowners to benefit from rehabilitating degraded rangelands, this option is more likely to be realised if the benefits of rehabilitation are investigated and made known. Ecosystem services are the benefits people obtain from ecosystems. This study evaluates the effect of rehabilitation on five ecosystem services: water infiltration, erosion control, forage production, nutrient cycling and plant diversity on degraded ostrich rangelands in the Succulent Karoo lowlands of the Little Karoo. The Landscape Functional Analysis (LFA) Indices of water infiltration, nutrient cycling and soil stability (Tongway & Hindley 2004) in conjunction with empirical/quantitative measurements were used to evaluate the performance of active rehabilitation as a management technique for enhancing ecosystem service provision. Within 18 months, rehabilitation enhanced water flow regulation as indicated by improved water infiltration rates (ml/s), and increased erosion control as indicated by an increase in percentage cover of resource sink zones such as micro-catchments and mulched areas. These resource sink zones capture nutrients, soil and water and are indicative of improved erosion control. After a year and half, however, rehabilitation had not been sufficient for increasing nutrient cycling (soil organic carbon and nitrogen), forage production (LSU/60ha) and plant diversity (species richness) ecosystem services. Empirical evidence was more useful in determining the effect of rehabilitation on landscape functioning and ecosystem services than the Landscape Functional Analysis Indices, which were not consistent with empirical results and were relatively complex to interpret. The LFA landscape organisation index (total patch length/length of transect) was, however, useful in evaluating the effect of rehabilitation on erosion control and correlated well with empirical measurements of water infiltration. This study has shown that using active rehabilitation in combination with improved management strategies (e.g. rest from grazing) may induce recovery of certain ecosystem services within practically relevant time scales. In the short-term, the small scale of rehabilitation outcomes along with high costs makes it difficult to promote rehabilitation as a tool for increasing ecosystem services. To alleviate financial constraints on farmers and in order to socially and financially boost depressed local communities, the focus for now should be on sourcing funds from government for rehabilitation implementation. Small responses to rehabilitation could accumulate over time. Further research is needed to evaluate the long-term effects of rehabilitation on ecosystem services in order to promote markets for these non-commodity resources.

4.2 Introduction

The rapid rate that natural ecosystems are transformed by human activities, and the increasingly reduced area occupied by untransformed ecosystems, has emphasised the importance of

conservation, particularly ecological rehabilitation, outside of formally protected areas (Young 2000; Cortina *et al.* 2006; Reyers *et al.* 2010). Rehabilitation seeks to halt degradation and aims to redirect disturbed land to a pre-disturbed functional state (Aronson *et al.* 1993). One way to motivate for rehabilitation on private land is to demonstrate the ecosystem services ("the benefits humans obtain from ecosystems") attained through recovering degraded semi-natural fragments (O'Farrell *et al.* 2009; O'Farrell *et al.* 2010; Reyers *et al.* 2010). Ecosystem services, which are important for supporting human livelihood, include carbon storage, regulation of climate and water flow, provision of clean water, and maintenance of soil fertility (MA 2005; Rey Benayas *et al.* 2009). The quantification of these benefits is useful for incentivizing private landowners towards rehabilitation of their land (Hobbs & Harris 2001; Herling *et al.* 2009; O'Farrell *et al.* 2009). It also serves as a motivation for the implementation of fiscal (incentives, Payments for Ecosystem Services) and regulatory methods (punitive actions, statutory requirements) from governing bodies and the private sector (Salzman 2005; O'Farrell *et al.* 2009; Kemkes *et al.* 2010). Creating markets for non-commodity ecosystem services (Payments for Ecosystem Services) is an increasingly mainstream tool for influencing land-use decisions on private land (Aronson *et al.* 2010a; Daniels *et al.* 2010; Farley & Costanza 2010). In South Africa, Payment for Ecosystem Service programs that simultaneously restore hydrological services as well as attempt to address local poverty issues have been extremely successful (Turpie *et al.* 2008).

The Succulent Karoo biome in South Africa is the most diverse arid environment in the world (Desmet 2007). It is one of three global biodiversity hotspots (Mittermeier *et al.* 2005) and is under significant pressure from overstocking and heavy grazing by ostrich, sheep, cattle and donkeys (Mucina *et al.* 2006). A major and immediate concern is the boom in the ostrich farming industry (Cupido 2005; Mucina *et al.* 2006). Poor ostrich grazing management practises have been concentrated within the Succulent Karoo plant communities of the Little Karoo, leaving less than 2% of the natural vegetation in good condition with approximately 27% severely degraded, 61% moderately degraded and 9% completely transformed by cultivation (Le Maitre *et al.* 2009; Thompson *et al.* 2009). Historically, ostriches moved between rainfall patches following watercourses and selected high quality forage (Milton *et al.* 1994b; Dean & Milton 1999). For farming purposes, they are now confined to grazing camps (300 ha) at high densities (1.5 ha/ostrich), a practice known as flock breeding (Milton *et al.* 1994b; Cupido 2005; Murray 2007). Trampling and overgrazing by ostrich has resulted in changes in plant species composition, loss of vegetation cover and destruction of the biological soil crust (Cupido 2005). This has been attributed to a loss in ecosystem services of forage production, water flow regulation, tourism and erosion control (Reyers

et al. 2009). The South African Ostrich Industry in 2009 identified the inadequate management of biodiversity and ecosystem services as a major threat to the industry's sustainability. A switch to a relatively environmentally friendly pen breeding or feedlot farming method for ostriches combined with the rehabilitation of degraded rangelands has been recognised as a way to decrease the vulnerability of the ostrich industry to market shifts, disease or floods as well as improve productivity and conserve biodiversity (Murray 2007; SAOBC 2009). Vegetation recovery of degraded areas by means of natural succession is very slow. Therefore, removing grazing pressure alone is not enough to reverse degraded land back to former productive states and active rehabilitation is usually required (Westoby *et al.* 1989; Milton *et al.* 1994a; Milton & Dean 1995). Establishing a government funded Payment for Ecosystem Services in the Little Karoo could highlight the private benefits of conserving ecosystems, thereby encouraging landowners to move towards an alternative ostrich farming system and resting or rehabilitating other parts of their land (Le Maitre *et al.* 2007; Herling *et al.* 2009). The uncertainty regarding how environments respond to rehabilitation interventions is an issue facing private landowners wishing to participate in conservation as well as governmental institutions expressing an interest in funding rehabilitation projects (Cupido 2005; Ribaudo *et al.* 2010).

There are two approaches to quantifying the effects of rehabilitation. Nominal or qualitative approaches are based on categorical and ordinal variables and do not directly measure soil properties. These methods are based on indicators of ecosystem function and have often been employed due to their compromise between accuracy and affordability, especially in arid and semi-arid ecosystems (Whitford 2002; Maestre & Cortina 2004). Empirical or quantitative approaches on the other hand, directly measure continuous variables such as soil carbon and nitrogen, erosion, infiltration and the water holding capacities of soils (Holm *et al.* 2002). Empirical methods are usually more costly and time consuming in comparison to nominal methods. Although important advances have been made in the development of nominal methodologies (Tongway & Hindley 1995; Milton *et al.* 1998; Tongway & Hindley 2004), empirical methods in South African landscapes are favoured in comparison to the more subjective nominal methods (Palmer *et al.* 2001; Petersen *et al.* 2004a; O'Farrell & Donaldson unpublished).

Those ecosystem services, which are responsible for soil retention and enhancing rainfall infiltration, are of major importance in semi-arid South African rangelands (O'Farrell *et al.* 2009). In this study, the Landscape Function Analysis (LFA), based on indices derived from soil surface characteristics (Tongway & Hindley 2004), as well as empirical measurements were used to investigate whether rehabilitating degraded ostrich rangeland in the little Karoo increases landscape function and

ecosystem services within one and a half years of rehabilitation application. The main goal of this study was to provide information on the performance of active rehabilitation as a management technique for enhancing ecosystem service provision on degraded ostrich rangelands. Field-level sampling was used to estimate the effects of rehabilitation on five ecosystem services: water flow regulation, plant diversity, forage production, nutrient cycling and erosion control. The relationship between the LFA indices and empirical data was also assessed in order to evaluate the value of using LFA within a Succulent Karoo semi-arid environment.

4.3 Methods

4.3.1 Study Site

The study was carried out in the Succulent Karoo lowlands of the Oudtshoorn basin (10,163 km^2) in the Little Karoo (19,730 km^2), South Africa. The Little Karoo is located between the coastal Langeberg and Outeniqua mountains and the inland Anysberg, Swartberg and Antoniesberg mountain ranges. It extends from the town of Montagu in the west to Uniondale in the east, and encompasses four local municipal areas, Ladismith, Calitzdorp, Oudtshoorn and Uniondale (O'Farrell *et al.* 2008). The Oudtshoorn basin falls between two mountain ranges, the Swartberg mountain range in the north and the Outeniqua mountain range in the South and is situated approximately between the towns of Calitzdorp in the west and Oudtshoorn in the east (Thompson *et al.* 2009) (Fig. 4.1).

Since 1865, the Oudtshoorn basin has been at the centre of the Little Karoo's ostrich industry (Beinart 2003; Cupido 2005). It is now considered the most important ostrich production region producing more than 70% of South Africa's ostrich products (Cupido 2005, Murray 2007). The climate is warm-temperate and sub-mediterranean. Mean annual rainfall ranges from 150 to 250 mm, increasing to 400 mm on the lower slopes of the mountains, and up to 1000 mm on the highest, coast-facing peaks (Thompson *et al.* 2009). Rain may fall at any time of the year, however the most predictably wet seasons are spring (September–October) and autumn (April–May). Summers are hot (daily maxima up to 40 °C are frequent) and usually dry (Thompson *et al.* 2009). The low-lying parts of the basin are dominated by the dwarf, succulent shrublands, mainly Gannaveld, Apronveld and Succulent Karoo transitional thicket vegetation types, associated with the Succulent Karoo biome. Lower slopes are covered in dense thicket associated with the sub-tropical thicket biome; the upper slopes consist of fire-prone shrublands and heathlands of the fynbos biome (Low & Rebelo 1996; Vlok *et al.* 2005; Thompson *et al.* 2009). Since European settlement in the 1730s, the Oudtshoorn

Basin has provided forage for large numbers of livestock (principally ostriches). Prior to this, the area was intermittently grazed by sheep and cattle belonging to Khoekhoe nomadic pastoralists (Smith 1999). Upland habitats, associated with the Fynbos biome, provide poor forage and are lightly, if at all, grazed by livestock (Thompson *et al.* 2009).
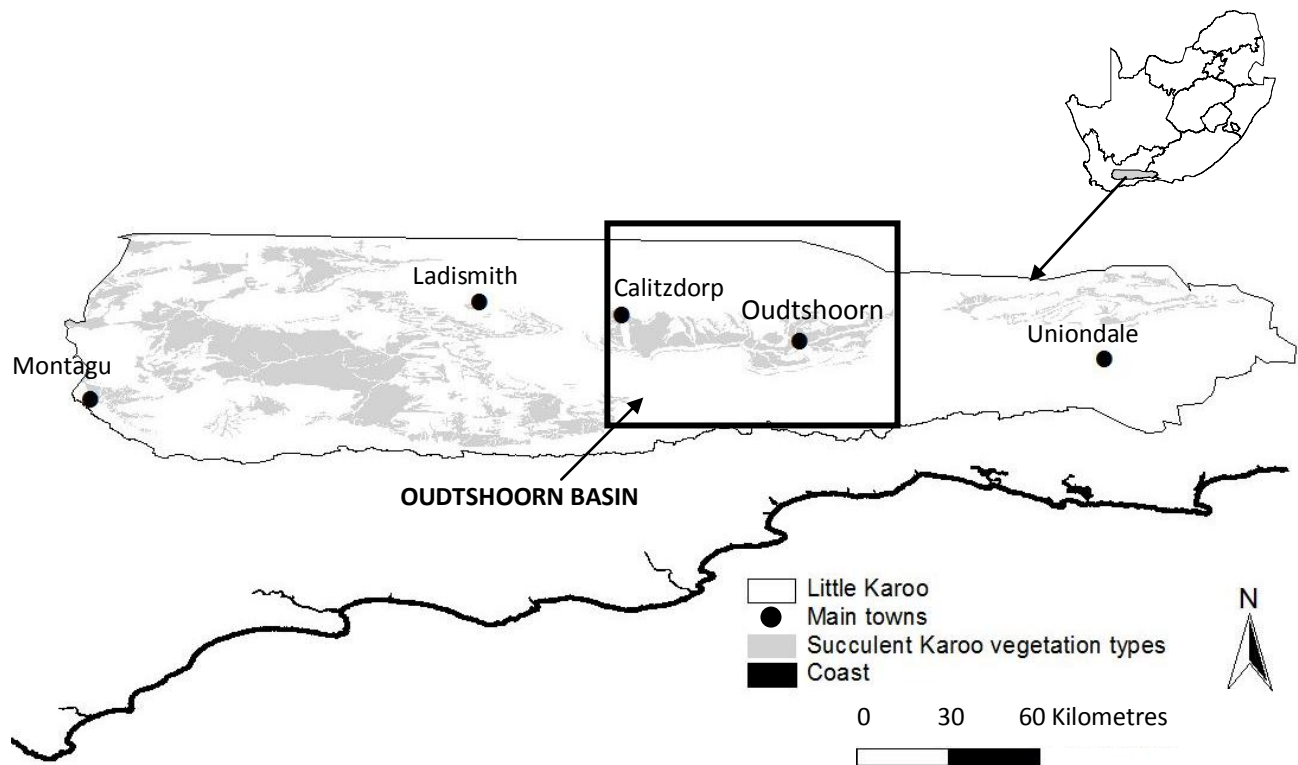
Figure 4.1 Map showing the Succulent Karoo lowlands of the Oudtshoorn basin in the semi-arid Little Karoo in South Africa (Vlok *et al.* 2005; Thompson *et al.* 2009)

4.3.2 Study Farms

Fieldwork was conducted 1 year and 8 months after rehabilitation was applied to four ostrich farms, Welbedag (33º33'54.2" S, 21º57'94.5"E), Greylands (33º35'83.1" S, 22º02'65.2"E), Witklip (33º42'21.3" S, 22º05'37.8"E) and Morestêr (33°35'38.90"S; 22° 1'35.94"E), falling within the Succulent Karoo lowlands in the Oudtshoorn basin. Surveys were also conducted on a benchmark site in the South African National Defence Force Military base near Oudtshoorn (Fig. 4.2).

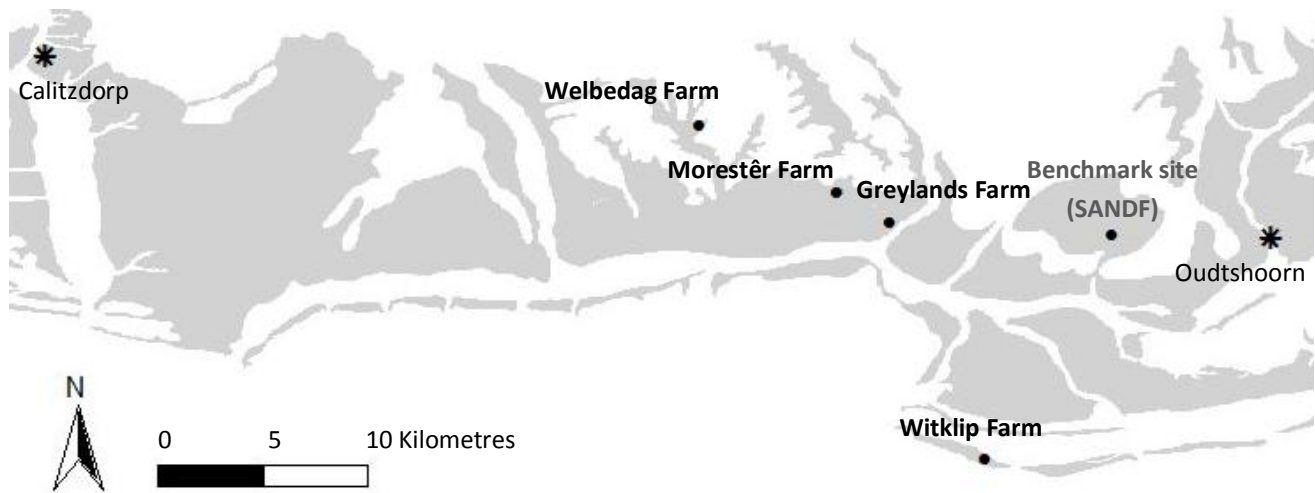**Figure 4.2** Study farms on which an ostrich camp was rehabilitated in February to March 2009. Grey areas represent the Succulent Karoo vegetation communities occurring on low-lying areas between Oudtshoorn and Calitzdorp, two towns of the semi-arid Little Karoo, South Africa (Vlok *et al.* 2005)

All farm ostrich camps are situated within the Succulent Karoo vegetation types of Apronveld, Gannaveld and Succulent Karoo thicket mosaics (Vlok *et al.* 2005; Thompson *et al.* 2009) (Table 4.1) and historically have been stocked at <6 ha/LSU. Welbedag and Morestêr have both been farmed for *c.* 40 years with ostriches, whereas Witklip and Greylands have only been farmed for one breeding season (*c.* one year) with ostriches. Apart from ostrich farming some of the camps have also supported springbok (*Antidorcas marsupialis*) and nguni cattle (*Bos taurus africanus*) in recent years (J. Ernst, H. Jonker, J. Potgieter pers. comm. 2010) (Table 4.2).

During 2008, study farms, which were considered to be transformed by either long or short-term ostrich farming pressure (S. Milton & K. Coetzee *in litt.* 2008), were selected for rehabilitation trials. Between February and March 2009 an area greater than 2500 m^2 of each available degraded ostrich camp per farm was rehabilitated using a combination of micro-catchments, mulching and seed treatments. Micro-catchments (i.e. holes *c.* 600 mm in depth and *c.* 500 – 900 mm in diameter) were made in the hard soil using a pick. Loosened material was shovelled into a low berm (micro-catchment wall) on the downhill side of the micro-catchment. Mulch consisted of pine and gum wood-chips approximately 100–150 mm in length. These were placed to form a dense mat (*c.* 100 mm deep) on the soil surface (Coetzee 2005; W. Stroebel pers. comm. 2010). Mulch was applied on compacted soil surfaces, ostrich paths and the floors of micro-catchments. A seed mix containing highly palatable, palatable and less palatable plant species was broadcast onto ostrich paths, mulched areas and within micro-catchments with mulched floors. Areas chosen for rehabilitation were generally the areas in the worst condition within the ostrich camp. Morestêr and Witklip were considered severely to moderately degraded in 2008 and Greylands and Welbedag were assessed as

less degraded or patchily degraded due to better vegetation cover and biological soil crusts (S. Milton & K. Coetzee *in litt.* 2008). The benchmark site consisted of an area on the South African National Defence Force Military base in Oudtshoorn (12 000 ha). The Military base has not been farmed with ostriches since the 1940s. However, a population of donkeys (*Equus africanus asinus*) has grazed on the area in recent years. In 2010, there were 300 donkeys on the base (M. Gush pers. comm. 2010). Although relatively degraded by military activities and grazing donkeys, the land had comparatively good biological soil crust and vegetation cover. The benchmark site was characteristic of Gannaveld vegetation (Vlok *et al.* 2005).

Table 4.1 Study farms and their associated habitat and vegetation description, dominant genera and environment characteristics (Vlok *et al.* 2005; Thompson *et al.* 2009).

Farm	Habitat	Description	Dominant genera	Environment
Morestêr Witklip	Gannaveld Gannaveld	Open, semi-succulent, low (\0.5 m) shrubland	*Salsola, Lycium, Tripteris, Pteronia*	Bottomlands; deep loamy sands; high salt content; 180–250 mmyr^{-1}
Greylands	Apronveld	Open, semi-succulent, low shrubland	*Pteronia, Eriocephalus*	Base of hills; shallow, rocky (colluvium) and clay-rich soils; 180–300 mmyr^{-1}
Welbedag	Succulent karoo thicket mosaics	Open, semi-succulent, low shrubland with emergent medium-high (1.5–2.0 m) shrubs	*Euclea, Rhigozum, Monechma, Drosanthemum*	Rocky slopes and ridges; shallow, rocky and clay-rich soils; 180–250 mmyr^{-1}

Table 4.2 Study farm history and stocking rates (J. Ernst, H. Jonker, J. Potgieter pers. comm. 2010)

Farm	Camp size	Stocking rate Ha/ostrich	Ostrich farming commenced	No. breeding seasons	Grazing regime	Ostrich grazing exclusion	Other livestock grazing
Morestêr	234ha	3	1940s	±23 seasons	Every 2/3 years	2007	Springbok - 4 years removed in 2005
Witklip	1062ha	1.5	2004	1 season	NA	2005	Rest 1999 -2004. Pre-2004 unknown
Greylands	359ha	3.5	1999	1 season	NA	2000	Nguni - 3 months in 2006; Springbok - 4 years removed in 2010. Pre-1999 unknown
Welbedag	1000ha	6	1940s	±23 seasons	Every 2/3 years	2008	NA

4.3.3 Experimental design, fieldwork and data processing

In October 2010, two sites, the rehabilitated and control (*c.* 60 m by 60 m) were marked out at three of the study farms, Welbedag, Greylands, and Witklip. Control sites were located in close proximity

to rehabilitated areas and were chosen to be representative of the rehabilitation site. Whereas areas >2500 m^2 on the latter three farms were rehabilitated with micro-catchments, mulching and seed treatments as deemed necessary, the rehabilitation on Morestêr was applied in an experimental crossed plot design (see Chapter 3). In order to include a rehabilitated and control site at this farm study plots were randomly chosen from the mulch, seed and micro-catchment treatments as well as the control plots. An area of *c.* 60 m by 60 m was marked at the military base to represent the benchmark site.

The landscape function of each site was assessed using the Landscape Function Analysis (LFA) (Tongway & Hindley 2004). Landscape Function Analysis uses soil surface indicators to assess the status of a given ecosystem in terms of functionality, that is, the degree to which resources tend to be retained, used, and cycled within the system. Output is given by three indices (stability, infiltration, and nutrient cycling) that summarize different components of ecosystem functionality (Maestre & Cortina 2004; Tongway & Hindley 2004). The stability index provides information about the ability of the soil to withstand erosive forces and to recover after disturbance. The infiltration index shows how the soil partitions rainfall into water available for plants, and run-off water that is lost from the system. The nutrient cycling index provides information about how efficiently organic matter is cycled back into the soil (Maestre & Cortina 2004; Tongway & Hindley 2004).

In order to conduct LFA surveys two 50 m transects were randomly located down slope within each *c.* 60 m by 60 m rehabilitated and control site on Welbedag, Greylands and Witklip ostrich camps as well as on the benchmark site. On Morestêr farm one 7.5 m transect was randomly located down slope in seven randomly chosen rehabilitated and control plots. A continuous record of patch and inter-patch zones were recorded along the transects. A patch was defined as a long-lived feature, able to collect water, sediments, and nutrients coming from run-off and separated by bare soil surface (inter-patches) from the next patch. Examples of patches include perennial plants and shrub branches contacting the soil (Tongway & Hindley 2004). Patches also include rehabilitation treatment structures like micro-catchments and mulch, as they are able to collect water, sediment and nutrients (Tongway & Hindley 2004; D. Tongway pers. comm. 2010). The soil surface of six (where available) randomly chosen patches per patch type (i.e. shrub clump, micro-catchment etc.) and six (where available) randomly chosen inter-patches per inter-patch type (i.e. bare soil, bare litter etc.) was assessed per transect. Eleven soil surface features were recorded within these patches and inter-patches following a semi-quantitative scale (Tongway & Hindley 2004) (Table 4.3).

Table 4.3 Soil surface features used to calculate the Landscape Function Analysis Indices (Maestre & Cortina 2004; Tongway & Hindley 2004)

Soil Surface Feature	Interpretation	Maximum Score	Index Used
Rainsplash protection/soil cover	Assesses vulnerability to physical crust formation	5	Soil Stability
Perennial vegetation cover	Assesses contribution of root biomass to nutrient cycling processes	4	Infiltration, nutrient cycling
Litter cover	Assesses vulnerability to physical crust formation		Soil Stability
Litter cover, origin and degree of decomposition	Assesses the availability of surface organic matter for decomposition and nutrient cycling	30	Infiltration, nutrient cycling
Biocrust cover	An indicator of surface stability, resistance to erosion, and nutrient availability	4	Soil stability, Nutrient cycling
Crust brokenness	Assesses loose crusted material available for wind ablation or water erosion	4	Soil Stability
Erosion type and severity	Assesses the nature and severity of recent or current soil loss	4	Soil Stability
Deposited materials	Assesses the quantity of soil accumulated from upslope sources	4	Soil Stability
Soil surface roughness/Micro-topography	Assesses surface roughness for water infiltration and flow disruption, seed lodgement	5	Infiltration, nutrient cycling
Surface resistance to disturbance	Assesses likelihood of soil detachment and mobilization by mechanical disturbance	5	Soil Stability, infiltration
Slake test	Assesses soil stability/dispersiveness when wet	4	Soil Stability, infiltration
Soil texture	Classify the texture which assesses infiltration rate and water storage	4	Infiltration

To obtain the three landscape functional indices (soil stability, infiltration and nutrient cycling) presented in this report as percentages, the scores that relate to the different three indices (Table 4.3; column four) for the different soil surface features were summed per patch and inter-patch surveyed. This LFA value was then divided by the maximum score that can be obtained for a given LFA index (40, 57 and 40 for the stability, infiltration, and nutrient cycling indices, respectively) See Tongway and Hindley (1995; 2004) for a complete description of score assignment and calculations. The combination of the measured soil surface features to obtain the LFA indices was performed with a Microsoft Excel template developed by David Tongway. LFA values are presented as percentages. The higher the values obtained, the better the status of the site for a given function. From the LFA transects the landscape organisation index (LOI) was calculated by dividing the sum of the patch zone lengths by the length of the transect in order to determine patch cover (%). This index describes how effectively the landscape regulates vital resources. It is considered a measure of how well the landscape traps nutrients, water and sediments and thus regulates soil erosion (Tongway & Hindley 2004).

A double ring infiltrometer (75 mm inner ring and 100 mm outer ring) was used to determine infiltration rate (ml/s) of water into the soil in control and rehabilitated sites. Using the same randomly selected patches and inter-patches that were assessed with LFA soil surface assessments,

six infiltrations were conducted per patch or inter-patch type. The double ring infiltrometer was inserted into the soil to a depth of 2 cm within the centre of the patch or inter-patch. Plants were removed from the soil surface by cutting them off at soil level, taking care to disturb the soil surface as little as possible. The soil surface was cleared of cut plants whereas the organic matter on top of the soil was left in place (Snyman 2003). The outer ring was completely filled where after 200 ml of water was poured into the inner ring. After ten minutes, the volume of water remaining in the inner ring was measured. If the volume of water infiltrated prior to the ten minute cut-off time, this time was then recorded. The outer ring's water level was kept constant to prevent lateral movement of water, thus ensuring one dimensional flow condition (Gupta et al. 2006). Measurements were converted into an infiltration rate (water infiltrated (ml)/time). A soil sample to the depth of 50 mm was taken next to the infiltration site for organic carbon, nitrogen (%) and soil texture analysis. Organic carbon (carbon) was determined using the Walkley-Black method, total nitrogen (nitrogen) was determined in a LECO nitrogen analyser, and soil texture was determined using the hydrometer method (BemLab, Somerset West 2011).

The two transects per site were also used to measure vegetation cover with the line intercept method. Cover of vegetation intercepted by the transect (m) was summed and divided by the total transect length (m) to calculate percentage vegetation cover. Vegetation cover data were used to determine the current grazing capacity (ha/LSU) of the site and was calculated according to the grazing index method (Du Toit 1995, Du Toit et al. 1995; Du Toit 2000; Du Toit 2002; Esler et al. 2006). Current grazing capacity was calculated as, (Benchmark value (500) / veld condition index) x (7.14). The cover of each individual species recorded was multiplied by its specific objective grazing index value, e.g. *Pentzia incana* (5.6% * 2.88 (OGIV)), in order to calculate the veld condition index (16.13). All these individual veld condition indices were summed, to give the total veld condition index for the site. This veld condition index was then compared to the known standard benchmark index (500) (Du Toit 2002; Esler et al. 2006) by dividing the benchmark index by the site veld condition index in order to calculate the grazing capacity of the veld in hectares per Small Stock Unit - ha/SSU (Merino sheep). This was then multiplied by a conversion factor of 7.14 to convert the grazing capacity into hectares per Large Stock Unit – ha/LSU (Du Toit 1995, Du Toit et al. 1995; Du Toit 2000; Du Toit 2002; Esler et al. 2006). Results were presented as the number of Large Stock Units (LSU) per 60 hectares. The recommended stocking rate for Succulent Karoo lowland vegetation types in the Little Karoo is 1 LSU/60ha (Cupido 2005). Objectively estimated grazing indices for plant species (OGIV) were obtained from the literature (Du Toit 1995, Du Toit et al. 1995; Du Toit 2000; Du Toit 2002; Esler et al. 2006). When objective grazing indices were not

available for a particular species its subjective index value was used (Du Toit *et al.* 1995). Plant species richness was determined as the total number of plant species recorded along the transect.

Different types of patches and inter-patches occurred between sites and farms. Control farms generally had three to four patch (e.g. shrub clumps, animal diggings) and inter-patch types (e.g. bare soil with litter, bare soil) whereas rehabilitated sites had many more due to the area containing rehabilitation treatments (e.g. micro-catchments, mulch). This resulted in different sample sizes for measured variables between control and rehabilitated sites on the four different farms (Table 4.4).

Table 4.4 Sample sizes for different variables measured for rehabilitated (R) and control (C) sites per farm including totals (T) and the South African National Defence Force military base benchmark site (B).
Variable cluster 1: Landscape functional analyses indices (Infiltration index, nutrient cycling index and soil stability index) and water infiltration rate (ml/s)
Variable cluster 2: Laboratory measured carbon, nitrogen and soil texture
Variable cluster 3: Landscape organisation index, vegetation cover (%), grazing capacity (LSU/60ha) and species richness.

	Welbedag			Greylands			Witklip			Morestêr			Totals		SANDF
	R	C	T	R	C	T	R	C	T	R	C	T	R	C	B
Variable cluster 1 (n=)	59	48	107	74	49	123	69	46	115	85	47	132	287	190	47
Variable cluster 2 (n=)	19	11	30	26	14	40	27	16	43	27	14	41	99	55	12
Variable cluster 3 (n=)	2	2	4	2	2	4	2	2	4	2	2	4	8	8	2

4.3.4 Statistical Analyses

Data were analysed for grouped rehabilitated and control sites across farms and compared to the benchmark site. Data were also compared between each rehabilitated and control site at each farm and then compared to the benchmark site. Normality and homogeneity of variances were checked using histograms and side-by-side box plots for multiple groups. Statistical differences between grouped LFA indices of soil stability, water infiltration and nutrient cycling, water infiltration rates ml/s, soil organic carbon (%), nitrogen (%) and soil texture for control and rehabilitated sites and the benchmark site were assessed with the non-parametric Kruskal-Wallace test for non-normal distributions. Statistical differences between LFA indices of soil stability, water infiltration and nutrient cycling, water infiltration (ml/s), soil organic carbon (%), nitrogen (%) and soil texture between each rehabilitated site and each control site per farm were assessed with the non-parametric Mann-Whitney-U test for comparing two groups with non-normal distributions. Differences between individual control sites and rehabilitated sites with the benchmark site were also assessed using the non-parametric Man-Whitney-U test. Correlations between the LFA indices and empirical measurements (water infiltration (ml/s); soil organic carbon and nitrogen (%)) as well as soil texture

were tested using the non-parametric spearman rank correlation coefficient (r_s) (Quinn & Keough 2002).

Vegetation cover (%), landscape organisation index (%), grazing capacity (LSU/60ha) and species richness were grouped for rehabilitated and control sites and compared using the non-parametric Man-Whitney-U test when distributions were non-normal and the t-test was used for normal distributions. Grouped variables were not statistically compared to the benchmark site due the benchmark site having a sample size of two. Lord's Range Test for the significance between the means of small samples (n=2-4) was used to compare vegetation cover (%), grazing capacity and species richness between rehabilitated and control sites per farm. Lord's Range test (L-statistic) was also used to compare rehabilitated and control sites to the benchmark site (Langley 1971). Lord's Range test statistic (L) was calculated by working out the mean of each sample group and subtracting the smaller mean from the larger mean. This answer was divided by the sum of the two ranges of each sample group. Significance levels for L are given in a table in Langley (1971). Larger values of L are more significant. Lord's Range tests analyses were performed in Microsoft Excel. All other analyses were conducted with the IBM SPSS Statistics 19 (SPSS Inc., Chicago, IL, U.S.A.). Means are reported in graphs for all data even though rank based non-parametric tests were mainly used. This is done for descriptive purposes.

4.4 Results

4.4.1 Water flow regulation

Water infiltration rate (ml/s) – double ring infiltrometer
The soils of the control sites at the study farms had extremely low mean infiltration rates of 0.04ml/s. Grouped water infiltration was significantly higher on rehabilitated sites (*K=-38.12, df=2, p=0.02*). Infiltration rate, however, was significantly higher on the benchmark site compared to control (*K=121, df=2, p<0.001*) and rehabilitated sites (*K=82.87, df=2, p=0.002*) (Fig. 4.3).

Increases in water infiltration were only significant for rehabilitated sites at Welbedag (*U=1016, p=0.028*) and Greylands (*U=1311.5, p=0.01*) whereas increases on rehabilitated sites at Witklip (*U=1477.5, p=0.32*) and Morestêr (*U=1706.5, p=0.16*) were not significant (Fig. 4.4). The Welbedag and Greylands rehabilitated sites were closer to the benchmark compared to Morestêr (*U=1357, p=0.002*) and Witklip (*U=763, p<0.001*) which were significantly lower than the benchmark. Control sites, Welbedag (*U=612, p<0.001*), Greylands (*U=653, p<0.001*), Witklip

(*U=547.5, p<0.001*) and Morestêr (*U=615, p<0.001*), water infiltration rates were significantly lower than the benchmark.
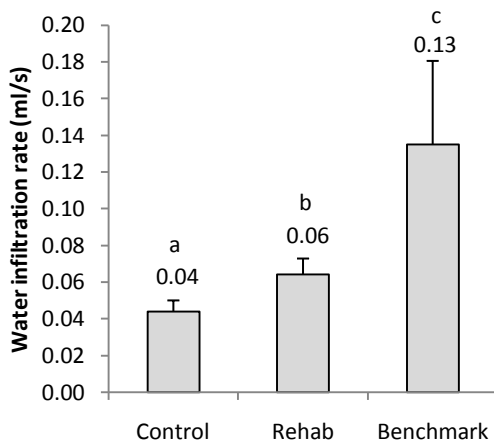

Figure 4.3 Grouped infiltration rate (ml/s) for control and rehabilitated sites compared to the benchmark site. Infiltration rates are expressed as means and error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.
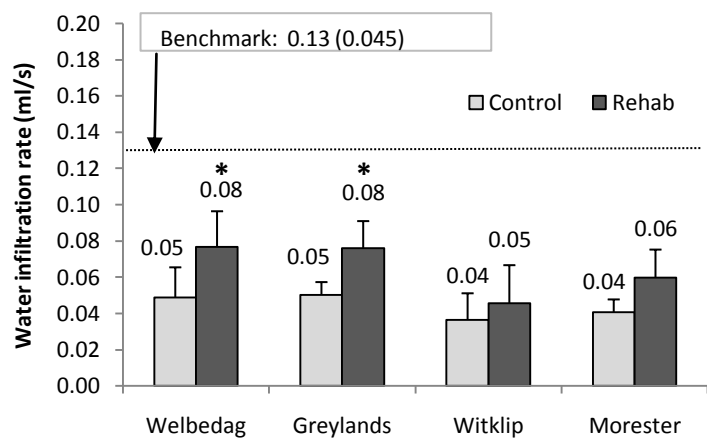
Figure 4.4 Water infiltration rate (ml/s) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Infiltration rates are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control sites per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

Landscape Functional Analysis water infiltration index (%)

Results determined from the landscape functional water infiltration index (%) were similar to those for the double ring infiltrometer measurements and indicated that rehabilitation had a significant effect on grouped water infiltration (*K= 31.65, df=2, p<0.001*). Water infiltration indices were significantly higher on rehabilitated sites in comparison to control sites (*K=-5.59, df=2, p<0.001*). However, unlike the double ring infiltrometer results, the LFA water infiltration index on the rehabilitation sites was similar to the benchmark site (Fig. 4.5).

The significant difference for higher infiltration indices on the rehabilitated sites compared to control sites were consistent across all four farms (Welbedag: *U=917.5, p=0.004*; Greylands: *U=1159, p=0.001*; Witklip: *U=1133.5, p=0.004*; Morestêr: *U=1483, p=0.014*) (Fig. 4.6). Although all rehabilitation sites had similar infiltration indices to the benchmark site, the Welbedag rehabilitated site showed a significantly higher index (*U=1030, p=0.023*).
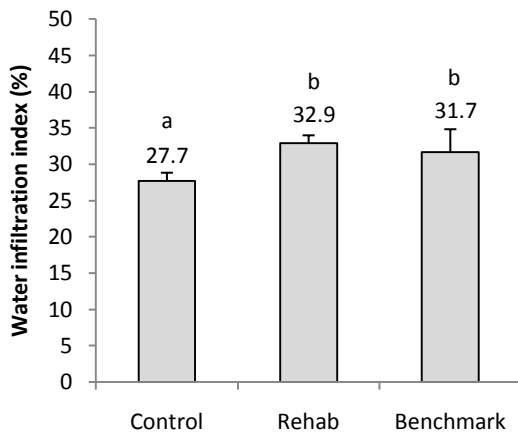
Figure 4.5 Grouped landscape functional water infiltration index (%) for control and rehabilitated sites compared to the benchmark site. Infiltration indices are expressed as means. Error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.
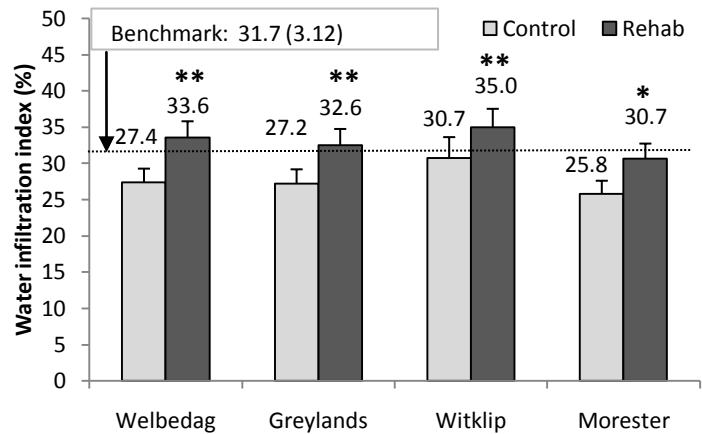
Figure 4.6 Landscape functional water infiltration index (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Infiltration indices are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

4.4.2 Nutrient cycling

Laboratory measured soil organic carbon (%)

There was no significant effect of rehabilitation on soil organic carbon (%). Control and rehabilitation sites did not differ significantly from the benchmark site or from each other in terms of soil organic carbon (%) (*K=4.38, df=2, p=0.112*) (Fig. 4.7). The trend for rehabilitation to have no significant effect on soil organic carbon (%) compared to control sites was consistent across three of the study farms. The Witklip rehabilitated site however, showed a significant decrease in soil organic carbon compared to the Witklip control site (*U=106.5, p<0.006*) and the benchmark site (*U=36.5, p<0.001*) (Fig. 4.8).

Laboratory measured soil nitrogen (%)

Rehabilitation had no effect on soil nitrogen compared to control sites (*K=7.71, df=2, p=0.02*). In addition, rehabilitation sites had significantly lower soil nitrogen compared to the benchmark site (*K=2.6, df=2, p=0.27*) (Fig. 4.9). Three of the farms showed rehabilitation to have no effect on soil nitrogen. The soils of Greylands control site, however, had significantly higher nitrogen than the rehabilitated site (*U=90, p=0.008*) (Fig. 4.10). Control sites across the four farms did not differ significantly from the benchmark. Apart from Witklip rehabilitated site (*U=36.5, p<0.001*), rehabilitated sites did not differ from the benchmark site.
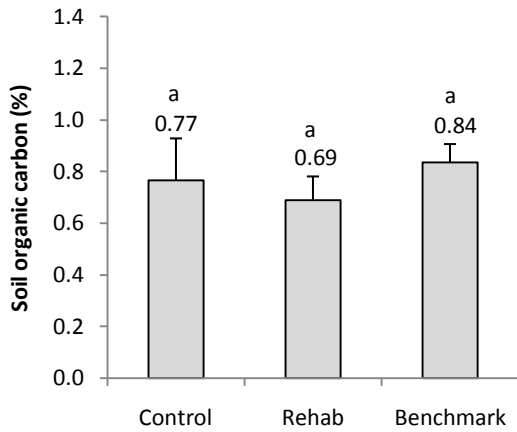
Figure 4.7 Grouped laboratory measured soil organic carbon (%) for control and rehabilitated sites compared to the benchmark site. Organic carbon percentages are expressed as means. Error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.

Figure 4.8 Laboratory measured soil organic carbon (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Soil organic carbon is expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

Figure 4.9 Grouped laboratory measured total nitrogen (%) for control and rehabilitated sites compared to the benchmark site. Soil nitrogen percentages are expressed as means. Error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.
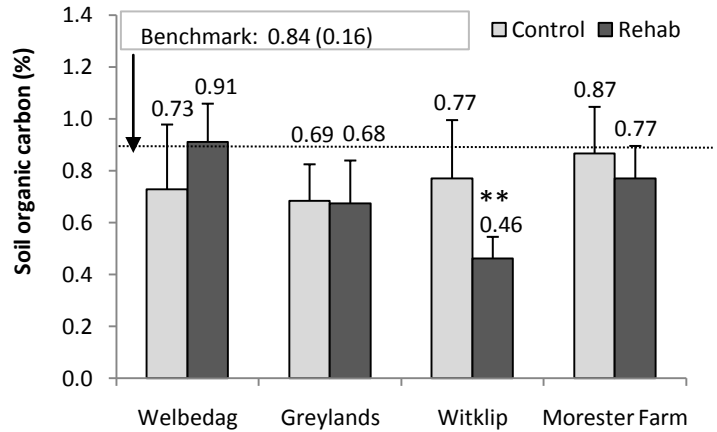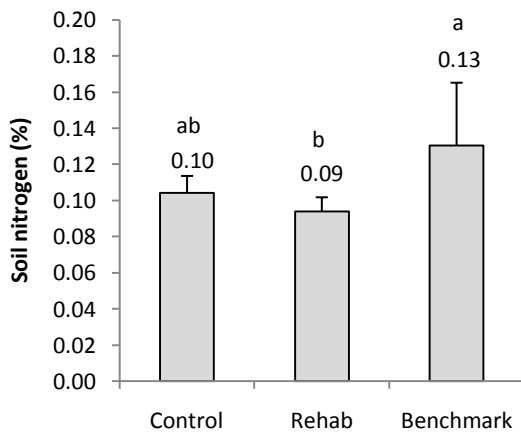
Figure 4.10 Laboratory measured total nitrogen (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Soil nitrogen percentages are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

Landscape Functional Analysis nutrient cycling index (%)

In contrast to the results for laboratory analysis of soil nitrogen and soil organic carbon, the LFA nutrient cycling index was significantly higher on rehabilitated sites compared to control sites (*K=-3.8, df=2, p<0.001*) although it did not differ from the benchmark site (Fig. 4.11). Higher nutrient cycling indices on rehabilitated sites compared to control sites were significant for three of the study farms (Welbedag: *U=1030, p=0.034*; Witklip: *U=1277, p=0.036*; Morestêr: *U=1495, p=0.017*). Rehabilitated and control sites on Greylands farm however, did not differ significantly (Fig. 4.12).

Control sites on Morestêr (*U=716, p=0.003*) and Witklip (*U=844.5, p=0.034*) farms were significantly lower in comparison to the benchmark. The Welbedag and Greylands control sites did not differ significantly. No rehabilitated sites across farms differed significantly from the benchmark.

Figure 4.11 Grouped landscape functional nutrient cycling index (%) for control and rehabilitated sites compared to the benchmark site. Nutrient cycling indices are expressed as means. Error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.
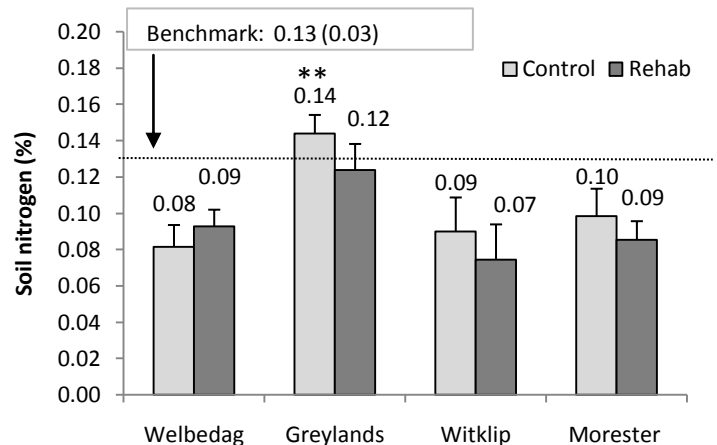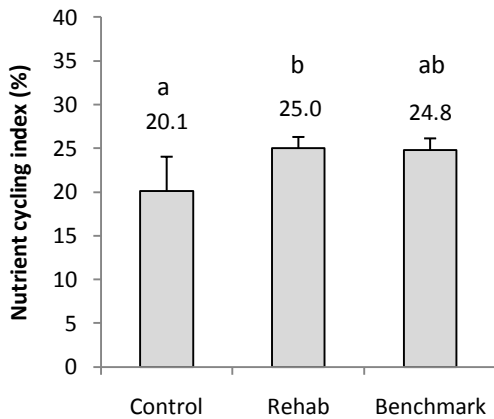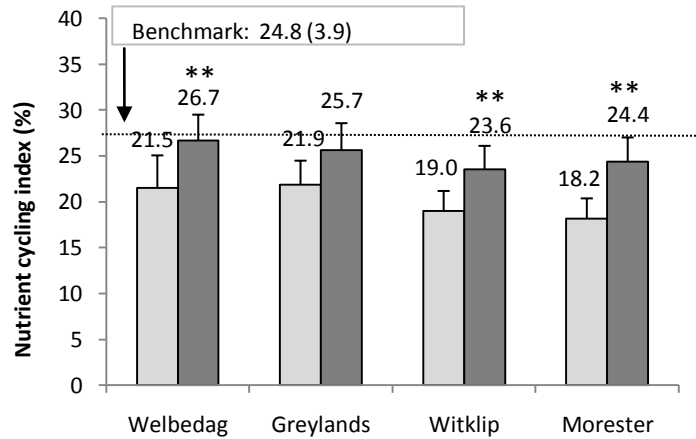
Figure 4.12 Landscape functional nutrient cycling index (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Nutrient cycling indices are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

4.4.3 Erosion control

Vegetation cover (%)

Rehabilitation did not affect the cover of perennial vegetation cover. Cover on rehabilitated sites did not differ significantly from control sites (*t=0.93, p=0.4*) (Fig. 4.13). This was consistent across all farms with no significant effect of rehabilitation on vegetation cover (Fig 4.14). Greylands (*L=33.52, p<0.05*), Morestêr (*L=41.02, p<0.05*) and Witklip (*L=37.85, p<0.05*) rehabilitated sites had significantly lower vegetation cover compared to the benchmark site. Welbedag rehabilitated and control sites showed no significant difference from the benchmark. Greylands control sites did not differ significantly from the benchmark, whereas Witklip (*L=30.31, p<0.05*) and Morestêr (*L=36.17, p<0.05*) control sites had significantly lower vegetation cover.

Figure 4.13 Grouped vegetation cover (%) for control and rehabilitated sites and the benchmark site. Vegetation cover percentages are expressed as means. Error bars denote upper 95% confidence intervals. No significant differences were found.
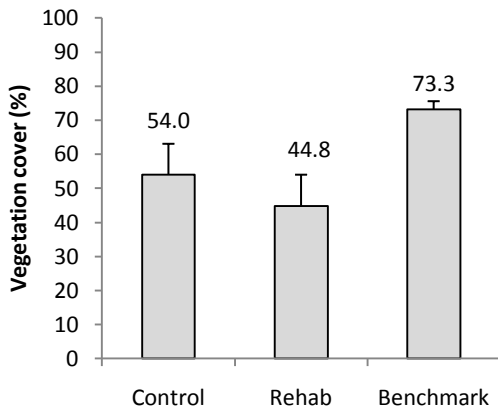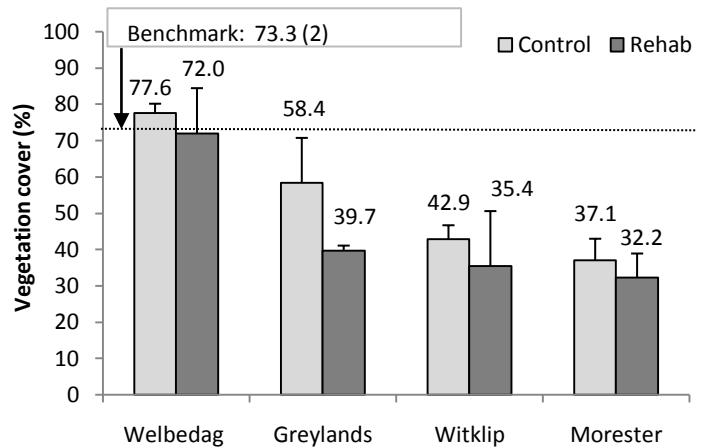

Figure 4.14 Vegetation cover (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Vegetation cover percentages are expressed as means and error bars denote upper 95% confidence intervals. No significance difference was found between rehabilitated and control sites per farm.

Landscape Functional Analysis landscape organisation index % (Cover of resource sinks)

The percent of area classified as actively trapping resources such as water and soil was significantly higher on the rehabilitated sites compared to control sites ($U=3, p<0.001$) (Fig. 4.15). This positive effect of rehabilitation was consistent for three of the farms, Welbedag ($L=1.92, p<0.05$), Greylands ($L=3.5, p<0.05$) and Morestêr ($L=5.6, p<0.05$), which all showed significantly higher landscape organisation indices compared to control sites (Fig. 4.16). There was however, no significant increase of the cover of patches trapping resources at Witklip rehabilitation site compared to the control site. Rehabilitated sites did not differ significantly from the benchmark and only Morestêr control site had significantly lower cover of resource sinks compared to the benchmark ($L=0.36, p<0.05$).

Landscape Functional Analysis soil stability index (%)

Rehabilitation had a significant effect on the landscape functional soil stability index with higher soil stability on rehabilitated sites compared to control sites ($K=-3.08, df=2, p=0.006$). Control sites were significantly lower than the benchmark ($K=3.16, df=2, p=0.005$), whereas rehabilitated sites did not differ significantly from the benchmark site (Fig. 4.17). This pattern was consistent for Welbedag ($U=1035, p=0.037$) and Morestêr ($U=1426, p=0.006$) but not for Greylands and Witklip rehabilitated sites (Fig. 4.18). Morestêr ($U=439, p<0.001$) and Witklip ($U=599, p<0.34$) control sites had significantly lower soil stability indices compared to the benchmark, whereas Welbedag and Greylands did not show significant differences. No rehabilitated sites differed significantly from the benchmark.
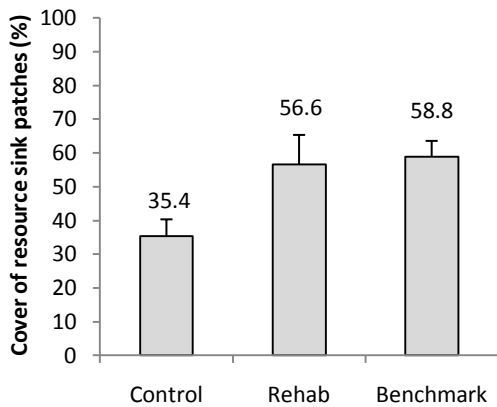
Figure 4.15 Grouped Landscape Functional Analysis landscape organisation index (%) (percent age of the area covered in patches which capture water and soil resources) for control and rehabilitated sites and the benchmark site. The landscape organisation indices are expressed as means. Error bars denote upper 95% confidence intervals.

Figure 4.16 Landscape Functional Analysis landscape organisation index (%) (percentage of the area covered in patches which capture water and soil resources) for control and rehabilitated sites on four ostrich farms in the Little Karoo. The landscape organisation indices are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).
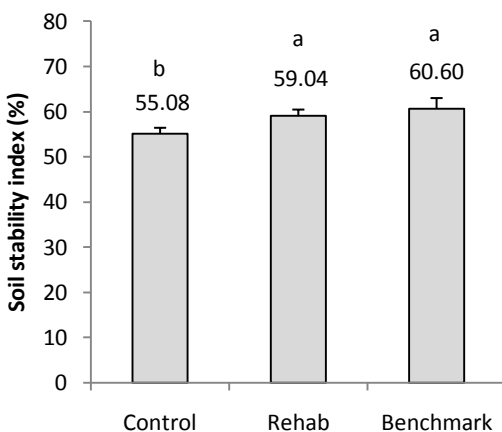
Figure 4.17 Grouped Landscape functional soil stability index (%) for control and rehabilitated sites compared to the benchmark site. Soil stability indices are expressed as means. Error bars denote upper 95% confidence intervals. Values not sharing a common letter are significantly different.
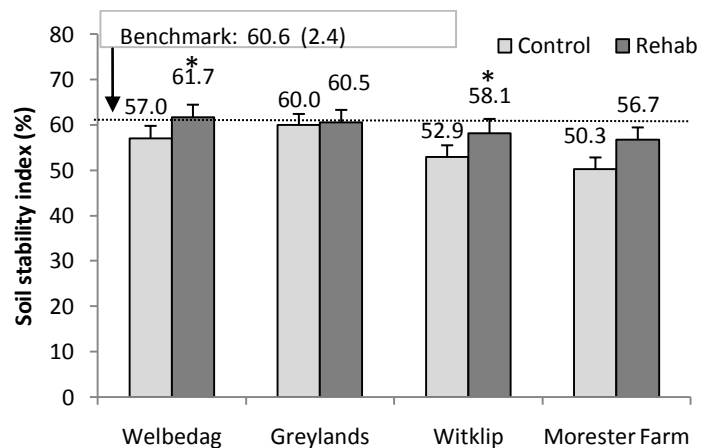
Figure 4.18 Landscape functional soil stability index (%) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Soil stability indices are expressed as means and error bars denote upper 95% confidence intervals. (Significant differences between the rehabilitated and control site per farm are indicated by *p < 0.05; **p < 0.01; ***p < 0.001).

4.4.4 Forage production

Current grazing capacity (LSU/60ha)

Rehabilitation had no significant effect on current grazing capacity ($t=0.36$, $p=0.7$) with control and rehabilitated sites having similar grazing capacities (Fig. 4.19). This was consistent across the four study farms (Fig. 4.20). Witklip ($L=0.02$, $p<0.05$) and Morestêr ($L=0.02$, $p<0.05$) control sites had significantly lower grazing capacities then the benchmark. Greylands and Welbedag control sites did

not differ significantly from the benchmark. Rehabilitated sites also did not differ significantly from the benchmark site.
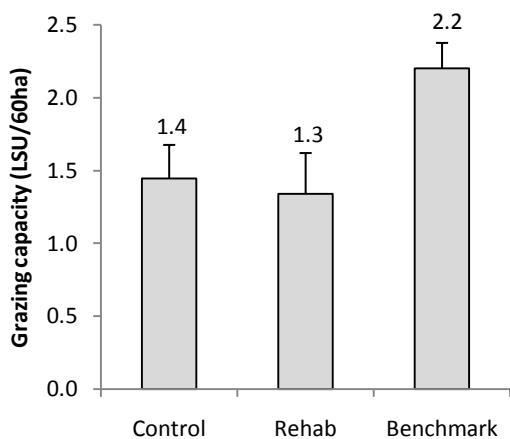
Figure 4.19 Grouped current grazing capacity (LSU/60ha) for control and rehabilitated sites and the benchmark site. Grazing capacities are expressed as means. Error bars denote upper 95% confidence intervals. No significant differences were found.
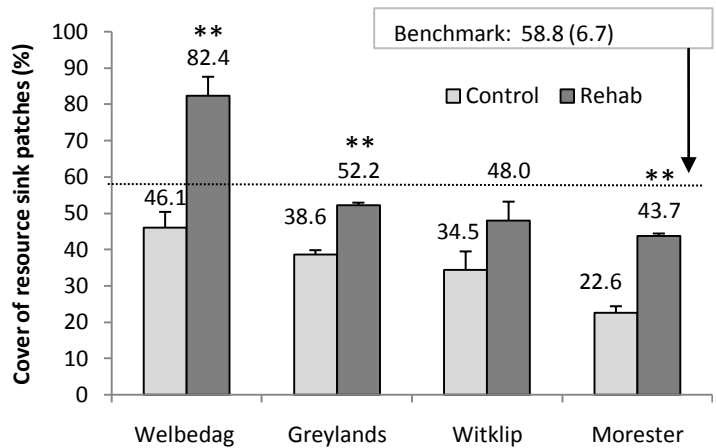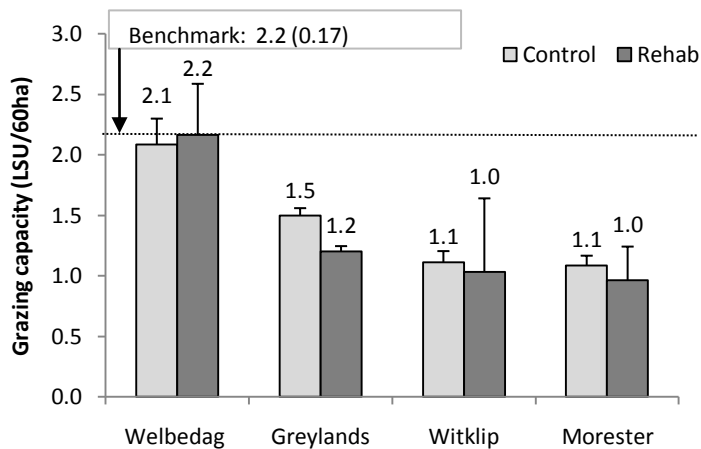

Figure 4.20 Current grazing capacity (LSU/60ha) for control and rehabilitated sites on four ostrich farms in the Little Karoo. Grazing capacities are expressed as means and error bars denote upper 95% confidence intervals. No significance difference was found between rehabilitated and control sites per farm.

4.4.5 Plant diversity

Species richness

Rehabilitation did not significantly increase species richness (Fig. 4.21) (*t=0, p=1*). This was consistent across the study farms (*L=5.5, L=9, L=1.5, L=2, p>0.05*) (Fig. 4.22). There were no significant differences between control and benchmark species richness as well as rehabilitated sites and benchmark species richness.
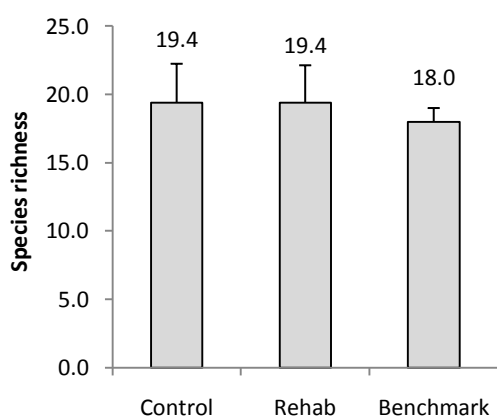
Figure 4.21 Grouped species richness for control and rehabilitated sites and the benchmark site. Species richness is expressed as means. Error bars denote upper 95% confidence intervals. No significant differences were found.
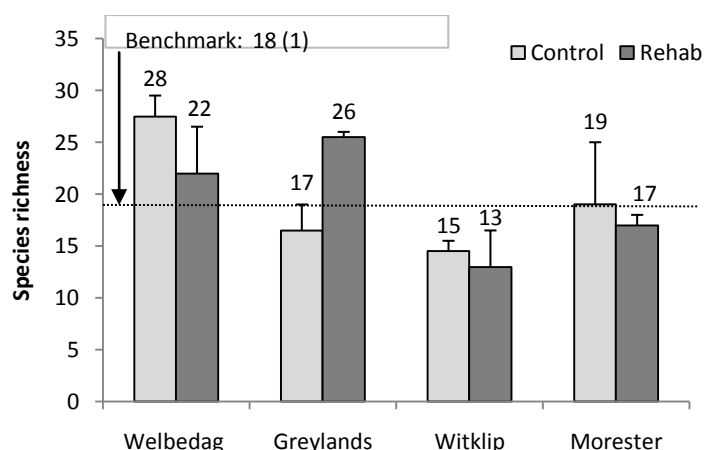
Figure 4.22 Species richness for control and rehabilitated sites on four ostrich farms in the Little Karoo. Species richness is expressed as means and error bars denote upper 95% confidence intervals. No significance difference was found between rehabilitated and control sites per farm.

4.4.6 Soil texture (sand, silt and clay %)

The distribution of sand, silt and clay did not differ between control and rehabilitated sites except for Greylands where the soil on the rehabilitated site had a significantly higher silt content compared to the control site ($U=97, p=0.02$). Soil clay (%) was significantly higher on the benchmark site compared to the Greylands control ($U=29, p=0.04$) and rehabilitated site ($U=76.5, p=0.01$), Morestêr control ($U=25, p=0.002$) and the rehabilitated site ($U=66.5, p=0.003$), Welbedag control ($U=27, p=0.015$) and the rehabilitated site ($U=65, p=0.044$). Greylands rehabilitated site also had significantly higher silt and significantly lower sand compared to the benchmark site ($U= 78, p=0.14; U=89, p=0.035$). Witklip control and rehabilitated site, had significantly higher silt (%) ($U=32, p=0.03; U=57, p=0.001$) and significantly lower sand ($U=34, p=0.04; U=64, p=0.003$) and clay ($U=47, p=0.23; U=78.5, p=0.003$) compared to the benchmark site (Table 4.5).

Table 4.5 Soil texture (clay, silt and sand (%)) (mean (SE)) between rehabilitated, control and the benchmark site.

	Welbedag		Greylands		Witklip		Morestêr		Military base
	Rehab	Control	Rehab	Control	Rehab	Control	Rehab	Control	Benchmark
Sand (%)	95.5(0.4)	94.7(0.8)	90.9(1)	94.4(0.8)	87.8(1.4)	86.2(2.2)	94.2(0.5)	95.2(0.5)	94.2(0.5)
Silt (%)	2.5(0.3)	3.2(0.8)	6.9(0.8)	3.7(0.9)	9.9(1.3)	12(2.1)	4.3(0.5)	3.4(0.5)	3.3(0.6)
Clay (%)	1.8(0.3)	2.2(0.1)	2.3(0.6)	1.9(0.2)	2.3(0.3)	1.8(0.3)	1.5(0.2)	1.3(0.2)	2.5(0.2)

4.4.7 Relationships between empirical measurements and LFA indices

There were only non-significant weak positive correlations between water infiltration rates (ml/s) and the LFA water infiltration index. There were, however, strong significant correlations between the infiltration rate (ml/s) and the LFA nutrient cycling index and LFA soil stability index. There was also a strong significant correlation between the cover of resource sinks (areas that capture nutrients, water and sediment) and the water infiltration rate (ml/s) (Table 4.6).

All three LFA landscape functional indices were correlated with the cover of resources sinks (landscape organisation index). There was however, no correlation between vegetation cover and the cover of resource sinks. There were no significant correlations between soil organic carbon (%) and soil nitrogen (%) with any of the variables measured. The correlation between the nutrient cycling index and lab measured soil nitrogen ($r_s=0.183, p=0.64$) was very weak and non-significant and there was no correlation between soil organic carbon (%) and the nutrient cycling index (%) ($r_s=0, p=1$).

Table 4.6 Spearman rank correlation matrix for variables investigated on all sites (n=9)

	Infiltration rate (mm/s)	Infiltration index (%)	Nutrient cycling index (%)	Soil stability (%)	Vegetation cover (%)	Cover of resource sinks (%)	Carbon (%)	Nitrogen (%)	Sand (%)	Silt (%)
Infiltration index (%)	0.36									
Nutrient cycling index (%)	0.88**	0.7*								
Soil stability (%)	0.85**	0.6	0.87**							
Vegetation cover (%)	0.33	-0.07	0.1	0.47						
Cover of resource sinks (%)	0.82**	0.77*	0.9**	0.9**	0.37					
Carbon (%)	0.17	-0.2	0.0	0.0	0.28	0.02				
Nitrogen (%)	0.43	-0.28	0.18	0.38	0.28	0.03	0.22			
Sand (%)	0.28	-0.39	0.06	0.18	0.38	0.1	0.57	0.21		
Silt (%)	-0.52	0.18	-0.25	-0.42	-0.65	-0.4	-0.57	-0.2	-.89**	
Clay (%)	0.43	0.53	0.42	0.61	0.39	0.62	-0.5	0.13	-0.39	0.04

*Significant correlations are indicated as *$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.*

4.5 Discussion

Rainfall infiltration, resistance to erosion and nutrient cycling are crucial processes for proper functioning of semi-arid areas (Whitford 2002; Maestre *et al.* 2006). Rehabilitation within 18 months on degraded ostrich study farms improved ecosystem services of water flow regulation and erosion control as indicated by increased water infiltration rates (ml/s) and an increase in the cover of areas which capture resources like soil, water and nutrients (resource sink zones). In addition, rehabilitated sites attained higher functional status compared to the control sites as a result of improvements in the three landscape functional indices of water infiltration, nutrient cycling and soil stability (Tongway & Hindley 2004). Lab measured soil organic carbon and total nitrogen, however, did not increase and remained similar at rehabilitated and control sites, indicating that rehabilitation did not affect the rate of litter turn over and decomposition within the soil and therefore, did not increase soil fertility and nutrient availability to plants. Rehabilitation, in addition had no effect on vegetation cover, species richness and grazing capacity. Therefore, results from the quantitative measures suggest that rehabilitation within 18 months on these study farms, despite being successful at increasing water flow regulation and erosion control ecosystem services, did not increase nutrient cycling, forage production and plant diversity ecosystem services.

Study farms might need a longer phase of recovery of ecosystem functions like water infiltration before increases in vegetation cover, species richness and grazing capacity can be attained (Whisenant 1999; Hobbs 2002). Studies evaluating the relationship between community composition

and ecosystem functioning have shown that straightforward relationships are not common (Huston *et al.* 2000; Maestre & Cortina 2004; Hooper *et al.* 2005; Cortina *et al.* 2006). Some authors suggest that the functional status of an ecosystem may limit early stages of establishment of plants and seedlings while the starting condition of a site can also influence the success of rehabilitation efforts (Milton 1994; Milton *et al.* 1994a; Whisenant 1999; Hobbs 2002). Increased water infiltration was observed at all four study farms although it was only significant at Welbedag and Greylands. Welbedag and Greylands were considered to be in much better condition at the start of the rehabilitation trials compared to Witklip and Morestêr which were considered to be far more degraded before rehabilitation was applied to the farms in 2009 (K. Coetzee & S. Milton *in litt.* 2008). It is possible that the starting conditions of these more degraded farms could have reduced the effect of rehabilitation here and therefore hampered the outcome at the rehabilitated sites (Whisenant 1999; Hobbs 2002).

4.5.1 Water flow regulation

Infiltration rates are predominantly influenced by vegetation cover and soil texture. In addition, soil organic carbon and nitrogen is known for stabilising soil aggregates and facilitating infiltration and nutrient cycling (Young & Onstad 1978; Wood *et al.* 1987; Le Bissonnais *et al.* 1995; Gutierrez & Hernandez 1996; Descroix *et al.* 2001; Calvo-Cases *et al.* 2003; Mills & Fey 2003; Descheemaeker *et al.* 2006; O'Farrell *et al.* 2009). In this study, there were no differences in soil texture or vegetation cover between rehabilitated and control sites. Rehabilitation also had no positive effect on soil organic matter (organic carbon and nitrogen) and there were no significant correlations between soil texture, vegetation cover, soil organic carbon and nitrogen with water infiltration rates.

Casenave and Valentin (1992) showed that infiltration capacity depends on surface characteristics such as surface crusts. The presence of physical and/or biological soil crusts alters characteristics of the soil surface and can play a defining role in many ecosystem functions (Belnap & Lange 2001). Soil compaction has been noted as an important reason for slower infiltration rates on soils of degraded rangelands (Gifford & Hawkins 1978; Du Preez & Snyman 1993; Stroosnijder 1996; Mwendera & Saleem 1997; Belnap & Lange 2001; Snyman 2005; Snyman & du Preez 2005; Du Toit *et al.* 2009). Non-biotic soil surface crusts, or physical crusts, which form on compacted soil surfaces, can seal and smooth surfaces, thus decreasing rainfall infiltration and increasing the volume and velocity of water run-off (Belnap & Lange 2001). It is possible that the infiltration rates on rehabilitation sites were influenced positively due to the physical effect of micro-catchments and mulch treatments had on the soil surface.

Digging micro-catchments loosens compacted soil surfaces and soil physical crusts particularly along the micro-catchment walls and micro-catchment floor. Mulching reduces soil physical crusts under the layer of mulch as the wood chip mulch becomes incorporated into the soil through the rough wood chips piercing the soil as well as partly through decomposition. Hanke *et al.* (2011) using a double ring infiltrometer in the winter rainfall Richtersveld region of the Succulent Karoo found an increase in water infiltration within the base of micro-catchment structures. Higher infiltration within micro-catchments was explained by the accumulation of coarse sand and organic material, which was assumed to decrease soil surface compaction. Hanke *et al.* (2011), in addition, ascribed increased infiltration after one year to the loosening of ground surfaces during the planting of shrubs.

Biological soil crusts (biocrusts) are a soil surface community made up of cyanobacteria, lichens, and moss. They are globally widespread and are an important part of the soil community in the Succulent Karoo biome (Belnap 2006; Büdel *et al.* 2009). Biocrusts, which stabilize the soil surface, however, have also been shown to reduce water infiltration (Schmiedel *et al.* 2001). Early successional crusts are composed of filamentous cyanobacteria and are brittle and less than 3 mm thick. Intermediate successional crusts are well-established cyanobacterial crusts up to 3.9 mm thick. If undisturbed, early and intermediate successional cyanobacterial crusts develop into a late successional crust that includes lichens and mosses (Büdel *et al.* 2009). Disturbances such as grazing and trampling, result in a transition from dark, late successional crusts to light, early or intermediate crust types (Grote *et al.* 2010). Within all four degraded ostrich control sites many of the bare soil patches were observed to be covered in a light layer of recovering cyanobacterial species typical of the early successional crusts. In some cases, even thicker layers of darker crusts similar to intermediate crusts were present. While grazing and trampling are known to disturb soil surface biocrusts, micro-catchments and mulch were also observed to have an effect. Digging micro-catchments removes soil along with surface biocrusts. Mulch covers, and therefore, shades existing biocrusts, thus inhibiting photosynthesis. In a study in the Succulent Karoo, soil covered by biocrusts always had the lowest infiltration rates and in most cases exhibited lower infiltration rates when compared to adjacent open soil sites (Schmiedel *et al.* 2010). It is possible that along with a physical reduction in soil compaction and physical soil crusts the reason for an increase in water infiltration on rehabilitated sites was due to the removal of mostly early and some intermediate successional biocrusts. This effect of rehabilitation on biological soil crusts should be investigated as the Succulent Karoo contains the highest cyanobacterial diversity (49 species) occurring in arid and semi-arid regions

worldwide (Büdel *et al.* 2009). These earlier successional crusts protect the soil surface from erosion and can build soil fertility in the future (Eldridge *et al.* 2002; Belnap 2006; Grote *et al.* 2010).

Biocrusts are also one of the indicators of rangeland and soil health used in the Landscape Functional Analysis (Tongway & Hindley 2004). The LFA soil surface feature that subjectively measures biocrust cover is however not used in the calculation of the LFA infiltration index. The soil surface features that are used in the calculation include perennial vegetation cover, litter cover, micro-topography, slake test and soil texture. It is possible that the biocrust cover is indirectly incorporated into the LFA infiltration index through the slake test because non-biotic physical crusts generally disperse in water whereas biocrusts continue to bind the soil during wetting. It was however noted in this study that soil with a high cover of early successional cyanobacterial biocrust was more likely to disperse in water in comparison to intermediate cyanobacterial or later successional lichen or moss covered crusts. Within the LFA method when determining biocrust cover it is based on a score out of 4 with 1=1% or less, 2=1 to 10%, 3=10 to 50% and 4=greater than 50% biocrust cover. Physical crusts are often colonized by biological soil crusts (Belnap & Lange 2001). In soil with heavy physical crusting, the surface morphology of crusts is mostly controlled by soil and chemical characteristics, with the biological components having limited effect (Belnap & Lange 2001). When using the LFA scoring system for measuring biocrust cover an initial cyanobacterial crust type that covers 30% of the soil surface can score equally to a later successional lichen or moss crust type, which covers 30% of the soil surface. The LFA method should put more emphasis on rating the type of crust within the scoring system.

The water infiltration index was significantly higher on rehabilitated sites compared to control at all farms and therefore, did not correlate well with empirical measures of water infiltration rate (ml/s). It is likely that high scores for micro-topography and litter cover LFA soil surface features contributed to the high LFA infiltration indices at rehabilitated sites. Micro-catchments (600 mm in depth) substantially increase surface relief within the rehabilitated areas and mulching aided in increasing litter cover percentages. Only fine particles of mulch were included as litter cover whereas larger chips were included as soil cover (D. Tongway pers. comm. 2010).

4.5.2 Nutrient Cycling

Measures of soil organic carbon and nitrogen content are generally used in the literature to indicate organic matter content (Snyman & du Preez 2005). Nitrogen is considered the most limiting nutrient for plant growth in arid and semi-arid ecosystems (Mazzarino & Bertiller 1999; Snyman & du Preez 2005). Soil organic carbon increases pore spaces and therefore surface area within the soil, which

subsequently retains more water and nutrients (Fynn *et al.* 2009). Loss of soil organic carbon and nitrogen leads to a reduction in soil fertility, land degradation and even desertification (Thurow *et al.* 1986; Reicosky *et al.* 1995; Whitford 1996; Teague *et al.* 1999; Mills & Fey 2003; Snyman & du Preez 2005; Snyman 2005). At this study site soil organic carbon and nitrogen were very low and typical of degraded semi-arid areas and sandy soils (<1%) (Mills & Fey 2003; Snyman & du Preez 2005). Most rehabilitated areas showed similar carbon and nitrogen levels to the control areas and in some cases rehabilitated sites showed a decline in carbon and nitrogen.

Shrubs in the Succulent Karoo have small-scale effects on their abiotic environment by creating fertile islands underneath their canopies (Stock *et al.* 1999). 'Fertile islands 'have been well described in the literature where falling leaf litter accumulates below shrub crowns enriching the topsoil with organic matter and higher course soil fractions (Hook *et al.* 1991). This effect is increased due to the redistribution of top soil material by wind and water, which is trapped below shrub canopies (Schmiedel *et al.* 2010). Rehabilitation had no effect on the vegetation cover and thus did not enhance this 'fertile island' effect. One and a half years, however, is a very short time-frame to expect an increase in vegetation cover with subsequent increases in carbon and nitrogen particularly in a semi-arid area where rainfall is less than 250 ml per annum. Detecting change in karoo vegetation is difficult because of the slow rate of population turnover (Yeaton & Esler 1990). Hanke *et al.* (2011) found no increases in carbon or nitrogen when planting adult plants after one year within the Richtersveld region of the Succulent Karoo.

It has been suggested that disturbed, semi-arid rangelands once retrogressed beyond a threshold of drought resilience will recover only by active intervention (Snyman & du Preez 2005; Milton *et al.* 2004a). Mulch is commonly applied worldwide in rehabilitation attempts to modify poor soil properties. Mulch directly affects soil properties like moisture, temperature, and nutrient availability (Bulmer 2000; Beukes & Cowling 2003; Athy *et al.* 2006; Schmiedel *et al.* 2010). The content of soil organic carbon is determined by losses of organic carbon (through decomposition, erosion of particles and losses through dissolved organic matter) and inputs of organic carbon (manures and plant litter) (Milne & Heimsath 2009). One would expect that the input of wood chip mulch at the rehabilitated sites should have resulted in an increase of soil carbon and decrease in soil nitrogen (Corbin & D'Antonio 2004). Several studies have discussed the effect of carbon additions (primarily sugar, sawdust or mixture of the two and hardwood mulch) as a tool for suppressing alien or weedy species for conservation and horticultural purposes (Gill & Jalota 1996; Sharma *et al.* 1998; Athy *et al.* 2006). Studies hypothesise that the addition of carbon reduces plant-available nitrogen and thereby increase the competitiveness of slower growing indigenous plants over alien or weedy

species (Corbin & D'Antonio 2004). Results have however not been consistent. Some studies have reported that carbon additions provided a competitive advantage to indigenous plants by reducing nitrogen availability (Morgan 1994; Zink & Allen 1998; Blumenthal *et al.* 2003; Averett *et al.* 2004; Perry *et al.* 2004; Prober *et al.* 2005). Others have found that carbon additions inhibited both indigenous and alien plants (McLendon & Redente 1992; Baer *et al.* 2004; Kulmatiski & Beard 2006; Lowe *et al.* 2002). However, it has also been reported that carbon additions did not inhibit either indigenous or alien plants (Wilson & Gerry 1995; Reever-Morghan & Seastedt 1999; Cione *et al.* 2002; Corbin & D'Antonio 2004).

Rehabilitation on the four study farms had no consistent effects on soil nitrogen levels with a significant decrease in nitrogen only at one rehabilitated site. In addition, rehabilitation showed no positive effect on soil organic carbon. Wood chips remain intact for a long period (Athy *et al.* 2006). Athy *et al.* (2006) found that after two years, areas treated with leaf mulch created measurable organic matter inputs, whereas hardwood mulched areas showed no change in soil organic matter. This is simply because leaves decay faster than wood chips (Miltner & Zech 1998). In most arid and semi-arid rangelands, litter and root turnovers are very slow (Whitford *et al.* 1988; Snyman & du Preez 2005). Similarly, to other studies which have investigated the effect of wood chip mulching on soil organic matter, this study confirmed that within one and a half years wood chip mulch had no positive effect on soil organic carbon as well as no consistent effects on soil nitrogen. Most studies have found a decrease in nitrogen or no change in nitrogen in the soil during first two years after woodchip mulching. A reduction in nitrogen under mulch is short-lived (± 2 years) and can lead to an increase in nitrogen in the third year (Holtz *et al.* 2005; Miller & Seastedt 2009). It is possible that as the Little Karoo is a semi-arid system given time nutrient cycling and soil organic matter could increase at these rehabilitated sites in the future.

Micro-catchments have been shown to act as sediment traps, which can enhance nutrient availability at the structures. This has been shown for traditional pits used in Niger in western Africa (Vohland & Barry 2009) as well as in the Anza Borrego Desert State Park in western United States (Bainbridge 2007). A study focussing on soil properties under half moon micro-catchments in Burkina Faso (western Africa) found no effects of structures on soil properties (Vohland & Barry 2009). Equally, this study showed no significant positive effects of rehabilitation on soil organic carbon and nitrogen. It is likely that numerous factors are involved in the response or lack of response of carbon and nitrogen in the soil of the rehabilitated sites (Mills & Fey 2003; Mills & Fey 2004). Factors include, oxygenation of soil through digging the micro-catchments (Tiessen *et al.* 1992; Gregorich *et al.*, 1994), greater microbial activity due to increased soil temperature utilising available nitrogen, and

increased rates of mineralisation of soil organic matter due to light rainfall during the study period (Birch 1958; Mills & Cowling 2010).

The LFA nutrient-cycling index in contrast to empirical measurements showed an increase in nutrient cycling on three rehabilitated sites. Soil surface features used to estimate the LFA nutrient-cycling index are perennial vegetation cover, litter cover, litter origin and degree of decomposition, biocrust cover and micro-topography (Tongway & Hindley 2004). The reason why the LFA nutrient-cycling index indicated a positive effect of rehabilitation on nutrient cycling at the study farms was most probably due to the presence of fine mulch chips, which were washed by rain and blown by wind onto bare areas. This resulted in increased litter cover scores for rehabilitated sites. In addition, trapped transported plant litter had combined with mulch layers on the study sites. Therefore, areas covered with mulch always scored higher for litter cover as well as decomposition. The degree of litter decomposition is based on litter fragment sizes and whether the litter is in close contact with the soil. Litter decomposition, in addition, also scores high if litter fragments are partially buried and if the soil surface under the layer of litter has a dark colour. It is possible, that mulching confused the process of assessing litter cover and decomposition in this study and probably overestimated the degree of decomposition of the woodchips. Micro-catchments created a micro-topography that was visually representative of one conducive for capturing nutrients within the system. This was also one of the reasons for the higher LFA nutrient cycling indices (Tongway & Hindley 2004).

4.5.3 Erosion control

Rainfall interception is one of the main reasons for enhanced infiltration and reduced water run-off (Casermeiro *et al.* 2004; O'Farrell *et al.* 2009; Bartley *et al.* 2010a; Bartley *et al.* 2010b). The only means of reversing the degradation cycle is to introduce management practices that capture and retain water (Whisenant 1999; Tongway & Hindley 2004). Resource sink zones include any type of patch within a landscape that is capable of capturing water runoff and thereby capturing sediments and nutrients (Tongway & Hindley 2004). Previous findings support the idea that an increase in cover of resource sink zones is important for reducing runoff (Ludwig *et al.* 2006). The LFA landscape organisation index is derived by dividing the sum of the length of patch zones i.e. shrub clumps, large branches in contract with the soil, mulched areas and micro-catchment basins by the length of the transect. Landscapes with higher resource sink zones have a higher functional status as they are able to capture sufficient rainfall and organic materials (Tongway & Hindley 2004). Rehabilitated sites were demonstrated to have a higher cover of patches that capture soil, water and nutrients (landscape organisation index). This is indicative of better erosion control in comparison to sites without rehabilitation. The increase in cover of resource sinks at the rehabilitation sites is due to

areas covered in mulch and micro-catchments and not vegetation cover. Vegetation cover remained similar between rehabilitated and control sites despite increases in water infiltration. The cover of resource sinks was highly correlated with the empirically measured water infiltration rate (ml/s). This is an indication of the influence of rehabilitation treatments such as micro-catchments and mulch on water infiltration at these sites. Micro-catchments and mulch, by enhancing water infiltration at a plot scale, as well as reducing the velocity of runoff at the site and collecting rainwater within the structures, have the ability to modify water flow over the landscape (Vohland & Barry 2009). In the short-term, the increase of resource sinks at the rehabilitation sites has not yet increased soil organic carbon, nitrogen or vegetation cover. It could, however, depending on rainfall conditions, result in noteworthy effects in the long-term. Micro-catchments and mulched areas create patches in which more water is available for plant growth (Hanke *et al.* 2011). Furthermore, micro-catchments in the Succulent Karoo could effectively harvest surface runoff and mitigate water erosion caused by heavier rains and particularly by thunderstorms. Mulching also has the potential to accumulate rainwater in the soil underneath protruding woodchip mulch pieces and this could capture wind driven rainfall droplets when rainfall intensities are low (Hanke *et al.* 2011).

The LFA soil stability index was only significantly higher at two of the rehabilitated sites. In addition, most LFA soil stability indices only differed by *c.* 4%. In all likelihood, this was due to high soil cover (wood chip mulch) on the rehabilitated sites versus high biocrust cover on the control sites, both of which contribute to high soil stability.

4.5.4 The benchmark site

In order to test whether rehabilitation has been successful it is suggested that the rehabilitated site should be compared with a benchmark reference site. Benchmark ecosystems are those not subjected to the environmental degradation that the rehabilitation was intended to redress and are supposed to represent the desired end-point of rehabilitation (Rey Benayas *et al.* 2009). Apart from having higher infiltration rates, the military base generally proved inefficient as a benchmark site for this rehabilitation study. The benchmark was badly degraded through military base activities in conjunction with donkey grazing. Water infiltration was higher on the benchmark site, although not significantly higher when compared to two of the rehabilitated sites. High infiltration rates on the benchmark site probably occurred as a result of broken soil surfaces caused by light donkey grazing as well as higher vegetation cover. Light trampling by grazing animals has been shown to loosen the topsoil sufficiently to increase the infiltration rates in the Nama Karoo biome (Du Toit *et al.* 2009). Du Toit *et al.* (2009) found that no grazing increased bulk density and soil compaction and decreased the infiltration rate. The military base is 12 000 ha and there were 300 donkeys on the base. Possibly

local conservation authorities could provide added information and assistance for the purposes of conservation where possible on military base land in order to sustain the land for future use. Only 2% of the Succulent Karoo lowland vegetation communities are in a relatively natural state with 98% either moderately or severely degraded (Le Maitre *et al.* 2009; Thompson *et al.* 2009). Therefore as experienced in this study, finding a benchmark site suitable for comparison for further rehabilitation efforts in the Little Karoo might be challenging.

4.5.5 The applicability of Landscape Functional Analysis in this study

One of the central objectives of the Landscape Function Analysis methodology is to provide a framework for the comparison of similar sites showing potential degradation or recovery (Tongway & Hindley 2004). Despite recent experimental work indicating that the infiltration index is a suitable descriptor of infiltration rate (Maestre *et al.* 2006) in this study, there was no correlation between the infiltration index and empirically measured water infiltration. There were, however, strong correlations between empirically measured infiltration rates and the other two indices of nutrient cycling and soil stability. This indicates that certain soil surface features used for the calculation of these two indices are important for estimating water infiltration rates on these study farms. Surface features such as soil cover, biocrust cover, crust brokenness and deposited materials are not used to calculate the infiltration index (Tongway & Hindley 2004). Biocrusts are known to have an effect on water infiltration (Belnap 2006; Schmiedel *et al.* 2010) and fractured soil surface crusts in addition influence infiltration rates (Du Toit *et al.* 2009). It is possible that the infiltration index overestimated infiltration rates on Witklip and Morestêr.

The strong correlations between the LFA soil stability index and nutrient cycling index with water infiltration rate (ml/s) suggests that these measurements could act as a surrogate for water infiltration rates. However, when individual ordinal scores are summed to form these indices, reporting and testing means and variance terms is questionable because the statistical properties are not known (Maestre *et al.* 2006). Although non-parametric tests were used to evaluate site differences in LFA indices and to correlate them with the infiltration rate results from statistical comparisons involving these indices should be interpreted with caution (Maestre *et al.* 2006). The landscape organisation index is based on continuous measurements and therefore the strong correlation between the cover of resource sinks (landscape organisation index) and water infiltration rates does suggest that the cover of rehabilitation treatments such as micro-catchments and mulch as well as shrub cover influenced the infiltration rate on the farms.

Studies using Landscape Functional Analysis have been carried out in South Africa mainly to measure the effects of degradation and not recovery. Similar to this study, other studies have found low levels of concurrence between the Landscape Functional Analysis Indices and empirical measurements. Petersen *et al.* (2004a) noted that the LFA technique was unable to draw a distinction between communal and commercial grazing strategies in the Succulent Karoo Biome. O'Farrell and Donaldson (unpublished) found that the LFA infiltration index highlighted differences in rainfall infiltration between study sites, which rainfall simulations did not detect. Palmer *et al.* (2001) found that the soil surface assessment technique was ineffective and described it as insufficiently sensitive to measure shifts in rangeland condition, which had been detected using vegetation field surveys and remote sensing methods. This study indicates, that this technique might be more useful if the eleven soil surface features assessed were analysed separately without summarizing them into the three soil indices of water infiltration, soil stability and nutrient cycling. For example, biocrusts are known to effect infiltration rate. However, they are left out of the calculation of the LFA infiltration index. Using these eleven-soil surface features one can postulate the effects on water infiltration rates, soil stability and nutrient cycling.

Landscape Function Analysis (LFA) is provided as a technique to rapidly determine broad biogeochemical processes occurring at the soil surface (Tongway & Hindley 2004). However, it has constraints, as it is an essentially subjective method because it relies to some extent on the surveyor's skills and experience. It is possible that with sufficient training courses conservation officials and extension officers could use the technique. However, due to the lack of congruence between empirical measurements and the LFA functionality indices it is not recommended for measuring the success of rehabilitation treatments in the short-term. Furthermore, it is not recommended as an efficient easy technique for private landowners. Although it might be less time-consuming than empirical techniques, it still involves many person-hours to collect field data and interpreting the results can be tricky and relatively complex. It is possible that LFA will be more effective for long-term monitoring of rehabilitation success of permanent sites rather than site comparisons (O'Farrell & Donaldson unpublished). The LFA landscape organisation index (total patch length / transect length), however, has proved quite successful and was highly correlated with water infiltration rates. This suggests that this index from the LFA technique could be used as a quick and efficient way to determine the broad functional status of the study sites. Other techniques, however, should also be considered for use in this area. The Quick Rangeland Health Assessment method is a semi-subjective, multi-criterion method developed for the Karoo. It was initially developed for rangeland managers to monitor veld condition change (Milton & Dean 1996; Milton *et al.* 1998) and has been

proven to have reasonable accuracy for assessing veld condition in the Succulent Karoo biome by a single user (Cupido 2005).

4.5.6 Payment for ostrich rangeland ecosystem services

Benefits associated with rehabilitation include rehabilitation of ecosystem services such as carbon sequestration, herbivore browse, flood control, biodiversity, control of soil erosion, and the provision of jobs in economically depressed rural areas (Mills & Cowling 2006). Carbon credits, poverty relief funds and biodiversity funds are all sources of funding for rehabilitation (Mills & Cowling 2006). The benefits obtained from rehabilitation in this study include increased erosion control and water flow regulation as well as potential socio economic benefits relating to the labour used for collecting and sowing seed, digging micro-catchments and applying mulch. However, within two years of the interventions the interventions had no significant effect on plant diversity, grazing capacity and nutrient cycling. Once water infiltration and erosion control are fully functional it is possible that other ecosystem services will follow suit. However, despite the success of rehabilitation at increasing certain ecosystem services, the high cost of active rehabilitation, which includes costs like R2.47/m^2 for mulching, R1.54/m^2 for micro-catchments with mulch and R0.37/m^2 for sowing indigenous plant species, will most probably inhibit its widespread implementation. Rehabilitation did not have an effect on grazing capacity and the rehabilitated sites in their current state cannot be farmed profitably.

High costs of rehabilitation programmes aimed at increasing ecosystem services around the world are the norm. China, for instance, has invested over 700 billion Yuan (1 Rand = 1.05 Yuan) in ecosystem service payments for active rehabilitation projects between 1998–2010 (Liu *et al.* 2008). Most of this money came from central and local governments and market-based mechanisms have not been used (Liu *et al.* 2008). In the Little Karoo, farmers could save on cost and do the work themselves using physical, natural and human resources already available on the farm. However, it is likely that rehabilitation is not a top priority for ostrich farmers. Incentives such as Payments for Ecosystem Services (PES) can be used to promote the uptake and adoption of environmentally friendly farming systems and rehabilitating lands. There is a potential for the development of PES between landowners and tourism operators, which can be coupled with product certification schemes, such as eco-labelling, sustainable grazing and meat production (Murray 2007; Le Maitre *et al.* 2009). The most successful PES schemes for the Little Karoo in the short-term would include government funded poverty alleviation programmes. Unemployment rates are standing at 40% in the Little Karoo (Murray 2007; SAOBC 2009) and labour intensive projects would act as a social and

financial boost for the local community. Outside funding could also alleviate financial constraints faced by landowners (Turpie *et al.* 2008).

4.6 Conclusion

It has been suggested that if important ecosystem services and benefits derived from rehabilitation can be demonstrated at a farm scale this could motivate for their rehabilitation (Edwards & Abivardi 1998; O'Farrell *et al.* 2009). This study demonstrated that active rehabilitation of degraded ostrich farms in the Succulent Karoo lowlands in the Little Karoo increased rainfall infiltration and the cover of resource patches, which capture rather than lose water, nutrients and soils. Empirical evidence has proved more useful in determining the effect of rehabilitation on landscape functioning and ecosystem services than the Landscape Functional Analysis Indices (Tongway & Hindley 2004). The LFA landscape organisation index (length of patches divided by total transect length) is based on continuous data and was found useful in characterising the sites in terms of resource sink zones and determining the effect of rehabilitation on erosion control. Before incorporating the LFA methodology into routine rehabilitation monitoring practices future studies are needed to validate the LFA indices.

It takes decades for degraded rangelands to recover (Hoffman & Rohde 2007, Botha *et al.* 2008). This study has shown that using active rehabilitation measures in combination with improved management strategies (rest from grazing) may induce recovery of certain landscape functions and ecosystem services within more realistically suited time scales. Although erosion control and water flow regulation benefits of rehabilitation have been demonstrated, one and a half years have not been sufficient for rehabilitation to positively affect nutrient cycling (soil organic carbon and nitrogen), forage production and species richness. It is possible that positive changes in infiltration and erosion control processes will facilitate rehabilitation success in the future. Long term monitoring, however, is needed to validate these assumptions and depending on rainfall and stocking densities, the recovery may be extended. In order for active rehabilitation to be effectively implemented within the degraded succulent Karoo lowlands of the little Karoo, market based mechanisms like Payments for Ecosystem services should be explored with assistance and support from the central government.

Chapter 5: Synthesis and conclusion

5.1 Motivation for the study

There is a growing awareness amongst conservationists of the need to focus efforts outside protected areas and on private and communal lands in order to conserve species and landscapes (Secretariat of the Convention on Biological Diversity 2010a). A fundamental challenge for conservationists, landowners and the public is learning how to strike the balance between the conversion of natural areas for sustained economic opportunity and the conservation of natural areas for the provisioning of ecosystem services (Carpenter *et al.* 2009; Farley & Costanza 2010). Ecosystem service research is conceptually a relative new field (O'Farrell 2005), particularly in arid and semi-arid ecosystems (O'Farrell 2005; Aronson *et al.* 2010a). However, highlighting ecosystem services that support human livelihoods attained through the rehabilitation of degraded lands has shown the potential to motivate for conservation on private lands (MA 2005; O'Farrell *et al.* 2009; Rey Benayas *et al.* 2009; Aronson *et al.* 2010a).

The South African Ostrich Business Chamber through their Long term Biodiversity Management Strategy has motivated for their members to switch to environmentally friendly farming systems and resting or actively rehabilitating degraded lands in order to ensure the long-term sustainability of the ostrich industry, conserve biodiversity and improve productivity (SAOBC 2009). The Little Karoo region is the core ostrich production region in the country and, therefore, the area that has been most affected by unsustainable management practises (Cupido 2005; Murray 2007). Degraded semi-arid rangeland states are resistant to change, relatively stable, provide little forage to livestock and rarely recover within timeframes relevant to land owners (Milton & Hoffman 1994; Milton & Dean 1995; Milton *et al.* 1994a; Wiegand & Milton 1996; Valone *et al.* 2002; Kinyua *et al.* 2009; Seymour *et al.* 2010). Conservation practises within these degraded agricultural landscapes could be implemented if the value in adopting a new farming system and rehabilitating degraded lands was made clear to farmers. Furthermore, demonstrating values of rehabilitation to society could facilitate the establishment of incentive structures like Payment for Ecosystem Services within the Little Karoo. This could in turn provide a financial incentive for landowners to rest and rehabilitate degraded lands.

The main aim of this study was to evaluate the short-term (*c.* one year to one-and-a-half years) potential of rehabilitation for increasing ecosystem services on degraded ostrich rangeland. Such an evaluation would provide the South African Ostrich Business Chamber with information on rehabilitation effects and success. This knowledge could be used to motivate ostrich farmers to keep

their land in a natural state and to rehabilitate degraded areas where necessary. Furthermore, the study was conceptualised to present information on the short-term effects of rehabilitation on ecosystem services. This was done to stimulate conversations around developing and implementing Payment for Ecosystem Service schemes, which would in turn incentivise farmers to rehabilitate degraded ostrich rangelands.

This final chapter of the thesis provides a brief synthesis of the key findings of the study. It describes the study's limitations as well as future research priorities for the science and practice of rehabilitation and ecosystem services in the Little Karoo. Key findings are ordered according to the four main objectives of the study that were set out in chapter 1:

(1) To evaluate the relative success of three different rehabilitation methods for returning indigenous vegetation and grazing services to a severely degraded ostrich camp.

(2) To determine the effects of rehabilitation on landscape functioning and ecosystems services on four disused and degraded ostrich camps.

(3) To assess the ability and usefulness of the Landscape Function Analysis technique for quantifying differences in water and organic matter retention within the Succulent Karoo lowland vegetation communities.

(4) To discuss the implications of results for management and rehabilitation of semi-arid degraded Succulent Karoo vegetation.

5.2 Key Findings

5.2.1 *The success of seed, mulch, ripping and micro-catchment rehabilitation treatments for returning indigenous vegetation and grazing services to a severely degraded ostrich camp*

This study found that micro-catchment, seed and mulch rehabilitation treatments all have short-term benefits for creating opportunities for further long-term recovery. Ripping (R0.13 per m^2 / R1 300 per ha) which was the least costly method to apply showed no positive results in the short-term. In terms of cost, this was followed by sowing seed (R0.37 per m^2 / R3 700 per ha), micro-catchments (R1.54 per m^2 / R15 400 per ha) and mulching (R2.47 per m^2 / R24 700 per ha). In the short-term (one-year), micro-catchments were the most successful rehabilitation treatment. Micro-catchments although being more expensive than ripping and sowing seed, significantly increased palatable plant density and palatable species richness. Seed treatments significantly increased palatable species richness. Mulching altered the ratio of palatable to less palatable plants from 1:12 to 1:3 on the degraded ostrich farm. Mulch created a barrier for seed germination and physically damaged highly

abundant perennial less palatable and annual species. Mulch however, did not reduce species richness. This therefore, increased the species evenness within plant communities and as a result increased the Shannon Index of plant diversity. Certain woody shrubs were capable of surviving or germinating and emerging from beneath the woodchips and therefore attained higher densities. In the long-term, this could create a competitive advantage for late successional species (McLendon & Redente 1992; Paschke *et al.* 2000; Blumenthal *et al.* 2003). Despite increases in palatable plant density, species richness and plant diversity, rehabilitation in the short-term did not present a return on investment for the private landowner. This is because micro-catchments, ripping or seed treatments did not provide tangible benefits in terms of increasing vegetation cover and grazing capacity (LSU/ha). Furthermore, mulching actually reduced less palatable vegetation cover and thereby decreased the grazing capacity (LSU/ha).

Studies in other parts of the Succulent Karoo biome have reported similar findings, where only two out of the six rehabilitation treatments were shown to have positive influences on perennial vegetation cover (Anderson *et al.* 2004; Simons & Allsopp 2007; Schmiedel *et al.* 2010). Treatments included the scattering of quartz stones on disturbed quartz fields (Schmiedel *et al.* 2010) and the planting of adult plants with functional roles such as facilitation of other plants or reduction of soil erosion (Anderson *et al.* 2004; Schmiedel *et al.* 2010). These functional plants act as ecosystem engineers and facilitate seedling establishment by reducing high temperatures near the soil surface as well as providing micro-habitats with a higher soil nutrient content (Turner *et al.* 1966; Humberto *et al.* 1996). In addition, Herling *et al.* (2009) assessed the landholder rehabilitation costs for Karoo vegetation, finding these to be in a range of R4 000 - R20 000 per hectare, making short-term rehabilitation not financially viable. The short-term value, however of an increase in palatable species richness, plant density and plant diversity as shown in this study is hard to determine and probably will be based on personal opinion and intrinsic value rather than an actual rand value (Farley & Costanza 2010; Ribaudo *et al.* 2010).

Almost all species present on heavily overgrazed Succulent Karoo rangelands are less palatable and the return interval for palatable plant recovery is more than two decades (Seymour *et al.* 2010). Therefore, increases in perennial palatable plant density and species richness within micro-catchments and seed treatments may be a satisfactory achievement over the one-year period. It implies an improvement of the veld, as more forage plants are available for grazing. In addition, increases in palatable plant density, species richness and plant diversity could provide future benefits. Micro-catchments, seed and mulch treatments have shown the potential in the short-term to enable the return of grazing capacity and system resilience. System resilience is important especially

in semi-arid areas where perturbations are a common phenomenon. Longer-term research is needed to substantiate these short-term findings.

5.2.2 The effect of rehabilitation on landscape functioning and ecosystem services on four disused and degraded ostrich camps

Semi-arid African rangelands are often characterized by threshold dynamics and alternate degraded stable states (Ellis & Swift 1988; Milton *et al.* 1994a) that rarely recover within timeframes relevant to land managers (Milton & Hoffman 1994; Milton & Dean 1995; Milton *et al.* 1994a). This study has shown that using active rehabilitation in combination with improved management strategies (rest from grazing) may induce recovery of certain ecosystem services within shorter and practically relevant time scales of 18 months. In this study, the short-term rehabilitation benefits included increased erosion control and water flow regulation as well as socio-economic benefits relating to the employment opportunities created by the rehabilitation application. Rehabilitation, using a combination of micro-catchments, mulching and seed treatments, enhanced water flow regulation as indicated by improved water infiltration rates (ml/s). Rehabilitation also increased erosion control as indicated by an increase in the percentage cover of resource sink zones that capture water, soil and nutrients. There was however, no effect of rehabilitation on plant diversity, grazing capacity and nutrient cycling.

Micro-catchments and mulching increased the percentage cover of resource sinks (LFA landscape organisation index) at the rehabilitated sites, which is indicative of higher erosion control (Tongway & Hindley 2004; Vohland & Barry 2009). The percentage of resource sinks was further highly correlated with empirical measurements of water infiltration rates (ml/s). Infiltration rates on rehabilitation sites were assumed to be positively influenced due to the physical effect of micro-catchments and mulch treatments on non-biotic soil surface crusts (physical crusts) and possibly recovering biological soil crusts (biocrusts). Both physical and biological soil crusts have been shown to reduce rainwater infiltration (Galun *et al.* 1982; Eldridge *et al.*, 2000; Kidron *et al.*, 2000; Belnap & Lange 2001; Warren 2001; Schmiedel *et al.* 2010; Xiao *et al.* 2010). However, most biocrusts on disturbed land are composed mainly of a thin layer of recovering cyanobacterial species, which have formed on top of the soil physical crust. In soil with heavy physical crusting, the surface morphology of the crust is mostly controlled by soil and chemical characteristics, with the biological components having limited effect (Belnap & Lange 2001). It is therefore possible, that increased infiltration rates at rehabilitated sites were mainly due the reduction of the soil physical crust and not biocrusts. This, however, needs further investigation.

Rehabilitation, despite increasing certain ecosystem services, did not have an effect on grazing capacity and therefore the four rehabilitated sites in their current state cannot be profitably farmed. In addition, after a year and half, rehabilitation has not been sufficient for increasing nutrient cycling (soil organic carbon and nitrogen) and plant diversity (species richness) ecosystem services. Once water infiltration and erosion control are fully functional it is possible that other ecosystem services will follow suit. For example, increased water infiltration rates on rehabilitated sites may result in positive feedbacks, which could result in increases in perennial vegetation cover and therefore increases in nutrient cycling and soil organic matter.

It is estimated that each year *c.* 75 billion metric tonnes of soil is lost around the world. This has an associated cost of US$400 billion (Myers 1993; Pimentel *et al.* 1995). In South Africa, *c.* three tonnes of topsoil per hectare is lost annually (Yeld 1993). Top soil loss promotes the formation of gullies and affects areas downwind and downstream where sediments are deposited. Silt and sand washed downstream can clog reservoirs and canals with sediments (Le Maitre *et al.* 2007; O'Farrell *et al.* 2009). Although there has been no increase in grazing capacity, there has, however, been an increase in water infiltration and erosion control on the rehabilitated sites, which does suggest that less water and soil are being lost. Water flow regulation and erosion control benefits could motivate private landowners to rest and rehabilitate certain denuded areas. However, despite the success of rehabilitation at increasing certain ecosystem services, the short-term high cost of active rehabilitation, which includes costs like R 2.47/m^2 for mulching, R 1.54/m^2 for micro-catchments with mulch and R 0.37/m^2 for sowing indigenous plant species, will most probably inhibit its widespread implementation by farmers. However, as accrued water flow regulation and erosion control benefits affect both the private landowner and the greater populace it is possible that this could motivate the provision of fiscal funds from governing bodies for rehabilitation on private lands.

5.2.3 The ability and usefulness of the Landscape Function Analysis technique for quantifying differences in water and organic matter retention within the Succulent Karoo lowland vegetation communities

Empirical evidence proved more useful in determining the effect of rehabilitation on landscape functioning and ecosystem services than the Landscape Functional Analysis Indices (Tongway & Hindley 2004). The LFA indices, which are based on subjectively scored ordinal data, were not consistent with empirical results and were relatively complex to interpret. The LFA landscape organisation index (LOI) (length of patches divided by total transect length), however, is based on continuous data and was found useful in characterising the sites in terms of resource sink zones and

determining the effect of rehabilitation on erosion control. The LOI was highly correlated with empirical measurements of water infiltration (ml/s) and could, therefore, provide a quick and efficient way to determine the broad functional status of the study sites in terms of erosion control and water infiltration. The other three LFA indices of water infiltration, nutrient cycling and soil stability showed inconsistent results and did not prove useful for quantifying differences in water and organic matter retention on the study farms.

Other studies, which used and tested the abilities of the LFA technique in South Africa have remarked on the unsuitability of the LFA for detecting changes in landscape functional attributes within the Succulent Karoo biome (Petersen *et al.* 2004a), Nama Karoo biome (O'Farrell & Donaldson unpublished) and within Valley Bushveld vegetation in the Eastern Cape (Palmer *et al.* 2001). Furthermore, some studies that used the LFA methods within South African landscapes did not include the data in the published results (Simons 2005; Allsopp & Simons 2007). Although the LFA approach might be a little quicker to conduct than empirical techniques, it is still time consuming and relies to some degree on the skills and knowledge of the person conducting the survey. It is therefore not recommended for extension officers and landowners on degraded ostrich farms. It is possible that the LFA will be more effective for long-term monitoring of rehabilitation success of permanent sites rather than site comparisons (O'Farrell & Donaldson unpublished). Before incorporating the LFA methodology into routine rehabilitation monitoring practices future studies are needed to validate the LFA indices. Furthermore, other techniques should also be considered for use in this area such as the Quick Rangeland Health Assessment method (Milton & Dean 1996; Milton *et al.* 1998; Cupido 2005).

5.2.4 Management and rehabilitation implications for semi-arid degraded Succulent Karoo vegetation in the Little Karoo

In this study, active rehabilitation has increased water infiltration, erosion control, palatable plant density and species richness in practically relevant timescales of one and a half years. Unfortunately, these benefits might not be tangible to landowners or society as of yet, due to the small-scale nature of rehabilitation application (four degraded farms). Furthermore, the willingness of private landowners to engage and invest in rehabilitation depends on the carrying capacity of their land and the resulting ecosystem service rewards (Danckwerts & Marais 1989; Milton *et al.* 1994a). In this regard, rehabilitation had no positive effect on grazing capacity or vegetation cover in the short-term.

It is possible, however, that the benefits from rehabilitation highlighted in this study will encourage private landowners to undertake rehabilitation projects on their own land for economic reasons or

even due to a sense of stewardship (Holl & Howarth 2000). Landowners are encouraged to use resources available on site to reduce costs. Seed can be collected on site, in areas which still harbour palatable species. Seed can be sown within micro-catchments within a thin layer of mulch. Seeding alone is not recommended (Milton & Dean 1995; Visser *et al.* 2004). Large winged seed and grass seed should be broadcast within micro-catchment treatments. Very small seeded succulent shrubs such as Mesembryanthemaceae do not appear to establish successfully. Instead, adult plants of succulent species from less degraded areas should be transplanted to degraded sites (Schmiedel *et al.* 2010). These plants, although mostly not palatable act as ecosystem engineers and facilitate seedling establishment (Turner *et al.* 1966; Humberto *et al.* 1996). Caution should be taken to allow for sufficient gaps between transplanted plants as established perennial plants have been shown to out-compete one another and seedlings in certain Karoo environments (Milton 1994, 1995b). Species that have shown to be successful as transplants within the Succulent Karoo biome include *Brownanthus pseudoschlichtianus*, *Cephalophyllum spissum* and *Cephalophyllum inaequale* (Anderson *et al.* 2004; Schmiedel *et al.* 2010). Furthermore, certain species have been successfully established from cuttings like *Portulacaria afra* (Mills & Cowling 2006; Mills & Cowling 2010).

Landowners could save on cost and rehabilitate degraded lands using physical, natural and human resources already available on the farm. All options however cost money and without initial incentives or highly effective communications among local conservationists, the South African Ostrich Business Chamber and the ostrich farmers, it is unlikely that farmers will take up the challenge to firstly make the switch to environmentally friendly farming systems and secondly to rest their land and thirdly to spend money on rehabilitation. If ecological systems are viewed as public property, the responsible party should pay for rehabilitation. However, this has been shown as ineffective and difficult to apply for numerous reasons (Salzman 2005; Kemkes *et al.* 2010). A stronger motivator in all likelihood for a switch to new environmentally friendly farming practises and resting or possibly rehabilitating lands is the development and implementation of Payment for Ecosystem Services in the Little Karoo (Turpie *et al.* 2008; Ribaudo *et al.* 2010). Funding for rehabilitation projects in the Little Karoo should be sourced from a combination of public and private sources.

5.3 Conservation and Payment for Ecosystem Services in the Little Karoo

The carrying capacity of the Succulent Karoo lowlands is generally quite low (1 LSU/60 ha) and even in its intact state cannot be sustainably farmed while making a profit from the land (Cupido 2005). This highlights the importance long-term biodiversity management strategy of the South

African Ostrich Industry, which proposes a switch from flock breeding to pen breeding as one of its key strategic initiatives. The SAOBC recognises the low carrying capacity of the lands and therefore believes that an industry wide adoption of the pen-based ostrich breeding system is appropriate. Alternatively, strictly environmentally regulated and monitored flock breeding farming systems involving trade-offs between ostrich farming on certain parts of the land and conservation on other parts is also being considered. The latter two options could provide a "win-win" solution for conservationists and farmers if it is possible to implement them on the large scale (SAOBC 2009).

It is likely that a PES similar to the 'working for wetlands/water' community programmes funded through poverty alleviation money could be the most successful scheme for the Little Karoo. These public-funded poverty relief programs that restore hydrological function set up as Payment for Ecosystem Services projects have shown potential in South Africa (Turpie *et al.* 2008). In an area where unemployment rates are high, the importance of community involvement in rehabilitation projects cannot be underestimated in terms of education and developing a sense of stewardship (Holloran 1996; Geist & Galatowitsch 1999; Turpie *et al.* 2008). Other PES schemes that have been successfully implemented in other countries include emissions trading, mitigation banking, eco-labelling, water quality trading, wetland mitigation, carbon cap-and-trade, over-the-counter carbon and fee hunting (Ribaudo *et al.* 2010). Specifically for the Little Karoo a PES scheme could be developed between landowners and tourism operators, which could compensate farmers for tourism appropriate management approaches. Production certification for sustainable grazing and meat production could also be explored (Murray 2007; Le Maitre *et al.* 2009). Unlike other biomes in South Africa, from an ecosystem services perspective, the Succulent Karoo biome is characterized by both the lack of dominance by a single ecosystem service and a general scarcity of services in the region (Le Maitre *et al.* 2007; Egoh *et al.* 2010; O'Farrell *et al.* 2010). This highlights the importance of bundling different services (water flow regulation, erosion control, biodiversity) in order to motivate for PES schemes (Farley & Costanza 2010; Farley *et al.* 2010; Wendland *et al.* 2010). This study provides relevant and important information for policy makers and farmers in order to stimulate a system where Payment for Ecosystem Services, monetary incentives, government subsidisation and poverty relief projects can be further investigated. These payment systems might be necessary in order to protect biodiversity, safe guard ecosystem services and enhance socio economics in the Little Karoo.

5.4 Future research priorities

A major advantage of this study is that it provides the crucial baseline data for follow up studies.

Most rehabilitation studies lack baseline data and this seriously hampers results and recommendations (Wagner *et al.* 2008; Deri *et al.* 2009; Herling *et al.* 2009). Rehabilitated and control sites on the study farms can be re-surveyed in future providing extremely valuable information on the long-term success of rehabilitation. Continued monitoring will show whether enhanced emergence of palatable perennial plants in micro-catchments and reduced competition in mulched areas will support or inhibit the recovery of other key perennial shrubs. These data will provide insights into important competition and facilitation processes among different plant strategy types and contribute towards an understanding of ecological processes within these Succulent Karoo lowlands. Ripping rehabilitation treatments showed no effects in the short-term, long-term research is needed to determine whether ripping has positive effects in the future (Kinyua *et al.* 2009). Monitoring control sites will aid in identifying and clarifying the appropriate resting periods required for vegetation recovery. This information is necessary for the development of appropriate and effective rangeland management strategies (Schmiedel *et al.* 2010).

Farmers in the Little Karoo already produce a wide variety of ecosystem services that are valued by society. Even in a degraded state farms produce ecosystem services that would be lost if these farms were completely transformed by development. Research should focus on quantifying ecosystem services like biodiversity, cultural information, tourism and temperature, flood and dust control (Ribaudo *et al.* 2010). This could possibly motivate for a fiscal framework in support of degraded land rehabilitation.

The possible negative effect of rehabilitation on recovering biocrusts should be investigated. Biocrusts promote the accumulation of organic carbon and nitrogen and increase soil stability (Grote *et al.* 2010). Furthermore, the Succulent Karoo has the highest diversity of cyanobacterial biocrust species in the world. It is of importance to understand the adverse effects of rehabilitation on recovering biocrust species diversity and the ecosystem services that these recovering biocrusts might be providing (Büdel *et al.* 2009).

Rehabilitation within the Succulent Karoo biome has generally not been effective at increasing perennial vegetation cover or grazing capacity. One method that has shown potential includes planting plants that function as seed and silt traps in the environment (Schmiedel *et al.* 2010). Transplanting adult plants from refuge patches of vegetation to degraded bare areas should be investigated as a method to increase vegetation cover and thus grazing capacity within degraded ostrich camps.

Woodchip applications are commonly used in conservation to control alien plant species (McLendon

& Redente 1992; Morgan 1994; Wilson & Gerry 1995; Gill & Jalota 1996; Sharma *et al.* 1998; Zink & Allen 1998; Reever-Morghan & Seastedt 1999; Cione *et al.* 2002; Lowe *et al.* 2002; Blumenthal *et al.* 2003; Averett *et al.* 2004; Baer *et al.* 2004; Corbin & D'Antonio 2004; Perry *et al.* 2004; Prober *et al.* 2005; Athy *et al.* 2006; Kulmatiski & Beard 2006). Therefore, the long-term influence of woodchip mulch on the trajectory of perennial and annual palatable and unpalatable species establishment, either directly or in interaction with the level of soil nitrogen availability, deserves more attention.

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Appendices

Appendix 3.1 Residual plots (three factor ANOVA models) for data on the effects of soil, mulch and seed treatments on species richness, plant density, vegetation cover, plant diversity and grazing capacity. (Although some of the formal Levenes tests suggest violations of homogeneity, residual plots show similar variances and no serious violations of homogeneity like variances increasing with mean suggesting no skewed distributions).
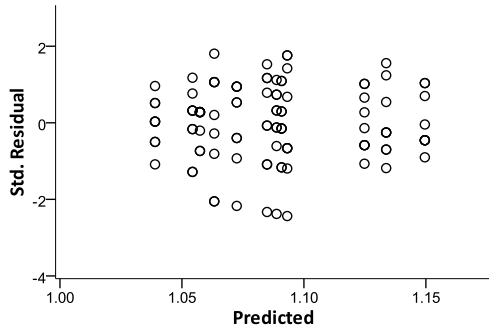

Figure 1 Log_{10} transformed palatable species richness residual plot (Levenes, F=2.34,p=0.013).
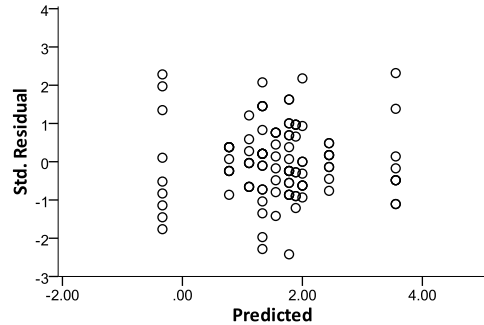
Figure 2 less palatable species richness residual plot (Levenes, F=2.41,p=0.01).
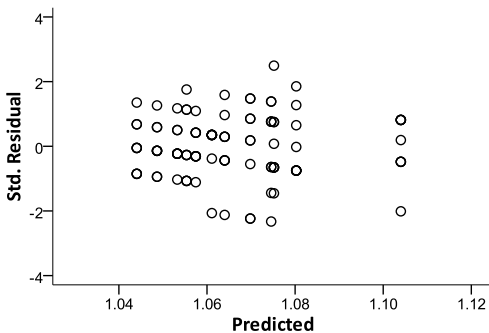
Figure 3 Log_{10} transformed annual species richness residual plots (Levenes: F=1.5, p=0.14).
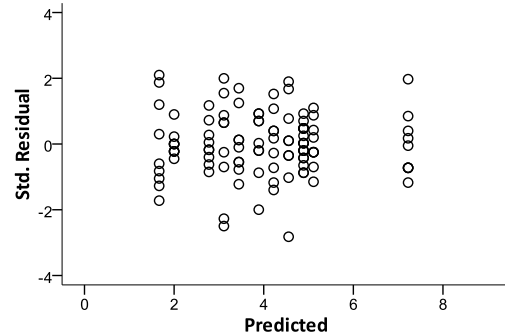
Figure 4 Total Plant species richness residual plots (Levenes: F=2.9, p=0.02).
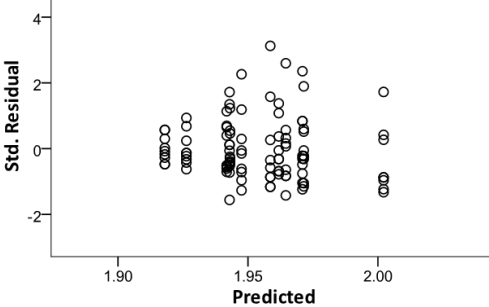
Figure 5 Log_{10} transformed palatable plant density residual plot (Levenes: F=1.71, p=0.073).
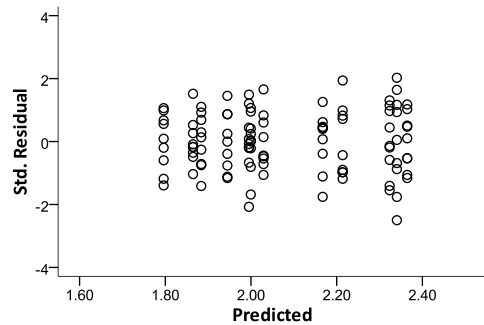
Figure 6 Log_{10} transformed less palatable plant density residual plot (Levenes: F=1.3, p=0.23)
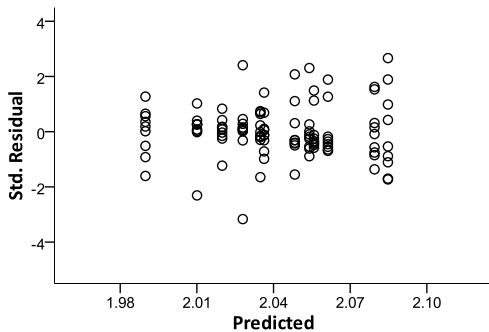
Figure 7 Log_{10} transformed annual plant density residual plot (Levenes: F=1.15, p=0.26)
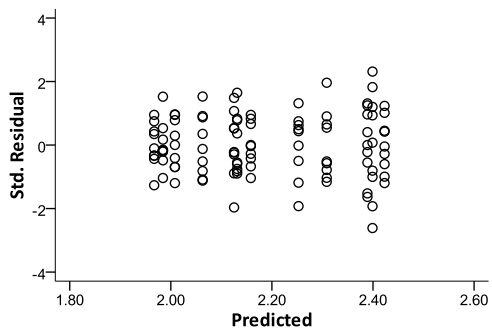
Figure 8 Log_{10} transformed total plant density residual plot (Levenes: F=2.03, p=0.03)
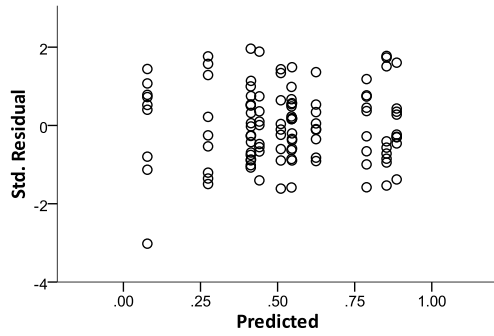
Figure 9 Perennial Shannon plant diversity (H) residual plot (Levenes: F=1.6, p=0.0.09)
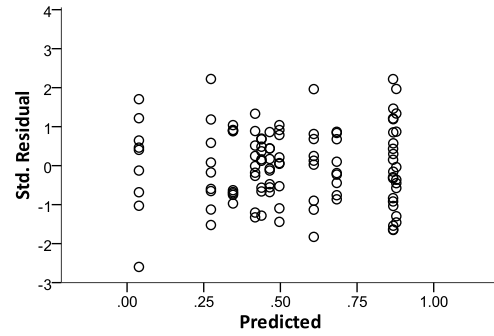
Figure 10 Total Shannon plant diversity (H - annual and perennial plants included residual plot (Levenes: F=1.6, p=0.0.09)
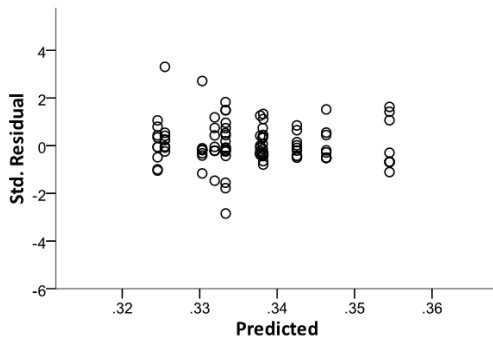

Figure 11 Arcsine and square root transformed perennial palatable vegetation cover residual plot (Levenes: F=1.01, p=0.43).
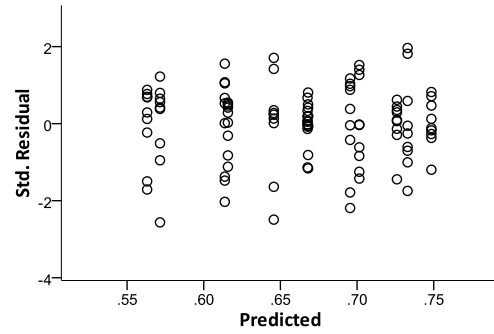

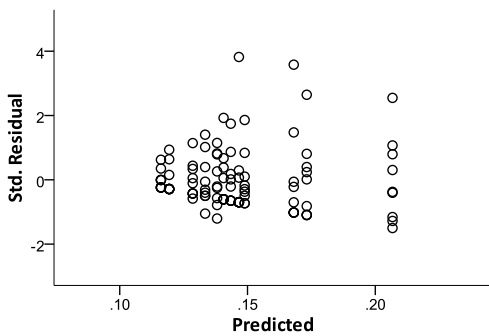**Figure 12** Arcsine and square root transformed perennial less palatable vegetation cover residual plot (Levenes: F=2.01 p=0.041)

Figure 13 Arcsine and square root transformed annual vegetation cover residual plot (Levenes: F=1.71 p=0.081)
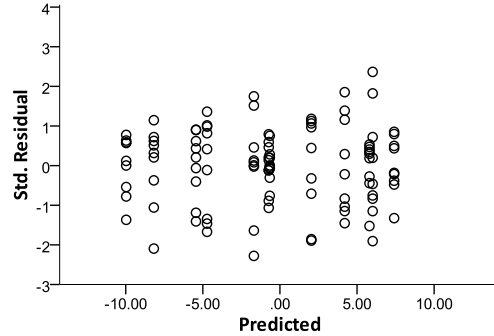
Figure 14 Total perennial vegetation cover residual plot (Levenes: F=2.4, p=0.1)
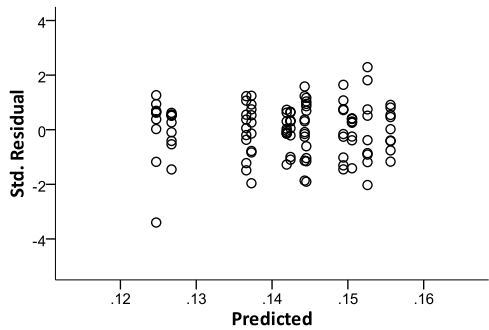
Figure 15 Arcsine and square root transformed grazing capacity (LSU/ha) only considering perennial plants residual plot (Levenes: F=2.01 p=0.033)
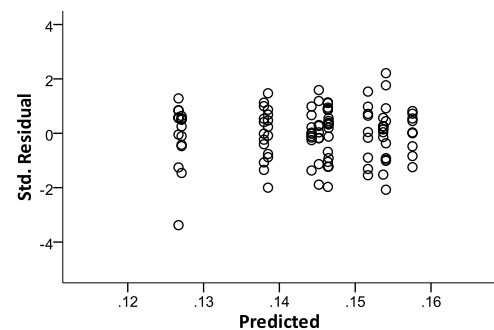

Figure 16 Arcsine and square root transformed total grazing capacity (LSU/ha) considering annual and perennial plants residual plot (Levenes: F=2.01 p=0.025)

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Appendix 3.2 Mean plant density changes (±SE) from 2009 to 2010, after rehabilitation for treated and untreated plots, on Morester farm - a degraded ostrich farm in the Little Karoo.

Palatability group	Plant species	Control (n=36)	Micro (n=36)	Rip (n=36)	Seed (n=54)	No seed (n=54)	Mulch (n=54)	No Mulch (n=54)
Highly palatable								
Dwarf woody shrub	*Atriplex semibaccata*	0.39 (0.11)	0.72 (0.24)	1.17 (0.67)	0.38 (0.15)	1.14 (0.45)	0.47 (0.14)	1.05 (0.46)
Dwarf woody shrub	*Limeum aethiopicum*	0.22 (0.1)	0.03 (0.03)	0.17 (0.1)	0.14 (0.06)	0.14 (0.07)	0.16 (0.06)	0.12 (0.08)
perennial forb	*Gazania krebsiana*	0.06 (0.04)	0.03 (0.03)	0 (0)	0.06 (0.03)	0 (0)	0.06 (0.03)	0 (0)
perennial forb	*Hermania pulverata*	0.06 (0.06)	0.11 (0.11)	0 (0)	0.07 (0.07)	0.04 (0.04)	0.11 (0.08)	0 (0)
perennial forb	*Lessertia annularis*	0.00 (0.07)	0.14 (0.07)	0.08 (0.06)	0.16 (0.07)	-0.01 (0.01)	0.08 (0.05)	0.06 (0.06)
Woody shrub	*Rhigozum obovatum*	0.06 (0.04)	0 (0)	0 (0)	0 (0)	0.04 (0.03)	0 (0)	0.04 (0.03)
Woody shrub	*Tripteris sinuata*	1.58 (0.51)	5.25 (1.65)	2.42 (0.79)	4.89 (1.09)	1.28 (0.61)	4.31 (1.18)	1.85 (0.47)
Palatable								
Annual	*Amaranthus thunbergii*	-2.03 (0.68)	-3.14 (1.64)	-2 (1.01)	-2.35 (1.06)	-2.43 (0.86)	-2.65 (0.94)	-2.13 (0.98)
Annual	*Chenopodium mucronatum*	6.22 (1.8)	6.44 (1.44)	9.25 (3.47)	5.68 (1.55)	8.94 (2.28)	2.75 (0.78)	11.86 (2.51)
Annual	*Gazania lichtensteinii*	0.17 (0.1)	0.08 (0.05)	0.03 (0.03)	0.11 (0.06)	0.07 (0.04)	0.06 (0.04)	0.13 (0.07)
Annual	*Ursinia nana*	0.06 (0.04)	0.08 (0.05)	0.22 (0.13)	0.07 (0.04)	0.17 (0.09)	0.22 (0.09)	0.02 (0.02)
Dwarf shrub succulent	*Drosanthemum liqui*	0.22 (0.2)	0.39 (0.18)	0.06 (0.17)	0.17 (0.14)	0.28 (0.16)	-0.11 (0.1)	0.56 (0.18)
Dwarf shrub succulent	*Euphorbia burmannii*	-0.19 (0.1)	0.17 (0.1)	-0.22 (0.13)	-0.06 (0.11)	-0.11 (0.08)	-0.06 (0.08)	-0.11 (0.11)
Dwarf shrub succulent	*Mesembryanthemum noctiflorum*	0.08 (0.12)	0.81 (0.23)	1.17 (0.83)	0.37 (0.17)	1 (0.55)	0.28 (0.14)	1.09 (0.56)
Dwarf woody shrub	*Anisodontea sp*	0 (0)	0 (0)	0.03 (0.03)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)
Dwarf woody shrub	*Bassia salsoloides*	0 (0)	-0.11 (0.07)	0 (0)	-0.06 (0.04)	-0.02 (0.02)	-0.02 (0.02)	-0.06 (0.04)
Dwarf woody shrub	*Helichrysum asperum*	0.89 (0.46)	0.56 (0.4)	0.08 (0.05)	0.61 (0.36)	0.41 (0.2)	0.15 (0.15)	0.87 (0.37)
Dwarf woody shrub	*Hypertelis salsaloides*	0.5 (0.31)	0.22 (0.11)	0.25 (0.1)	0.39 (0.21)	0.26 (0.09)	0.15 (0.08)	0.5 (0.22)
Dwarf woody shrub	*Tetragonia fruticosa*	1.83 (0.56)	2.5 (0.86)	0.33 (0.21)	1.6 (0.43)	1.51 (0.58)	2.12 (0.65)	0.99 (0.27)
perennial forb	*Chaenostoma subnudum*	1.72 (0.8)	3.03 (1.23)	1.67 (0.75)	1.17 (0.32)	3.11 (1.04)	1.04 (0.33)	3.24 (1.03)
perennial forb	*Kedrostis nana*	0 (0)	0.06 (0.06)	0 (0)	0 (0)	0.04 (0.04)	0.04 (0.04)	0 (0)
Perennial grass	*Fingerhutia africana*	0.5 (0.2)	1.67 (0.6)	1.08 (0.3)	1.89 (0.4)	0.28 (0.21)	0.91 (0.37)	1.26 (0.29)
Perennial grass	*Stipagrostis obtusa*	0 (0)	0.03 (0.03)	0 (0)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
Succulent Shrub	*Zygophyllum retrofractum*	0.19 (0.15)	0.22 (0.31)	-0.39 (0.17)	0.12 (0.21)	-0.1 (0.15)	0.12 (0.18)	-0.1 (0.19)

Woody shrub	*Felicia filifolia*	0.06 (0.04)	0.03 (0.03)	0 (0)	0.06 (0.03)	0 (0)	0.06 (0.03)	0 (0)
Woody shrub	*Pentzia incana*	0.03 (0.03)	0.11 (0.08)	0.03 (0.03)	0.11 (0.06)	0 (0)	0.07 (0.04)	0.04 (0.04)
Woody shrub	*Pteronia glauca*	0.08 (0.08)	0 (0)	0 (0)	0 (0)	0.06 (0.06)	0 (0)	0.06 (0.06)
Woody shrub	*Putterlickia pyracantha*	0.14 (0.11)	0.03 (0.03)	0 (0)	0 (0)	0.11 (0.08)	0.02 (0.02)	0.09 (0.08)
Woody shrub	*Salsola aphylla*	0.03 (0.03)	0 (0)	-0.03 (0.05)	0 (0.03)	0 (0.03)	-0.02 (0.02)	0.02 (0.03)
less-palatable								
Annual	*Atriplex lindleyi*	0.03 (0.03)	0.19 (0.1)	0.03 (0.03)	0.07 (0.04)	0.09 (0.06)	0.09 (0.05)	0.07 (0.06)
Annual	*Chenopodium murali*	2.86 (1.02)	0.56 (0.27)	2.53 (0.71)	1.89 (0.63)	2.07 (0.59)	2.43 (0.67)	1.54 (0.54)
Annual	*Tetragonia echinata*	0 (0)	-0.06 (0.03)	0 (0)	-0.01 (0.01)	-0.03 (0.02)	-0.03 (0.02)	-0.01 (0.01)
Dwarf shrub succulent	*Augea capensis*	-0.61 (0.53)	-3.25 (1.12)	0.5 (0.69)	-1.3 (0.72)	-0.94 (0.68)	-1.44 (0.54)	-0.8 (0.82)
Dwarf shrub succulent	*Delosperma peersii*	-0.31 (0.11)	0.11 (0.08)	-0.03 (0.05)	-0.07 (0.06)	-0.07 (0.08)	-0.04 (0.07)	-0.11 (0.07)
Dwarf shrub succulent	*Drosanthemum hispidum*	0.44 (0.34)	0.03 (0.03)	0.17 (0.07)	0.09 (0.05)	0.33 (0.23)	0.06 (0.04)	0.37 (0.23)
Dwarf shrub succulent	*Leipoldtia schultzii*	9.06 (3.95)	4.89 (2.42)	-0.36 (0.19)	5.32 (2.72)	3.73 (1.61)	4.47 (1.6)	4.58 (2.73)
Dwarf shrub succulent	*Malephora lutea*	13.75 (3.9)	19.06 (5.21)	15.44 (5.95)	15.54 (3.98)	16.63 (4.31)	4.74 (1.23)	27.43 (5.3)
Dwarf shrub succulent	*Ruschia pungens*	0.36 (0.23)	0.64 (0.41)	0.03 (0.03)	0.25 (0.17)	0.44 (0.27)	0.21 (0.18)	0.47 (0.26)
Dwarf shrub succulent	*Ruschia spinosa*	-0.61 (0.28)	0.89 (0.43)	-0.11 (0.19)	0.15 (0.27)	-0.04 (0.27)	-0.19 (0.16)	0.3 (0.34)
Dwarf succulent	*Piaranthus germinatus*	0 (0)	0 (0)	-0.06 (0.03)	-0.03 (0.02)	-0.01 (0.01)	-0.01 (0.01)	-0.03 (0.02)
Dwarf succulent	*Senecio radicans*	0.03 (0.03)	0 (0)	0 (0)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)
Dwarf woody shrub	*Aptosimum indivisum*	0.03 (0.12)	0.28 (0.25)	-0.14 (0.17)	0.12 (0.15)	-0.01 (0.15)	-0.14 (0.17)	0.25 (0.13)
Dwarf woody shrub	*Galenia filiformis*	0.69 (0.27)	0.67 (0.26)	1.22 (0.53)	0.52 (0.26)	1.2 (0.34)	0.2 (0.08)	1.52 (0.41)
Dwarf woody shrub	*Lampranthus haworthii*	0 (0)	0.03 (0.03)	0 (0)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)
Dwarf woody shrub	*Peliostoma leucorrhizum*	0.06 (0.06)	0 (0.05)	0.03 (0.03)	0.01 (0.02)	0.05 (0.05)	-0.03 (0.02)	0.08 (0.05)
Dwarf woody shrub	*Selago sp1*	-0.69 (0.84)	0.47 (0.44)	0.17 (0.17)	-0.15 (0.47)	0.11 (0.44)	-0.39 (0.28)	0.35 (0.57)
Geophyte	*Othono alba*	0 (0)	0.08 (0.06)	0.06 (0.04)	0.06 (0.03)	0.04 (0.04)	0.06 (0.03)	0.04 (0.04)
perennial forb	*Hermannia cernua*	0 (0)	0.03 (0.03)	0.03 (0.03)	0.02 (0.02)	0.02 (0.02)	0.04 (0.03)	0 (0)
perennial forb	*Lepidium africanum*	0.36 (0.36)	0 (0)	0 (0)	0 (0)	0.24 (0.24)	0.24 (0.24)	0 (0)
Perennial grass	*Cynodon incompletus*	0.03 (0.03)	0.03 (0.03)	0.03 (0.03)	0.04 (0.03)	0.02 (0.02)	0.04 (0.03)	0.02 (0.02)
Woody shrub	*Exomis sp*	0 (0)	0 (0)	0.03 (0.03)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
Woody shrub	*Rhus burchellii*	0 (0)	0 (0)	0.03 (0.03)	0.02 (0.02)	0 (0)	0.02 (0.02)	0 (0)
Unpalatable								

Annual	*Conyza sp*	0.03 (0.03)	0 (0)	0.06 (0.04)	0.04 (0.03)	0.02 (0.02)	0.04 (0.03)	0.02 (0.02)
Annual	*Mesembryanthemum crystalinum*	1.17 (2.39)	3.03 (1.62)	2.61 (2.28)	2.99 (1.84)	1.55 (1.61)	-0.92 (1.3)	5.45 (1.98)
Annual	*Oligocarpus calendulaceus*	3.53 (1.94)	1.78 (0.85)	1.42 (0.59)	2.13 (0.79)	2.35 (1.24)	3.41 (1.37)	1.07 (0.48)
Annual	*Onconosiphon piluliferum*	0 (0)	0.22 (0.13)	0.08 (0.05)	0.15 (0.09)	0.06 (0.04)	0.07 (0.04)	0.13 (0.08)
Annual	*Tribulus terrestris*	0.03 (0.03)	0 (0)	0 (0)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)
Annual grass	*Schismus barbatus*	0.19 (0.12)	1.17 (0.56)	0.5 (0.22)	0.63 (0.26)	0.61 (0.33)	0.81 (0.35)	0.43 (0.23)
Dwarf shrub succulent	*Crassula tetragona*	0.03 (0.04)	0.11 (0.11)	0 (0)	0.08 (0.08)	0.01 (0.02)	0.03 (0.03)	0.06 (0.07)
Dwarf shrub succulent	*Euphorbia heptagona*	0 (0)	0.06 (0.04)	0 (0)	0.02 (0.02)	0.02 (0.02)	0.02 (0.02)	0.02 (0.02)
Dwarf shrub succulent	*Mesembryanthemum articulatum*	0.97 (0.79)	1.67 (1.19)	0.78 (0.53)	0.37 (0.12)	1.91 (1)	0.35 (0.11)	1.93 (1)
Dwarf shrub succulent	*Mesembryanthemum junceum*	2.92 (1.31)	0.81 (0.44)	1.31 (0.44)	1.83 (0.88)	1.52 (0.43)	0.52 (0.2)	2.83 (0.93)
Dwarf shrub succulent	*Mesembryanthemum splendens*	0.14 (0.07)	0.19 (0.12)	-0.03 (0.04)	0.1 (0.04)	0.1 (0.09)	0.06 (0.06)	0.14 (0.07)
Dwarf shrub succulent	*Othonna cylindricus*	0 (0)	0 (0)	0.03 (0.03)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
Dwarf shrub succulent	*Ruschia crassa*	0 (0)	-0.06 (0.03)	0 (0)	-0.01 (0.01)	-0.03 (0.02)	-0.03 (0.02)	-0.01 (0.01)
Dwarf shrub succulent	*Ruschia intricata*	0.03 (0.03)	0 (0)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)	0 (0)
Dwarf shrub succulent	*Sarcostemma viminalus*	-0.06 (0.03)	0 (0)	0 (0)	-0.01 (0.01)	-0.03 (0.02)	-0.03 (0.02)	-0.01 (0.01)
Dwarf succulent	*Aloe varigata*	0 (0)	0.11 (0.13)	0 (0)	0.06 (0.08)	0.02 (0.04)	0.02 (0.04)	0.06 (0.08)
Dwarf succulent	*Crassula expansa*	36.92 (12.69)	15.31 (4.66)	20.19 (12.22)	16.2 (5.3)	32.07 (10.91)	5.48 (1.58)	42.8 (11.57)
Dwarf succulent	*Crassula muscosa*	0 (0)	0.06 (0.04)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)	0.04 (0.03)
Dwarf succulent	*Crassula subaphylla*	0.22 (0.13)	0.11 (0.11)	-0.22 (0.09)	0.04 (0.11)	0.04 (0.08)	0.04 (0.06)	0.04 (0.12)
Dwarf woody shrub	*Galenia pubescens*	19.47 (8.89)	16.25 (7.02)	16.94 (8.6)	14.44 (6.66)	20.67 (6.66)	-7.24 (3.25)	42.35 (7.44)
Dwarf woody shrub	*Galenia sp1*	0 (0)	-0.39 (0.16)	-0.17 (0.07)	-0.17 (0.07)	-0.2 (0.09)	-0.2 (0.09)	-0.17 (0.07)
Dwarf woody shrub	*Lycium cinerium*	1.75 (0.36)	1 (0.22)	0.58 (0.22)	0.93 (0.23)	1.3 (0.23)	1.15 (0.19)	1.07 (0.27)
Dwarf woody shrub	*Solanum tomentosum*	0.06 (0.04)	0.06 (0.04)	0.03 (0.03)	0.06 (0.03)	0.04 (0.03)	0.06 (0.03)	0.04 (0.03)
Geophyte	*Dipcadi sp*	0.08 (0.06)	0 (0)	0.03 (0.03)	0 (0)	0.07 (0.04)	0.07 (0.04)	0 (0)
Geophyte	*Drimia anomale*	0.03 (0.03)	0 (0)	0 (0)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
Geophyte	*Ornithogalum sp*	0 (0)	0.06 (0.06)	0 (0)	0 (0)	0.04 (0.04)	0 (0)	0.04 (0.04)
Geophyte	*Trachyandra sp*	0.03 (0.03)	0 (0)	0 (0)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
perennial forb	*Lactuca serriola*	0.03 (0.03)	0 (0)	0 (0)	0.02 (0.02)	0 (0)	0.02 (0.02)	0 (0)
perennial forb	*Malva parviflora*	0.17 (0.12)	0.19 (0.12)	0.06 (0.04)	0.24 (0.11)	0.04 (0.03)	0.26 (0.11)	0.02 (0.02)
perennial forb	*Salvia clandestina*	0 (0)	0 (0)	0.03 (0.03)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)

perennial forb	*Senecio angustifolia*	0 (0)	0 (0)	-0.06 (0.03)	-0.03 (0.02)	-0.01 (0.01)	-0.01 (0.01)	-0.03 (0.02)
Succulent Shrub	*Cotyledon orbiculatum*	0.03 (0.03)	0.08 (0.08)	0.03 (0.03)	0.07 (0.06)	0.02 (0.02)	0 (0)	0.09 (0.06)
Woody shrub	*Asparagus capensis*	0.03 (0.06)	0.06 (0.05)	0.08 (0.08)	0.04 (0.03)	0.07 (0.07)	0.02 (0.04)	0.09 (0.06)
Woody shrub	*Asparagus suavelens*	0.06 (0.06)	0.03 (0.03)	0 (0)	0.06 (0.04)	0 (0)	0 (0)	0.06 (0.04)
Woody shrub	*Athanasia trifurcata*	0 (0)	0 (0)	0.03 (0.03)	0 (0)	0.02 (0.02)	0.02 (0.02)	0 (0)
Woody shrub	*Carissa haematocarpa*	0.06 (0.04)	0 (0.03)	0 (0)	0.03 (0.03)	0.01 (0.02)	0.03 (0.03)	0.01 (0.02)
Woody shrub	*Chrysocoma ciliata*	2.06 (0.92)	2.97 (1.48)	0.5 (0.39)	1.81 (0.81)	1.88 (0.88)	2.34 (1.07)	1.34 (0.54)
Woody shrub	*Lycium oxycarpum*	0.03 (0.03)	-0.06 (0.03)	0.03 (0.03)	-0.01 (0.03)	0.01 (0.02)	0.01 (0.02)	-0.01 (0.03)
Woody shrub	*Pteronia pallens*	0.17 (0.09)	0.33 (0.2)	-0.06 (0.09)	0.19 (0.13)	0.11 (0.1)	0.17 (0.12)	0.13 (0.11)
Woody shrub	*Rosenia humilis*	0.06 (0.04)	0.11 (0.05)	-0.03 (0.04)	0.03 (0.04)	0.06 (0.04)	0.03 (0.03)	0.06 (0.05)

Appendix 3.3 Mean vegetation cover changes (±SE) from 2009 to 2010, after rehabilitation for treated and untreated plots, on Morester farm - a degraded ostrich farm in the Little Karoo.

Palatability group	Plant species	Control (n=36)	Micro (n=36)	Rip (n=36)	Seed (n=54)	No seed (n=54)	Mulch (n=54)	No Mulch (n=54)
Highly palatable								
Dwarf woody shrub	*Atriplex semibicata*	0.05 (0.14)	0.28 (0.08)	0.16 (0.07)	0.14 (0.08)	0.18 (0.07)	0.16 (0.02)	0.16 (0.08)
perennial forb	*Hermannia pulverata*	0 (0)	0 (0.01)	-0.01 (0.01)	-0.01 (0)	0 (0.01)	0 (0)	-0.01 (0)
Woody shrub	*Rhigozum obovatum*	0 (0)	0 (0.06)	0.06 (0.04)	0.04 (0)	0 (0.04)	0 (0)	0.04 (0)
Woody shrub	*Tripteris sinuata*	0.31 (0.1)	0.23 (0.18)	0.49 (0.13)	0.46 (0.1)	0.23 (0.12)	0.4 (0.05)	0.29 (0.11)
Palatable								
Annual	*Amaranthus thunbergii*	-0.04 (0.01)	0.01 (0.05)	0.05 (0.01)	-0.01 (0.03)	0.03 (0.03)	-0.02 (0)	0.03 (0.01)
Annual	*Chenopodium mucronatum*	0.9 (0.37)	1.05 (0.56)	1.76 (0.3)	1.22 (0.43)	1.25 (0.49)	0.55 (0.17)	1.92 (0.14)
Annual	*Ursinia nana*	0 (0)	0 (0.02)	0.02 (0.01)	0.01 (0)	0 (0)	0.01 (0)	0 (0.01)
Dwarf shrub succulent	*Drosanthemum liqui*	0.18 (0.08)	-0.14 (0.13)	0.11 (0.08)	-0.18 (0.16)	0.28 (0.14)	-0.02 (0.01)	0.12 (0.11)
Dwarf shrub succulent	*Euphorbia burmannii*	-0.04 (0.11)	-0.17 (0.24)	-0.24 (0.16)	-0.21 (0.06)	-0.08 (0.17)	-0.03 (-0.02)	-0.26 (0.03)
Dwarf shrub succulent	*Mesembryanthemum noctiflorum*	0 (0.07)	-0.07 (0)	0 (0.05)	-0.05 (0)	0 (0.05)	0 (0)	-0.05 (0)
Dwarf woody shrub	*Helichrysum asperum*	0.01 (0.06)	0.09 (0)	0 (0.03)	0.04 (0.03)	0.03 (0.04)	0 (0)	0.06 (0)
Dwarf woody shrub	*Hypertelis salsaloides*	0 (0.03)	0.03 (0)	0 (0)	0 (0.02)	0.02 (0)	0.02 (0)	0 (0.02)
Dwarf woody shrub	*Tetragonia fruticosa*	0.14 (0.42)	0.73 (0.19)	0.33 (0.19)	0.56 (0.28)	0.25 (0.2)	0.33 (0.05)	0.47 (0.27)
perennial forb	*Aptosimum procumbens*	-0.01 (0.01)	-0.01 (0.01)	-0.01 (0.01)	-0.01 (0.01)	-0.01 (0)	-0.02 (0)	0 (0.01)
perennial forb	*Chaenostoma subnudum*	0.03 (0.07)	0.12 (0)	0 (0.01)	0.01 (0.04)	0.09 (0.05)	0 (0.01)	0.1 (0)
Perennial grass	*Fingerhuthia africana*	0.04 (0.05)	0.08 (0.15)	0.18 (0.11)	0.18 (0.01)	0.01 (0.04)	0.13 (0.01)	0.07 (0.1)
Succulent Shrub	*Zygophyllum retrofractum*	0 (0.15)	-0.12 (0.26)	0.06 (0.22)	-0.2 (0.12)	0.17 (0.18)	-0.16 (0)	0.13 (0.17)
Woody shrub	*Euclea undulata*	0.2 (0)	0 (0)	0 (0)	0 (0.14)	0.14 (0)	0.14 (0.01)	0 (0.14)
Woody shrub	*Pentzia incana*	0 (0.02)	0.02 (0)	0 (0.02)	0.02 (0)	0 (0)	0.02 (0)	0 (0.02)
Woody shrub	*Putterlickia pyracantha*	0.06 (0.01)	0.01 (0.35)	-0.41 (0.23)	-0.23 (0.06)	0.01 (0.23)	0.01 (-0.02)	-0.24 (0.06)
Woody shrub	*Salsallo aphylla*	0 (0)	0 (0.07)	0.07 (0.04)	0.04 (0)	0 (0.04)	0 (0)	0.04 (0)
less-palatable								
Annual	*Atriplex lindeyi*	0 (0)	0 (0.01)	0.01 (0.01)	0.01 (0)	0 (0.01)	0 (0)	0.01 (0)
Annual	*Chenopodium murali*	0.05 (0)	0 (0.09)	0.1 (0.01)	0.01 (0.06)	0.09 (0.01)	0.09 (0.01)	0.01 (0.06)
Annual	*Pharnaceum sp*	0 (0)	0 (0.06)	0.08 (0)	0 (0.04)	0.05 (0)	0.05 (0)	0 (0.04)
Dwarf shrub succulent	*Augea capensis*	-0.05 (0.28)	0.07 (0.14)	-0.07 (0.2)	0.11 (0.13)	-0.14 (0.2)	-0.18 (0)	0.15 (0.13)

Dwarf shrub succulent	*Delosperma peersii*	0.22 (0.36)	0.27 (0.19)	-0.32 (0.33)	0.33 (0.17)	-0.21 (0.32)	0.09 (0.01)	0.03 (0.2)
Dwarf shrub succulent	*Drosanthemum hispidum*	0.14 (0.07)	-0.07 (0)	0 (0.01)	0.01 (0.11)	0.04 (0.09)	-0.04 (0)	0.08 (0.05)
Dwarf shrub succulent	*Drosanthemum vespertinum*	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Dwarf shrub succulent	*Leipoldtia schultzii*	-3.91 (1.21)	-3.56 (0.37)	-0.75 (1.01)	-3.59 (0.66)	-1.89 (0.57)	-3.66 (-0.37)	-1.82 (1.06)
Dwarf shrub succulent	*Malephora lutea*	1.15 (0.47)	1.98 (0.34)	1.27 (0.31)	1.47 (0.32)	1.46 (0.38)	0.79 (0.2)	2.13 (0.2)
Dwarf shrub succulent	*Ruschia pungens*	-1.31 (0.65)	-0.68 (0)	0 (0.46)	-0.47 (0.58)	-0.86 (0.43)	-0.68 (-0.09)	-0.64 (0.6)
Dwarf shrub succulent	*Ruschia spinosa*	-0.05 (0.16)	-0.32 (0.06)	0.05 (0.1)	-0.22 (0.09)	0.01 (0.09)	-0.11 (-0.01)	-0.1 (0.1)
Dwarf woody shrub	*Selago sp1*	-0.4 (0.17)	-0.16 (0.1)	-0.14 (0.16)	-0.35 (0.08)	-0.11 (0.17)	-0.09 (-0.03)	-0.37 (0.06)
Geophyte	*Othonno alba*	0 (0.01)	0.01 (0.02)	0.02 (0)	0 (0.02)	0.02 (0)	0.02 (0)	0 (0.02)
Unpalatable								
Annual	*Mesembryanthemum crystalinum*	0.1 (0.09)	0.07 (0.1)	0.21 (0.09)	0.17 (0.07)	0.08 (0.1)	0.09 (0.02)	0.16 (0.06)
Annual	*Oligocarpus calendulaceus*	0 (0)	0 (0.01)	0.01 (0)	0 (0.01)	0.01 (0)	0.01 (0)	0 (0.01)
Annual grass	*Pentaschistis airoides*	0.03 (0.04)	0.04 (0)	0 (0.02)	0.02 (0.03)	0.03 (0.03)	0 (0)	0.05 (0)
Annual grass	*Schismus barbatus*	0.09 (0.05)	0.1 (0.02)	0.03 (0.04)	0.07 (0.03)	0.08 (0.03)	0.09 (0.01)	0.06 (0.04)
Dwarf shrub succulent	*Crassula tetragona*	0 (0)	0 (0.02)	-0.02 (0)	0 (0.01)	-0.01 (0)	-0.01 (0)	0 (0.01)
Dwarf shrub succulent	*Mesembryanthemum articulatum*	0 (0.02)	0.02 (0.21)	0.03 (0.09)	-0.07 (0.11)	0.11 (0)	0.03 (0)	0 (0.14)
Dwarf shrub succulent	*Mesembryanthemum junceum*	0.66 (0.25)	-0.29 (0.24)	-0.17 (0.17)	0.04 (0.25)	0.09 (0.14)	-0.23 (0.01)	0.36 (0.26)
Dwarf shrub succulent	*Mesembryathemum splendens*	0.39 (0.23)	-0.23 (0.05)	0.06 (0.17)	-0.05 (0.11)	0.2 (0.09)	-0.03 (0.01)	0.18 (0.18)
Dwarf shrub succulent	*Othonna cylindrica*	0.02 (0.11)	0.11 (0)	0 (0.01)	0.01 (0.07)	0.07 (0.07)	0.01 (0.01)	0.07 (0.01)
Dwarf shrub succulent	*Ruschia perfoliata*	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Dwarf shrub succulent	*Sarcostemma viminale*	-0.06 (0)	0 (0)	0 (0)	0 (0.04)	-0.04 (0.04)	0 (0)	-0.04 (0)
Dwarf succulent	*Aloe variegata*	0 (0)	0 (0.02)	-0.02 (0)	0 (0.01)	-0.01 (0)	-0.01 (0)	0 (0.01)
Dwarf succulent	*Crassula expansa*	0.51 (0.2)	0.49 (0.13)	0.28 (0.13)	0.34 (0.17)	0.5 (0.2)	0.1 (0.06)	0.75 (0.03)
Dwarf succulent	*Crassula muscosa*	0.01 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Dwarf woody shrub	*Galenia pubescens*	1.21 (1.16)	1.08 (0.98)	-2.88 (0.88)	0.71 (0.97)	-1.1 (0.96)	-2.17 (-0.03)	1.77 (0.82)
Dwarf woody shrub	*Galenia sp1*	0.11 (0.19)	0.31 (0.19)	0.36 (0.12)	0.22 (0.14)	0.3 (0.16)	0.03 (0.04)	0.5 (0.08)
Dwarf woody shrub	*Lycium cinereum*	0.69 (0.6)	-0.19 (0.55)	-0.59 (0.43)	0.17 (0.51)	-0.23 (0.33)	0.04 (0)	-0.1 (0.59)
Dwarf woody shrub	*Solanum tomentosum*	0.02 (0)	0 (0)	0 (0)	0 (0.01)	0.01 (0.01)	0 (0)	0.01 (0)
perennial forb	*Malva parviflora*	0 (0.01)	0.01 (0.02)	0.02 (0)	0 (0.02)	0.02 (0.02)	0 (0)	0.02 (0)
Succulent Shrub	*Cotyledon orbiculatum*	0.03 (0)	0 (0)	0 (0)	0 (0.02)	0.02 (0.02)	0 (0)	0.02 (0)

Woody shrub	*Asparagus capensis var. capensis*	-0.11 (0.23)	-0.33 (0)	0 (0.16)	-0.27 (0.09)	-0.02 (0.08)	-0.2 (-0.02)	-0.1 (0.17)
Woody shrub	*Asparagus suaveolens*	0.18 (0)	0 (0)	0 (0.06)	0.06 (0.06)	0.06 (0.06)	0.06 (0.01)	0.06 (0.06)
Woody shrub	*Athanasia trifurcata*	0 (0)	0 (0.07)	0.07 (0)	0 (0.05)	0.05 (0)	0.05 (0)	0 (0.05)
Woody shrub	*Carissa haematocarpa*	-0.05 (0.14)	-0.08 (0)	0 (0.09)	-0.05 (0.04)	-0.04 (0.09)	-0.04 (-0.01)	-0.05 (0.04)
Woody shrub	*Chrysocoma ciliata*	-0.08 (0.29)	-0.29 (0.18)	0.25 (0.21)	-0.03 (0.15)	-0.05 (0.18)	0.1 (-0.01)	-0.18 (0.19)
Woody shrub	*Prosopis sp*	0 (0)	0 (0.09)	-0.09 (0.06)	-0.06 (0)	0 (0)	-0.06 (0)	0 (0.06)
Woody shrub	*Pteronia pallens*	0.17 (0.56)	1 (0.45)	-0.63 (0.45)	0.1 (0.21)	0.25 (0.39)	0.08 (0.02)	0.28 (0.31)
Woody shrub	*Rosenia humilis*	0 (0.04)	-0.04 (0)	0 (0)	0 (0.02)	-0.02 (0.02)	0 (0)	-0.02 (0)