LIVESTOCK IMPACT ASSESSMENT AND RESTORATION STRATEGIES IN THE SEMI-ARID KAROO

PETRUS CORNELIUS BEUKES

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This thesis is dedicated to the three M's in my life:
Marisé, Michelle and Monique

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ABSTRACT

Karoo rangelands exhibit spatial and temporal patterns that have important implications for the livestock industry. Spatially, there are gradients, often abrupt, in ecosystem structure and functioning, while plant composition and productivity are highly variable over time. A predictive understanding of these patterns, and the processes that cause them, is a prerequisite for developing appropriate restoration strategies. This thesis comprises several studies that attempt to relate vegetation patterns and processes to restoration strategies in southern Africa's Succulent- and Nama karoo ecosystems.

One hypothesis is that small-scale changes in soil physical and chemical properties are responsible for the fine-scale patterning evident in winter-rainfall Succulent karoo ecosystems. Alternatively, these patterns could be the result of area-selective grazing by livestock. To evaluate these hypotheses, plant and soil data were collected along soil- and grazing gradients radiating from a watering point in a Succulent karoo landscape. Results indicated that properties influencing soil hydrology and nutrient status are important determinants of pattern, and that long-term area-selective grazing can permanently change some of these properties. The hypothesis that the stasis of severely degraded patches in this piosphere is a consequence of poor water infiltration and seed limitation was tested in a restoration experiment. It appeared that natural seed availability is not limiting, but water infiltration has to be improved to initiate the restoration process. Removal of shrub material in long-ungrazed and moribund areas on the outer perimeter of the piosphere, had a positive impact in releasing resources for more seedlings and young growth, but did not alter plant species richness.

Stocking rate, composition and management of livestock profoundly influence the dynamics and composition of summer-rainfall Nama karoo vegetation. Proponents of non-selective grazing (NSG) argue that the periodic concentration in high densities of livestock in small areas, followed by long resting periods, improves vegetation composition as a consequence of low grazing selectivity, and enhances vegetation productivity and soil ecosystem processes as a result of intense hoof-action, dunging and urination impacts. Despite its application in farming systems, no studies have yet tested the predictions of NSG. I evaluated the effects of NSG on the soils and vegetation of Nama karoo rangeland in a fully replicated experiment. NSG did not alter the fertile-patch matrix, but improved soil infiltration. Plant compositional and cover changes could not be related to NSG. Rainfall was a much stronger driving force. I also explored the economics of NSG at the farm scale under different rainfall and stocking scenarios. An ecological-economic model predicted that NSG would be a viable option in higher rainfall (>200mm) areas because of the forage buffering capacity which enables the manager to maintain livestock through unpredictable droughts.

Restoration strategies for the Succulent Karoo have to focus on the resource-retention capacity of the soils. Livestock can reduce this capacity; low-stocking, flexible farming systems are therefore recommended for these more fragile ecosystems. Livestock in the more resilient Nama Karoo can be managed in a NSG system that can lead to an improvement in ecosystem functioning and maintain productivity in times of drought.

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DISCLAIMER

Unless acknowledgements are made to the contrary, the work in this thesis is my own. It has not been submitted to any other university.

P.C. Beukes

GENERAL INTRODUCTION

THESIS STRUCTURE

In this chapter I explain the layout of the thesis, followed by a general background to the Karoo region, motivation and objectives as per chapter, and information about particular study areas where field work was conducted.

The thesis starts by addressing the relative importance of soil properties and livestock impacts in determining plant community structure in the Succulent Karoo (Chapter 1). This is important for understanding the heterogeny of these rangelands and also the changes over time; knowledge which is essential for developing cost-effective restoration strategies (Chapter 2). The proponents of non-selective grazing (NSG) systems claim that livestock, correctly managed, can be used as "engineers" for reversing retrogression in karoo rangelands. In Chapter 3 and 4 I evaluate the impacts of livestock in a NSG system on soils and vegetation of the Nama Karoo. A further claim is that the costly multi-camp infrastructure of the NSG system is useful in counteracting unpredictable climatic conditions. In Chapter 5 I use a computer model to evaluate the profitability of NSG under different rainfall and stocking scenarios.

Each chapter was written to exist as an independent scientific paper, and therefore develops its own rationale, sets its own objectives and draws its own conclusions. Despite this independence each chapter is a contribution to the central issue of this thesis: the relationships between livestock, degradation and restoration in the Karoo.

THE SEMI-ARID KAROO

The semi-arid Karoo in South Africa can be distinguished from the arid desert biome by having a broader range of mean annual rainfall of between 50 and 600 mm compared to a narrow range below 100 mm (Desmet & Cowling 1999). The Karoo can be divided into two distinct biomes based on climatic variables and life form spectra; namely the Succulent Karoo and Nama Karoo (Rutherford & Westfall 1986) (Figure 1).

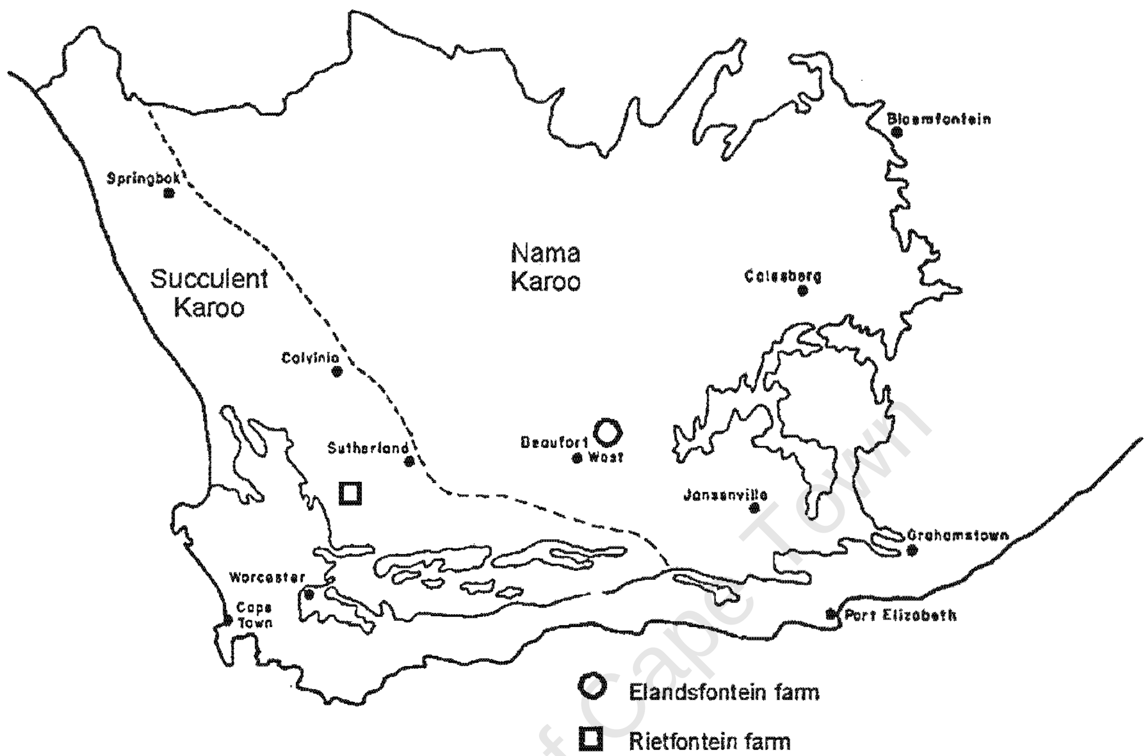

Figure 1 Principle localities in and around the Karoo (adapted from Cowling 1986). The location of farms where the research was done is also indicated.

Rainfall across the Karoo decreases from east to west and from south to north. Annual rainfall variability, expressed as coefficient of variation (CV), follows a similar trend (Desmet & Cowling 1999). The Succulent Karoo forms the southwestern border of the Karoo and receives a low, but relatively predictable moisture supply in the form of winter rainfall. Absolute and average minimum temperatures are high compared to other parts of the Karoo (Desmet & Cowling 1999). Dwarf, succulent shrubs of the families Mesembryanthemaceae, Crassulaceae, and Euphorbiaceae are prominent. Annuals (mainly from the family Asteraceae) may form an important component, especially on disturbed landscapes in spring (Hoffman & Cowling 1987). Grasses are rare, except in some sandy areas, and are mostly of the C_3 type (Low & Rebelo 1996). Most of the Nama Karoo occurs on the high-lying, central plateau of the western half of South Africa and is associated with higher maximum temperatures, and bimodal or strongly seasonal summer rains (Desmet & Cowling 1999). The low and unpredictable rainfall, and high summer temperatures makes this a harsh environment. The vegetation is dominated by C_4 type grasses and dwarf, woody shrubs including such prominent genera as *Pentzia* and *Rosenia* (Hoffman & Cowling 1987).

MOTIVATION AND OBJECTIVES

The Succulent Karoo is known for its often complex patchwork of plant communities, exhibiting high levels of diversity at many spatial scales (Cowling *et al.* 1989). Previous studies have focussed on the growth form, and species mixes of these communities (Cowling *et al.* 1994; Rubin 1998), as well as on processes like competition, facilitation, and phenology (Yeaton & Esler 1990; Esler 1993). There is well-documented evidence of the role of the edaphic environment in determining vegetation patterns on landscape- and local scales (Palmer *et al.* 1999). Rainfall has been shown to be an important driving force. Particularly the amount, seasonality, distribution, and type of rainfall determines the outcome of many processes in these arid ecosystems (Milton *et al.* 1997; Milton *et al.* 1999). Similarly the impacts of large grazing herbivores have been well studied with an emphasis on their role in changing the species composition of the pasture to a less-palatable, less-productive state (Milton 1992; Milton *et al.* 1997; Milton *et al.* 1999). The numbers and movements of large herbivores (e.g. livestock) have also been implied in a number of investigations into desertification of the Karoo (Dean *et al.* 1995a; Dean *et al.* 1995b). Grazing can act in combination with a complex array of environmental factors to induce vegetation change. It is therefore extremely difficult to make wide generalizations regarding the role of grazing as a determinant of change (Palmer *et al.* 1999). Nevertheless, both Milton *et al.* (1997) and Palmer *et al.* (1999) present summaries of a number of hypotheses for the mechanisms of grazing-induced changes. There is, however, a lack of understanding of the rate and magnitude of soil changes as a result of large herbivore impacts (Palmer *et al.* 1999). This understanding is vital for the development of improved livestock management practices and restoration strategies (Stokes 1994). In Chapter 1 I address the question of the impacts of long-term overgrazing on soils, how this degradation reflects in the plant community, and how the degradation can be identified by the abundance of certain key plant species.

Research into the conceptual framework and strategies of restoring degraded karoo rangeland has been a priority for many years (Milton *et al.* 1994; Milton & Dean 1995). It has been shown that a sufficient seed source and microsites for germination are two important factors for perpetuating healthy populations of preferred forage species (Milton & Hoffman 1994). Furthermore it has been shown that the emergence and survival of seedlings in these arid lands are closely linked to competition from established plants as well as total rainfall, and distribution of rainfall (Milton 1995). A number of reseeding experiments with palatable karoo shrubs and grasses have shown poor to variable results mainly because of highly degraded topsoil environments and lack of follow-up rainfall (Milton 1994). There is a need for understanding the ecosystem processes which have to be rekindled before large scale reseeding can be considered (Milton *et al.* 1997). Australian researchers have made significant contributions in this regard by showing the importance of recreating the fertile patch matrix of the landscape with consequent improvement in the capacity of the rangeland to capture and conserve soil, water, and nutrients (Ludwig & Tongway 1996). In Chapter 2 I address the question of the relative importance of seed availability, topsoil environment, and rainfall in revegetating eroded, bare areas in the Succulent Karoo. I also evaluate the results of reducing the competition from long-lived woody plants in an attempt to kick-start a transition from under-utilized, moribund veld to a more species-rich, productive vegetation.

Transitions in the structure of the grassy shrubland of the Nama Karoo have been well documented. The major driving forces of these transitions are stochastic events which influence plant demographic processes and selective defoliaton by large herbivores (Roux 1966; Hoffman *et al.* 1990; Milton & Hoffman 1994). Natural events like droughts and insect outbreaks often result in large scale deaths of mature karoo shrubs and grasses, and in so doing, release resources for a new generation of productive, young plants (Milton *et al.* 1995). Sufficient seed sources and healthy seed banks are, however, essential for regeneration after these events (O'Connor 1991). The Nama karoo vegetation shows a high resilience to grazing, suggesting an evolutionary history in which large herbivores played a significant role (Roux & Theron 1987). However, the expansion of the livestock industry since the beginning of this century created different pressures on karoo soils and vegetation (Milton & Dean 1995). Today much is known about the selective feeding behaviour of sheep, cattle, and goats and how this selectivity has changed species-rich karoo veld into woody, unpalatable shrub-veld, with a low capacity to support stock, and therefore low economic return (Roux & Theron 1987; Fuls 1992; Palmer *et al.* 1999). Many research projects were initiated to focus on the development of grazing management strategies which could stop this degradation (Roux 1968; Hoffman 1988). While most recommended light rotational grazing with strategic stock reductions, few really succeeded in controlling the detrimental impacts of selective defoliation (Acocks 1966). Palmer & Hoffman (1997) point to the dearth of experimental tests to support the rationale and purported successes of the different grazing management strategies. There is a need to address the role of the grazing animal as an "engineer" in promoting essential ecosystem processes which could alter the direction of transitions in the Nama karoo vegetation. In Chapters 3 and 4 I evaluate a grazing management system which was designed to combat selectivity, stimulate biogeochemical cycling, and improve primary productivity and, ultimately, the yield of animal products from the land. The ecological merits of non-selective grazing (NSG) is evaluated in terms of the impacts of grazing animals in such a system on the soils and vegetation of the Nama Karoo.

Stock owners in the Karoo have to make a living under adverse conditions, namely a highly variable rainfall and often lengthy droughts. The important contribution which these pastoral communities make to the economy of the country has resulted in many research and support projects which aimed at improving the viability of the industry (Vorster 1994). Ecological research mostly focussed on the impacts of stocking rate and grazing management on karoo plant communities, animal performance, and accelerated soil erosion (Roux & Vorster 1983). Currently, the most acceptable approach is to rotate animals in a simple, flexible grazing system and to destock in a timely manner in the event of a drought (Roux 1968; Roux *et al.* 1981). At first glance the arguments behind this approach appear sound. However, Hoffman (1988) points out, the approach is based on models of karoo dynamics which are unsubstantiated, and does not combat selectivity by the grazing animal (Acocks 1966). Drought, and the consequent decline in the forage reserve, is another major economic barrier to overcome, and most farmers will admit that it is easy to farm when it rains but a different story when it becomes dry (Howell 1978). In Chapter 5 I use a computer model in an attempt to show that concentrating livestock in a non-selective grazing (NSG) system not only makes ecological sense (Chapters 3 and 4) but can improve the pastoralists capacity to cope with droughts and, therefore, also makes economic sense. The aim is to optimize farm infrastructure, stocking rate, species mix, and supplementary feeding for maximum financial returns within the capacity of the land. I also evaluate the conditions under which simpler, cheaper and more flexible grazing systems are recommended.

The key questions per chapter can be listed:

<u>Chapter 1</u>

1. Does long-term concentrated stock activity in the Succulent Karoo result in soil changes?
2. Which plant species can be used to identify degraded Succulent karoo veld?
3. How can this knowledge of degraded veld contribute to reversing the process of veld degradation?

<u>Chapter 2</u>

1. Can degraded bare areas in the Succulent Karoo be revegetated by soil surface treatments and reseeding?
2. Does severe pruning of moribund, ungrazed Succulent karoo veld result in a transition to a more species-rich, productive vegetation?

<u>Chapter 3</u>

1. Does non-selective grazing (NSG) of Nama karoo veld result in improved soil quality?

<u>Chapter 4</u>

1. Does non-selective grazing (NSG) of Nama karoo veld result in improved plant cover, species composition and production?
2. How does rainfall influence these vegetation parameters?

<u>Chapter 5</u>

1. Is it an economically viable option to invest resources in the multi-camp infrastructure required for a NSG system?
2. What is the optimum number of camps for a given farming unit at different levels of veld productivity?
3. How can the animal compliment be manipulated to maximize profits?

STUDY AREA

Research was undertaken on two farms, Rietfontein in the Succulent Karoo and Elandsfontein in the Nama Karoo (Figure 1).

Rietfontein (4074 ha, 32°52'S/19°54'E) is situated 90 km north-east of the town of Ceres in the succulent Ceres Karoo. The land consists of a gentle slope with fanning erosion rills from about 570 m altitude in the south-eastern boundary to about 530 m in the north west. Run-off water is chanelled along these erosion rills ("washes") to the ephemeral Ongeluks river to the north of the farm. The rock types comprise shales, tillites and sandstones of the Dwyka and Ecca formations of the Karoo System. Geomorphologically, erosion is the dominant force in the landscape; hence, soils are young and skeletal (Ellis & Lambrechts 1986). Soils have a relatively low clay content (14 % in the topsoil), are lime-rich and alkaline (pH is 8.0), and have high electrical resistance (655 ohm) and sodium content (Exchangeable Sodium Percent = 3 – 4 %) (Chapter 1). The process of clay illuviation and mechanical impact of raindrops often leads to crusting and low infiltration rates (Ellis & Lambrechts 1986).

According to Schulze (1997) the mean annual rainfall for the study area is between 100 and 200 mm; the coefficient of variation of annual rainfall is between 35 and 40 %; and the mean annual A-pan equivalent potential evapotranspiration is 2400 – 2600 mm. A mean rainfall of 139 $mm.yr^{-1}$ was recorded on Rietfontein for the period 1995 to 1998. Most rain is recorded in the winter months (May – August) (Figure 2) when precipitation is derived from relatively predictable westward-moving low pressure cells or cold fronts (Desmet & Cowling 1999). The warm season rainfall is unpredictable and is associated with west coast troughs, thunderstorms and the autumnal northerly flow of moist, tropical air (Desmet & Cowling 1999). Summers are relatively hot with average maximum temperatures of > 30°C from December to February (Figure 2). The coincidence of high temperatures and low rainfall over this period results in extreme moisture stress for shallow-rooted plants. The summer aridity also results in lack of summer food for livestock and the low carrying capacity (8 ha/sheep) of these pastures. In some places it is still a common practice for pastoralists to stock their holdings in the Ceres Karoo only for 4 – 5 months during the winter (Rubin 1998).

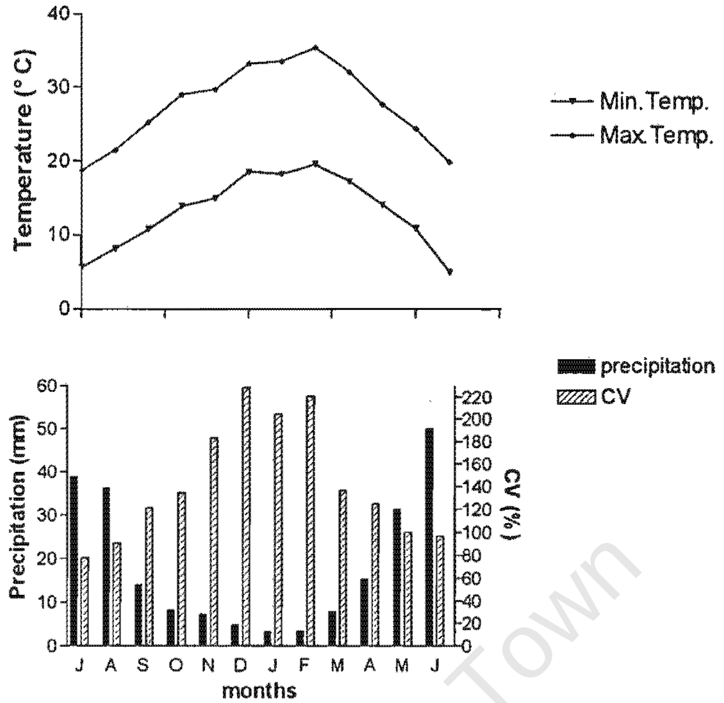

Figure 2 Mean monthly precipitation and temperatures for the rainfall station Mertenhof (32°09'S/19°12'E, 338 m altitude) representative of the homogenous climate zone 62 (Dent *et al.* 1989) in which the farm Rietfontein falls. Rainfall data have been collected over 78 years and temperature data over 6 years (data from Computing Centre for Water Research, University of Natal).

The vegetation of the study area is classified by Acocks (1975) as veld type 31, Succulent Karoo; and by Low & Rebelo (1996) as Lowland Succulent Karoo. It is a low succulent shrubland, dominated by dwarf leaf-succulent members of the Mesembryanthemaceae, especially species of *Ruschia, Drosanthemum, Malephora* and *Delosperma*. Annuals and geophytes may be common after good rains but perennial grasses are scarce (Figure 3).

A perennial spring in the centre of Rietfontein provides water to both humans and livestock. The water flows from cracks in the rocky bed of one of the washes and accumulates in a pool of approximately 200 m^2. Water can be pumped to five paddocks on a rotational basis. There was no livestock present during the study period as the farm was totally destocked in the early 1990's.

Elandsfontein (7000 ha) is situated 32 km north-east of Beaufort West in the Nama Karoo (32°15'S/22°45'E). A series of doloritic rocky outcrops run across the farm with the rest of the landscape consisting of level to near-level pediments. Soils are of mixed origin, derived from dolerite weathering products and Karoo System shales or shale-derived pedisediments (Ellis & Lambrechts 1986). The reddish coloured duplex soils have a coarse sandy loam texture, are relatively rich in most plant nutrients (topsoil P > 20 mg/kg; topsoil exchangeable K > 7 %), and have low levels of salts (electrical resistance > 1000 ohms), exchangeable Na (ESP < 1 %), and organic carbon (OC = ± 0.2 %).

Mean annual rainfall for the region ranges between 200 and 300 mm and the coefficient of variation of annual rainfall is between 35 and 40 % (Schulze 1997). The mean for the farm for the period 1987 – 1998 was 212 $mm.yr^{-1}$ with a CV of 47 %. Severe droughts occur regularly in the Nama Karoo with the last one being the 1990 – 1994 drought when Elandsfontein recorded a mean of 139 $mm.yr^{-1}$. Fieldwork at Elandsfontein commenced in the drought-breaking year of 1995 (238 mm) after the very dry 1994 (33 mm). There is a trend for more rainfall, with a higher reliability, to fall during summer months (Figure 4). Frontal systems bring light rain in winter while summer rains are often characterized by brief cloudbursts resulting in rapid run-off. The mean annual A-pan equivalent potential evapotranspiration is 2400 – 2600 mm (Schulze 1997).

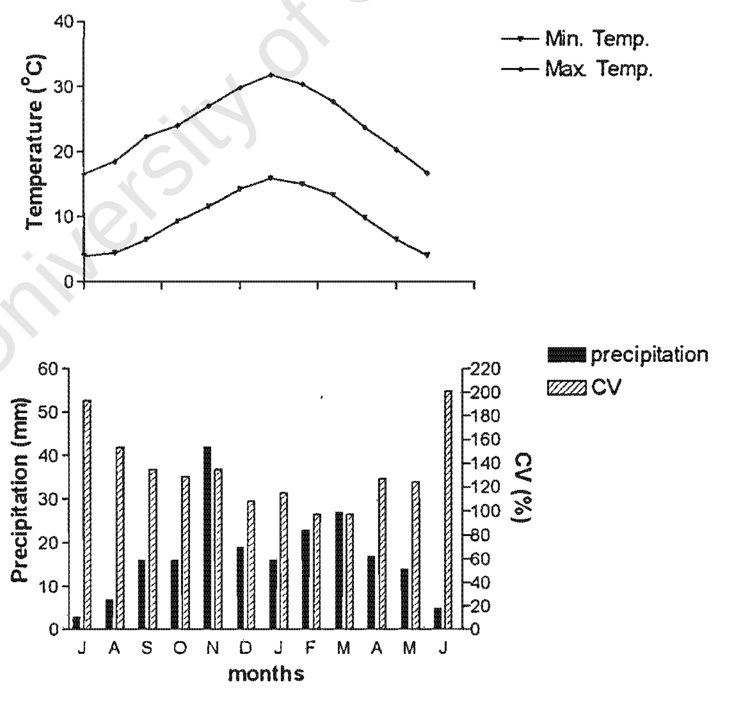

Figure 4 Mean monthly precipitation for Elandsfontein (32°15'S/22°45'E, 950 m altitude) and mean monthly temperatures for Stolshoek (32°20'S/22°30'E, 1067 m altitude). Rainfall data have been collected over 12 years and temperature data over 11 years. (Temperature data supplied by Agromet, Institute of Soil Climate and Water, Stellenbosch).

The vegetation is classified by Acocks (1975) as veld type 26, Karroid Broken Veld; and by Low & Rebelo (1996) as Central Lower Karoo. This semi-arid grassy shrubland is dominated by grasses, mainly species of *Stipagrostis* and *Eragrostis*, while *Pentzia* and *Rosenia* are dominant dwarf shrub genera. Rocky sites include a sparse, tall shrub to low tree stratum where the principle genus is *Rhus*. A highly variable ephemeral, and scarce succulent component exists (Figure 5).

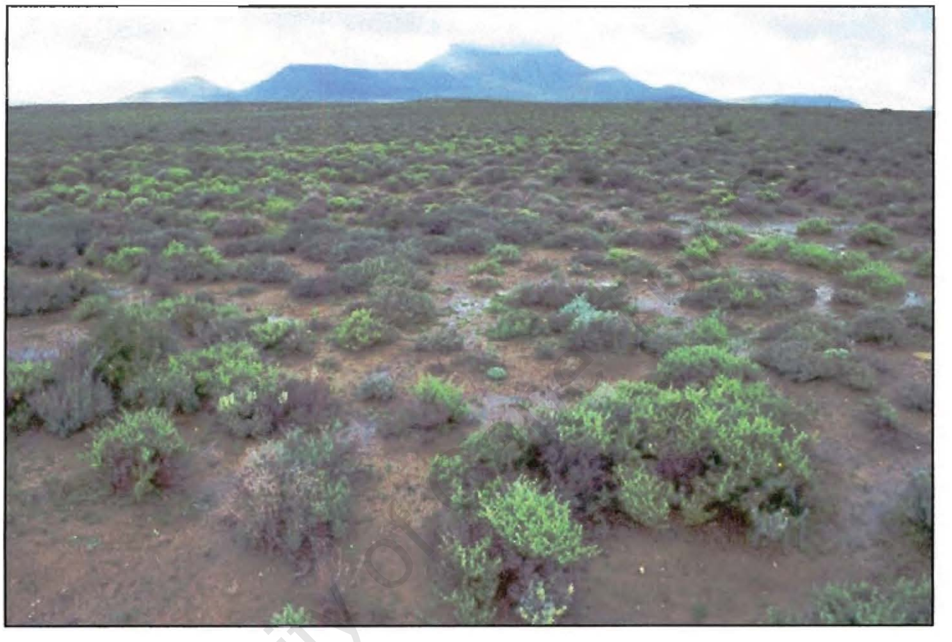

Figure 3 The Rietfontein landscape (Photo: D.M. Richardson)

Figure 5 The Elandsfontein landscape

Over a period of 30 years the landowner subdivided the farm into more, smaller paddocks. Only since the early 1990's were these paddocks sufficiently small to achieve the high grazing pressures of the non-selective grazing system. Currently the infrastructure on Elandsfontein consists of 147 paddocks arranged in wagon-wheel layouts around 38 permanent watering points. Large mixed herds of Nguni cattle, Merino sheep, and Boer goats (in an average ratio of 1:14:3) are rotated through the paddocks in a non-selective grazing system. The number of herds are reduced as far as possible in order to concentrate animals and minimize camp time (< 14 days). Grazing pressures of 40 – 60 Large Stock Unit Grazing Days per hectare (LSU*days/ha) are applied, after which a paddock is rested for at least a full year. A rumen stimulant in the form of cut saltbush (*Atriplex nummularia*) is supplied to improve animals' intake of poorer quality fibre towards the end of a period of occupation.

Wild animals that were observed at both study areas which might have an impact on soils and vegetation were herbivores like steenbok (*Raphicerus campestris*), hares (*Lepus*), tortoises (*Psammobates*), and several types of invertebrates belonging to various feeding groups. Mammals (mainly aardvark *Orycteropus afer*, porcupine *Hystrix austro-africanae*, bat-eared fox *Otocyon megalotis* and suricate *Suricata suricatta*) which excavate holes in the hard-capped karoo soils, are present in both study areas.

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CHAPTER 1

A STUDY OF SOILS AND VEGETATION ACROSS A GRAZING GRADIENT IN THE SUCCULENT KAROO

INTRODUCTION

Rangelands are currently the single most important economic resource of the Ceres Karoo agricultural sub-region in the winter-rainfall Succulent Karoo of South Africa. This is an area of low rainfall and skeletal, saline soils, which makes it suitable only for extensive pastoral use (Mackay & Zietsman 1996). The sustainable use of these rangeland resources requires the measurement and interpretation of vegetation patterns (Stafford Smith & Pickup 1990) and the capability to assess and monitor the condition or conservation status of these resources (Graetz & Ludwig 1978; Bosch & Gauch 1991; Mackay & Zietsman 1996).

A number of studies have shown the value of a systematic collection of soil and vegetation measurements around watering points. For instance, Tolsma *et al.* (1987) and Perkins & Thomas (1993) found changes in plant species composition and soil nutrient status as a result of cattle grazing around artificial watering points in the Kalahari Desert region of Botswana. Jeltsch *et al.* (1997) modelled this degradation around watering points in the Kalahari and showed that the recovery potential of shrub-encroached zones is negligible over a time span of 100 years. This kind of information was important for evaluating the environmental impacts of increasing borehole density as well as for policy recommendations regarding beef production systems in Botswana (Perkins 1996).

Work by Bayer & Stokes (1993) on fence-line contrasts in the Ceres Karoo highlighted the shortcomings of previous projects aimed at quantifying vegetation changes as a result of stock farming in the Succulent Karoo. They showed that the abundance of certain key species could be explained by grazing impacts. However, there was no attempt to relate vegetation patterns to soil factors. Therefore, the objectives of this study were:

- to describe the vegetation changes across a study area in the Ceres Karoo;
- to relate these vegetation changes to historical land use patterns, and possible soil changes as a result of stock activities around the central watering point;
- to comment on veld condition and identify indicator species that could be useful in veld condition assessment;
- to broaden the knowledge base which is essential for efficient rehabilitation of a degraded ecosystem.

STUDY AREA

The study was conducted on the farm Rietfontein (4 074 ha) which is situated 90 km north east of the town of Ceres in the succulent Ceres Karoo (32°52'S / 19°54'E). See the General Introduction chapter for a detailed description of the study area.

Historical Background

The spring on Rietfontein has been visited by large herbivores for many decades. Initially, before the first livestock farmers arrived in the Ceres Karoo in 1727 (Smuts & Alberts 1988), the springbok (*Antidorcas marsupialis*) was the most abundant grazer which, in some years, migrated into the Ceres Karoo in huge numbers (Skinner 1993). Although the springbok is known to be relatively independent of free water they must have concentrated their grazing and movement activities around water and could have been responsible for bare patch and pan formation, and general erosion (Roux & Opperman 1986).

Since the 18th century the Ceres Karoo has been settled by livestock farmers, who kept mainly sheep, goats and donkeys, but also cattle in the more grassy areas. Many were "trekboere" who migrated from the Koue Bokkeveld on top of the nearby mountains down into the warmer Ceres Karoo basin during winter months (Lichtenstein 1806 as cited in Smuts & Alberts 1988). The winter rains stimulated production in the mainly succulent vegetation and, together with the higher salt levels in the soils, winter grazing was believed to be healthy for livestock (Burchell 1811 as cited in Smuts & Alberts 1988). One can expect that the first permanent farmers in the region settled around the water sources, and with Rietfontein being one of the strongest flowing springs in the area (J. Dauth pers. comm.), it was permanently occupied by at least late 1800s (he oldest graves date back to that time) (pers. obs.). During this time the route through Karoopoort across the Ceres Karoo to Kimberley became a very busy one when people and their animals were drawn to the wealth of the diamond mines (Smuts & Alberts 1988).

The spring at Rietfontein has always been open to any trekboer to water his stock, and even today the spring can still be used without permission from the owner (I. Theunissen pers. comm.). Because of the water, the veld around Rietfontein has always been heavily stocked. For example, flocks of approximately 1 200 Afrikaner sheep were herded around this spring in the 1950's (I. Theunissen pers. comm.). There were no fences at that time and the herdsmen moved across farm boundaries with their stock but had to return to the spring at least once a day to water the stock. There were water-for-grazing agreements between neighbouring farmers (I. Theunissen pers. comm.) and one can expect that the combined effects over many decades of the landowners' stock, herds from neighbouring farms, and those of trekboere, had a severe impact on the vegetation around the spring. The farm was fenced in the 1970's and the recommended carrying capacity of ± 500 adult sheep was maintained for about 20 years until it was totally destocked in the early 1990's (G. du Plessis pers. comm.). Although the veld around the spring is at present not as bare as it use to be (G. du Plessis pers. comm.), there are still very clear vegetation and soil changes as one moves from the spring to the outer boundaries of the farm.

METHODS

Using the 1972 stocking data of 510 adult sheep (small stock units = SSU) and taking into account the radial nature of grazing around a point source, it was possible to calculate the changing grazing resource availability (ha) and the changing stocking levels [Herbivore Utilization Intensities (HUI) = SSU/ha] for the annuli represented by distance intervals from the point source (Perkins & Thomas 1993). From Figure 1 it is clear that from a distance of about 1500 m from the watering point, HUI begins to increase exponentially towards the watering point. This distance was then used to distinguish between lightly utilized areas further than 1500 m from the watering point (HUI < 1.0 SSU/ha), heavily utilized areas between 1500 m and 400 m from the watering point (HUI between 1.0 and

13.5 SSU/ha), and extremely heavily utilized areas closer than 400 m to the watering point (HUI > 13.5 SSU/ha).

Three belt transects consisting of a total of 101 quadrats (5m x 5m) were sampled from lightly utilized areas (further from the watering point) to heavily utilized areas (closest to the watering point) (Figure 2). Quadrats were systematically spaced 150 m apart along the three transects. Within each quadrat the canopy diameter of every perennial plant was measured along two axes (L and W) at right angles to each other. Percentage canopy cover was calculated using the formula $\pi\frac{L.W}{4}$. In each quadrat a soil profile was described and two soil samples (one topsoil and one subsoil) were collected for measuring percentage stone (> 2 mm) content, soil electrical resistance (as a measure of electrolyte concentration in the soil); $pH_{(water)}$, and lime. Forty one quadrats, representing the major plant communities, were selected from the total of 101 quadrats and these soils were analysed for $pH_{(KCl)}$, Calcium, Magnesium, Sodium, Potassium, and particle size by the Department of Soil Science, University of Stellenbosch using the techniques described in the Handbook of standard soil testing methods (1990). Of the 41 selected quadrats, 25 were in lightly utilized areas (> 1500m from the watering point), and 16 were in heavily utilized areas (< 1500m from the watering point).

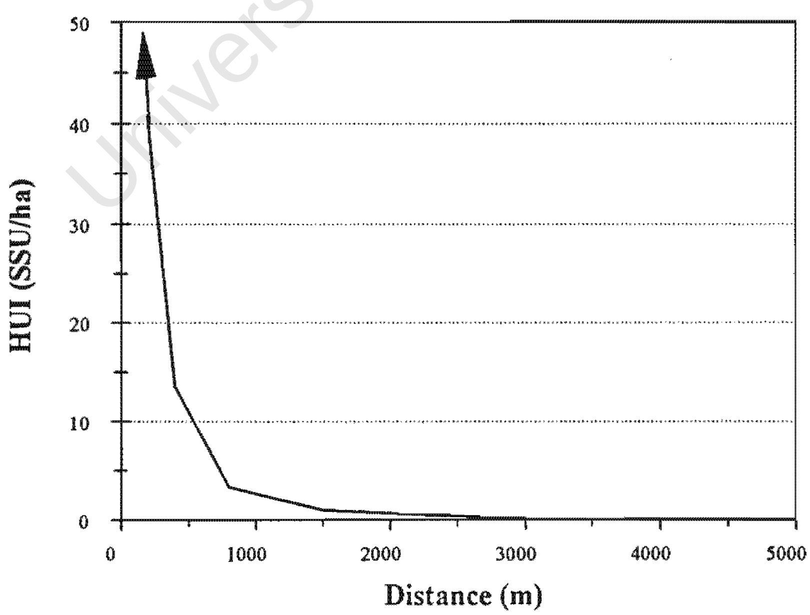

Figure 1 The relationship between distance from a watering point and Herbivore Utilization Intensity (HUI). The actual HUI values depend on the stocking rate, in this case 510 Small Stock Units (SSU).

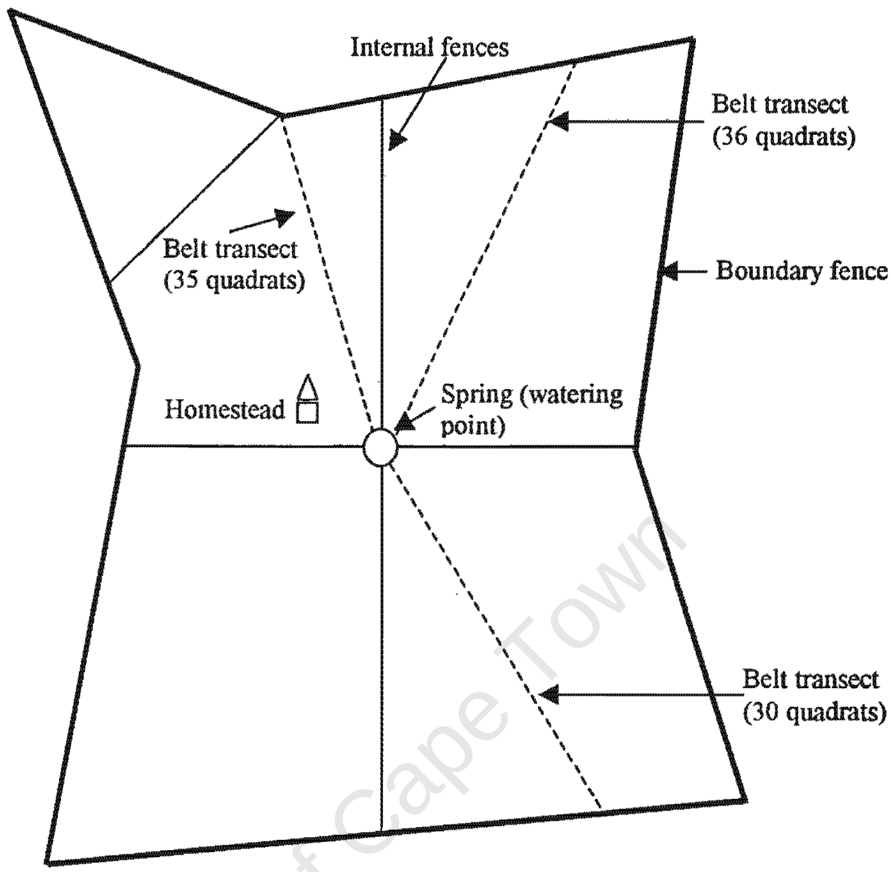

Figure 2 The spatial arrangement of the three belt transects at Rietfontein (not according to scale).

The patterns of variation in the soil variables were summarized using principle components analysis (PCA) in the CANOCO package (Ter Braak 1987a). Variables analysed were: depth of topsoil and subsoil; resistance of topsoil and subsoil (log transformed); clay content of topsoil and subsoil; $pH_{(water)}$ of topsoil and subsoil; stone (> 2 mm) content of topsoil and subsoil; presence of lime in the topsoil and subsoil; stone (> 6 mm) cover; and gravel (< 6 mm) cover. In the presentation of the biplot (soil variables and quadrats) (Figure 3) HUI classes were superimposed onto the quadrat ordination. This indicated possible relationships between grazing intensity and soil variables.

Plant cover data (arcsine transformed) were ordinated using an indirect (detrended correspondence analysis, DCA) gradient analysis (Ter Braak 1987a). The data set was reduced by removing species that occurred in less than six quadrats. In DCA, axes are extracted from the plant data alone. Quadrats are plotted as weighted averages of their component species and species are plotted according to their optimal abundances. Groupings of quadrats in Figure 4 can be related to groupings of species in Figure 5. HUI classes were also superimposed onto the quadrat ordination to indicate possible relationships between grazing intensity and vegetation patterns.

The species and soil data were analysed using canonical correspondence analysis (CCA) (Ter Braak 1987b). In this analysis axes are presented as linear combinations of soil variables, and quadrats are plotted according to their component species (Figure 6), and species plotted according to their abundances (Figure 7). CCA is designed to detect patterns of variation in species data that are best explained by the measured environmental variables. The end product is a biplot that expresses not only variation in species distributions, but also the main relationships between soil variables and species (Esler & Cowling 1993). In the quadrat biplot (Figure 6), HUI classes were superimposed onto the ordination. This was done to generate hypotheses regarding the relative roles of soils and utilization intensities in explaining floristic patterns. On inspecting the CCA species biplot three species groupings were identified (Figure 7).

RESULTS

A total of 45 species in 14 families was sampled in the 101 25 m^2 quadrats (Table 1). On average 8 species (min = 3; max = 18) were recorded per quadrat. Most species (56 %) were dwarf, leaf succulent shrubs belonging to the Mesembryanthemaceae. The prostrate succulent, *Drosanthemum eburneum*, was most abundant in terms of frequency while *Antimima hantamensis, Ruschia spinosa, Prenia tetragona,* and *Delosperma* sp. contributed most in terms of canopy cover (Table 1).

Table 1 (continued)

Family	Plant Species	Key to abbreviations in Fig. 4 & 6	Growth Form[1]	Frequency (%)	Mean % Canopy Cover (SD)
Chenopodiaceae	*Atriplex semibaccata*		PPH	1.0	0.001 (0.01)
Mesembryanthemaceae	*Hammeria salteri*		DLSS	1.0	0.003 (0.04)
Asphodelaceae	*Bulbine* sp.		Tuberous, contracted, stemless	1.0	0.001 (0.01)
Mesembryanthemaceae	*Phyllobolus splendens* subsp. *splendens*		DLSS	1.0	0.006 (0.06)
Asparagaceae	*Asparagus capensis*		DS	1.0	0.007 (0.07)
Poaceae	*Stipagrostis obtusa*		Graminoid	1.0	0.002 (0.02)
Crassulaceae	*Crassula deltoidea*		DLSS	1.0	0.0001 (0.001)
Asphodelaceae	*Aloe variegata*		DLSS	1.0	0.0003 (0.003)
PER QUADRAT					**19.1 (9.6)**

[1] **Explanation of growth form abbreviations:**

DLSS	=	**Dwarf, leaf succulent shrub (< 60 cm high)**
MHS	=	**Mid-high shrub (60 – 200 cm high)**
DS	-	**Dwarf shrub (< 60 cm high)**
DSSS	=	**Dwarf, stem succulent shrub (< 60 cm high)**
SA	=	**Succulent annual**
PPH	=	**Prostrate, perennial herb**

In general soils of the study site were relatively shallow, stony, low in clay, and high in pH and soil electrical resistance. However, resistance was found to be extremely variable. Lime is a common feature in these soils (Table 2).

Table 2 Mean values (SD) for soil variables measured at the farm Rietfontein, Ceres Karoo (n = 101 unless otherwise indicated)

Soil variable	Topsoil	Subsoil
Stone (> 2mm) content %	16.5 (7.8)	22.9 (14.1)
pH (water)	8.0 (0.56)	7.5 (1.06)
Electrical resistance (ohm)	644 (700)	583 (535)
Clay content (%) (n = 41)	14.8 (5.1)	24.7 (7.0)
Depth (mm)	46.6 (17.0)	206.2 (65.4)
Lime (presence as % frequency)	58	58

The variable loadings, indicating the importance of each axis, from the PCA are presented in Table 3. Axis 1 contributed 92 % of the total of 99.8 % of the variation in the data that was accounted for. The high loadings for topsoil and subsoil depth characterize axis 1 as a soil depth gradient. Other variables which make important contributions to explaining the variation in the data are stoniness, electrical resistance, and surface cover. The first biplot (Figure 3) shows the quadrat distribution in relation to the first two PCA axes. The HUI classes are superimposed onto this quadrat ordination. There is a trend for the soils of quadrats falling into the high HUI class (> 13.5 SSU/ha) to have shallow top- and subsoils; to be stony in both horizons, and to have a high gravel (< 6 mm) cover (Figure 3).

Table 3 Variable loadings (eigenvectors) from the principle components analysis (PCA) of soil data for the first four components (axes)

Variable	AX_1	AX_2	AX_3	AX_4
Stone (> 2mm) content of the topsoil	-132	-425	227	863
Stone (> 2mm) content of the subsoil	-567	-474	662	-126
Topsoil pH	-1	104	74	-291
Subsoil pH	193	4	-64	-100
Electrical resistance of topsoil	-357	203	132	-482
Electrical resistance of subsoil	-220	252	37	-429
Clay in topsoil	316	88	26	233
Clay in subsoil	259	53	29	139
Topsoil depth	633	693	341	54
Subsoil depth	999	-43	3	-4
Gravel cover (< 6mm)	-429	5	76	-362
Stone cover (> 6mm)	255	-283	-92	499
Lime in topsoil	92	-55	-64	-32
Lime in subsoil	81	23	-59	-3
Proportion of total variance	**92**	**4.2**	**2.5**	**1.1**

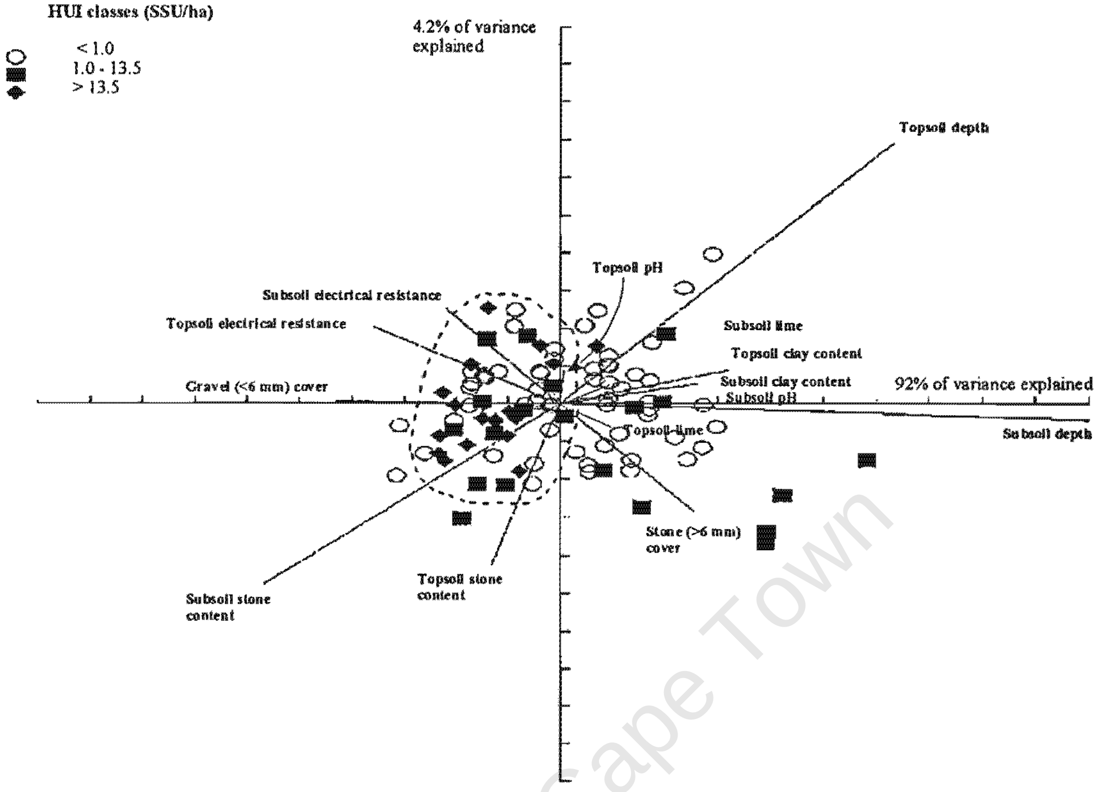

Figure 3 Quadrat ordination plot using principle component analysis (PCA) on 101 quadrats with 14 soil variables from Rietfontein farm. HUI classes are superimposed.

A plot of quadrat ordination using DCA on cover values of 29 plant species is shown in Figure 4. HUI classes are superimposed. The gradient of HUI classes is clear with low values on the left and high values on the right. Species which increase in abundance under high utilization intensity are *Galenia africana, Prenia tetragona, Zygophyllum retrofractum,* and *Delosperma* sp. (Figure 5). Figure 5 also shows that species which dominate in less utilized veld are *Crassula muscosa, Crassula subaphylla, Euphorbia karroensis, Ruschia spinosa* and *Antimima hantamensis.*

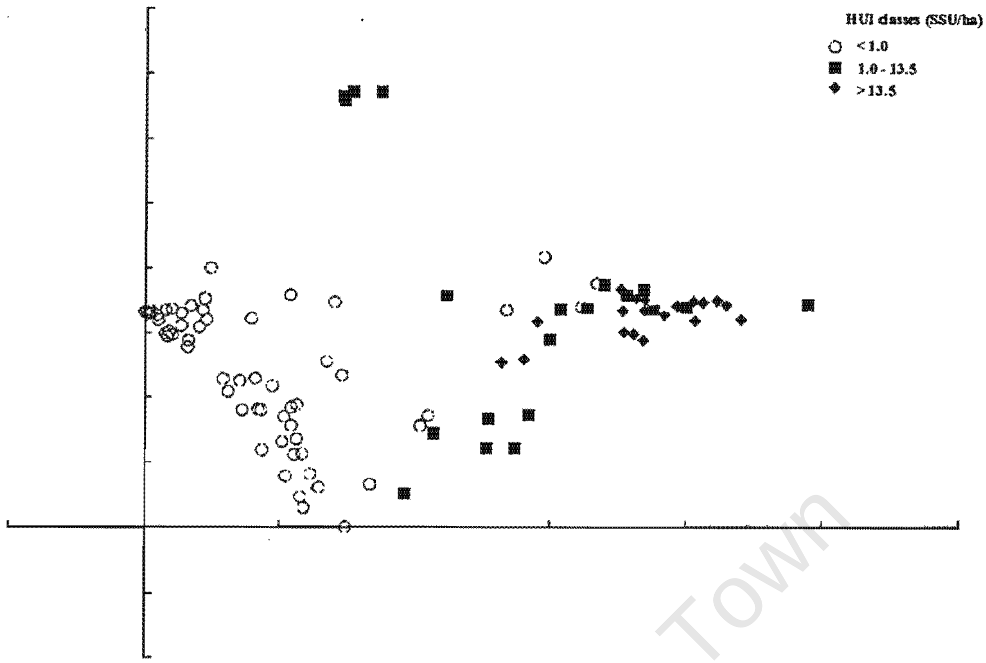

Figure 4 Quadrat ordination plot using detrended correspondence analysis (DCA) on 101 quadrats with 29 plant species from Rietfontein farm. HUI classes are superimposed.

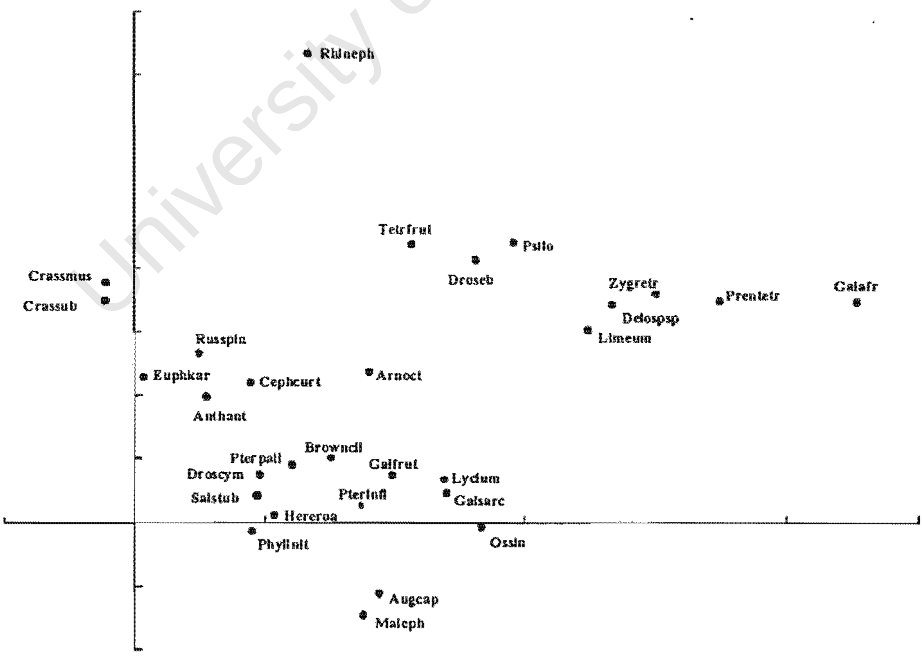

Figure 5 Species ordination plot using detrended correspondence analysis (DCA) on 101 quadrats with 29 plant species from Rietfontein farm. (See Table 1 for the key to the abbreviations).

Figure 6 presents the quadrat ordination plot using CCA on 14 soil variables and 29 plant species. HUI classes are superimposed, and soil variable gradients are indicated. Areas with high HUI values tend to have higher pH and lime levels and the soils are shallower and more stony. The CCA biplot of the species ordination diagram together with the soil variables (Figure 7) shows the species distribution very clearly, with various groupings of species emerging.

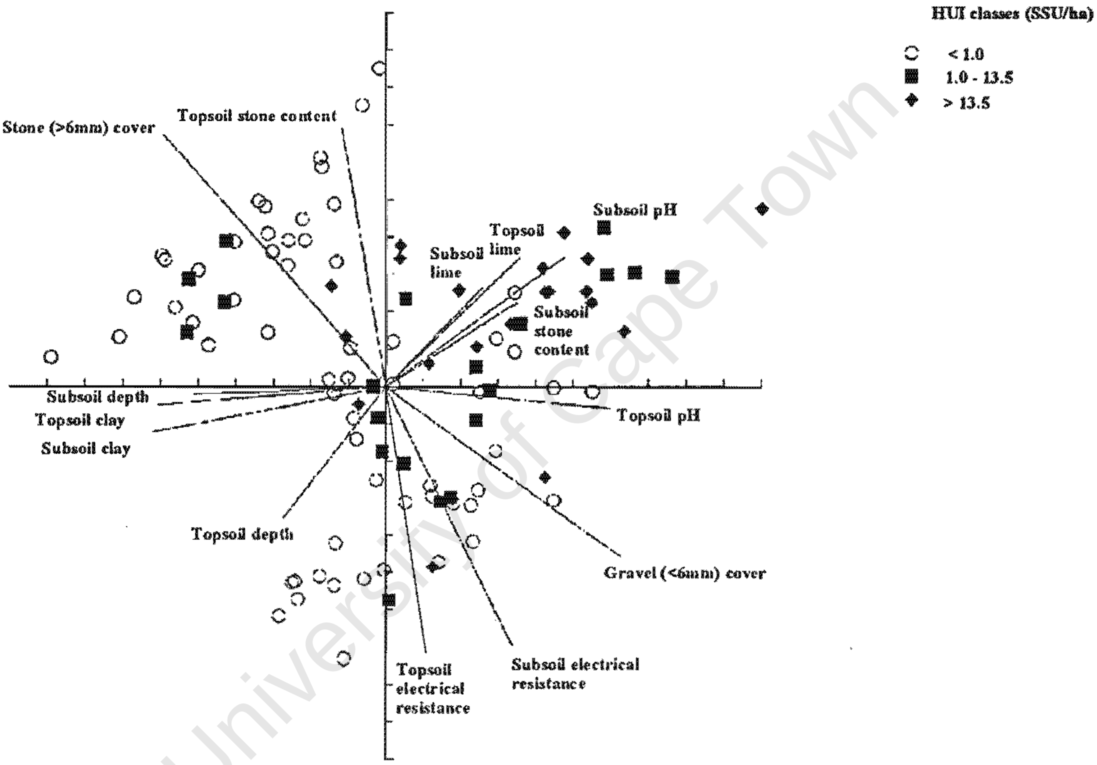

Figure 6 Quadrat ordination plot using canonical correspondence analysis (CCA) on 101 quadrats with 14 soil variables and 29 plant species from Rietfontein farm. HUI classes are superimposed.

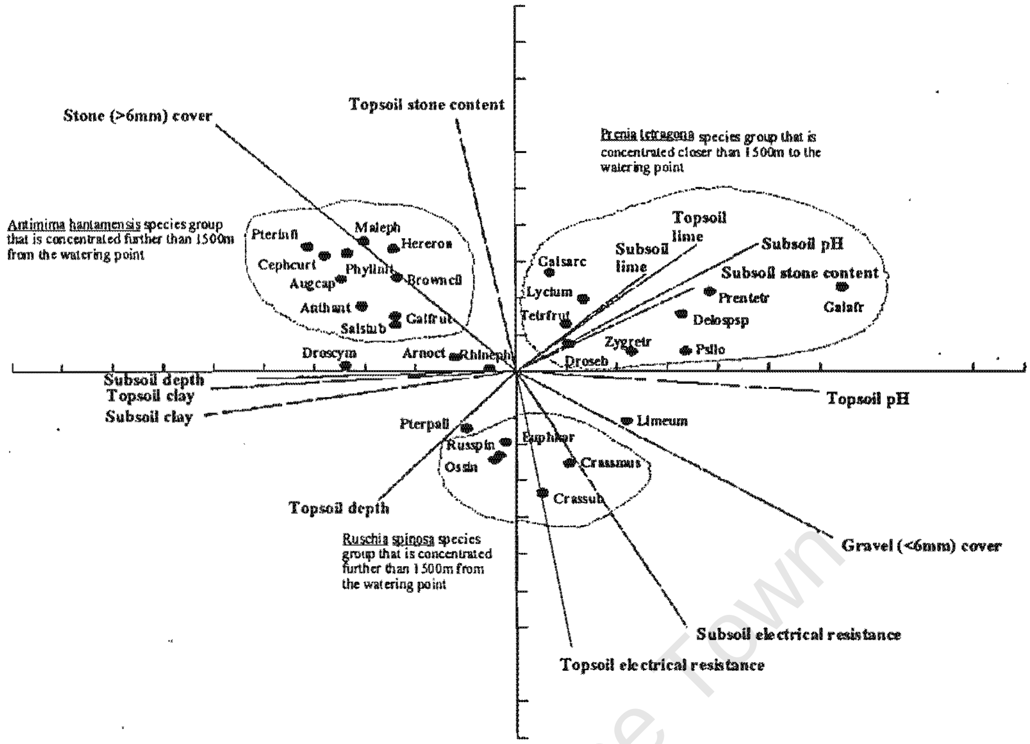

Figure 7 Species ordination plot using canonical correspondence analysis (CCA) on 101 quadrats with 14 soil variables and 29 plant species from Rietfontein farm. (See Table 1 for the key to the abbreviations).

Species associated with the *Prenia tetragona* group are *Galenia africana, Delosperma* sp., *Psilocaulon cf. dinteri, Zygophyllum retrofractum, Tetragonia fruticosa, Lycium afrum, Galenia sarcophylla,* and *Drosanthemum eburneum.* Species belonging to this group were concentrated closer than 1 500 m to the watering point. Further away from the watering point two species groupings can be identified. The one group comprises *Crassula subaphylla, Crussula muscosa, Euphorbia karroensis, Osteospermum sinuatum* and the dominant *Ruschia spinosa.* The other plant group which was found further than 1 500 m away from the watering point is more species-rich, and dominated by *Antimima hantamensis* with associated species like *Pteronia inflexa, Malephora crassa, Phyllobolus nitidus, Cephalophyllum curtophyllum, Hereroa odorata, Augea capensis, Brownanthus ciliatus, Galenia fruticosa,* and *Salsola tuberculata.*

Table 4 shows the relationships between HUI and plant and soil variables. There is evidence that high utilization intensities result in a decline in plant cover and species richness. Species like *Prenia tetragona* and *Delosperma* sp. increase while *Antimima* *hantamensis* and *Ruschia spinosa* decrease under high utilization intensities. The soils close to the watering point are shallower and more stony with a gravel (< 6 mm) cover, and pH and lime levels are higher. Table 5 shows the results of 41 selected soil samples classified into two HUI classes and analysed for chemical properties. The topsoil close to the watering point have lower levels of Ca, Mg, Na, and K with consequent higher electrical resistance compared to the topsoils further away from the water. The exchangeable sodium percent (ESP = Na as a proportion of total exchangeable bases) did not differ significantly between the two HUI classes.

Table 4 Mean values (± standard deviation) of soil and plant variables in three herbivore utilization intensity classes (SSU/ha). Significant differences among means was determined using one way parametric ANOVA (denoted by superscript A) and Kruskal-Wallis test (denoted by superscript B). Plant cover values were arcsine transformed. Means with similar superscript letters do not differ significantly.

	Herbivore utilization intensity classes (SSU/ha)				
Variable	**< 1.0** **n = 61**	**1.0 – 13.5** **n = 20**	**> 13.5** **n = 20**	**Test statistics**	**Significance**
Total plant canopy cover^{A}	13.2 ± 5.1^{a}	8.0 ± 4.5^{b}	7.2 ± 3.7^{b}	16.066	* *
Plant species richness^{A}	9.1 ± 3.7^{a}	7.4 ± 2.1^{ab}	6.2 ± 2.2^{b}	6.822	* *
Cover of *Prenia tetragona*^{A}	0.1 ± 0.3^{a}	0.9 ± 1.3^{b}	2.0 ± 1.8^{c}	25.498	* *
Cover of *Delosperma* sp.^{A}	0.4 ± 1.2^{a}	1.5 ± 1.7^{ab}	2.9 ± 3.2^{b}	13.958	* *
Cover of *Antimima hantamensis*^{A}	1.3 ± 1.8^{a}	0.01 ± 0.02^{b}	0^{b}	9.499	* *
Cover of *Ruschia spinosa*^{A}	6.8 ± 6.7^{a}	0^{b}	0^{b}	62.961	* *
Topsoil $pH_{(water)}^{A}$	7.9 ± 0.6^{a}	8.2 ± 0.46^{b}	8.16 ± 0.35^{ab}	3.891	*
Subsoil $pH_{(water)}^{A}$	7.3 ± 1.15^{a}	7.8 ± 0.85^{ab}	8.0 ± 0.66^{b}	4.010	*
Stone (> 2mm) in topsoil (%)^{A}	16.7 ± 7.8	14.2 ± 6.5	18.4 ± 7.9	1.558	NS
Stone (> 2mm) in subsoil (%)^{A}	20.2 ± 11.9^{a}	23.5 ± 17.1^{ab}	30.8 ± 13.5^{b}	4.583	*
Topsoil depth (mm)^{A}	50.3 ± 18.7^{a}	43.3 ± 13.1^{ab}	38.8 ± 10.2^{b}	4.157	*
Subsoil depth (mm)^{A}	213.4 ± 56.1^{a}	233.5 ± 87.4^{a}	157.0 ± 28.3^{b}	9.031	* *
Electrical resistance of the topsoil (log ohm)^{A}	2.44 ± 0.62	2.57 ± 0.53	2.70 ± 0.27	1.777	NS
Electrical resistance of the subsoil (log ohm)^{A}	2.50 ± 0.50	2.64 ± 0.32	2.69 ± 0.33	1.684	NS
Presence of lime in topsoil (1 = present; 0 = absent)^{B}	43.8	54.3	69.5	16.244	* *
Presence of lime in subsoil (1 = present; 0 = absent)^{B}	44.7	51.8	69.5	14.824	* *
Gravel cover (< 6mm) (1 = present; 0 = absent)^{B}	38.6	62.3	77.5	41.213	* *
Stone cover (< 6mm) (cover classes: 0; 1; 2; 3; 7)^{B}	57.3	49.9	33.0	11.134	* *

* * P < 0.01
* P < 0.05
NS not significant

The species listed in Table 6 are significantly correlated with distance from the watering point in three species groups. If distance from the water can be regarded as related to grazing pressure and therefore veld condition, these species could be valuable indicators of veld condition.

Table 5 Mean values (± standard deviation) of soil variables in two herbivore utilization intensity classes (SSU/ha). Data from 41 selected soil samples which were analysed for chemical properties. Significant differences between means was determined using one way parametric ANOVA

	Herbivore utilization intensity classes			
Soil Variable	**> 1.0 SSU/ha** (< 1500m from watering point) **n = 16**	**< 1.0 SSU/ha** (> 1500m from watering point) **n = 25**	**Test statistics**	**Significance**
Topsoil $pH_{(KCl)}$	7.64 ± 0.6	7.52 ± 0.7	0.296	NS
Subsoil $pH_{(KCl)}$	7.01 ± 0.9	6.92 ± 1.4	0.053	NS
Topsoil electrical resistance (log ohm)	2.70 ± 0.2	2.36 ± 0.5	6.211	*
Subsoil electrical resistance (log ohm)	2.42 ± 0.2	2.32 ± 0.4	0.832	NS
Clay in topsoil (%)	16.5 ± 4.6	13.6 ± 4.9	3.215	NS
Clay in subsoil (%)	26.5 ± 6.1	23.5 ± 7.1	1.882	NS
Exchangeable sodium in topsoil (%)	3.1 ± 1.8	3.5 ± 1.9	0.282	NS
Exchangeable sodium in subsoil (%)	4.96 ± 3.1	5.0 ± 2.9	0.005	NS
Ca in topsoil (me/100g)	0.19 ± 0.17	1.24 ± 2.06	3.885	*
Ca in subsoil (me/100g)	0.46 ± 0.39	0.85 ± 1.13	1.585	NS
Mg in topsoil (me/100g)	0.03 ± 0.03	0.52 ± 0.68	7.220	*
Mg in subsoil (me/100g)	0.07 ± 0.07	0.39 ± 0.49	5.679	*
Na in topsoil (me/100g)	0.21 ± 0.14	0.86 ± 0.88	7.330	*
Na in subsoil (me/100g)	0.43 ± 0.27	0.87 ± 0.83	3.650	NS
K in topsoil (me/100g)	0.01 ± 0.004	0.03 ± 0.02	9.181	* *
K in subsoil (me/100g)	0.009 ± 0.004	0.016 ± 0.02	1.822	NS

* * $P < 0.01$
* $P < 0.05$
NS not significant

Table 6 Ecologically important species for Rietfontein and their significance as indicator species in three groups. Direction of correlation with distance from the watering point is shown in brackets.

	Prenia tetragona	*Ruschia spinosa*	*Antimima hantamensis*
Malephora crassa	High (+)	-	-
Rhinephyllum macradenium	High (+)	-	High (-)
Antimima hantamensis	-	-	High (+)
Delosperma sp.	-	-	High (-)
Ruschia spinosa	-	High (+)	High (+)
Drosanthemum eburneum	-	High (-)	-

DISCUSSION

The Rietfontein piosphere

Sampling along three belt transects radiating from a single watering point in the Ceres Karoo constitutes a pseudoreplicated sampling design. This limits the study to being a descriptive one where hypotheses on grazing impacts, soils, and vegetation patterns could be generated. The main hypothesis that the soils close to the watering point (< 1500 m) have been changed permanently by the effects of concentrated large herbivore activity over many decades is supported by the results of a number of workers (Roux & Opperman 1986; Fuls 1992; Ludwig & Tongway 1995; Landsberg *et al.* 1997). The decline in plant cover (Cowling 1986; Roux & Theron 1987) together with hoof action loosening the topsoil (Roux & Opperman 1986) have resulted in soils being readily washed and blown away (Acocks 1975; Roux & Opperman 1986). Several results point to the loss of topsoil close to the watering point. The soils are shallower and because of the smaller volume of soil the stone (> 2 mm) content per unit volume of soil is higher. As a result of the loss of topsoil the saprolite in the subsoil is exposed on the soil surface as gravel (< 6 mm) cover. Removal of the nutrient rich topsoil has also resulted in lower levels of Ca, Mg, Na and K with consequent higher soil electrical resistance. The higher lime levels close to the watering point could be related to topographical features with lime being leached towards the lower lying spring. However, this is more likely the result of topsoil erosion as a result of grazing impacts on vegetation cover. Roux and Opperman (1986) reported on an experiment on karoo plains where continuous grazing was applied for 25 years and where the orthic A horizon of about 15 cm deep was eroded away, leaving exposed a B horizon rich in lime. Perkins and Thomas (1993) measured an increase in pH, nitrate, and phosphate within a 50 m radius from boreholes in the Kalahari Desert region of Botswana. They attributed this to the concentration of livestock urine and dung. However, utilization intensities have to be extreme to register any such increases (Perkins & Thomas 1993). The marginally higher pH closer to the watering point in this study is probably more related to the accumulation of lime than to dung and urine concentration.

The plant species which characterize the vegetation close to the watering point (*Prenia tetragona*, *Galenia africana*, *Zygophyllum retrofractum*, and *Delosperma* sp.), are all positively correlated with herbivore utilization intensity. These are pioneer species which increase under high levels of continuous selective defoliation, trampling and dunging, and survive on shallow, lime-rich, and nutrient-poor soils. The development of this less-palatable, species-poor, and sparse community in the inner 1500 m zone around the watering point is in agreement with other workers who found that grazing impacts are greatly diminished beyond 1 – 2 km from a watering point (Foran 1980; Perkins & Thomas 1993; Fusco *et al.* 1995). Studies by Landsberg *et al.* (1997) around artificial water sources in Australia also showed that patches close to water are characterized by a decrease in palatable perennial plants, an increase in unpalatable plants, particularly shrubs, and an increase in the proportion of bare ground where the soils are different and have lost their ability to capture and store water and nutrients. More studies have shown a decrease in plant cover (Thrash *et al.* 1993) and species-richness (Tolsma *et al.* 1987), and an increase in the abundance of short-lived plants (Andrew & Lange 1986; Thrash *et al.* 1993) around watering points regularly used by large herbivores. Plant species that increase towards the watering point are often exotics (Landsberg *et al.* 1997) like *Atriplex lindleyi* and *Salsola kali* in this study (pers. obs.). These fugitive, short-lived plants may create a different habitat and act as important food plants for wildlife (Fusco *et al.* 1995) e.g. quail and sandgrouse. This might be seen as an advantage coming from the vegetation changes around watering points. However, this pioneer community has a low biomass and water use efficiency (Le Houérou 1984) with a consequent reduced year-round productivity (Fuls 1992).

A noteworthy finding is the abundance of the highly palatable shrub, *Limeum aethiopicum*, on sites of high utilization intensity close to the watering point. Similarly Perkins & Thomas (1993) found in their study in the Kalahari desert of Botswana that the highly nutritious perennial grass, *Digitaria eriantha*, remained abundant close to the boreholes. These palatable plant species are probably adapted to cope with regular defoliation as well as survive harsh environmental conditions.

The mechanism underlying the degradation of the ecosystem close to the watering point with the consequent loss of topsoil and development of bare areas, is probably related to the breakdown of the fertile patch matrix of the karoo veld (Yeaton & Esler 1990). Fertile patches are areas where resources like soil, water and litter accumulate and may be due to terrain features (e.g. flats or depressions on gentle slopes and plains) and vegetative and other biological features (e.g. animal diggings). The fertility of these patches are high because they are "top-dressed" from time to time with litter, topsoil and water which blow or flow into the patch from inter-patch areas (Tongway 1994). These features reduce the net transporting or eroding capacity of wind and water by slowing flows, causing turbulence or creating a tortuous path (Tongway 1994). Water in particular is retained in the patch by shading and mulching. Soil-living fauna such as lizards, spiders, ants, termites and springtails commonly live in fertile patches because of the more favourable micro-climate and plentiful food supply. Their hole-forming activities aerate the soil and improve infiltration rates for rainfall (Tongway 1994). There is a positive feedback in that the denser the fertile patches and the better their condition, the more effective the ecosystem is in trapping and retaining scarce resources (Tongway 1994).

With frequent concentrated animal activities around the watering point, floristic composition changed (Hoffman & Cowling 1987; Roux & Theron 1987; Milton 1990; Milton *et al.* 1994) with amongst others the loss of mound-building mesembs (Yeaton and Esler 1990; Esler *et al.* 1992) like *Ruschia spinosa, Malephora crassa, Brownanthus ciliatus,* and *Rhinephyllum macradenium*. This together with a reduction in total plant biomass remaining on the land altered the nature of the protection of the soil (Roux & Oppermann 1986; Bosch 1987). The size, number, spacing, and efficiency of fertile patches were altered with a consequent leaking of soil, nutrients, and water from the system (Fuls 1992; Tongway 1994; Ludwig & Tongway 1995). As electrolytes (this study) and organic matter are lost and clay levels in the surface layers increase, mechanical dispersion of clay caused by raindrop impact is a major factor causing crust formation (Greene & Tongway 1989). This, together with the loss of cryptogamic material on the surface (Graetz & Ludwig 1978; Andrew & Lange 1986), results in a dense, compacted surface layer with low infiltration of water (Roux & Opperman 1986; Greene & Tongway 1989; Thurow *et al.* 1988; Fuls 1992), and more run-off and therefore leaking of resources (Tongway 1994; Landsberg *et al.* 1997). As the degradation continues, the small erosion cells or source areas link up to form large cells which feed water, soil and nutrients into a few large sinks (Pickup 1985). These large erosion cells may become totally devoid of vegetation with the compacted surface forming extremely unfavourable conditions for seed entrapment, germination and establishment (Walters 1951; Roux & Theron 1987; Stafford Smith & Pickup 1990). Relatively high exchangeable sodium percent (this study), hoof action of livestock and splash erosion contributes to the break down of the soil structure, de-flocculation of the clay fraction, and sealing of the surface layer (Walters 1951; Greene & Tongway 1989).

Jeltsch *et al.* (1997) modelled shrub encroachment around artificial waterholes in the semi-arid Kalahari, and argue that rainfall and grazing pressure are important factors which influence the rate of formation and growth of these piosphere zones. This raises the question whether degraded veld close to a watering point will recover by removing livestock activities and resting it for an indefinite period. Although there are still several extensive bare areas near the watering point, G. Du Plessis (pers. comm.) states that there has been a perceivable improvement in plant cover since the fencing of the farm in the 1970's. This is probably due to some form of rotational grazing system which allowed for resting periods. The key plant species which are most likely responsible for this increase in cover are the short lived *Prenia tetragona,* and the longer lived, unpalatable *Galenia africana*. *P. tetragona* is well adapted to shallow, stony soils with high levels of lime and is probably a very important factor in the rehabilitation of this degraded veld to a more productive, species-rich vegetation. Many believe that revegetation is not possible with a loss of topsoil and changes in physical factors such as soil infiltration and nutrient status (Acocks 1975; Cowling 1986; Roux & Opperman 1986; Roux & Theron 1987; Tongway 1994). Because it involves species losses and therefore changes in facilitative and competitive interactions (Fowler 1986; Milton *et al.* 1994), these transitions in arid rangelands are not always reversible (Westoby 1980; Westoby *et al.* 1989; Friedel 1991; Fuls & Bosch 1991; Laycock 1991). Pioneer plants like *Prenia tetragona* are valuable in re-establishing the small-scale fertile patches which is necessary to slow down the cascading effects of resource losses by acting as natural filters (Ludwig & Tongway 1995) and as microhabitats for new plants (Danin 1978; Eckert *et al.* 1986) and beneficial soil-dwelling animals (Milton and Dean 1992). Together with mechanical techniques for rehabilitating degraded soils (Walters 1951; Fuls 1992) adapted plants have to be promoted by reseeding (Friedel 1991), and creating favourable conditions for establishment (see Chapter 2).

Indicators of change

Monitoring the complex impacts of the disturbances associated with grazing which radiates out from a source of water requires useful indicators (Landsberg *et al* 1997). Landsberg *et al.* (1997) found in their study of Australian piospheres that species composition and abundance of understorey plants, birds, and ants were easy to measure and also sensitive to grazing intensity. If these indicators of change are relatively efficient to sample and if they co-vary with changes in other biotic groups (Landsberg *et al.* 1997) they could be useful for the selection of one or a combination of "key" species. This could reduce the subjectivity in choice of "what" to measure as part of range assessment and monitoring (Graetz & Ludwig 1978; Schmidt *et al.* 1994).

Prenia tetragona and *Ruschia spinosa* are the two dominant plant species which are sensitive to one of the main environmental parameters, distance from the water, affecting veld composition in this study site. *Prenia tetragona* shows an increase in abundance closer to the watering point, together with species like *Zygophyllum retrofractum, Delosperma* sp., and *Galenia africana*. The abundance of these key species is a good indicator of poor veld condition as can be seen around the watering point (B. Bayer pers. comm.). The abundance of *Ruschia spinosa, Antimima hantamensis, Phyllobolus nitidus, Hereroa odorata, Pteronia inflexa*, and *Euphorbia karroensis* increases further away from the watering point and can therefore be regarded as indicators of veld subjected to lower levels of grazing pressure. *Ruschia spinosa* is a key species not only because of its abundance in the Succulent Karoo , but it forms the bulk of sheeps' diet during the dry summer when a small rainfall event may cause the leaves of this succulent shrub to swell (Le Roux *et al* 1994; Shearing 1994; I. De Jong pers. comm.). Many believe that this plant is expanding its range in the Succulent Karoo because of its spinescent nature and therefore ability to survive increased grazing pressure (Hoffman & Cowling 1987; Roux & Theron 1987; Le Roux *et al.* 1994). It is debatable whether the large tracks of land further away from the watering point dominated by mature, woody individuals of *Ruschia spinosa* should be labelled "good condition" veld (B. Bayer pers. comm.). This is a species-poor community which is probably in a stable state (Hoffman & Cowling 1987; Westoby *et al.* 1989) and will require a special input to alter species composition (George *et al.* 1992) (see Chapter 2). Milton (1990) shows that a spinescent *Ruschia* species was less common on heavily grazed compared to moderately grazed karoo plains. This species can therefore disappear from heavily grazed areas, such as around watering points, with possible further cascading species losses. Mounds develop around species of Mesembryanthemaceae such as *Ruschia* as a result of the capture of wind and water transported soil and organic material at their bases. The stems and branches of the canopies are sparsely packed so that light levels beneath the canopy are relatively high. These plants provide ideal refuges for palatable woody shrubs such as *Osteospermum sinuatum* and *Tetragonia* spp. (Yeaton & Esler 1990; pers. obs.).

A key species which could be of value for monitoring veld condition is *Malephora crassa*. This creeping mesemb is relatively resistant to heavy defoliation (Milton 1990) but does decline under extreme utilization intensities (Table 6). *Rhinephyllum macradenium* is probably an intermediate species in a successional sequence (Esler *et al.* 1992) and dominates sites of intermediate grazing intensity (Figure 7 and Table 6). Another mat-forming mesemb which shows a similar pattern is *Drosanthemum eburneum* (Figure 7).

CONCLUSIONS

The results of this study confirm the findings of many workers that heavy grazing within 2 km of a central watering point removes palatable species (e.g *Ruschia spinosa,* and *Antimima hantamensis*) and those species sensitive to trampling damage, which allow "increaser" species to establish. These increasers are typically species with ephemeral life histories (e.g. *Prenia tetragona*) which flourish after rain, or unpalatable perennial shrubs (e.g. *Galenia africana*). Changes in vegetation also have an impact on soil properties. With the decline in cover and loss of species the fertile patch matrix of the veld breaks down which leads to accelerated erosion and the consequent redistribution of water and nutrients in the landscape. Soils of continuously overgrazed areas are therefore shallow, stony, and nutrient poor.

Apart from grazing induced soil changes which play an important role in determining plant patterns, other soil variables which are important in the Succulent Karoo are texture, abundance of stone in the profile and on the surface, pH, lime, and soil nutrient levels. These physical and chemical soil properties relate to moisture and nutrient availability which is primarily the cause of vegetation pattern. An understanding of how grazing animals alter the water and nutrient flows on a patch scale will be essential for developing management techniques aimed at conserving the fertile patch matrix of the Succulent Karoo.

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CHAPTER 2

AN EVALUATION OF SOME RESTORATION STRATEGIES FOR THE SUCCULENT KAROO

INTRODUCTION

The degradation of arid and semi-arid rangelands is a worldwide phenomenon (Call & Roundy 1991; Milton & Dean 1995; Ludwig & Tongway 1996). Degradation is generally evident in a decline in productivity at all spatial scales and has a direct bearing on the capacity of the rangeland to support grazing animals and to provide a sustainable income to the landowner. One model, which attempts to conceptualize this degradation process, was presented by Westoby *et al.* (1989). In this state-and-transition model of vegetation change there is a set of discrete states of vegetation and a set of discrete transitions between states, rather than a steady vegetation change along a single continuum. It follows that in order to reverse degradation it may be necessary to force the system to jump, or cross a threshold from one stable state to another (Hobbs & Norton 1996). Management input may be required to reduce the leaking of resources (soil, moisture, nutrients) (Whisenant & Tongway 1995), restore propogule pools (Milton 1994), and establish the belowground components of the ecosystem, including a healthy soil microbial community (Whitford *et al.* 1988; Zink & Allen 1998).

Milton *et al.* (1994) developed a state-and-transition model for the Karoo that invokes stepwise degradation, starting with plant demographic changes, followed by plant and animal losses, and eventually major changes in soil condition and processes. According to the model grazing animals and their hooves alter the outcome of competitive interactions between plants, differentially influence the reproductive output of forage and nonforage species, and influence microsite availability for seedling establishment (Milton & Hoffman 1994). Milton & Hoffman (1994) further argue that there are important interactions between climatic factors and transition processes, which need to be understood for successful restoration of karoo rangelands. The challenge is to reverse or alter the course of this stepwise degradation process, bearing in mind that the more degraded the land, the greater the management and costs required to return the system to previous states (Milton *et al.* 1994; Hobbs & Norton 1996).

The Ceres Karoo (also known as the Tanqua Karoo) is part of southern Africa's winter-rainfall succulent karoo biome (Milton *et al.* 1997). The region has been subjected to occasional but heavy grazing bouts in winter by nomadic pastoralists for at least 2000 years (Rubin 1998). Large herds of springbok (*Antidorcas marsupialis*) migrated from the summer-rainfall Nama Karoo to the east into the Ceres Karoo during the winter months when food was relatively abundant (Skinner 1993). These pulsed grazing events probably had limited impact on the vegetation and soils of the region. Degradation and accelerated soil erosion were initiated by heavy continuous grazing when graziers permanently occupied the region since the late 19th century (Rubin 1998). Today, much of the Ceres Karoo is in a seriously degraded state (Mackay & Zietsman 1996). Stokes (1994) postulates that much of this vegetation change has occurred since the 1940's, but since 1971, after the introduction of a stock withdrawal programme, no further degradation trends were evident. The plant communities dominated by ephemerals and plants of low forage value are in a stable state, with no, or very little, recovery even with rests of up to 50 years (Stokes 1994).

Possible constraints on the passive recovery of bare areas and poor veld in the Karoo include inadequate supply of seed, availability of suitable microsites for plant establishment, altered soil properties, and the truncation of key soil biotic processes (Milton & Hoffman 1994; Stokes 1994). Here we investigate the possibility of kick-starting the restoration of bare areas by soil surface treatments with gypsum and/or organic mulch. We also apply an exogenous seed source to test the hypothesis that seed availability limits autogenic recovery (Whisenant & Tongway 1995). Gypsum improves the hydraulic conductivity of the topsoil (Ilyas *et al.* 1993) while mulching improves infiltration (Ludwig & Tongway 1996), and the frequency of favourable microsites (Abusuwar 1995; Milton 1995). We also test the hypothesis that the severe pruning of moribund, woody shrubs in ungrazed karoo veld will result in increased resource availability and therefore increased seedling establishment and vigour of existing plants. The drastic disturbance of shrub cutting might be necessary to kick-start a transition from a species-poor, unpalatable state to a more diverse state with a higher grazing value.

STUDY AREA

Research was undertaken on the farm Rietfontein (4074 ha, 32°52'S/19°54'E), situated 90 km north-east of the town of Ceres in the Ceres Karoo. See the General Introduction for more detail on the study area. The dominant plant community in the vicinity of the restoration trials for this study corresponds to the *Zygophyllum microcarpum – Lycium cinereum* open plains community described by Rubin (1998) for Tankwa Karoo National Park, 50 km north of Rietfontein The vegetation where the defoliation experiment was conducted corresponds to Rubin's (1998) *Rhinephyllum macradenium – Ruschia spinosa* succulent dwarf shrubland community of the low-lying Dwyka shales at TKNP (Rubin 1998).

METHODS

Gypsum-mulch-reseeding experiment

In April 1994 four unvegetated bare areas were identified within 400 m of the perennial Rietfontein spring, an area that had been subjected to heavy continuous grazing for many years. The bare areas selected were subjectively assessed as similar in terms of size (ca 0.1 ha), slope (< 5 %), soil surface condition (bleached A horizon), gravel cover (> 80 %), and surrounding plant community (*Prenia tetragona* – dominated). A 10m x 10m plot, subdivided into eight blocks (5 m x 2.5 m), was demarcated on each bare area, and in a randomized design, a zero treatment, and different combinations of gypsum, mulch and reseeding treatments were allocated to each block. The corners of the blocks were clearly marked with steel pegs, and furrows (ca. 7 cm deep) were dug around the blocks to reduce the fluvial movement of gypsum across the blocks. An equivalent of 5 $t.ha^{-1}$ gypsum ($CaSO_4$) was hand strewn onto designated blocks and lightly raked into the topsoil to reduce wind dispersion. All remaining blocks were subsequently lightly raked to avoid bias. In the blocks designated for reseeding, three lines, 600 mm apart were marked with steel pegs and furrowed to approximately 1cm depth. Approximately 2000 seeds of *Ruschia spinosa* (ca. 50 % viable) were hand sown along one furrow, 1800 *Chaetobromus dregeanus* seeds (ca. 50 % viable) along another, and 600 *Osteospermum sinuatum* seeds (ca. 33 % viable) along the third furrow. The furrows were closed and lightly compacted by hand.

These three species were chosen in an attempt to rebuild a productive community from an agricultural perspective. *Ruschia spinosa* (hereafter *Ruschia*) is a pioneer, mound-forming shrub that can establish in the open (Yeaton & Esler 1990), *Osteospermum sinuatum* (hereafter *Osteospermum*) is a winged, wind-dispersed and highly palatable species favouring more sheltered microsites (Milton 1995), and *Chaetobromus dregeanus* (hereafter *Chaetobromus*) is a palatable perennial grass that was thought once to be more common in the Ceres Karoo (B. Bayer pers. comm.). An organic mulch in the form of thatching reed (Restionaceae), derived from discarded roofing material, was spread onto designated blocks. *Acacia karroo* branches were stacked on top of the reeds to minimize relocation by wind. Thatch was used as mulch because it is unlikely to contain any seeds which might confound the results of the experiment. The 10 m x 10 m plots were fenced with a 60 cm high mesh wire to exclude steenbok, tortoises, hares, and rodents.

In June 1994, three months after the treatments and the morning after a 6 mm rainfall event, infiltration depth was determined in each block. Five vertical soil profiles per block were dug and the depth of water infiltration visually assessed by looking at soil colour. The difference between moist and dry soil was easily observed, and the depth of infiltration could be measured to the nearest millimetre.

The monitoring of seedling abundance started in October 1994 and continued with at least one visit per year until a final assessment in December 1998. During each visit the abundance of live seedlings and established plants was determined. All seedlings and established plants in the marked lines were attributed to the exogenic seed source (i.e. reseeded). After a severe drought in 1995, when the majority of seedlings succumbed, reseeding of similar volumes of seed along the same lines was repeated in April 1996. This was done to extend the experiment.

Also during each visit the seedling and/or established plant volume originating from the endogenic seed source was subjectively assessed and scored for each block. In assessing the plant volume per block, the horizontal and vertical projections of the phytomass were considered. Blocks were scored for phytomass volume on an eight point scale where the block per plot with the largest volume received a score of eight. Because these scores were independent of the exogenic seed source, the scores for the blocks, reseeding, was not considered as a treatment in the analysis. No attempt was made to assess the species composition of the seedling cohorts but casual observations were made on dominant ephemeral and perennial plant species, size and vigour of plants, endogenic seed abundance and distributions, as well as invertebrate activities.

The highest seedling and surviving plant counts were recorded in April 1995, October 1996, and June 1997. The count data for these three survey dates were square root transformed and the effects of treatments analysed by means of parametric two-way analyses of variance. The plant volume scores for each survey date were subjected to non-parametric Kruskal-Wallis analyses of variance by ranks to determine treatment effects.

Cutting experiment

In April 1994 a homogenous patch of *Ruschia spinosa* – dominated veld was identified about 2.5 km away from the central watering point at Rietfontein. The size of the *Ruschia* mounds and the moribund canopies indicated that this patch was seldom utilized by livestock, mainly because of the distance from the water. A plot 10 m x 25 m was demarcated and subdivided into 10 blocks (5 m x 5 m). The corners of the blocks were permanently marked. Five blocks were randomly selected as the control whereas the other five received the cutting treatment. All the plants in the latter blocks were cut with a brushcutter to a height of ca. 50 mm. The larger portions of the cut plants were removed from the blocks, although a substantial amount of plant debris remained on the ground.

In October 1994 I estimated the percentage litter cover, counted the number of flowering *Crassula subaphylla* plants per block, counted the number of *C. subaphylla* plantlets in a randomly positioned 2890 cm^2 frame, and counted the number of growth tips per 217 cm^2 of *C. muscosa* canopy. Both *C. subaphylla* and *C. muscosa* are dominants in this community and both were virtually always found on *Ruschia* mounds.

During the April 1995 visit it was realised that the abundance of *Crassula subaphylla* plantlets in the cut blocks was clearly an artefact of the brushcutting, with many leaves breaking off and rooting on contact with the ground. It was decided to exclude *C. subaphylla* when counting the number of seedlings per 2890 cm^2 of mound (raised soil underneath a shrub canopy) and intermound microhabitats. After good autumn rains both *Ruschia* and *C. muscosa* showed signs of active growth and the number of light green, young leaves per 723 cm^2 of *Ruschia* canopy, and the number of growth tips per 217 cm^2 of *C. muscosa* canopy were counted.

With fewer seedlings present in July 1996, each full block (25 m^2) was searched for mound and intermound seedlings. After a relatively long, dry spell the average percentage of a *Ruschia* canopy which appeared dead was estimated, and the number of *Ruschia* shrubs which showed any sign of recent growth was expressed as a percent of the total number of *Ruschia* plants per block. Young growth on *Ruschia* plants is easily identified, as the stems are a pinkish colour and the young leaves a light green.

In October 1996 mound and intermound perennial seedlings were again counted covering the full blocks (25 m^2). With the good winter rains of 1996 there was an abundance of annuals. The number of annual plants per block was counted. The numbers of *Ruschia* canopies with flowers and young growth were counted and expressed as a percent of the total number of *Ruschia* plants per block.

During a final visit to this experiment in December 1998 the number of perennial plant species per block was counted, as well as the number of young leaves per 723 cm^2 of *Ruschia* canopy.

The measurements taken during the different surveys were analysed for treatment effects by means of two-tailed *t*-tests.

RESULTS

Gypsum-mulch-reseeding experiment

The deepest rain water infiltration occurred in the gypsum treatments ($P < 0.001$), although mulching also improved infiltration significantly ($P < 0.05$), and showed a significant interaction with gypsum ($P < 0.05$) (Table 1).

Table 1 Infiltration depth (mm) after a 6 mm rainfall event on bare soils under different treatments. n = 40.

Treatment	Mean	Std. Error	F	Significance level
Control	34.6	2.4		
Mulch	46.8	3.6	6.5	.0117
Gypsum	53.4	2.1	25.5	.0000
Mulch + Gypsum	54.8	2.3	4.1 (interaction)	.0440

No seedlings of *Ruschia*, *Chaetobromus*, or *Osteospermum* were observed in October 1994 after a very dry winter. After good rains in March 1995 (40 mm) seedlings of all three reseeded species emerged the following April (Figure 1). Of the three reseeded species, the highest number of seedlings per 1000 viable seeds sown was observed for *Osteospermum*. For all three species the highest number of seedlings were observed in the gypsum plus mulch treatment (Figure 1). Mulching, and, in the case of *Osteospermum*, gypsum treatment resulted in significantly higher numbers of seedlings compared to controls (Table 2). The winter of 1995 was very dry and most established seedlings had perished by the following summer. The good rains in December 1995 (45 mm) resulted in very little establishment since few seedlings were counted in January and April 1996. Reseeding in April 1996 was followed by good winter rains. Despite this, few seedlings had emerged in mid-winter (July 1996), but a relatively good emergence was recorded in spring (October 1996), especially for *Osteospermum* (Figure 1). For both *Ruschia* and *Osteospermum*, significantly more seedlings were counted in blocks where the soil was treated with either gypsum or mulch, or a combination of the two (Figure 1, Table 2). For the rest of the study period no seedlings of the two small-seeded species, *Ruschia* and *Chaetobromus,* were observed. The larger seeded *Osteospermum* showed another good germination and emergence event in late autumn – early winter (June 1997) (Figure 1). The highest seedling abundance was recorded in the gypsum plus mulch treatment. However, the gypsum treatment did not differ significantly from the control (Table 2).

Table 2 Results of two-way analyses of variance of seedling counts of three species reseeded under two treatments. Count data were square root transformed

		Mulch		Gypsum		Interaction	
Survey date	**Species**	**F-value**	**P-value**	**F-value**	**P-value**	**F-value**	**P-value**
April 1995	Ruschia	10.0	0.008	2.69	0.127	2.69	0.127
April 1995	Chaetobromus	17.19	0.001	0.97	0.344	0.97	0.344
April 1995	Osteospermum	21.05	<0.001	7.74	0.017	7.74	0.017
October 1996	Ruschia	8.63	0.012	1.76	0.21	0.12	0.74
October 1996	Chaetobromus	0.1	0.752	1.25	0.285	0.37	0.554
October 1996	Osteospermum	3.79	0.075	11.93	0.005	0.07	0.79
June 1997	Osteospermum	6.62	0.024	0.48	0.504	0.23	0.64

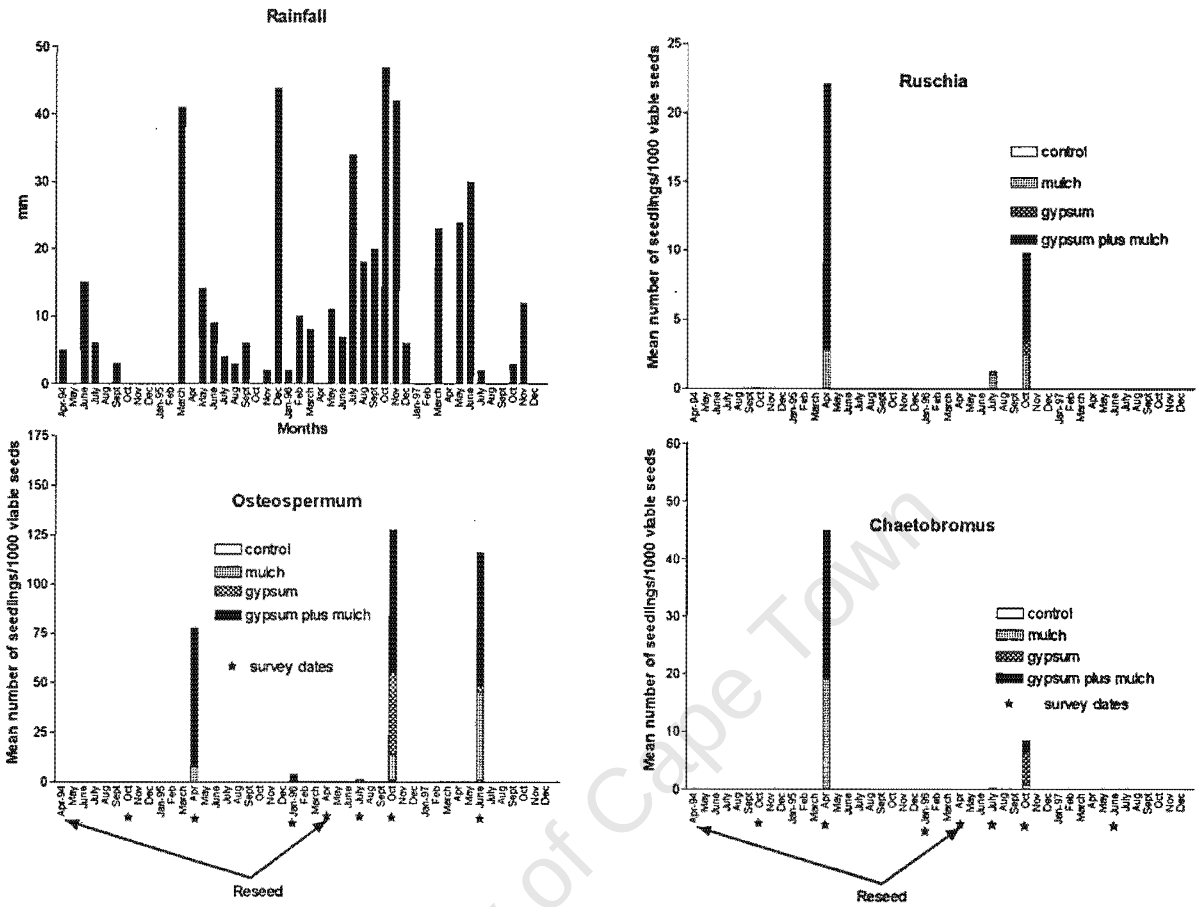

Figure 1 Mean number of seedlings counted for three reseeded species in different treatments. Monthly rainfall is also shown.

The survival of the cohorts of seedlings that emerged from October 1996 was monitored until the end of the study period (Figure 2). With good rains in autumn 1997 (40 – 50 mm) some plants survived until the June 1997 survey. Thereafter the highly irregular winter rainfall of 1997 and 1998 resulted in the death of most plants, except a few *Ruschia* (3) and *Osteospermum* (2) individuals. Although the mulch alone, or in combination with gypsum, often showed more emerging seedlings, survival was not as good as in the gypsum only treatment (Figure 2). However, the effect of gypsum on survival was only significant for *Osteospermum* (Table 3). In June 1997 85 *Osteospermum* plants, and 18 *Chaetobromus* individuals were encountered in a single gypsum-treated block with the highest counts.

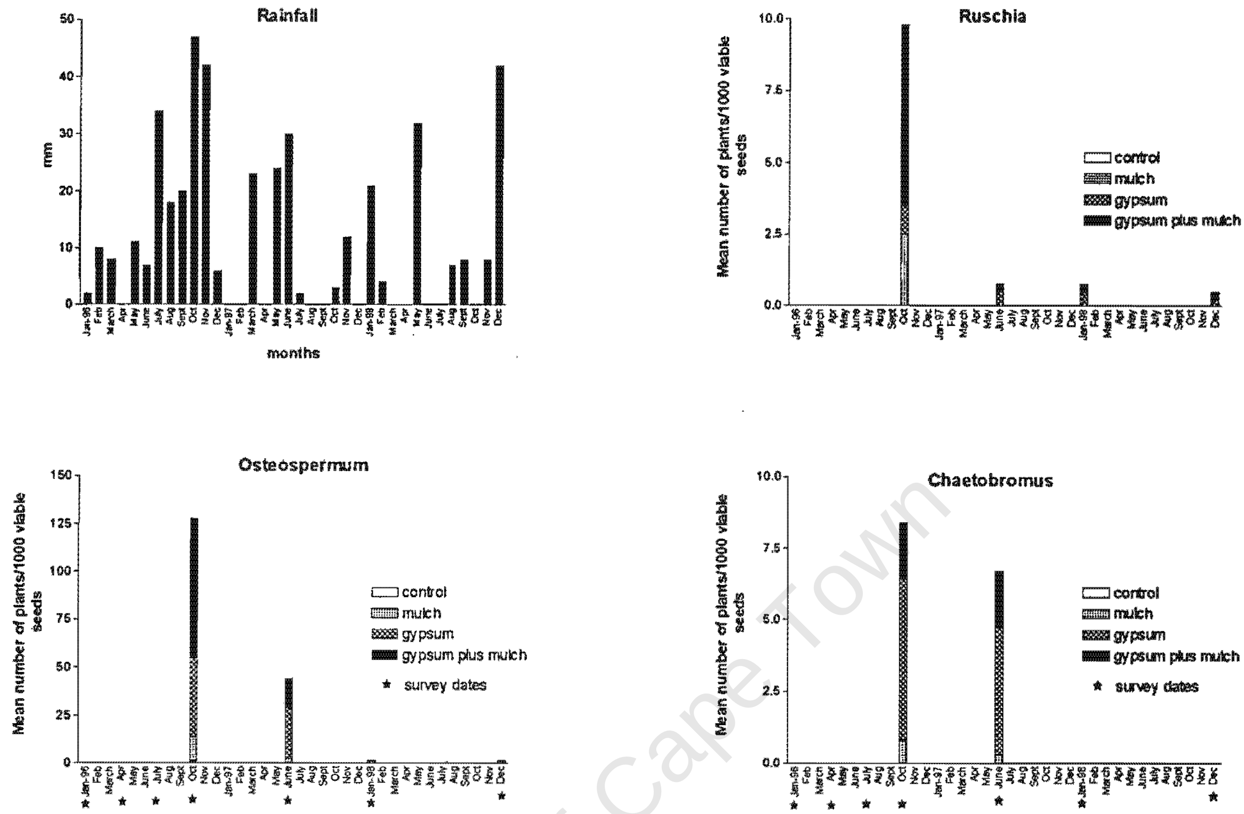

Figure 2 Mean number of surviving plants counted for three reseeded species in different treatments. Monthly rainfall is also shown.

Table 3 Results of two-way analyses of variance of surviving plant counts of three species reseeded under two treatments. Counts were conducted in June 1997 and data were square root transformed. Insufficient numbers of *Ruschia* individuals were available for statistical analysis

Species	Mulch F-value	P-value	Gypsum F-value	P-value	Interaction F-value	P-value
Chaetobromus	0.14	0.712	2.48	0.141	0.0	0.983
Osteospermum	0.01	0.912	5.0	0.045	0.53	0.482

Gypsum and/or mulching resulted in significantly greater endogenic seedling and established plant volume scores for most of the survey dates (Figure 3). Plant deaths in some of the blocks towards the end of the study period caused greater variability in the scores; therefore, non-significant treatment effects were recorded for the last two survey dates. Throughout the study period the highest plant volumes occurred in the gypsum plus mulch treatment. There were no consistent differences between plant volumes of the gypsum alone and mulch alone treatments, although both treatments showed higher plant volumes compared to the control blocks (Figure 3).

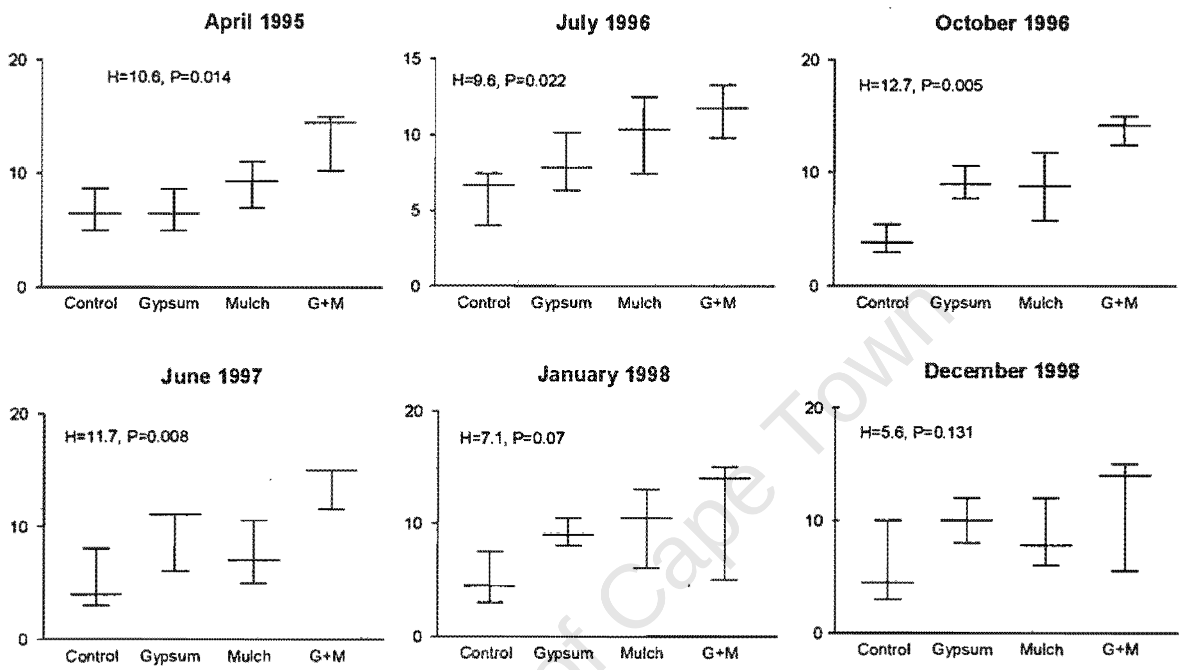

Figure 3 Box-and-whisker plots of plant volume scores for different treatments of bare areas. Results of Kruskal-Wallis analysis of variance by ranks are also given.

Cutting experiment

Severe defoliation (brushcutting) of moribund *Ruschia* veld initially resulted in reduced vigour of two dominant species, *Crassula subaphylla* and *C. muscosa*, which both grow exclusively under *Ruschia* canopies (Table 4). After good autumn rains in 1995, more seedlings were observed, especially on *Ruschia* mounds, in the cut treatment compared to controls, while both *Ruschia* and *C. muscosa* showed vigorous resprouting in the cut treatment (Table 5). By July 1996 a large proportion of each cut *Ruschia* canopy was still dead, although more resprouting *Ruschia* canopies showed signs of young growth in the treatment compared to the controls (Table 6). In the spring of 1996, after a wet winter, significantly more seedlings on mounds, annual plants, and actively growing *Ruschia* plants were counted in the treatment compared to the control plots (Table 7). No seedlings were present in December 1998, after a very dry winter and spring 1998. About 40 mm of rain in early December 1998 resulted in some growth of *Ruschia*, more so in the treatment than in the control plots. Species richness was not significantly higher in the treatment compared to the controls (Table 8).

Table 4 Measurements of vegetation parameters in treatment and control blocks (25 m^2 per block) of the cutting experiment at Rietfontein during the October 1994 survey. Means (n = 5), standard deviations (±) and results of t-tests are given.

Vegetation parameters	Treatment	Control	t-value	P (two- tail)
Percentage litter cover	21 ± 8.6	14.5 ± 5.4	1.43	0.190
Number of flowering *Crassula subaphylla* plants per block.	0.8 ± 1.3	6.8 ± 4.8	2.69	0.028
Number of *Crassula subaphylla* plantlets per 2890 cm^2	70.8 ± 41.6	27.6 ± 28.6	1.92	0.092
Number of *Crassula muscosa* growth tips per 217 cm^2 of canopy.	1.6 ± 2.1	7.0 ± 4.1	2.65	0.029

Table 5 Measurements of vegetation parameters in treatment and control blocks (25 m^2 per block) of the cutting experiment at Rietfontein during the April 1995 survey. Means (n = 5), standard deviations (±) and results of t-tests are given.

Vegetation parameters	Treatment	Control	t-value	P (two- tail)
Number of mound seedlings per 2890 cm^2	57.4 ± 16.1	25 ± 9.3	3.89	0.005
Number of intermound seedlings per 2890 cm^2	25.8 ± 11.9	17.6 ± 14.6	0.97	0.358
Number of young *Ruschia* leaves per 723 cm^2 of canopy	114 ± 28.5	30.2 ± 14.7	5.84	0.000
Number of *Crassula muscosa* growth tips per 217 cm^2 of canopy	64.4 ± 28.6	17.6 ± 9.6	3.46	0.009

Table 6 Measurements of vegetation parameters in treatment and control blocks (25 m^2 per block) of the cutting experiment at Rietfontein during the July 1996 survey. Means (n = 5), standard deviations (±) and results of t-tests are given.

Vegetation parameters	Treatment	Control	t-value	P (two- tail)
Number of mound seedlings per 25 m^2	11.2 ± 6.8	22.6 ± 14.0	1.63	0.141
Number of intermound seedlings per 25 m^2	0.6 ± 1.3	2.2 ± 4.9	0.7	0.503
Average percentage per *Ruschia* canopy which appears dead	82.4 ± 6.7	33 ± 6.6	11.69	0.000
Percentage of *Ruschia* canopies with young growth	78 ± 13	20 ± 12.2	7.25	0.000

Table 7 Measurements of vegetation parameters in treatment and control blocks (25 m^2 per block) of the cutting experiment at Rietfontein during the October 1996 survey. Means (n = 5), standard deviations (±) and results of t-tests are given.

Vegetation parameters	Treatment	Control	t-value	P (two- tail)
Number of mound seedlings per 25 m^2	6.8 ± 4.0	1.4 ± 2.6	2.55	0.034
Number of intermound seedlings per 25 m^2	3.0 ± 4.5	2.2 ± 2.2	0.36	0.731
Number of annual plants per 25 m^2	160.4 ± 18.2	45.4 ± 13.7	11.28	0.000
Percentage of *Ruschia* canopies flowering	52.4 ± 18.7	58.2 ± 9.7	0.62	0.555
Percentage of *Ruschia* canopies with young growth	54.8 ± 16.5	11.5 ± 5.3	5.59	0.001

Table 8 Measurements of vegetation parameters in treatment and control blocks (25 m^2 per block) of the cutting experiment at Rietfontein during the December 1998 survey. Means (n = 5), standard deviations (±) and results of t-tests are given.

Vegetation parameters	Treatment	Control	t-value	P (two- tail)
Number of perennial species per 25 m^2	5.4 ± 1.3	5.0 ± 1.6	0.43	0.678
Number of young *Ruschia* leaves per 723 cm^2 of canopy	81.6 ± 37.5	27.2 ± 11.4	3.1	0.015

DISCUSSION

Gypsum-mulch-reseeding experiment

The existence of extensive bare areas in close proximity to the central watering point on the farm Rietfontein can be attributed to the impacts of longterm overutilization by livestock (see Chapter 1). The reduced plant cover and trampled topsoil have probably aggravated the negative impact of raindrops on the soil surface (Scott 1988). Soil particles were suspended and moved by rainwater. Where these fine particles were deposited they clogged the pores between aggregates and formed "washed in zones" of decreased porosity. With the gradual loss of the topsoil through water and wind erosion, more of the clayey subsoil was exposed. The relatively high exchangeable sodium levels (3 – 4 %) of these soils together with low electrolyte (especially Ca) concentrations (Chapter 1) resulted in the swelling and dispersion (deflocculation) of the clay complex (Sharma 1972; Du Plessis & Shainberg 1985). This chemical sealing probably contributed to the formation of impenetrable, compacted soil surface crusts with poor structure and hydraulic conductivity, where few or no perennial plants can establish and survive (Walters 1951; Russell 1973; Greene & Tongway 1989; Ilyas *et al.* 1993).

Although mulching significantly improved water infiltration into these bare area soils by reducing raindrop impact and ponding run-off water, chemical sealing remained a problem. This is confirmed by the fact that the infiltration depth into the gypsum treated soils was significantly higher than in the mulch treatment. Gypsum is a slow-dissolving salt that readily contributes Ca-electrolytes to the solution (Du Plessis & Shainberg 1985), which can displace some of the exchangeable sodium adsorbed to the clay colloids (Russell 1973). In this process the excess sodium which causes clay dispersion and crusting, may be leached to deeper layers provided electrolyte levels remain high, there is sufficient water flow through the profile, and there are no impermeable layers near the surface (Ilyas *et al.* 1993). Gypsum, therefore, reduced the chemical sealing of these sodic, alkaline soils by partially displacing sodium and adding to the electrolyte concentration. Work by Sharma (1972) showed that gypsum increases infiltration rate and water storage in the profile by creating a more stable pore size distribution. Processes like evaporation, drainage, and movement of water to plant roots should be affected by this improvement in the soil structure (Sharma 1972). The results support the finding of Du Plessis &

Shainberg (1985) and Loveday (1976) that gypsum, applied at a rate of 5 $t.ha^{-1}$, can provide substantial improvement in infiltration. It is expected that permanent amelioration of soil surface conditions of these bare areas will only be achieved with repeated gypsum treatments at the above rate (Loveday 1976). The incorporation of the gypsum into the topsoil by the raking action probably contributed to the dissolution of the salt and resulted in a quicker response to the treatment (Schuman *et al.* 1994).

Although not all seasons were surveyed the seedling counts for all three species (Figure 1) appears to support the findings of Esler (1993) and Milton (1995) that germination events in the Succulent Karoo are determined by the availability of cool season (autumn to spring) moisture. Despite adequate rain in autumn 1997 no seedlings of the small-seeded *Ruschia* or *Chaetobromus* were counted in June 1997, 14 months after reseeding in April 1996. This lends support to the results of Esler (1993) and Milton (1995) that persistent seed banks are of minor importance in the re-establishment of perennial plants in the Succulent Karoo. However, there was a relatively good germination event for the larger-seeded *Osteospermum* in June 1997, which indicates that there might be some degree of seed bank persistence in this species. Of the three reseeded species, the best results were obtained for *Osteospermum* with a maximum count of 150 seedlings / 1000 viable seeds sown. Germination and emergence of *Ruschia* and *Chaetobromus* were generally poor with maximum counts of 40 and 51 seedlings per 1000 viable seeds respectively.

With no established plants competing for resources on these bare areas, the only explanation for the poor emergence in the reseeded-only treatment is the unfavourable soil surface conditions. Only *Osteospermum* emerged at all in the reseeded-only treatment with a maximum count of 2 seedlings/1000 viable seeds. This supports the findings of a number of workers (Joubert & Van Breda 1976; Abusuwar 1995; El-Shorbagy & Suliman 1995) that some form of soil treatment, which improves rainfall efficiency, generally results in improving the success of reseeding operations. Although gypsum had the greatest impact on water infiltration into these bare areas, the highest average number of seedlings for all three species were counted in the gypsum plus mulch treatment (Figure 1). Mulching alone or in combination with gypsum significantly increased seedling emergence in all three species (Table 2). The implication is that water infiltration is not the only problem to be addressed in revegetating denuded areas (Tongway & Ludwig

1996). Sheltered microhabitats (Craig 1985; Eckert *et al.* 1986; Aguiar *et al.* 1992; O'Connor 1997), which can provide more than water to the germinating seed and young seedling, have to be created. Organic mulches have been shown to capture sediment, litter, and seeds (Ludwig *et al.* 1994; Milton 1995); absorb raindrop impact; reduce evaporation; insulate the soil (Abusuwar 1995); and recreate fertile patches (Ludwig & Tongway 1996). The combination of gypsum and mulch in this experiment therefore improved soil moisture conditions and created more favourable microhabitats.

In a similar vegetation type south east of the study area, Milton (1994) also found that most seedlings died within a year of emergence, mainly because of lack of follow-up rainfall. At Rietfontein, only 5 mm of rain fell between July and October 1997. Wiegand *et al.* (1995) suggested that both *Ruschia* and *Osteospermum* require more rain during this period for seedling survival. The better survival in the gypsum only treatment, especially for *Osteospermum* (Figure 4), is possibly because these seedlings received more light and were generally stronger. The poor recruitment in this experiment begs the question how populations of *Osteospermum* and *Ruschia* species are maintained in the matrix vegetation. The answer probably lies in a combination of recruitment site availability (Eriksson & Ehrlén 1992; Esler 1993), and rare, favourable sequences of rainfall events (Gutterman 1981). Using a simulation model Wiegand *et al.* (1995) predicted that rainfall would be sufficient for recruitment of *Ruschia* only in 22 % of years and for *Osteospermum* only in 28 % of years. *Ruschia* and *Osteospermum* are both relatively long-lived woody species for which high levels of seedling mortality do not have a major impact on population dynamics (Esler 1993). Other species which showed good emergence and survival in the surrounding degraded veld, especially through the very dry winter of 1995 (30 – 35 mm), were *Aptosimum procumbens*, *Limeum aethiopicum*, *Galenia fruticosa*, and *Galenia sarcophylla*. These species could be considered for future reseeding experiments in the Succulent Karoo.

The results of this experiment support the findings of a number of workers (e.g. Schuman *et al.* 1994; Milton 1995; Ludwig & Tongway 1996) that by treating the topsoil for improving moisture conditions, the biomass and cover of naturally seeded annuals and perennials will increase. Ephemerals like *Euryops annuus* (Asteraceae) and *Eurystigma clavatum* (Mesembryanthemaceae) benefited from the gypsum-improved water infiltration and the seed capture and protection capacity of the mulch. Biennuals and short-lived perennials, including *Prenia tetragona*, *Delosperma* sp., *Psilocaulon* cf. *dinteri*, *Drosanthemum eburneum*, *Brownanthus ciliatus*, and *Malephora crassa* also established from soil stored seed banks, or seeds which were washed or blown into the plots (Figure 5). The composition of this pioneer community reflected the composition of the surrounding vegetation, and supports Esler (1993) who found that these mat-forming Mesembryanthema of the early successional stages have larger canopy and/or soil seed banks and dormant seeds. The relatively large plant volumes of this naturally seeded community in the gypsum and mulch treatments indicate that seed is not a limiting factor in the revegetation of these bare areas. When favourable microhabitats are created by managing the physical environment (Milton *et al.* 1994), the natural seed abundance may be sufficient to recreate vegetation patches which will act as sink areas (Ludwig *et al.* 1994) that further capture and conserve scarce resources (Whisenant & Tongway 1995), and become fertile patches which expand over time. The positive impacts of mulches might be achieved quicker by allowing livestock hoof action to fragment, and trample the organic matter into the topsoil (Chapter 3). Pitting by hooves, dunging and urinating might further contribute to autogenic revegetation (Howell 1976). However, defoliation of seedlings is detrimental to their growth and survival (Milton 1994), and some form of protection, with for instance thorny branches, or stock withdrawal at some stage, will be necessary. According to Ludwig *et al.* (1994) it may not be necessary to cover the whole bare area with mulch. Patches of mulch could be deposited while keeping in mind the direction of water flow across the bare area. These mulch patches will act as sink areas for water, sediment and organic matter, and if 40 % of the bare area is covered like this the maximum amount of rain water will be conserved under conditions of low rainfall (160 mm) (Ludwig *et al.* 1994).

Cutting experiment

The hypothesis tested in this experiment was that the long-ungrazed *Ruschia*-dominated veld was moribund, and required defoliation in order to restore diversity and stimulate growth. Active vegetation manipulation needs to be attempted, if it can be done cost-effectively (Milton *et al.* 1994). This community, dominated by long-lived *Ruschia* shrubs,can be regarded as highly stable, and will take a great deal of disturbance to force a

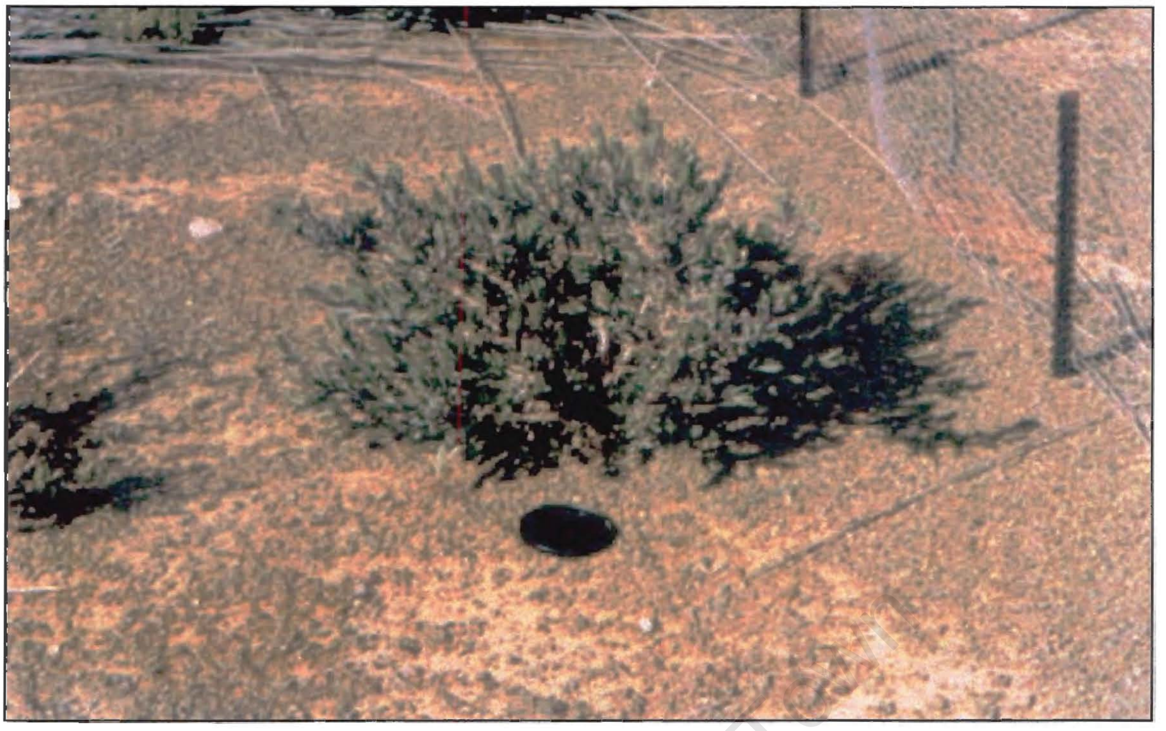

Figure 4 A reseeded *Osteospermum* plant on gypsum-treated soil.

Figure 5 Naturally seeded plants in a gypsum-plus-mulch treated block (centre right) compared to a control block (bottom right)

transition across a system threshold to a more diverse and productive state (Hobbs & Norton 1996). Possible thresholds could be the long-lived nature of the shrubs, the space they occupy, competition for resources, the lack of seeds of more palatable and productive species, and lack of suitable microsites for seedling establishment. Following this line of argument it would be possible to force a transition primarily by removing established plants in order to free space and resources, followed by reseeding with palatable plants, and by creating microsites.

A number of studies have shown that the establishment and survival of seedlings derived from exogenous and endogenous seed can generally be improved by the partial or total removal of existing vegetation (Harrington 1991; Passera *et al.* 1992; Milton 1994; 1995; Biedenbender & Roundy 1996). This experiment supports these findings with generally more seedlings encountered in the cut treatment, especially on the cut *Ruschia* mounds, than in the controls. The regular patterns of these *Ruschia* covered mounds, points to the importance of competition for resources between the established plants (Yeaton & Esler 1990). With their well established shallow lateral root systems, the adult *Ruschia* plants will collect most of the water in the topsoil leaving little for seedlings that have established in intermound areas. With the release of soil moisture as a result of cutting the adult plants (Milton 1995), more seed germinated, emerged, and survived on the cut compared to the uncut mounds. An important observation is that all the cut mounds remained intact at least untill the end of the study period. The intact mounds and the level topography prevented drastic increases in runoff and soil loss as was reported by Snyman & Fouche (1991) when vegetation cover was reduced.

The leaves of *Crassula subaphylla* fall off very easily (Shearing 1994) and may form roots where they come in contact with the soil. Initially the cutting action resulted in numerous vegetatively propagated plantlets of palatable *C. subaphylla*, probably because they were fragmented during the cutting process. *C. muscosa* is a palatable succulent species which initially suffered reduced vigour but then recovered a year after the cutting, and after good autumn rains, produced more growth tips per canopy surface area than in the uncut plants. The *Ruschia*, *Rhinephyllum macradenium*, *Galenia fruticosa*, and *Psilocaulon* cf. *dinteri* individuals which established in the intermound areas of both cut and uncut blocks is consistent with other data that show that these small-seeded species are well suited to colonize open areas. In contrast, the young individuals of *C. subaphylla*, *C. muscosa*, *Euphorbia karroensis*, *Anacampseros lanceolata*, and *Osteospermum* were only encountered on mounds, which shows that these species require moderated microhabitats for establishment.

A significantly higher cover of annuals, including palatable legumes, was recorded in the cut compared to the control treatments after the good winter rains in 1996. This pattern is consistent with other studies, both in the Karoo (Milton 1995) and elsewhere (Eckert *et al.* 1986). There are many possible reasons for this, including impacts of greater temperature fluctuations on germination (Quinlivan 1966), improved moisture conditions (Milton 1995), reduced allelopathic inhibiting effects (Gutterman & Herr 1981), and soil disturbances (Van Rooyen & Grobbelaar 1982).

Throughout the study period, on several survey occasions, the resprouting canopies of the cut *Ruschia* plants showed more active growth compared to the uncut canopies. Compensatory growth is influenced by amongst others the level of defoliation of a plant (Oesterheld & McNaughton 1991). Although the level of defoliation in this experiment was unnaturally high, there were clear signs of more young growth in the resprouting *Ruschia* branches. This appears to be in contrast to the findings of Van der Heyden and Stock (1995) that simulated browsing of *Ruschia spinosa* mostly caused a decrease in total non-structural carbon pools which plays a potentially important role in regrowth. However, Van der Heyden and Stock (1995) conclude that regrowth of karoo shrubs appears not to be limited by carbon reserves, but that localized nutrient return, water availability, and meristimatic considerations could be the important factors. The cutting in autumn, the beginning of the growing season for many winter rainfall karoo shrubs, could also have resulted in a larger proportion per shrub dying and slower recovery in the living parts (Venter 1962). A further negative aspect of this severe defoliation is that small palatable plants, like the succulent *Anacampseros lanceolata*, which are normally well protected against browsing animals inside the thorny canopies of *Ruschia*, become more vulnerable to grazing impacts.

Economic Considerations

Without a soil treatment that improves moisture conditions, reseeding of bare areas is doomed to be a failure. Any mechanical soil treatment is costly, and at about R900 per ha (at an application rate of 5 $t.ha^{-1}$, transport costs of 200 km from the fertilizer factory included), the gypsum treatment is economically feasible only at a patch scale. Mulching is probably cheaper, especially if locally cut material (e.g. old *Ruschia* shrubs) is used, but will still be a labour-intensive task. In this regard it would be possible to combine brushcutting of moribund veld with mulching of bare areas. The ability of the surrounding, mainly non-forage, ephemeral community to expand into the bare areas when environmental conditions are ameliorated has to be exploited. This will avoid the costs of an exogenous seed source, as well as the necessity to protect highly palatable young forage plants from domestic and wild herbivores. The time it takes for any financial investment to reap benefits is important. It is postulated that the time this revegetation process will take depends firstly on soil depth, and secondly on the size of the bare area, and therefore distance from a seed source. Some bare areas that are several hectares in size and are in such an eroded state that the bedrock protrudes in places, should perhaps be regarded as practically irreversible features.

CONCLUSIONS AND RECOMMENDATIONS

Mulching of bare areas in the Succulent Karoo enhanced rainfall efficiency and improved germination and emergence of reseeded and naturally seeded annuals and perennials. This restoration method has the potential to recreate vegetated areas that will further capture and conserve scarce resources. The gypsum treatment also resulted in improved water infiltration and seedling emergence, but might not be a cost-effective option mainly because of transport costs to these remote arid areas. Even with some form of soil treatment reseeding bare areas in this low rainfall part of the Karoo will remain a risky operation mainly because of the low probability of follow-up rainfall in late-winter and spring, which is essential for seedling survival.

Brushcutting old *Ruschia* veld created more favourable conditions for seeds from the surrounding uncut community as well as from annual forbs to germinate and emerge. The removal of moribund, woody material also stimulated young growth in a number of palatable shrubs. The cut veld, with periodically more annual weeds ("opslag") and young growth from forage plants, is probably more acceptable to browsing livestock, but there was no evidence that a single drastic treatment like this will provide the driving force to alter the community from its moribund, low-grazing-value state into a more diverse and forage-producing state. A possible threshold which must still be overcome is the apparent lack of seed of a greater diversity of palatable plants.

A worthwhile research avenue would be to evaluate the role of livestock as restoration tools in the treatment of bare areas in the Karoo. There are indications that feeding seed-bearing roughage and herding animals onto these bare areas might be an alternative approach to restoration (Howell 1976; Tainton 1984). Experimental sowing of wing-seeded forage species like *Osteospermum* and *Tetragonia fruticosa* into cut veld at the beginning of the cool season when seedling emergence occurs, could increase the ratio of forage to non-forage seedlings. The relatively large seeds of these species could facilitate their recruitment into the opened canopies of the cut shrubs.

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CHAPTER 3

THE IMPACTS OF NON-SELECTIVE GRAZING (NSG) ON SOIL PROPERTIES OF THE NAMA KAROO

INTRODUCTION

Livestock producers are increasingly interested in practices that maintain or restore the productivity of their rangelands. This has stimulated the interest of range scientists worldwide in improving our understanding of the effects of grazing on vegetation and soils (e.g. Webb & Stielstra 1979; Biondini & Manske 1996; Dormaar *et al.* 1997; Biondini *et al.* 1998). The Karoo region of South Africa is no exception. Owing to its size, biological diversity, and importance to the livestock industry, the grazing systems and their impacts in the Karoo have attracted much attention (see review by Hoffman 1988). However, Hoffman (1988) found that most of the recommended grazing management systems for the Karoo lack adequate empirical data to support them. Recent studies by Milton (1992), Milton *et al.* (1994), and Dean *et al.* (1995) have made a substantial contribution to our understanding of the role of herbivory and climate in structuring karroid communities. They have shown that environmental changes in the Karoo are difficult to explain with predictable, deterministic ecosystem models (e.g. classical succession) but that conceptual models which incorporate the complex interactions between grazing disturbances and moisture availability in an event-driven system, are closer to reality.

Grazing animals primarily affect rangeland soils by direct impacts through trampling and dunging (Smoliak *et al.* 1972), and indirectly by altering plant community structure (Thurow *et al.* 1988; Dean *et al.* 1995; Dormaar *et al.* 1997). Soil quality, or productive potential, as characterized by, amongst others, infiltration rate, nutrient status, and stability (resistance to erosion), determines the flows of nutrients and water between soils and plants (Tongway & Hindley 1995). These soil qualities may be changed by management either to strongly conserve resources (soil, nutrients, water) within the system, or to degrade into a leaky system (Tongway & Hindley 1995). Biologically mediated soil processes such as decomposition and mineralization, production and maintenance of macropores, production of soil aggregates, and fixation, are critical for the maintenance and restoration of soil production potential (Whitford & Herrick 1996). Grazing animals are the "tools" by which the rangeland manager can either maintain or reduce productive potential, depending mainly on stocking rate and grazing systems applied. The correct use of these tools can have long-term economic implications for the livestock operation (Savory 1983; Biondini *et al.* 1998).

There is an extensive literature on the merits of different stocking rates and grazing systems. Some reviews (e.g. O'Reagain & Turner 1992) suggest that there is little evidence to support rotational grazing in favour of continuous grazing. Others have postulated that grazing systems which combine short periods of high grazing intensity with prolonged periods of rest (≈ high-intensity, low-frequency grazing, HILFG ≈ non-selective grazing, NSG), simulates the herding of wild ungulates and may play an important role in influencing range condition (Acocks 1966). McNaughton *et al.* (1988) have shown that removal of this type of grazing in the Serengeti grasslands leads to an increase in N immobilized in litter and standing dead biomass, and a reduction in soil microbial turnover rates and net soil N mineralization. This stimulated McNaughton's (1993) grazing optimization theory - grazing stimulates soil and plant processes which, in turn, maximize primary production through plant compensatory growth mechanisms. However, this theory is still being intensively debated (Painter & Belsky 1993; Biondini *et al.* 1998).

A number of workers have shown that heavy grazing leads to plant compositional changes, a reduction in total organic cover, and changed physical and chemical properties of the topsoil layers with a resultant reduction in infiltration rates (Smoliak *et al.* 1972; McCalla *et al.* 1984; Graetz & Tongway 1986; Thurow *et al.* 1986; Biondini & Manske 1996; Mworia *et al.* 1997; Biondini *et al.* 1998). The literature, however, is replete with contradictory results on the effects of grazing on soils, possibly a consequence of the different environments, soils and grazing management systems of the trials (Lavado *et al.* 1996). One of the reasons for these contradictions is the various interpretations of "heavy grazing". Although heavy grazing is generally understood to mean that grazing animals are kept in an area (paddock) until they have removed up to 90 % of the aboveground biomass (Biondini *et al.* 1998), this can be achieved by either keeping relatively few animals for a long time in the paddock (which translates into short rest periods) or by stocking high densities for a short time in numerous small paddocks (which translates into longer rests depending on the number of paddocks available). I argue that these differences are important and will result in different impacts on soils and vegetation. I postulate that heavy grazing under a HILFG/NSG system would maintain or improve soil quality through its concentrated but short duration impacts on ecosystem processes (McNaughton *et al.* 1988; Savory 1991).

This study was designed to test this hypothesis by comparing the impacts of NSG (treatment) vs. no grazing (control) on certain soil quality indicators viz: (1) total soil organic carbon (OC) and its spatial distribution; (2) soil microbial respiration rates; (3) total nitrogen (TN); (4) aggregate stability, infiltration rate, erodability, and cumulative infiltration; (5) invertebrate activity. In each case I predicted that the grazing treatment would increase OC and change its spatial distribution, and would increase microbial respiration rates, TN, infiltrability of the soil, and invertebrate activity.

STUDY AREA

Fieldwork was conducted on the farm Elandsfontein in the Nama Karoo. See the General Introduction for a detailed description of the study area.

METHODS

Experimental design

In April 1995 four paddocks (A, B, E and F) on Elandsfontein were subjectively assessed as similar in terms of topography, soils, and vegetation, and were identified as the four replicates for this study. Although the paddocks differed in size (A = 34 ha, B = 87, E = 108, F = 32) the treatment, grazing pressure (measured as Large Stock Unit Grazing Days/ha = LSU*days/ha), were kept similar across the replicates by adjusting camp time (days). A five strand stock fence was used to erect a 50 m x 50 m exclosure in each paddock. The exclosure fences were positioned away from watering points and stock paths, and excluded livestock grazing but not herbivory by steenbok (*Raphicerus* *campestris*), hares (*Lepus* spp.), tortoises (*Psammobates* spp.), porcupine (*Hystrix austro-africanae*), and invertebrates. The fenced areas were regarded as controls whereas the adjacent areas were regarded as the grazing treatment.

The Treatment

After the exclosures were set up in April 1995, each paddock received a non-selective grazing (NSG) treatment each year thereafter (until 1998) by stocking a combination of Nguni cattle, Merino sheep, and Boer goats at grazing pressures varying from 40 to 60 LSUGD/ha. Grazing pressures varied between years because of food availability, and animal reproductive status and condition, but were always kept as similar as possible across replicates by adjusting camp time (days). The treatment was not applied during any particular season, but the aim was always to herd as many animals together as was available, and to keep camp time within the maximum of about 2 weeks (it varied between 2 and 16 days). These high grazing pressures were achieved by herding up to 348 LSU in a 32 ha paddock for 5 days (1998 treatment). In order to extend the period of occupation by a couple of days, and thereby forcing the animals to ingest more fibrous, less palatable material, they were supplied with 1.5 – 2 tonnes of cut saltbush (*Atriplex nummularia*) every alternate day. The woody portion (> 50 %) of the saltbush shoots does not get utilized and remain on the land as litter. The leaf material with an average 22 % protein, helps to maintain rumen function and improves intake of low quality fibre (Barnard 1986). For experimental purposes the treatment was applied annually, but under normal circumstances a paddock would receive a rest period of more than a year (147 paddocks at ± 9 days per paddock converts into a theoretical rest period of 3 – 4 years).

Soil organic carbon (OC)

In June 1998, after the treatment, three of the four replicate paddocks were selected for collecting soil samples for OC measurements. 5 m x 5 m plots were randomly positioned inside control and treatment areas of each paddock. Care was taken to avoid stock paths. Each plot was subdivided into a grid with one cell measuring 0.5 m x 0.5 m. Using this grid, 100 soil samples (top 5 cm) were systematically collected in each plot. After passing each soil sample through a 2 mm sieve to remove larger fractions, the OC content (%) was determined using the Walkley and Black method (Nelson & Sommers 1982).

Soil microbial respiration rates

Every year since 1996, after the treatment had been applied, soil samples were collected from control and treatment areas of the four replicate paddocks for measuring microbial respiration rates. Samples were randomly taken from the top 5 cm of open, unvegetated areas between shrub and grass clumps, as well as from mound soils underneath shrubs (mainly *Pentzia incana*). Samples were passed through a 2 mm sieve, moistened with 10 g water per 100 g air-dry soil, and stored in a polythene bag for one week. Bags were shaken every day to aerate the soil. After the incubation period of one week, CO_2 release as a result of microbial respiration was determined using NaOH to react with the CO_2 in respiration flasks. After approximately one week (the exact time was noted) in the respirometers, the free NaOH was titrated with HCl. Titrations were standardized against a control respirometer with sand instead of soil. The water content of the moist soil at the time the respirometers were set up was determined by drying sub-samples overnight at 105°C. Soil microbial respiration rates were expressed as g $CO_2.g^{-1}$ air-dry soil.s^{-1} (for more detail see Rowell (1994) pp. 117 – 119).

Total nitrogen (TN)

Subsamples were taken and pooled from the 100 soil samples per 5 m x 5 m grid collected for OC measurements. TN was determined for each of the six pooled samples using the Kjeldahl method. The pooled samples were regarded as representative and were used to compare the TN content of control with treatment soils after a treatment.

Aggregate stability, infiltration rate, erodability, and cumulative infiltration

In June 1998, after the grazing treatment, 20 kg soil samples were collected from the top 10 cm of open, unvegetated areas in treatment and control areas of all four paddocks. Soils were passed through a 5 mm sieve and then subjected to a rain event equivalent to 44 $mm.h^{-1}$ in a rainfall simulator. Aggregate stability (%), final infiltration rate ($mm.h^{-1}$), erodability ($g.m^{-4}.h^{-1}$), and cumulative infiltration (mm) were determined. After drying, a second rain event of 44 $mm.h^{-1}$ was simulated on the soils which were now partially sealed as a result of physical and chemical processes initiated by the first event and the drying process. Similar soil parameters were measured after the second rain event.

Invertebrate activity

Descending-point plant surveys were conducted as part of a study on the impacts of NSG on plant cover and composition (Chapter 4). While conducting these surveys it was noted when the descending steel rod struck any soil or litter signs related to invertebrate activity. Five surveys were conducted over the study period and for each survey (n = 500 points) the number of strikes of invertebrate activity was converted to percentage frequency.

Statistical analysis

Analysis of data using classical statistical methods requires independence between samples. Since it was expected that spatial correlation exists in the OC data, these methods were not appropriate and a geostatistical approach was used (Rossi *et al.* 1992). Data were graphically presented using interpolated perspective plots, while the degrees of spatial correlation in treatment and control soils were compared using variograms. A variogram models the average degree of similarity between the data points of a matrix as a function of their separation distance (lag distance). For patterned data the degree of similarity for data points with short lags is high, and this degree of similarity (continuity) decreases with an increase in lag distance. Conversely random data produces a constant variogram, meaning that, on average, the variance between data points does not change with distance (Rossi *et al.* 1992).

Microbial respiration rates were measured on more than one occasion. An analysis of variance for repeated measurements was performed to test the null hypothesis that the change over time in the measured parameter would not differ significantly between treatment and control. The AREPMEASURES procedure in GENSTAT (version 5) was used.

For TN, the rainfall simulator results, and invertebrate activity, paired t-tests were used to test the null hypothesis that there was no significant difference in the values of the observed variable between treatment and control.

RESULTS

Soil organic carbon (OC)

Because of the high variation between paddocks, the data for each paddock were treated separately. Although the difference was only significant in paddock A, there was a trend for OC content to be lower in treatment compared to control soils (Table 1). The frequency distributions of the treatment data were more asymmetrical about their means, indicated by the higher "skewness" measures. Treatment data were more positively skewed towards lower OC values, meaning a higher frequency of low OC values compared to control data (Figure 1).

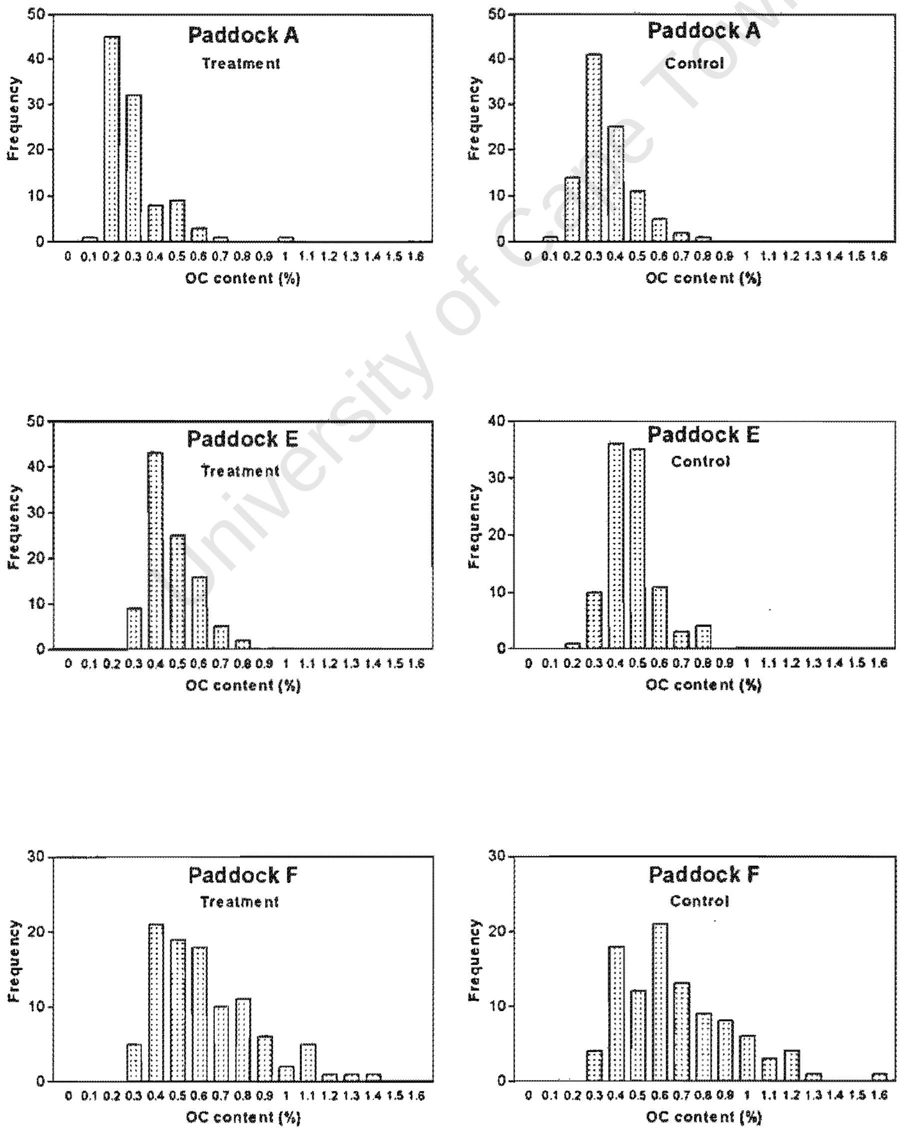

Figure 1 Frequency distributions of soil organic carbon measurements in treatment and control areas of three paddocks (A, E and F).

Table 1 Summary statistics for soil organic carbon (%) measurements from three paddocks (A, E and F) at Elandsfontein in June 1998. Results of t-tests between treatment and control soils are also given.

Statistic	PADDOCK A Treatment	A Control	E Treatment	E Control	F Treatment	F Control
Mean	0.26	0.31	0.42	0.43	0.58	0.63
Median	0.22	0.28	0.39	0.42	0.53	0.59
Mode	0.16	0.25	0.31	0.47	0.37	0.39
Standard Deviation	0.13	0.12	0.11	0.11	0.24	0.25
Skewness	2.10	1.05	0.89	0.82	1.03	0.99
Minimum	0.10	0.02	0.25	0.18	0.24	0.26
Maximum	0.93	0.75	0.76	0.76	1.38	1.54
Coefficient of Variation	50.0	38.0	26.8	26.6	40.7	39.7
t	3.128		0.622		1.315	
P	0.002		0.535		0.19	

Figure 2 shows that there was a patterned distribution in OC content of both treatment and control soils. There were "peaks" of OC under and around clumps of perennial grasses and shrubs, and "valleys" of low levels of OC of the interclump (open) areas. Apart from the apparent greater extent of "valley" in the treatment areas compared to the control, the non-selective grazing treatment appeared to have had very little impact on total OC as well as OC distribution on a 5 m scale after three years of the NSG treatment.

The variograms (Figure 3) show that data points separated by lag 0 – 2 (0 – 1 m) have lower variogram values than the general variances for the full data sets, and are therefore more alike, or spatially continuous. In most of the treatment and control data sets the variogram values remained constant from about lag 3 (1.5 m) which means that on average the variance between data points separated by more than this distance did not change. The exception was the higher variogram values between lag 4 – 5 (2 –3 m) in the treatment soils of paddock A. It means that data points separated by this distance were spatially more discontinous. This pattern is consistent with the pattern in the perspective plots of paddock A (Figure 2) which shows more and deeper "valleys" for the treatment area.

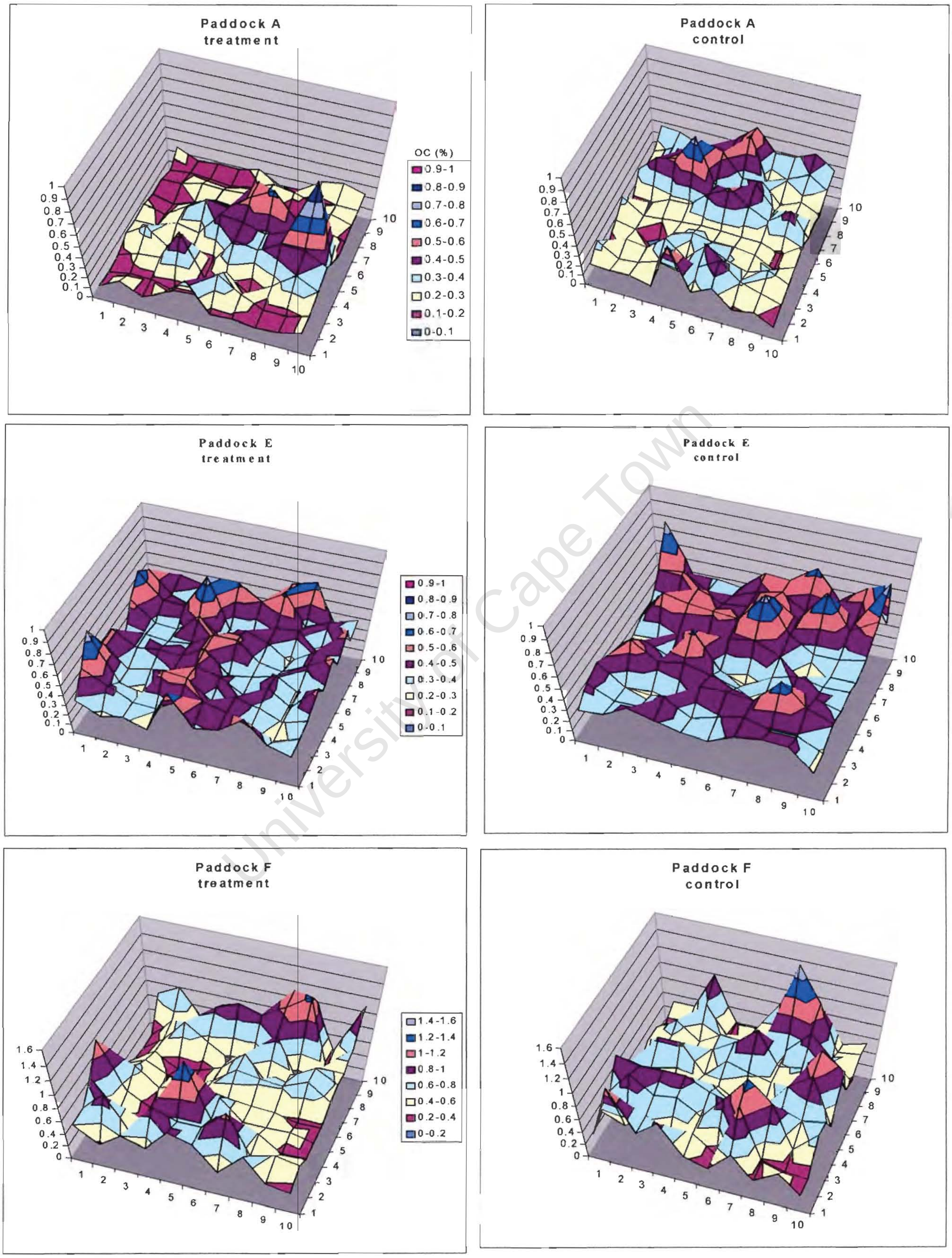

Figure 2 Perspective plots of organic carbon content (OC %) of treatment and control soils of three paddocks.

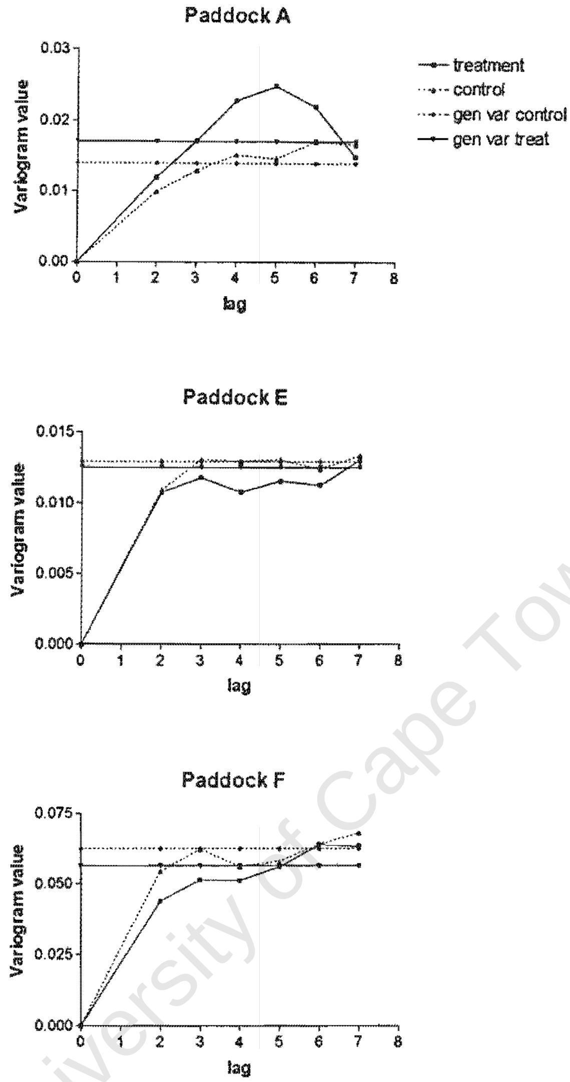

Figure 3 Omnidirectional variograms for the patterned spatial distributions of organic carbon shown as perspective plots in Figure 2 (1 lag = 0.5 m)

Soil microbial respiration rates

There appeared to be a positive relationship between microbial respiration rates of both treatment and control soils and the rainfall index (Figure 4). Respiration rates were always significantly higher in mound soils compared to intermound soils irrespective of treatment. Treatment had no significant impact on microbial activity in mound soils, but resulted in a significant increase in microbial respiration rates in intermound soils (Figure 4). The time effect and the time x treatment interaction were not significant.

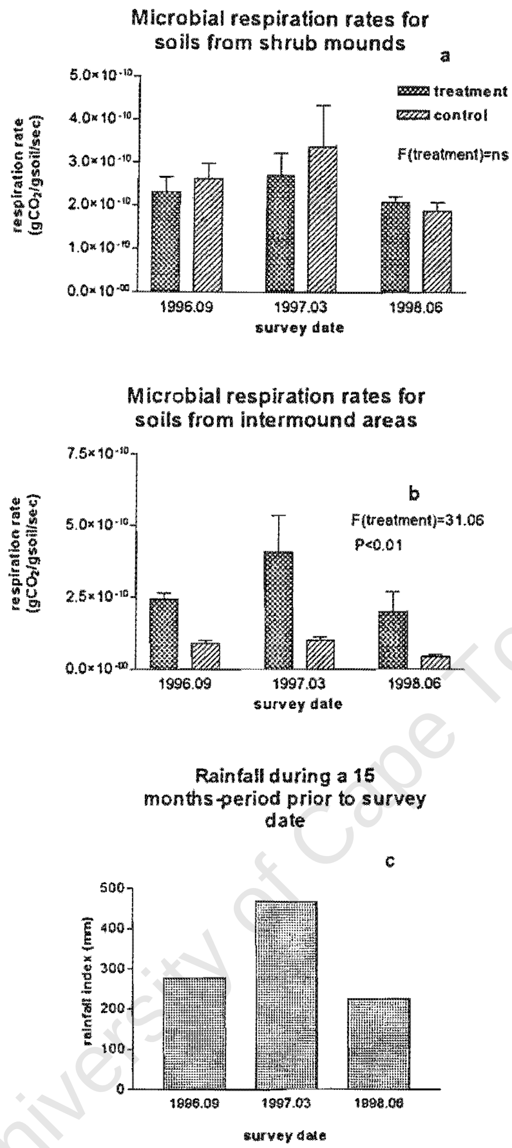

Figure 4 Soil microbial respiration rates for mound soils (a), intermound soils (b), and rainfall index (c) over the study period.

<u>Total nitrogen (TN)</u>

TN (%) was significantly lower in the treatment compared to the control soils (Table 2).

Table 2 Total soil nitrogen content (%) of six pooled samples collected in three control and grazing treatment paddocks in June 1998. The result of a paired t-test is shown.

Paddock	Treatment	Control	Paired t-test
A	0.062	0.07	t = 25
E	0.078	0.087	P < 0.01
F	0.118	0.126	

Aggregate stability, infiltration rate, erodability, cumulative infiltration

The rainfall simulator results showed that soils from grazed areas were more stable, resistant to erosion, and had a better water infiltration capacity compared to control soils when a rain event was simulated on sieved (disturbed) soils (Table 3). However, not all these differences were significant at the $P \leq 0.05$ level. When a rain event was simulated on sealed soils, which is closer to field conditions most of the time, the same pattern emerged, with treatment soils having better stability and infiltrability than control soils. However, with this second event no differences were significant.

Table 3 Characteristics (mean ± SD) of soils collected from open areas in control and grazing treatment paddocks (n = 4) in June 1998. Soils were subsequently subjected to a 44 $mm.hr^{-1}$ rainfall event using a rainfall simulator. Results of paired t-tests are shown.

		Treatment	Control	t-value	P
1	Initial rain shower on disturbed soils (44 $mm.hr^{-1}$)				
	Aggregate stability (%)	46.6 ± 12.5	17.3 ± 3.7	3.67	0.03
	Infiltration rate ($mm.hr^{-1}$)	14.4 ± 7.0	4.2 ± 0.0	2.91	0.06
	Erodability ($g.m^{-4}.hr^{-1}$)	25.8 ± 10.3	38.6 ± 6.3	1.75	0.18
	Cumulative infiltration (mm)	26.3 ± 8.1	9.4 ±2.9	3.13	0.05
2	Second rain shower on sealed soils (44 $mm.hr^{-1}$)				
	Aggregate stability (%)	23.4 ± 9.6	9.8 ± 1.7	2.49	0.09
	Infiltration rate ($mm.hr^{-1}$)	5.3 ± 2.7	2.1 ± 0.0	2.32	0.10
	Erodability ($g.m^{-4}.hr^{-1}$)	25.1 ± 7.7	35.1 ± 2.2	2.44	0.09
	Cumulative infiltration (mm)	13.0 ± 6.5	5.5 ± 0.5	2.21	0.11

Invertebrate activity

There was significantly more sign of invertebrate activity in the treatment compared to the control areas (Table 4).

Table 4 Percentage frequency (mean ± SD) of invertebrate activity encountered in 20 descending-point surveys of control and grazing treatment paddocks between 1995 and 1998. The results of a paired t-test is shown.

	Treatment	Control	t-value	P
Frequency of invertebrate activity (%)	0.4 ± 0.5	0.17 ±0.3	2.09	0.049

DISCUSSION

Soil organic carbon (OC), microbial respiration rates and total nitrogen (TN)

The patterned spatial distribution of OC corresponds with the results of other workers (Jackson & Caldwell 1993; Whitford & Herrick 1996) who found that litter accumulates under shrub canopies, resulting in patches of soil habitat that differ considerably from the unvegetated patches. Root die-back and litter are the primary sources of organic carbon (Rowell 1994; Whitford & Herrick 1996). The spatial distribution and morphology of shrubs, and the distribution and turnover of especially grass roots, might have played an important role in the observed patterns of OC distribution. The average lag distance of 3 (1.5 m) between OC "peaks" and "valleys" in these results is in agreement with Jackson & Caldwell (1993) and Lavado *et al.* (1996) who postulated that spatial autocorrelation in OC of soils is present at shorter distances (< 3m) and is dependant on the structure of the vegetation, and the impacts of large herbivores on litter dynamics.

After having measured higher levels of litter cover and fewer standing-dead plants in the grazed areas (Chapter 4), it was expected that soils of these areas would have had higher OC values. Although not significant in paddocks E and F the trend was actually for soils of the grazed areas to have lower OC values mainly because of higher frequencies of low OC readings i.e. more "valleys" (Figures 1 and 2). The apparent incidence of more "valleys" in the grazed areas is not consistent with the hypothesis that the wastage from feeding animals and hoof action onto mounds would distribute the organic matter into the open areas resulting in lower "peaks" and filled in "valleys" in the grazed areas. Clearly the visible coarse litter measured in the vegetation surveys (Chapter 4) was not reflected in the OC readings probably because soil samples were passed through a 2 mm sieve before recording OC content, or there was not sufficient time for the coarse litter to be fragmented.

There are a number of possible explanations for the observed trends in OC. The pulsed input of organic matter from the grazed vegetation, which has on average a lower above-ground standing biomass over a full grazing cycle compared to the ungrazed vegetation, might have been lower than the slower but more continuous input from the ungrazed vegetation. Organic matter input is related to standing biomass (Garcia-Miragaya & Cáceres 1990). Although the literature dealing with the effects of grazing on soil organic matter is sometimes contradictory, several workers (Graetz & Tongway 1986; Thurow *et al.* 1986; Thurow *et al.* 1988; Dormaar *et al.* 1997; Eldridge & Robson 1997) found that heavy grazing reduced the biomass and litter base, and therefore OC of the soils. It is possible that with the removal of > 50 % of the forage with non-selective grazing, organic matter input over time was lower in the grazed compared to ungrazed areas.

The frequency of invertebrate activity in the grazed areas were significantly higher compared to the control areas. A great diversity of invertebrates are associated with Karoo systems and are amongst others responsible for the fragmentation and redistribution of organic matter (Dean & Milton 1995). In sub-tropical arid and semi-arid rangelands a large portion of litter and herbivore dung is processed by termites which results in a strong negative correlation between activity and abundance of termites and soil organic matter (Nash & Whitford 1995). It is therefore possible that the greater incidence of invertebrate activity in the grazed areas has led to the redistribution of OC from open areas to nest sites, which are often below shrub canopies. OC could also have been redistributed to a greater depth than was sampled.

The third possible explanation for the trend of lower OC, especially in the open areas, of grazed plots compared to control plots is related to the significantly higher microbial respiration rates recorded in the open areas of the grazed plots (Figure 4). It is postulated that the pulse of organic matter input from feeding and trampling animals stimulated a rapid increase in microbial biomass (McNaughton *et al.* 1988; Tongway & Ludwig 1996; Whitford & Herrick 1996). Soil respiration rate (CO_2 production) depends on the activities of microbial organisms which is controlled by organic matter content, O_2 supply, temperature, soil water content, and nutrient supply (Rowell 1994). Aeration of soils may be limited by surface structural barriers e.g. horizontally dispersed clay layers generated by raindrop impact (Graetz & Tongway 1986). This crust, of especially the open, unvegetated area, was thoroughly broken by the hoof action of concentrated livestock herds during non-selective grazing treatment (Eldridge & Robson 1997). The top layers, down to approximately 10 cm, were mixed with a resultant increase in aeration (Thomas 1960), and possibly temperature (Johnston *et al.* 1971) and moisture content (Table 3). This impact was not as pronounced in mound soils because of the already loose and friable structure of these soils. The pulse of organic matter into the open areas together with aeration and a possible increases in temperature and moisture, resulted in a highly active microbial and microbivore community, which rapidly turned over soil OC (McNaughton *et al.* 1988; Tongway & Ludwig 1996). Smoliak *et al.* (1972) also found higher in vitro decomposition rates in soils from heavily grazed areas compared to ungrazed areas because of the greater amount of total belowground plant biomass, total C, NO_3-N, and the polysaccharide content of belowground plant materials. The greater frequency and size of litter trains in the grazed compared to the ungrazed areas (Chapter 4) helped to check evaporative water losses and induced higher water content in the grazed, open areas (Tongway & Hindley 1995), which further promoted microbial activities.

The higher respiration rates recorded in the mound soils compared to the open soils are probably related to the higher OC and more aerated soils of this habitat. It can be inferred that decomposition and mineralization of vital plant nutrients (e.g. N and P) would be more rapid in these mound soils as a result of the activities of the soil fauna. This nutrient release near the absorbing surfaces of plant roots is important for nutrient cycling. Because of the continuous input of organic matter into these soils, there is no pulse of rapid microbial growth that tends to immobilize nutrients (Whitford & Herrick 1996). On the other hand the pulsed, discontinuous input of fragmented organic matter, dung and urine into the grazed, open areas resulted in a rapid increase in microbial activities which may result in nutrient, especially N, immobilization (Shariff *et al.* 1994). When usable carbon is plentiful relative to N, microbial demand for N and soil N immobilization potentials are high (Holland & Detling 1990). This may make N temporarily unavailable to plants, but hold the nutrient in the system (Seastedt *et al* 1988). Drying will kill a large part of the microbial population but on rewetting, populations multiply very rapidly and have available to them C an N in the dead microbial cells (Rowell 1994). The balance between immobilization and mineralization is important for the maintenance and slow release of nutrients in the soil.

The results appear to confirm the basic link between OC and TN (Lavado *et al.* 1996). Moreover, it is consistent with the findings of Holland and Detling (1990), Willms *et al.* (1990), and Dormaar *et al.* (1997) that total soil C and N decline with grazing, probably because of decreases in C and N returns to the soil and increases in litter turnover rates. The lower TN in the grazed areas could be related to nutrient uptake by more productive plants in the grazed areas (Chapter 4), but also to a net export of nutrients via animal products (Lavado *et al.* 1996).

Aggregate stability, infiltration rate, erodability, and cumulative infiltration

The higher infiltration rate and cumulative infiltration, and lower erodability of the treatment compared to the control soils are all related to the significantly greater aggregate stability of the treatment soils (Greene & Tongway 1989). Since soil samples for the rainfall simulator measurements were collected from open unvegetated areas, it is possible to relate the rainfall simulator results to OC and respiration rates of open area (intermound) soils. The pulsed input of fragmented litter, dung and urine together with mixing and aeration of the top layers of treatment open soils leads to a rapid increase in soil biotic processes. While soil moisture is sufficient to maintain a film of water on the surfaces of soil particles, many taxa of soil microflora and microfauna remain active. As the soil dries out and the films of water disappear, protozoans encyst and nematodes enter an anhydrobiotic state. Bacteria also require water films to remain active while some fungi may continue activity in very dry soils (Whitford 1989). These soil biotic processes are largely responsible for aggregate creation and stabilization, macropore formation, and spatial-temporal patterns of soil organic matter and nutrients (Whitford & Herrick 1996). While microbial debris and humic materials are important components of aggregates, these aggregates are frequently held together by a "bag" of stringy fungal hyphae and plant roots (Oades 1993). Microbial polysaccharides (Chaney & Swift 1986) and root exudates (Pojasak & Kay 1990) have also been shown to stabilize macroaggregates. Stable aggregates and the pores within and between them are vital for the capacity of the soil to store and release water and nutrients and for plant roots to access these resources (Thurow *et al.* 1988; Tongway & Ludwig 1996; Whitford & Herrick 1996; Eldridge & Robson 1997). Raindrop impact on the structurally less stable aggregates of the control open soils probably resulted in clay dispersion and the formation of horizontally disposed clay layers (Graetz & Tongway 1986). These surface seals can act as physical barriers to gas exchange and water infiltration (Graetz & Tongway 1986). The greatest demand for O_2 is in the surface layers where the maximum numbers of roots, micro-organisms and animals occur (Rowell 1994). When the supply of O_2 does not meet the demand, soil biotic activities will have to decrease. This in turn may have a chain effect by further reducing the soil's structural stability and therefore infiltrability and resistance to erosive forces.

The results of this study show the opposite trends to what was found by a number of workers (e.g. Van den Berg *et al.* 1976; McCalla *et al.* 1984; Thurow *et al.* 1986; Fuls 1992; Eldridge & Robson 1997). These studies showed that high stocking rates, which resulted in a reduction in cover, standing crop, litter, and in structurally unstable, compacted soil surfaces, led to lower infiltration rates compared to soils under zero to moderate grazing. A possible explanation for the contradicting results of this study is that the high intensity – low frequency grazing of the non-selective grazing system did not reduce cover significantly over a full grazing cycle (see Chapter 4), whereas mulching, soil biotic activities, structural stability and ultimately infiltrability increased. This corroborates the findings of Wood and Blackburn (1981) and Thurow *et al.* (1988) that infiltration rates of pastures grazed under a high intensity – low frequency system was similar to moderately continuously grazed pastures. The well documented phenomenon of compaction as a result of trampling (Thomas 1960; Van den Berg *et al.* 1976; Webb & Stielstra 1979; Dean 1992) was not evident in the infiltration results, probably because of the dry, sandy nature of the study site soils (Smoliak *et al.* 1972), and the short duration and low frequency of trampling (Thomas 1960).

Short duration grazing, which is a grazing system where a high density group of animals is quickly rotated through a number of camps, has also been shown to have negative impacts on pasture soils (McCalla *et al.* 1984; Thurow *et al.* 1988; Willms *et al.* 1990). It would appear that the frequency of disturbances is the key. Continuous or frequent disturbances may impact negatively on the type and amount of plant cover, litter deposition and soil bulk density. Infrequent, albeit heavy, grazing ensures a pulsed but adequate below-ground input of organic matter as a substrate for soil organisms, while soils are loosened and aerated but not compacted (Figures 5 and 6). The trend for both treatment and control soils to seal, and thereby reduce infiltrability, after an initial rain event (Table 3) is important in confirming field observations that intermound soils become hard and devoid of life when grazed areas are rested for too long (Savory 1983). The longer these unvegetated areas are exposed to raindrop impact and the baking sun, the more their physical structure can be expected to deteriorate. The challenge is therefore for the manager to avoid returning animals to a paddock too soon, and thus impact negatively on plant productivity, or wait too long and thus reduce rainfall effectiveness.

CONCLUSIONS

Four non-selective grazing treatments applied over three years, resulted in the maintenance of fertile mound habitats, with higher levels and turnover rates of OC. This is an important positive result in that any grazing system should aim at conserving, and expanding, the fertile patch matrix in order to retain scarce resources (water, nutrients) in the ecosystem. The high rates of respiration in the grazed open areas compared to the controls indicated an active microbial community, which had a number of consequences in the ecosystem. There was a trend for OC turnover to be more rapid in grazed open areas compared to controls, which is important for nutrient dynamics. Soil aggregates were stabilized, which led to an improved infiltrability and resistance to erosive forces. There appears to be a negative impact of NSG on TN, but whether this translates into a degradation in soil quality is as yet unresolved.

Thus far in this monitoring experiment the results tend to support the views of McNaughton *et al.* (1988) that intense herbivory results in positive consequences which reverberate through the ecosystem. It is however postulated that the duration and frequency of such herbivory is of utmost importance in terms of ecosystem impacts. Short duration, low frequency, intensive herbivory appears to be best suited to African ecosystems which evolved under conditions of large mammalian herbivory (McNaughton *et al.* 1988).

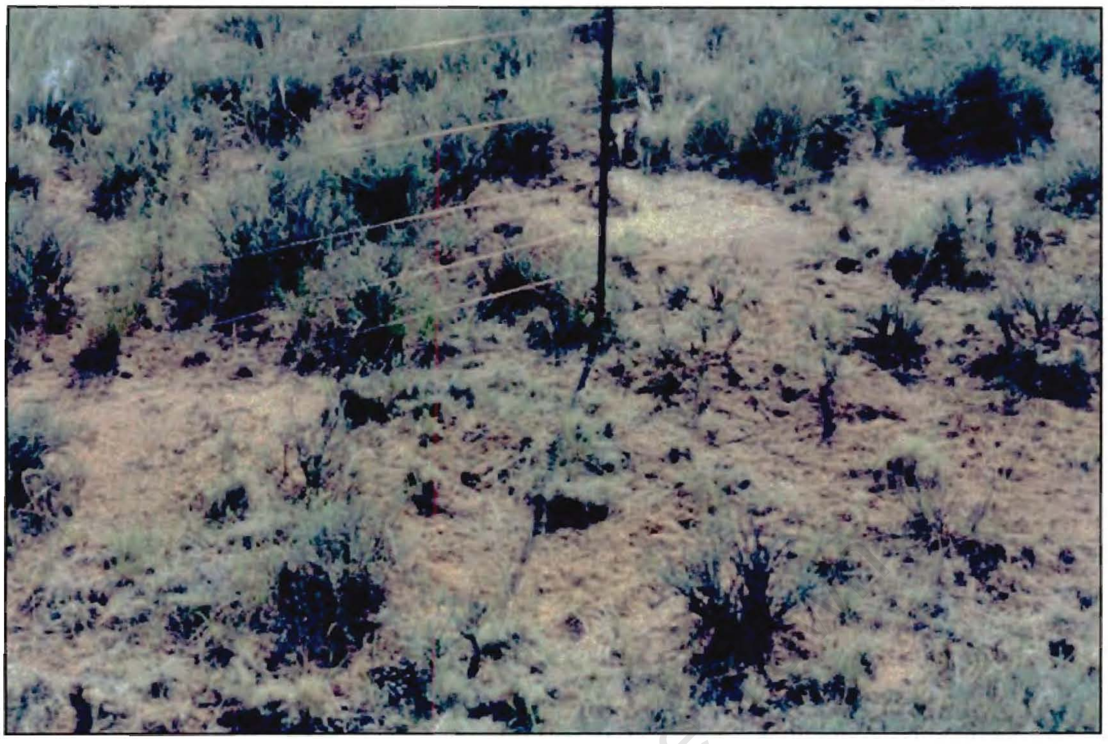

Figure 5 Heavily grazed and trampled karoo veld (bottom) compared to an exclosure (top)

Figure 6 Signs of hoof action and dunging after a NSG treatment.

Finally, soil biological (e.g. microbial respiration rates) and physical (e.g. aggregate stability / infiltrability) parameters are potentially useful integrated indicators of overall ecosystem health. Grazing affects these parameters and measuring them is therefore an appropriate way to detect changes in the soil-plant relationships (Lavado *et al.* 1996) and can help to avoid ecologically unsuitable grazing systems.

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CHAPTER 4

THE IMPACTS OF NON-SELECTIVE GRAZING (NSG) ON PLANT COMPOSITION, COVER AND PRODUCTION OF THE NAMA KAROO

INTRODUCTION

Grazing animals are highly selective in terms of area, growth forms (grass vs. shrubs), plant species, and plant parts (Danckwerts 1987). If animals are stocked at a low density in a large paddock, they will only select the most palatable, most nutritious young growth, and if left in that paddock for a long time (several weeks to months) they will regularly return to the young regrowth of the grazed plants (Danckwerts 1987). Less palatable and unpalatable grasses and shrubs will grow and propagate unhindered and eventually dominate the vegetation. This is evident in large parts of the Nama Karoo where shrubs like *Chrysocoma ciliata, Rosenia humilis, Pteronia* spp., and *Rhigozum trichotomum* have come to dominate vast areas of selectively grazed veld (Acocks 1953). Owing to the long-lived nature of these plants, lack of propagules of more preferred species, and reduced ecosystem functioning of the topsoil (Milton & Hoffman 1994), veld in this poor condition occupies a state from which it is unlikely to recover by resting alone.

Theoretically the ill-effects of selective grazing can be prevented in two ways; by forcing animals to graze non-selectively through the application of heavy grazing pressures; or by removing animals from the veld before they graze regrowth, and not allowing them to return to the same pasture until such time that grazed plants have recovered. In the first option less-palatable plants will either be ingested or trampled with a resultant turnover in aboveground biomass, whereas in the second option the less-palatable plants will remain untouched. The nutrients bound in this biomass and the space these plants occupy will ultimately lead to a moribund plant community where both rates of nutrient cycling (McNaughton *et al* 1988) and productivity are relatively low.

The present direction in grazing management is to combine high grazing intensity for short periods of time (a form of herding; see Savory 1987) with prolonged periods of rest (Biondini & Manske 1996). The two most common grazing systems developed within this paradigm are high-intensity, low-frequency grazing (HILFG also called non-selective grazing NSG) and short-duration grazing (SDG also called time-controlled grazing) (Manley *et al.* 1997). NSG has been advocated as a grazing system that reduces selective defoliation by forcing animals to eat more species including the less palatable ones; that reduces the number of times regrowth is removed by shortening occupation time; and that improves plant vigour and forage production (Acocks 1966). NSG mainly differs from SDG in that the rest periods exceed the relatively short 30 – 60 days recommended for SDG (Savory 1988). In fact rest periods of a year or longer are recommended for NSG (Acocks 1966), which obviously entails an unconventionally large number of small paddocks.

The premise with these high density systems is that by concentrating a large number of animals with different feeding habits (e.g. mixed herds of cattle, sheep, and goats) into a small paddock for a short time, animals will be forced to defoliate most of the plants, irrespective of palatability. This will reduce the competitive advantage that the less-palatable species have under low density grazing systems. The hoof-action (trampling/treading), dung and urine deposition resulting from high density grazing is believed to increase biomass turnover rate, improve the topsoil environment, stimulate production, and allow more rapid shifts in species composition (Acocks 1966; McNaughton *et al.* 1988; Savory 1991). The relatively long rests should give defoliated plants ample time to recover, flower, and set seed, and for seedlings to recruit in the microsites created by hoof-action. On the other hand, Danckwerts (1987) has argued that species selective grazing persists, even under very high grazing intensities. Roux (1967) has vigorously opposed NSG, stating that it will bring about a reduction in plant cover, an overutilization and local extinction of palatable plants, destruction of new cohorts of seedlings, and soil compaction.

In general the studies reported by O'Reagain & Turner (1992) and the experiments designed by, amongst others, Willms *et al.* (1990) and Manley *et al.* (1997) to test the merits of these high density grazing systems, did not implement a sufficient number of small paddocks to concentrate animals effectively, and to allow for long enough rest periods. They also did not stock mixed herds to reduce selectivity.

It is postulated that grazing animals, correctly manipulated, can be used as "tools" to maintain and improve biological diversity and production, and stimulate ecosystem processes necessary for veld restoration. This holds important consequences for rangeland management in the Karoo, which will be revolutionized by the development of different techniques for concentrating and moving animals instead of using costly multi-camp infrastructures.

Apart from grazing animals and the system in which they are managed, rainfall amount and seasonality is another major determinant of veld condition in these arid and semi-arid areas (Westoby *et al.* 1989). It is therefore important when evaluating the sustainability of a particular grazing system to consider the relative roles of weather versus grazing regime in influencing the dynamics of the plant communities. A lack of understanding of this interaction may lead to non-sustainable management decisions (Walker 1993).

This study was designed to compare the impacts of a high density grazing system (NSG) with no grazing (control) in terms of the following vegetation parameters: (1) perennial, emphemeral, and litter cover; (2) perennial species composition; (3) aboveground grass and shrub production; (4) abundance of standing-dead shrubs; (5) survival, plant size, number of shoots and flower heads of the dominant shrub *Pentzia incana*; and (6) seedling abundance. Some of these parameters were also related to rainfall during the study period.

STUDY AREA

Fieldwork was conducted at the farm Elandsfontein which is described in more detail in the General Introduction to the thesis.

METHODS

See Chapter 3 for a description of the experimental design and treatment.

Plant cover and species composition

Vegetation surveys were conducted annually before the treatment was applied. In 1997 a second survey was conducted approximately 6 months after the treatment. Five 50 m transects were spaced 10 m apart and permanently marked inside each exclosure as well as in the 50 m x 50 m survey areas adjacent to the exclosures. Care was taken to avoid stock paths which tend to develop against fence lines. Surveys were conducted inside exclosures and outside by placing a line, with knots every ½ m, between the markers of each transect and descending a sharp pointed metal rod vertically from directly above each knot. This gave a total of 500 descending points per survey. According to Brady *et al.* (1995) this sampling intensity is sufficient to detect a 4 % change in plant basal cover with an 80 % probability. If the rod descended within the canopy spread of a plant (defined as "the area of ground encompassed by the vertical projection of an imaginary line, circumscribing the perimeter of the plant's canopy, onto the ground surface" (Roux 1963)) it was noted as a canopy spread strike next to the specific species. When overlapping canopies were encountered each species contributing to the overlap was noted as a strike. When the rod struck the basal parts of a perennial plant it was noted as a basal strike next to the specific species. In a similar way strikes of bare ground, prostrate plant litter, and ephemeral plants were also noted. Strikes of litter, and ephemerals underneath the canopies of perennial plants were also noted.

Percentage cover was determined by expressing the number of strikes as a percentage of total points. Relative contribution to the grass and shrub guilds, respectively, was expressed as the number of strikes per species as a percentage of the total grass or shrub strikes. This simple survey technique is suitable for long-term monitoring of major vegetation shifts but is unlikely to detect subtle compositional changes involving rarer species (Novellie & Strydom 1987).

Aboveground plant production

After the treatment was applied to all four replicates in May 1997, fifteen 1 m x 0.5 m quadrats were positioned in a systematic random manner inside and outside the exclosure of paddocks A, E and F. All the perennial plant biomass inside each quadrat was cut to stubble height, and grass and shrub material placed into separate bags, dried at 80°C for 24 hours, and weighed. After a five month growth period the same procedure was repeated in October 1997, while care was taken to avoid re-sampling any of the original quadrats. In a similar way grass and shrub standing crop biomass was determined after the treatment in June 1998, and after a growth period of 5 months, aboveground standing crop biomass was again measured in November 1998.

Survival and plant size of *Pentzia incana*

In June 1996 ten *Pentzia incana* individuals were selected in a systematic random manner from inside and outside each exclosure. Basal diameter, canopy height, and two perpendicular canopy diameter readings (to the nearest 10 mm) were measured on each individual using a steel ruler, and each plant was marked with an aluminium tag tied to the basal parts. The fate and size measurements of these individuals were then monitored in subsequent surveys.

Flower heads and shoots of *Pentzia incana*

On two occasions during the study period, when *Pentzia incana* was actively growing and flowering, ten individuals were selected in a systematic random manner from inside and outside the exclosures. After having taken the basal diameter, canopy height, and two canopy diameter readings, a wire quadrat (85 mm x 85 mm) was lowered onto the canopy of each individual, and the total number of flower heads and shoots counted inside the canopy volume framed by the quadrat. Actively growing shoots were identified by their soft and lighter-shade green tips.

Seedling abundance

A 2m x 2m plot was permanently marked inside and outside each exclosure in June 1996. The two plots per paddock were demarcated as close as possible to each other with the exclosure fence separating them. This was done to reduce the effects of environmental variability on seedling emergence. However, cognisance was taken of a possible stock path developing next to the fence. Into each plot 50 g of seed of the shrub *Osteospermum sinuatum* (bietou) and 100 g of seed of the grass *Chaetobromus dregianus* (hartebees grass) were hand sown. Although the grass species does not occur naturally at Elandsfontein, both are arid zone species of which abundant seed was available (H. Botha pers. comm.), and are therefore suitable species for evaluating the effect of the treatment on seedling emergence. Seeds were also collected from a shrub (*Tetragonia sarcophylla*) which grows at Elandsfontein and the same amount was hand sown into all the seedling plots in June 1998 after the treatment was applied to all four replicates.

On four occasions seedlings were counted in the plots. Initially only *Osteospermum* and *Chaetobromus* seedlings were counted but on subsequent occasions all seedlings, sown and naturally occurring, were noted. The abundance of *Tetragonia* seedlings will be monitored in the future.

Aboveground standing-dead shrubs

In June 1995, after the first treatment was applied with the exclosures in position, twenty plots (25 m^2) were positioned in a systematic random manner inside and outside the exclosure of paddock B. All standing dead shrubs inside the plots were noted, and their basal diameters measured with a steel ruler to the nearest 10 mm.

Statistical analysis

The overall null-hypothesis I tested was that change over time in the measured parameters would not differ significantly between the treatment (non-selective grazing outside the exclosures) and the control (no grazing inside exclosures) areas. The treatment was repeated through the experiment, and with the exception of number of standing-dead plants, all the other parameters were measured on more than one occasion. This design lends itself to analysis of variance for repeated measurements in order to describe the way in which the treatment effects change differentially with time (Payne 1995). Analyses were performed using the AREPMEASURES procedure in GENSTAT (version 5).

To test the hypothesis that rainfall determines pattern rather than grazing, a rainfall index was calculated by adding the rainfall of the three months prior to the survey date to the total rainfall of the calendar year preceding the survey date. The rainfall index was used in regression analyses with total, grass, and shrub canopy cover.

RESULTS

Plant cover and species composition

Low total canopy cover values were measured in 1995 after the severe drought year of 1994. After this, values increased significantly over time (Figure 1a) in both treatment and control areas and followed a trend that correlated significantly with the rainfall index (Figure 2a). The grazing treatment had no significant effect on total canopy cover. Being the major contributor to total canopy cover, the grass guild showed the same patterns as total cover (Figure 1c & 2b) whereas the shrub guild showed a significant increase in canopy cover over time (Figure 1e) but not as dramatically and as strongly correlated with rainfall index as grass (Figure 2c). Treatment had no significant effect on shrub canopy cover. However, there was a trend for a decrease in shrub canopy cover in the treatment plots; the difference between treatment and control was significant in 1998 ($t = -3.28$, $P < 0.05$). The slopes and intercepts of regression lines for treatment and control data sets did not differ significantly for any of the relationships shown in Figure 2, although cover in the treatment areas always showed a stronger relationship with rainfall index compared to that of the control areas (Figure 2a – c). Although there were no significant trends in basal cover values (Figure 1b, d, f), there was a trend for grass basal cover to increase and shrub basal cover to decrease over the sampling period. Treatment had no significant effect on any of the basal cover measurements.

There was significantly more litter recorded in the treatment compared to the control areas ($F = 33.6$, $P < 0.01$) (Figure 3a). There was a significant relationship between litter and rainfall index for control ($r^2 = 0.53$; $P < 0.01$) but not for treatment ($r^2 = 0.05$; ns). Treatment had no significant impact on the plant cover of ephemeral species (Figure 3b).

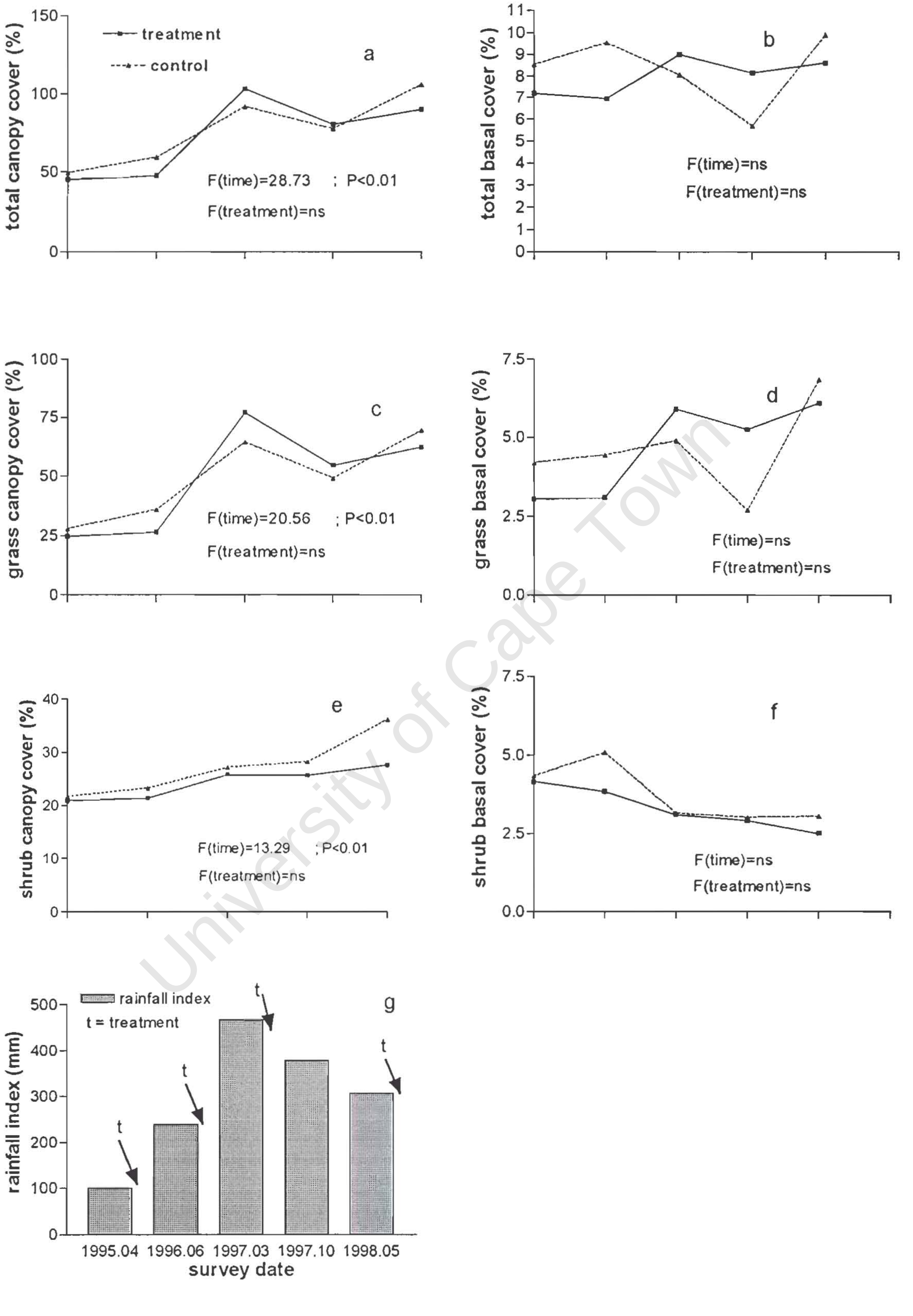

Figure 1 Total canopy cover and basal cover (a + b), grass canopy and basal cover (c + d), shrub canopy and basal cover (e + f), and rainfall index (g) over time (five survey dates). Results from analyses of variance are also given.

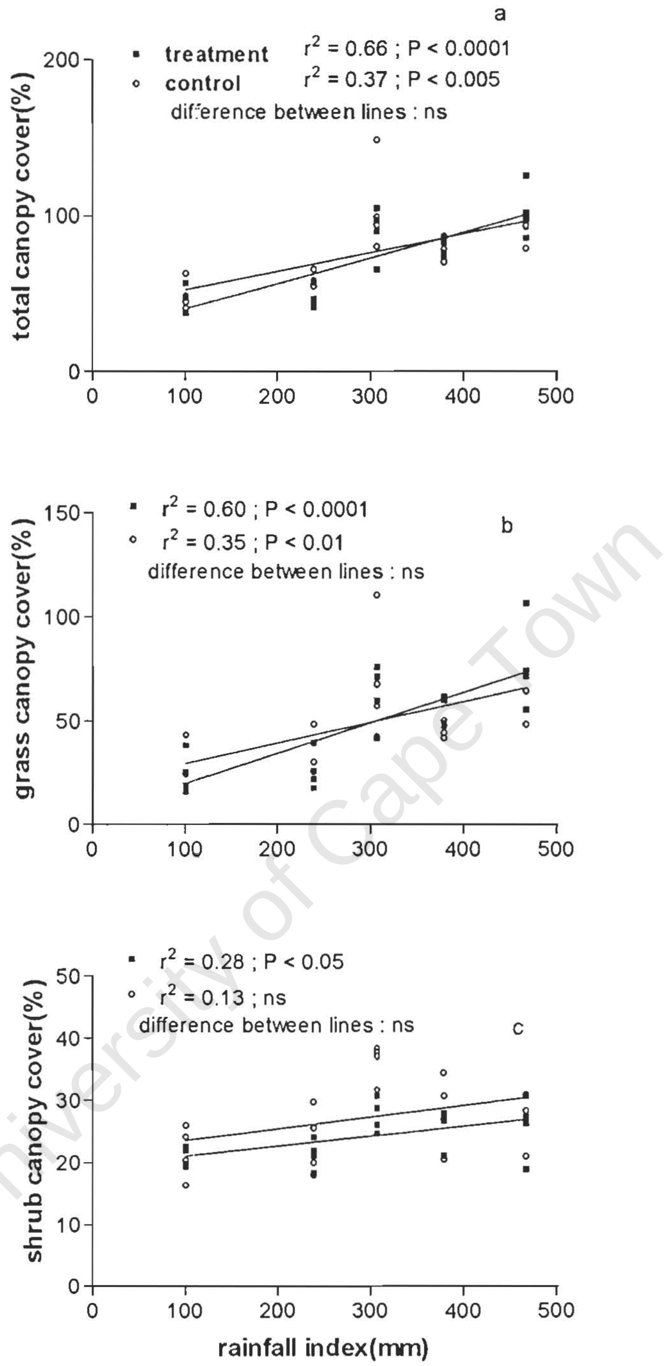

Figure 2 Regression of the relationships between rainfall index and total canopy (a), grass canopy (b), and shrub canopy (c) cover. Statistical results of goodness of fit, and differences between treatment and control regression lines are also given.

The most abundant grass species in the study area, *Eragrostis lehmanniana*, showed a significant increase in relative basal cover over time in both treatment and control areas (Figure 4a). *Stipagrostis ciliata* showed a significant decrease in relative basal cover over time (Figure 4b), whereas *Aristida congesta* spp. *congesta* also showed a decline in relative abundance over time but this was only significant in the canopy cover measurement (Figure 4c). These trends in grass species composition were independent of treatment. In the shrub guild, canopy cover of the dominant *Pentzia incana* increased, whereas cover of *Lycium cinereum* decreased (Figure 5a – c). Again this trend was independent of treatment.

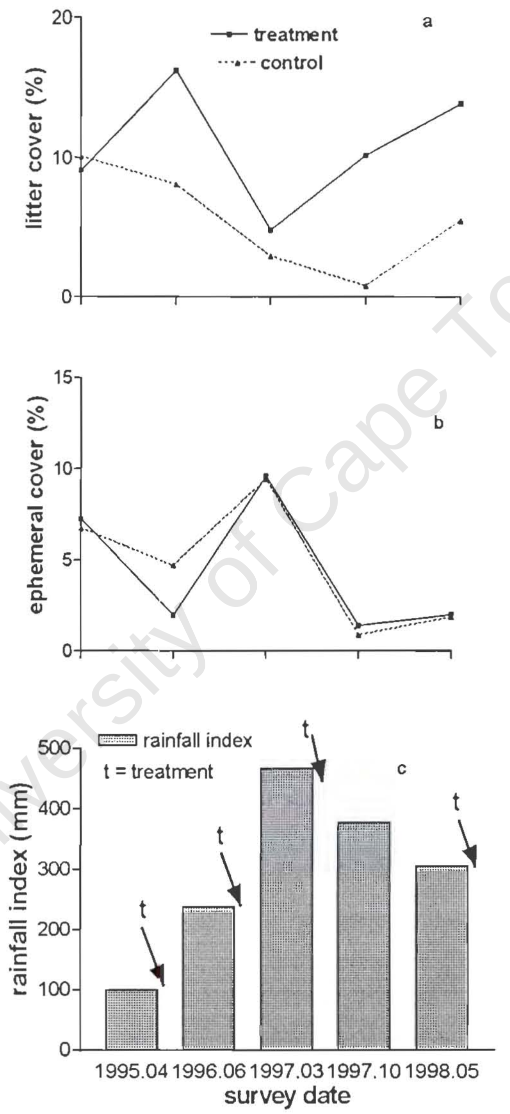

Figure 3 Litter cover (a), ephemeral cover (b), and rainfall index (c) over time (five survey dates).

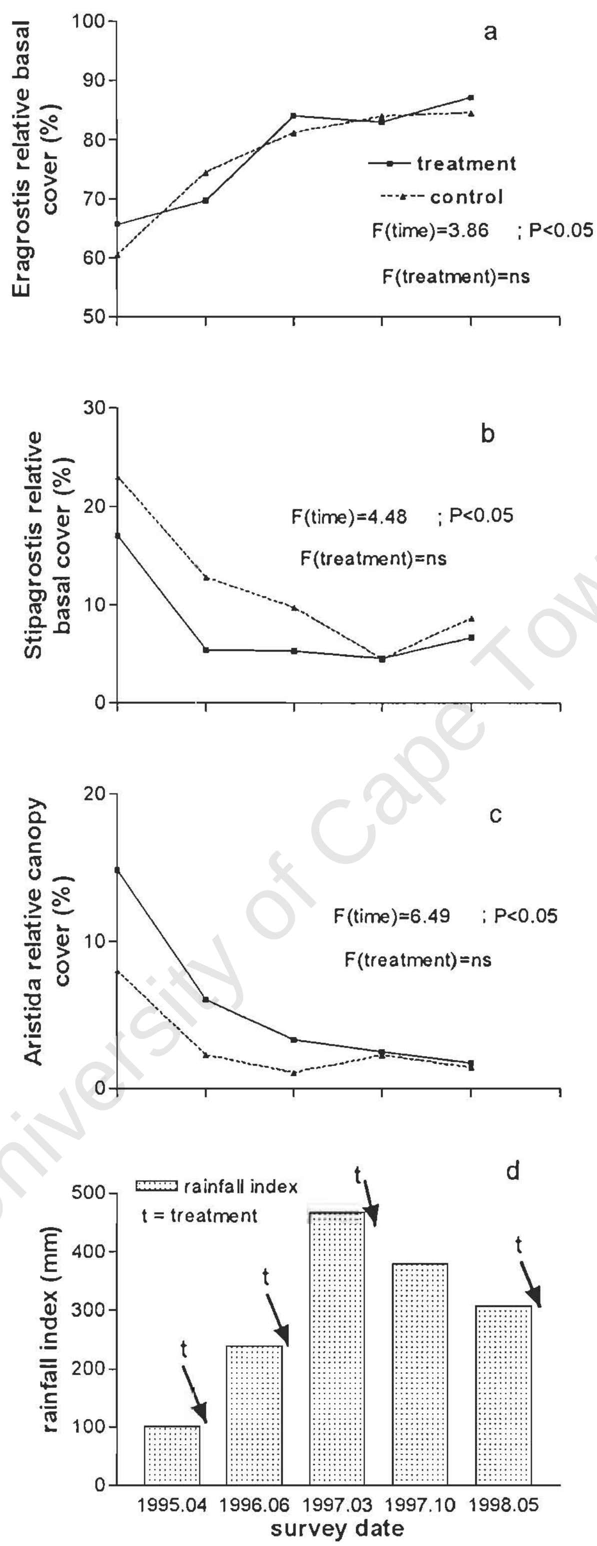

Figure 4 Changes in *Eragrostis lehmanniana* and *Stipagrostis ciliata* relative basal cover (a + b), *Aristida congesta* spp. *congesta* relative canopy cover (c), and rainfall index (d) over time (five survey dates). Results from analyses of variance are also given.

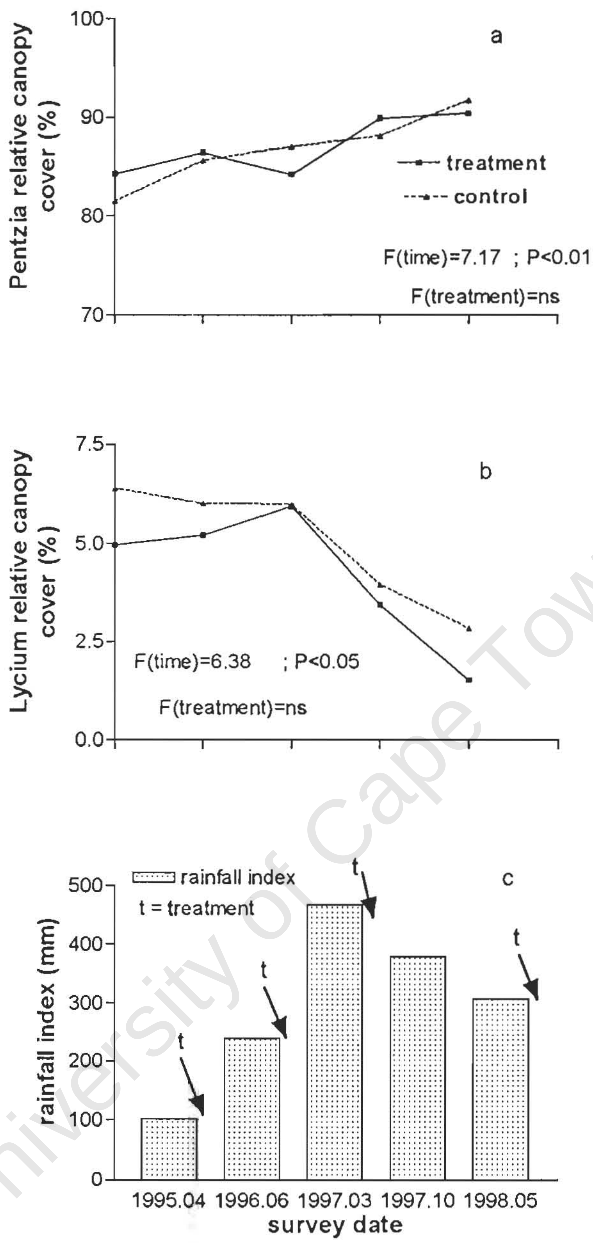

Figure 5 Changes in *Pentzia incana* (a) and *Lycium cinereum* (b) relative canopy cover, and rainfall index (c) over time (five survey dates). Results from analyses of variance are also given.

Aboveground plant production

Although both 1997 and 1998 had exceptionally dry winters, there was, on average, a greater grass biomass accumulation in the treatment compared to the control areas (Table 1). However, there were no significant trends in shrub biomass. Aboveground plant biomass in the treatment areas showed the potential to recover to control levels within five months after a non-selective grazing treatment.

Table 1 Grass and shrub aboveground biomass (gDM/0.5 m^2) for treatment and control areas in three paddocks at Elandsfontein. Statistical analysis is a paired t-test (one-tail).

Sample Date	Paddock	GRASS		SHRUB	
		Treatment	Control	Treatment	Control
May 1997	A	27.9	39.9	71.4	59.7
	E	57.2	91.5	58.4	48.1
	F	52.4	68.2	82.7	62.4
October 1997	A	33.5	36.9	73.6	65.2
	E	77.8	62.0	76.8	60.9
	F	53.9	45.2	76.3	60.1
Production (May – Oct 1997)	A	5.6	-2.5	2.2	5.5
	E	20.6	-29.5	18.4	12.8
	F	1.5	-23.0	-6.4	-2.3
		$t = 2.26$, $p = 0.08$		$t = 0.19$, $p = 0.43$	
June 1998	A	4.0	20.7	28.5	67.5
	E	15.7	38.5	66.9	45.4
	F	7.4	52.9	52.1	67.8
November 1998	A	17.6	30.6	49.0	90.8
	E	47.9	38.3	42.2	53.0
	F	30.6	32.8	67.7	68.0
Production (June – Nov 1998)	A	13.6	9.9	20.5	23.3
	E	32.2	-0.2	-24.7	7.6
	F	23.2	-20.1	15.6	0.2
		$t = 2.24$, $p = 0.08$		$t = 0.47$, $p = 0.34$	

Survival and plant size of *Pentzia incana*

After having received three treatments since tagging in 1996, five of the initial 40 tagged *Pentzia incana* individuals (12.5 %) were not found in the 1998 survey. All 40 tagged shrubs were found in the control areas. *Pentzia incana* showed significant growth in canopy surface area and canopy height between the 1996 and 1998 surveys, but treatment did not significantly alter this growth pattern in the grazed shrubs (Figure 6). Basal diameter was not influenced by time or treatment.

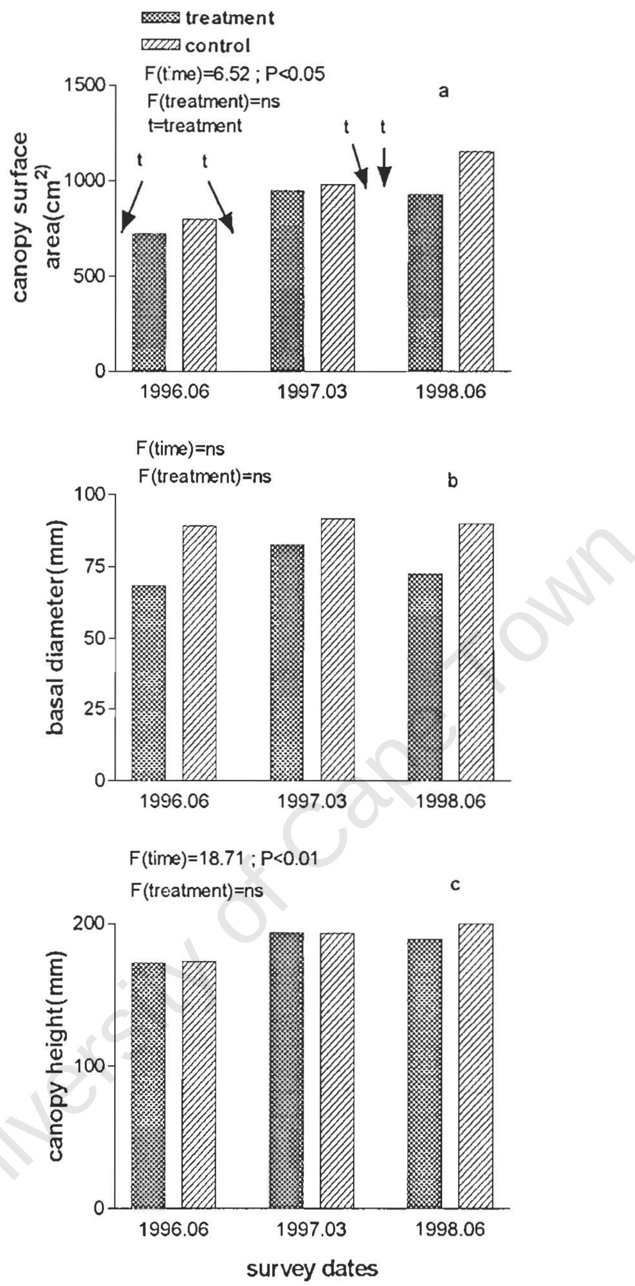

Figure 6 Plant size measurements for tagged *Pentzia incana* plants on three survey dates. Results from analyses of variance are also given.

<u>Flower heads and shoots of *Pentzia incana*</u>

The number of flower heads on *Pentzia incana* varied significantly with time (Figure 7a). There was a trend for both the number of flower heads and actively growing shoots of this species to be higher in the treatment than the control on both survey dates (Figure 7a, b).

Seedling abundance

Throughout the survey, seedling abundance in the treatment was significantly higher than in the control (Figure 7c). In both treatment and control plots none of the hartebees grass (*Chaetobromus dregianus*) seedlings survived long after the September 1996 survey. A considerable number of bietou (*Osteospermum sinuatum*) seedlings survived until the 1997 survey, but none was present in 1998.

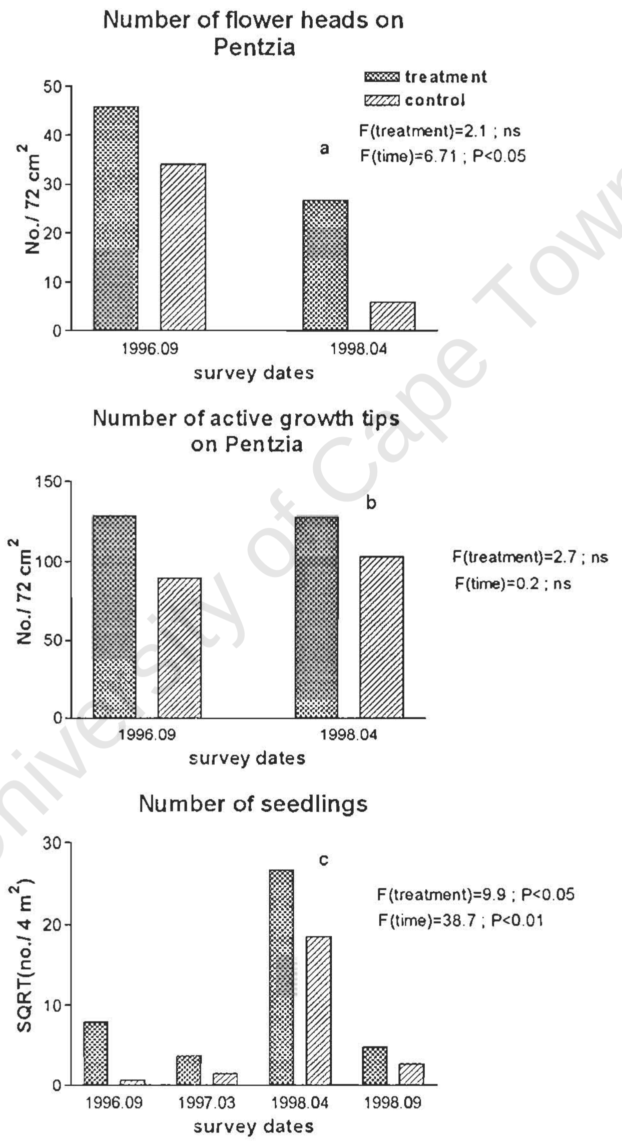

Figure 7 Numbers of flower heads (a) and active growth tips (b) on *Pentzia incana*, and seedling numbers (c) on different survey dates. Results from analyses of variance are also given.

Aboveground standing-dead plants

While the number of standing-dead shrubs was significantly lower in the treatment than the control, the average basal diameter was significantly higher (Table 2).

Table 2 Numbers of standing-dead shrubs counted in June 1995 in treatment and control areas of paddock B after a non-selective grazing treatment in April 1995.

	Treatment		Control			
	Mean	Standard error	Mean	Standard error	t-value	Significance
Number of standing-dead shrubs per ha	4440	350	8340	482	6.549	P < 0.001
Basal diameter of standing-dead shrubs (mm)	76	4.5	66	2.3	2.154	P < 0.05

DISCUSSION

The results show that NSG increased the amount of prostrate litter, possibly as a result of the trampling into the ground of standing-dead plants by livestock. NSG also increased germination and emergence of seedlings, and stimulated aboveground grass production, and flower and shoot production of the dominant shrub species in the study area. The observed changes in perennial grass and shrub canopy cover appeared to be mainly a response to rainfall. There are also indications that rainfall rather than grazing, influenced certain biotic interactions which, in turn, may have altered vegetation composition.

Plant cover, composition, and production

It is well established that rainfall is a strong driving force in the dynamics of arid ecosystems (Norton 1978; Hoffman *et al.* 1990; Turner 1990; O'Connor 1991; O'Connor & Roux 1995; Alzérreca-Angelo *et al.* 1998). In this study the grasses, with their characteristic ability to respond rapidly to rainfall events (Novellie & Strydom 1987; Hoffman *et al.* 1990; O'Connor & Roux 1995), showed a more closely correlated response to rainfall compared to shrubs. Rainfall over the longer term (e.g. annual) builds or slows momentum in canopy cover - more so in grasses than in shrubs - probably through influencing establishment, or death of tufts and shrubs. Superimposed onto this pattern of cover dynamics, short-term rainfall (e.g. quarterly) affects cover through increasing or decreasing plant size. NSG did not influence these changes in cover.

It can be expected that the drought of the early 1990's resulted in the death of many mature plants in the study area (Milton *et al.* 1995), creating space in an environment where competition for resources is great (Esler & Cowling 1993). When the drought was broken by substantial rains, particularly summer and autumn rains, in the four years following 1994, the population of *Eragrostis lehmanniana* increased (as measured by basal cover) independent of grazing treatment, while *Stipagrostis ciliata* and *Aristida congesta* ssp. *congesta* appear to have declined in relative abundance. *E. lehmanniana* can set seed within the first growing season, produce large amounts of small seeds on each plant, and maintain large soil seed banks (Sumrall 1990 as cited in McClaran & Anable 1992). This species is generally regarded as a vigorous competitor in disturbed areas, but also as a valuable pasture grass in the very dry parts of the country (Van Oudtshoorn 1991). *S. ciliata* is a valuable "climax" pasture grass which decreases with over-utilization, while *A. congesta* is a pioneer species of lower grazing value (Van Oudsthoorn 1991). The invasive characteristics of *E. lehmanniana* (McClaran & Anable 1992) together with the ability of this species to increase in cover under high summer temperatures when moisture is not limiting (Hoffman *et al.* 1990), explains the significant increase in relative cover compared to the less-competitive *S. ciliata* and *A. congesta*. Although the two species which supposedly are indicators of retrogression, *E. lehmanniana* and *A. congesta*, show opposite patterns, a disturbing factor is the decline in *S. ciliata* and *S. obtusa* populations (A. Lund pers. comm.). The abundant summer rains could be seen as a singular extreme event (Westoby 1980), and that *S. ciliata* and *S. obtusa* will overcome the competition from *E. lehmanniana* in due time. It remains to be seen whether this will happen, and if and how grazing will affect it.

The dominant karoo shrub in the study area, *Pentzia incana*, is of average palatability and the staple forage on most farms in the Nama Karoo (Shearing 1994). As in *Eragrostis lehmanniana* the significant response in *P. incana* cover over time is probably driven by rainfall (Roux & Theron 1987). The ability of *P. incana* to propagate itself vegetatively (Hoffman & Cowling 1987) and the infrequent nature of the heavy defoliations (Hobson & Sykes 1980) may be responsible for the lack of impact of NSG on this response. The basal cover of *P. incana* remained the same throughout the recovery phase after the drought. This means that the recovery strategy is based on survival of individuals with vigorous canopy growth when resources become available. The population explosion of *E. lehmanniana* probably inhibited the recruitment of *P. incana* seedlings. The smaller canopy areas of the grazed *Pentzia* plants (Figure 6a) is reflected in the lower canopy cover towards the end of the study period (Figure 1e). This may be beneficial in that young forage plants may be more productive than moribund ones (Milton 1992a). One can only speculate about the apparent concomitant decline in cover of the thorny shrub, *Lycium cinereum*. *L. cinereum* is well utilized by goats (Shearing 1994). It is regarded as an undesirable problem plant when it becomes too dense. According to the landowner *L. cinereum* is on the decline across the farm because of goat grazing and cattle trampling. However the result of this study indicate that this decline is independent of grazing.

The absolute dominance of two species in this vegetation, *E. lehmanniana* in the grass guild and *P. incana* in the shrub guild, indicates that this veld is in a state where it is resilient to infrequent heavy defoliations, and where rainfall causes substantial changes in the density and cover of the grass, and in the cover of the shrub species (Williams 1968). Thus far, the data cannot support the argument that non-selective grazing eliminates the competitive advantage of less-palatable species (Acocks 1966), thereby resulting in improved veld composition. Highly palatable succulent species like *Delosperma* sp. did not disappear from the grazed areas but also did not show any significant increase in abundance.

This study shows a strong case for compensatory growth after NSG, especially in the grass guild. It supports Dormaar *et al.* (1997) who found in their study of the impact of rotational grazing on mixed prairie vegetation in Canada, that stocking at levels designed to remove 50 % of standing crop had no significant impact on biomass or species composition. In fact they, as well as other researchers (see McNaughton 1979 and McNaughton *et al.* 1988), found a trend towards greater plant vigour and productivity on rotation-grazed compared to zero-grazed areas. However, the relationship between grazing and productivity is highly controversial (Belsky 1986). The dominant grass in our study area, *Eragrostis lehmanniana*, is tolerant of high grazing intensities (McClaran & Anable 1992) and shows signs of compensatory growth after defoliation during the slow-growing, cool season. This could be attributed to the removal of moribund material, and the improvement of water and nutrient cycling by litter deposition (Willms *et al.* 1993; Chapter 3), and the faster turnover of biomass (West *et al.* 1979). The insignificant biomass accumulation of shrubs in the treatment compared to control may be explained by the grazing being applied in autumn – early winter, the "initiating" phase of shrub growth (Venter 1962), and by the dry winters of 1997 and 1998. Du Preez (1972 as cited in Hoffman & Cowling 1987) has shown that defoliation during or immediately after a growth period greatly decreased dry matter yields of a number of karoo species. It is also possible that with the greater spatial discontinuity of shrub biomass, the sample size of 15 quadrats (1 m x 0.5 m per quadrat) was insufficient to detect any significant changes in shrub biomass.

When some of the long-lived karoo shrubs die, the plant skeleton may stay upright for a long time. In our study more than 90 % of tagged standing-dead shrubs in the control areas were still upright after two years. This material is woody, and is avoided by grazing animals. It represents immobilized nutrients which will be released more rapidly for re-cycling when in contact with decomposer organisms in the topsoil (Savory 1991). Concentrated hoof-action (livestock trampling) as applied in this experiment resulted in much of the dead material returning to the ground where it contributed to the very important litter layer. The larger litter load in the grazed areas could have important positive consequences for ecosystem processes (McNaughton *et al.* 1988; Savory 1991; Milton 1992a; Tongway 1994; Chapter 3). Hoof-action also resulted in some old shrubs with spreading canopies being killed (pers. obs.), which is important for the establishment of young individuals through gap formation (Gibson *et al.* 1987; Milton & Hoffman 1994). The short periods of heavy trampling did not result in an increase in ephemerals at the expense of perennials. This is important since sustained secondary production is strongly dependent on conserving, and improving key perennial elements of the vegetation (Curry & Hacker 1990).

Survival, plant size, shoots, and flower heads of *Pentzia incana*

Although *Pentzia incana* has a higher feeding value than lucerne (Henrici 1945) it is of average palatability, probably because of its bitter, pungent smell (Shearing 1994). It forms a vital part of sheep's diet and is therefore of economic importance (Le Roux *et al.* 1994). Consistent with West *et al.* (1979) and Milton (1992a), the grazing treatment in this study did not significantly influence the survivorship of established plants. After three treatments of severe defoliation and trampling, the majority of the tagged shrubs survived, and canopy surface area and canopy height increased by an average of 29 % and 10 %, respectively. Although lower than 45 % and 15 % respectively for the ungrazed shrubs, it was not significantly so. *P. incana* recovers well after defoliation (Venter 1962); recovery depends on the season and frequency of defoliation, both of which affect root reserves (Venter 1962; Hobson & Sykes 1980). A grazing frequency of once a year appears to be tolerable; however, the non-selective grazing treatment in autumn 1997 might have been during an active growth phase which could have affected the root reserves of *P. incana* negatively (Venter 1962). Van der Heyden (1992) found that another important forage species, *Osteospermum sinuatum*, is also tolerant of defoliation and regained normal proportions when protected for a single growing season. Contrary to our findings, workers like Crisp (1978) have found a decline in the population of *Atriplex vesicaria* under heavy grazing in arid rangelands of Australia, while Webb & Stielstra (1979) showed that individual *Ambrosia dumosa* plants were up to 65 % smaller in volume in heavily grazed areas in the Mojave desert. I postulate that these, and other contradictions on the impacts of heavy grazing on arid rangelands originates from different concepts of "heavy grazing" and also the grazing system under which "heavy grazing" is applied. It is not severe defoliation (up to 80 % of above-ground biomass removed) *per se* that kills grazing–tolerant plants (most palatable species in arid areas) but how it is applied (when, for how long, and how frequent).

Although not significant the higher numbers of actively growing shoots per unit canopy surface area in grazed *Pentzia* plants compared to ungrazed plants, probably points to compensatory growth in the grazed plants. Whether plants respond positively or negatively to herbivory is a highly controversial point (see Trlica & Rittenhouse 1993) but probably depends on the pre- and post-harvest conditions of the plants and their environment (Trlica & Rittenhouse 1993). Ungrazed *Pentzia* and many other karoo shrubs becomes moribund and woody, and tend to be avoided by animals. This happens under low grazing pressures when only certain shrubs are selected and animals return to those individuals for a "second bite" because of the young growth. Herding a large number of animals into a small paddock forces the animals to defoliate most, if not all, of the plants. Depending on the frequency of this treatment the plants do not get a chance to accumulate old woody biomass. Moribund biomass is indicative of a deteriorating forage condition, and tends to clog the system with immobilized nutrients (McNaughton *et al.* 1988).

Flowering and seed set are vital for replenishment of seed banks and for the recruitment of new individuals (Hoffman & Cowling 1987; Eriksson & Ehrlén 1992; Milton 1992b). It would seem that *Pentzia incana* flowering is not restricted to a specific season but is more influenced by cool season rainfall. Milton & Dean (1990) found that continuous, heavy grazing by sheep in the Succulent Karoo reduced the canopy size and flowering of preferred forage species. Although canopy sizes of *P.incana* individuals were reduced by grazing in our experiment, flowering was stimulated. It is possible that the defoliation frequency of once a year was low enough to stimulate shoot production from lateral buds, with a consequent increase in flower heads (Garrison 1953). This has the potential to increase the availability of viable *P. incana* seeds, and therefore proportional representation in the seedling assemblage and ultimately in the vegetation.

The non-selective grazing applied in this experiment resulted in the annual removal of a substantial proportion of the aboveground shrub biomass. However, there is no evidence that this treatment had negative short-term impacts on the survival, growth and reproduction of the dominant shrub species *Pentzia incana*. In fact, it appears that *P. incana* shows a flush of compensatory growth after short-duration, heavy defoliation which is reflected in increased shoot and flower head production. The long-term repercussion of this could be that instead of stimulating plant diversity by defoliating non-selectively, this grazing system could lead to an even greater dominance by *P. incana* (Brown 1985). Severe drought, which kills adult *Pentzia* plants (Milton *et al.* 1995), might be necessary to create gaps for other more palatable species to establish.

Seedling abundance

The herding of mixed livestock for short timespans resulted in higher seedling densities in treatment compared to control areas. A high seedling mortality was measured (see 1998 surveys, Figure 7c) but contrary to the results of Salihi & Norton (1987) and Defossé *et al.*

(1997) this could not be related to grazing impact. Milton (1995) also found that seedling densities were greater in karoo sites influenced by grazing, insects, litter and shade compared to undisturbed, exposed sites. She also found that the size of perennial seedlings were influenced by sheep grazing but not their survival. Depending on seedling survival rate, the higher seedling densities in the grazed areas should in time be reflected by higher cover, especially basal cover, readings. This together with the pitted, and litter covered soil surface could have important consequences in reducing run-off, conserving topsoil, and revegetating problem patches of veld (Howell 1976; Gibson *et al.* 1987).

CONCLUSIONS

Thus far NSG had very little impact on the cover dynamics, and species composition of the vegetation at the study area. Rainfall is a more important driving force than grazing treatment of the fluctuations in plant cover. There is no evidence that this grazing system promotes or reduces diversity. The few species shifts observed were more likely the result of competitive interactions. After years of selective grazing the density and reproductive output of the more palatable shrubs are low. It will take a long time for these populations to increase in abundance to such an extent that it is measurable with our survey techniques. The competition from grazing-tolerant plants occupying space, is another restriction for major species shifts to occur.

This study shows that the grassy Nama Karoo veld can support more animals and higher grazing pressures than what is generally accepted, as long as selective defoliation is reduced, and occupation time is kept short by the grazier (± 2 weeks maximum) with a rest of up to a year or even longer. It was, however, observed that if rest periods become too long the beneficial impacts of non-selective defoliation with concomitant hoof-action were slowly lost and the veld lost "momentum" in terms of litter deposition and plant vigour.

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CHAPTER 5

AN ECOLOGICAL ECONOMIC SIMULATION MODEL OF A NON-SELECTIVE GRAZING (NSG) SYSTEM IN THE NAMA KAROO

INTRODUCTION

Annual precipitation is known to be the major determinant of primary production in arid and semi-arid regions (Seely 1978; Rutherford 1980). The Nama Karoo biome of South Africa is characterised by a low (100 – 300 mm) and variable annual rainfall (Venter *et al.* 1986). This uncertainty, together with the large variety of terrain and soil characteristics of these karroid landscapes (Visser 1986), results in grass and shrub biomass production which is highly variable in space and time. Variability in total rainfall and seasonality of rainfall results in a rangeland which shows episodic, event-driven dynamics, which may include shifts in the relative biomass of grasses and shrubs (Milton & Hoffman 1994). The frequency of disaster-magnitude droughts (Vogel 1994) with the consequent losses of perennial grasses and shrubs, invasion by alien weeds, and drastic decline in carrying capacity (Milton *et al.* 1995) adds further uncertainty (Ellis 1994) to the survival of land owners who have to make a sustainable living in these extensive pastoral areas. In former times the stochastic availability of food and water to domestic livestock was partially overcome by nomadic farming practices (Vorster 1994; Dean *et al.* 1995). With the establishment of the mining industry and accompanying industrial development, a large market for food and other agricultural commodities was created. This initiated the change from subsistence to commercial stock farming on privately owned rangelands (Vorster 1994). Today many commercial farmers are faced with some form of dryland degradation (Dean *et al.* 1995), decreasing carrying capacities, lower product income, and higher input costs and interest rates (Vorster 1994). They cannot afford to expand their farming enterprises by buying more land and, therefore, have to find other ways to cope with the environmental variability.

Solutions to these problems include the establishment of simple, low-input farming systems using adaptive and opportunistic management (O'Reagain & Turner 1992; Scoones 1994; Vorster 1994; Wiegand *et al.* 1995); better adapted animal genotypes; invader plant control; drought insurance policies; and drought fodder crops (Vorster 1994). Various farming systems for the Karoo have been devised by agriculturists over the years; however, most are based on conceptual models that have no or inadequate empirical support (Hoffman 1988; O'Reagain & Turner 1992). None has really succeeded in coping with the high spatial and temporal variation in productivity. The realities are that optimistic assessment of veld condition and carrying capacities, overstocking, profitable but maladapted animal types, and state subsidies for supplementary feeds, infrastructure, and stock reduction (Milton & Dean 1995) have created a situation of pseudo-prosperity (Vorster 1994). With the phasing out of financial aid, it has become essential to investigate alternative farming strategies to reduce poverty and depopulation of rural areas, and so avoid major socio-economic problems (Vorster 1994). One such an approach is to devise a system which allows for an accumulation of forage reserves (Pickup & Stafford Smith 1993) on the one hand, and the optimum utilization of forage on offer (Nolan 1996) on the other. The prerequisite for any sustainable system is that the long term capacity of the biological system to produce forage from rainfall must be maintained, and the system must produce an acceptable financial return for the owner and dependents, thereby providing an acceptable standard of living (Pickup & Stafford Smith 1993).

The owner of the farm Elandsfontein near Beaufort West designed a system based on the principles of Acocks (1966) and Savory and Parsons (1980). The system incorporates a multi-camp infrastructure (147 camps on a 7000 ha unit) with a diversity of animal types (sheep, cattle and goats). This non-selective grazing (NSG) system is based on the premise that by concentrating a large number of animals with different feeding habits into a small paddock for a short time, animals will be forced to defoliate most of the plants, irrespective of palatability. This should reduce the competitive advantage of the less-palatable species when animals graze more selectively. The concomitant hoof-action (trampling/treading), dunging, and urination is believed to increase biomass turnover rate, improve the topsoil environment, stimulate production, and allow more rapid shifts in species composition (see Chapters 3 and 4). The relatively long rests of this grazing system allow ample time for recovery. The landowner also feed rumen stimulants to further improve the efficient, non-selective utilization of the forage on offer. The system requires a high investment in capital and operating costs. Since costs can rapidly exceed the returns, careful assessment of viability is needed (Pickup & Stafford Smith 1993).

This paper reports on a computer model which was developed to simulate the ecological processes, and to test the economic merits of investing resources into infrastructure. Given that investing in fences helps to dampen the impacts of environmental stochasticity, what then is the optimum number of camps at different levels of veld productivity, and how can the animal compliment be manipulated to maximize profits?

THE ELANDSFONTEIN SCENARIO

See the General Introduction for a detailed description of the study area.

The farm has been occupied by the Lund family since 1923 when there were no internal fences. When the current owner, A. Lund, inhereted the farm in 1964, it was already subdivided into 40 camps. At this stage the legacy of continuous selective grazing in the past was still very much evident in the form of shrub encroachment (*Lycium*), signs of soil loss, and extensive bare areas. The landowner reasoned that in order to combat selective defoliation and utilize the available forage resources more efficiently, he needed to develop the infrastructure and diversify the stock compliment. Over a period of 30 years the farm was subdivided into 147 camps, arranged around 38 watering points, using the wagonwheel layout. Homogenous vegetation units were kept separate as far as possible to reduce area selectivity by livestock. An indigenous cattle breed (Nguni) was stocked together with Merino sheep in order to utilize coarse grass material more efficiently. Boer goats were introduced to utilize more of the available shrub forage. The number of herds are reduced as far as possible in order to concentrate the impact of feeding and hoof action on soil and vegetation processes (see Chapters 3 and 4). Animals are supplied with cut saltbush (*Atriplex nummularia*) shoots, which grows wild on the farm and acts as a cheap, but satisfactory rumen stimulant. The saltbush improves rumen fermentation and therefore enables animals to ingest low quality, fibrous material. Animals can be retained in a camp for a few days longer while utilizing more of the available forage and still remain in an acceptable condition. With this system the landowner stocks ± 500 LSU (total) on a permanent basis on a unit with a recommended carrying capacity of ± 300 LSU. Even during the 1991 – 1994 drought there was a sufficient forage buffer on the farm to avoid destocking. It is therefore possible to buy or sell stock whenever the market prices are favourable. According to the landowner the ecological benefits of non-selective grazing are already apparent in that grass cover is increasing, shrub species composition is changing, bare areas have recovered and soil loss is reduced. An intensive monitoring programme is in progress in order to test some of these claims. However, the production figures of Elandsfontein looks impressive enough and it is this constant production, overriding to a large extent the high variability of the environment, which stimulated this study.

MODEL DESCRIPTION

A dynamic model was built with the aim of integrating knowledge of Nama Karoo ecological processes with management options and consequent financial implications. The model was developed in STELLA (High Performance Systems 1993), a high-level programming language, which facilitated the interactive and collaborative development of the model (Costanza *et al.* 1993). The user-friendly interface of STELLA will allow resource managers, agricultural planners and economists to experiment with the model. The model simulates the ecological processes per hectare of karoo veld and assumes the outflows to be similar per hectare for the rest of the farming unit. Although this is obviously a gross generalization, considering the heterogeneity of karoo landscape geomorphology and spatial distribution of rainfall, given the aims of the study it was regarded as acceptable. An annual time step was selected in order to simulate the impact of annual rainfall on fibre and animal production, and finances.

The model comprises three interactive sub-models, namely vegetation, production, and financial (Figure 1). Grass and shrub biomass production are determined by rainfall. Biomass is removed by decomposition and grazing and browsing animals. If it is a graze year for the modelled hectare, the biomass of grass and shrubs removed is determined by the density of animals, the intake per animal, and the time they spend in the camp i.e. on the hectare. Camp time is affected by grass biomass available when the animals enter the camp, the density and intake of animals, and the proportion of the available grass biomass that can be removed, which in turn depends on camp size, and whether rumen stimulants are provided or not. The time animals spend on the hectare (camp time) determines the recovery period for the hectare, which, in turn, determines when it is a graze year for the hectare.

Meat and wool production per year are calculated from density, intake per animal, camp time, and the area grazed per year (hectare), which in turn depends on farm size and recovery period. The financial sub-model converts the meat and wool production into monetary terms using the current meat and wool prices. Expenses comprise set-up and maintenance costs of the infrastructure (which is determined by camp numbers), rumen stimulant costs, and the costs of stocking the farm. Annual profit can be calculated from gross income minus expenses. By manipulating management inputs (Figure 1) model output in terms of annual profit can be maximized for a given set of conditions. Each of the sub-models is discussed in more detail below. Parameter names, units, symbols and estimates are listed in Appendix A.

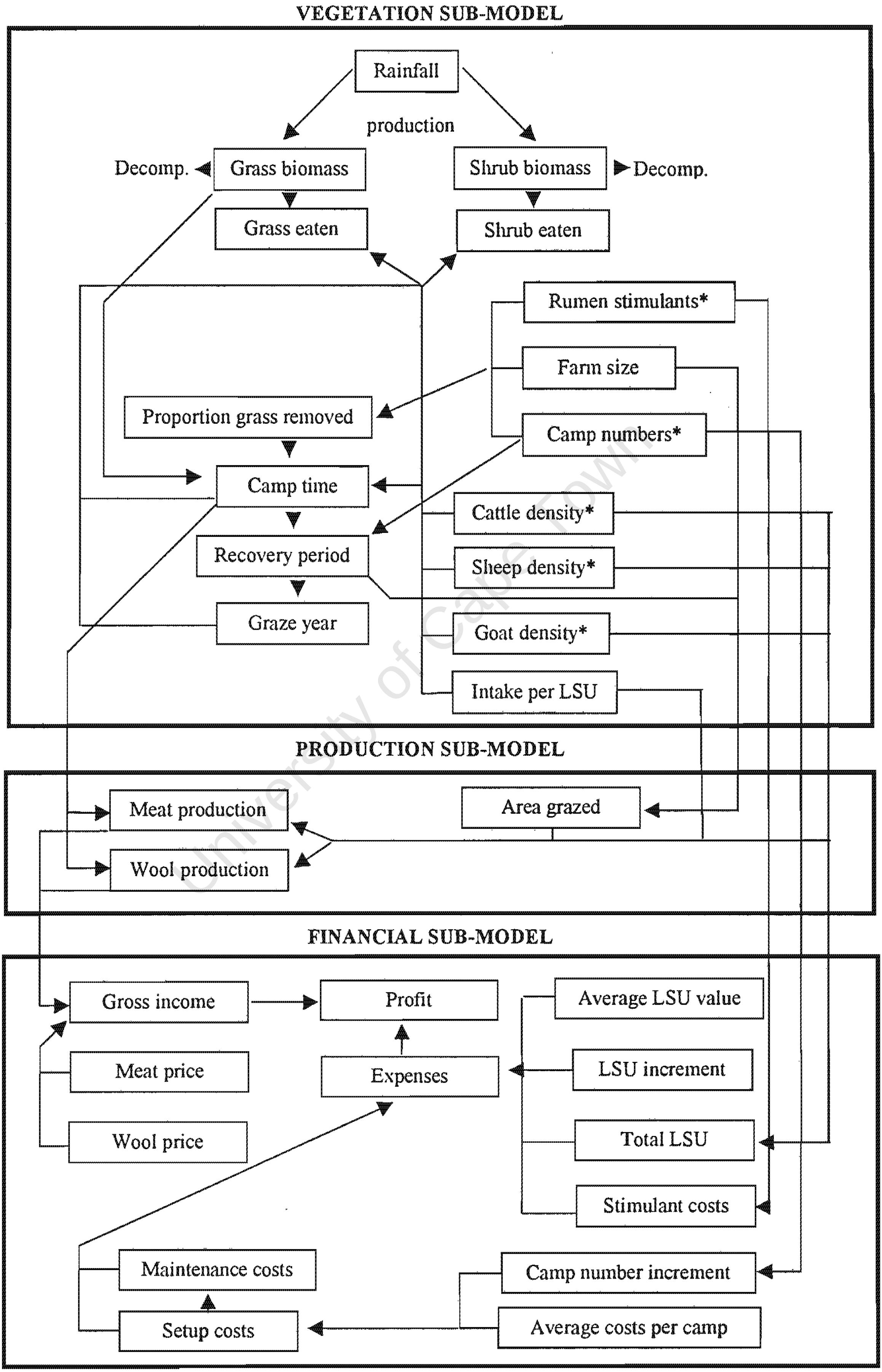

Figure 1 Conceptual model of the grazing system

VEGETATION SUB-MODEL

In this sub-model an annual rainfall value (P, mm) is generated using mean rainfall (P_{av}, mm/year) and coefficient of variation of rainfall (CV) as follows:

$$P = \mathrm{NORMAL}\,(P_{av}, P_{av}{*}CV) + 0.1 * P_{av} * \mathrm{SINWAVE}\,(1, 20) \tag{1}$$

Annual grass production (G_p, kg/ha) and shrub production (S_p, kg/ha) are estimated using P such that:

$G_p = 1.9 * P$

$S_p = 0.9 * P$

The production factors used in the above formulae were derived from the results of above-ground biomass harvesting trials conducted at Elandsfontein (Table 1; see Chapter 4 for methods). The only plots which showed an accumulation of grass biomass were the grazed plots which accumulated grass at an average annual rate of 1.9 * 250 mm (being the long term mean annual rainfall for Elandsfontein).

Table 1 Grass and shrub biomass figures (kg DM/ha) for the farm Elandsfontein, Beaufort West

		GRASS					
			Ungrazed plot			**Grazed Plot**	
Cutting date		**May '97**	**Oct '97**	**June '98**	***May '97**	**Oct '97**	***June '98**
Paddock	A	786	736	414	558	670	80
	E	1830	1240	770	1144	1556	314
	F	1364	904	1058	1048	1078	148
Average		1326	960	748	916	1102	180

*Non-selective grazing treatment (with rumen stimulants) applied before these cutting dates

		SHRUB					
		Ungrazed plot			Grazed Plot		
Cutting date		May '97	Oct '97	June '98	*May '97	Oct '97	*June '98
Paddock	A	1194	1304	1350	1428	1472	570
	E	962	1218	908	1168	1536	1338
	F	1248	1202	1356	1654	1526	1042
Average		1134	1240	1206	1416	1512	982

*Non-selective grazing treatment (with rumen stimulants) applied before these cutting dates

According to the data in Table 1 the shrub biomass accumulated in both the ungrazed and grazed plots between May 1997 and October 1997 at an average annual rate of 0.9 * 250 mm. The total aboveground biomass production for Elandsfontein, therefore, approximates 3 $kg.ha^{-1}mm^{-1}$, which is lower than the mean value of 5 $kg.ha^{-1}.mm^{-1}$ obtained by both Seely (1978) and Rutherford (1980) for arid and semi-arid areas of Southern Africa. However, Rutherford (1980) states that plant production in semi-arid areas are usually much lower, and was found to range down to about 1 $kg.ha^{-1}.mm^{-1}$.

Annual loss of grass (G_b) and shrub biomass (S_b) through decomposition (G_d and S_d, kg/ha) is calculated as a ratio of the standing crop biomass:

$G_d = 0.4 * G_b$
$S_d = 0.08 * S_b$

These decomposition factors were derived from the decline in grass and shrub biomass values in the ungrazed plots over the period October 1997 to June 1998 (Table 1), which incorporates normally warm and wet spring and autumn periods when decomposition will be maximal. The factors were proportionally adjusted to reflect decomposition losses on an annual basis.

The proportion of available aboveground grass biomass which can be removed by grazing animals (G_{bprop}) is determined by camp size (C_s, ha) which in turn is a function of farm size (F_s, 7000 ha) and camp numbers (C_n, adjustable management input):

$$C_s = \frac{F_s}{C_n} \text{ and}$$

$$G_{bprop} = 0.6 * e^{(-0.0038 * Cs)} + 0.2 \quad (2)$$

The relationship between G_{bprop} and C_s is assumed to be exponential. Equation 2 was derived from data obtained from two biomass harvesting trials at Elandsfontein. In the one, the aboveground plant production in non-selectively grazed was compared with production in ungrazed plots (Table 1). In another harvesting trial the proportion of aboveground grass removed at different grazing intensities (without the feeding of rumen stimulants) were compared. In the second trial a "large" camp size (C_s) was simulated by reducing the stocking rate but lengthening the time of stay in a 45 ha camp (Table 2). The data from these harvesting trials (Tables 1 and 2) were used to derive the relationship.

Table 2 Grass biomass values (kg DM/ha) before and after grazing at different intensities.

Grazing intensity	Before grazing	After grazing	Proportion removed (%)
46 ha camp grazed for 4 days by 222 LSU ∴ 19.3 LSGD/ha (without rumen stimulants)	1602	1032	36
45 ha camp grazed 7.7 days by 175 LSU ∴ 30 LSGD/ha (without rumen stimulants)	1602	858	46
45 ha camp grazed 22 days by 61 LSU ∴ 30 LSGD/ha (without rumen stimulants)	1322	1236	7

LSU = Large Stock Unit
LSGD = Large Stock Grazing Days

The largest value for G_{bprop} (ca. 0.8) was obtained by comparing grass biomass of ungrazed plots with those of grazed plots after the June 1998 harvesting exercise (Table 1). The grazed plots in paddocks A and F (both ca. 30 ha in size) were subjected to grazing before cutting by ± 350 LSU (Large Stock Units) for 5 days (equals a grazing intensity of 50 – 60 LSGD/ha; LSGD = Large Stock Grazing Days). The value of 0.8 was obtained with the feeding of rumen stimulants whereas the largest value obtained without feeding rumen stimulants was in the order of 0.4 (Table 2). The lowest value of G_{bprop} (in the order of 0.1) was obtained in the harvesting trial where the "large" camp was simulated (Table 2). One can, however, expect that the value of 0.1 can at least be doubled by feeding rumen stimulants. A third point on the graph (G_{bprop} = ca. 0.6) was obtained by comparing grass biomass of the ungrazed plot with that of the grazed plot after the June 1998 harvesting exercise in a 108 ha camp (paddock E in Table 1) where the grazed plot was also subjected to a grazing intensity of 50 – 60 LSGD/ha. Because of the size of the camp the same animal compliment had to stay longer (16 days) to achieve the same grazing intensity as in paddocks A and F (30 ha camps). However, the proportion of grass removed dropped from ca. 0.8 to ca. 0.6 (one can expect these values to be halved without providing rumen stimulants). Animals are more concentrated in smaller camps where a high grazing intensity (LSGD/ha) can be maintained with a short period of stay. By concentrating animals they are forced to feed less selectively, thereby taking a wider spectrum of species (Acocks 1966) as well as more biomass per plant (Danckwerts 1987; Du Toit 1996). The advantage of a short period of stay (5 – 10 days) is that it minimizes the number of times that regrowth is removed (Acocks 1966; Danckwerts 1987). Rumen stimulants improve fermentation rate, and therefore intake of low quality forage (Leng 1984). It was assumed that rumen stimulants would increase the proportion of grass removed by 50 %. Therefore:

If $R_s = 1$ then G_{bprop} else $G_{bprop} * 0.5$

Camp time (C_t, days) is determined by the available grass biomass (G_b) when animals enter the camp, the proportion of grass biomass which can be removed (G_{bprop}), the density of grazers (cattle [C_{dens}]; sheep [S_{dens}] in LSU/ha), and intake per Large Stock Unit per day (I_{LSU}, kg/LSU/day). Densities depend on the numbers stocked by the grazier and C_s, while intake was taken as 10 kg/LSU/day (ARC 1980).

It was assumed that a sheep's diet consists, on average over a year-long period, of 50 % shrub and 50 % grass (Du Toit *et al.* 1995), while cattle eat only grass, and goats concentrate mainly on shrubs and non-grassy herbaceous plants (Botha *et al.* 1983). Thus:

$$C_t = {(G_b * G_{bprop})} \Big/ {[(C_{dens} * I_{LSU}) + (S_{dens} * I_{LSU} * 0.5)]} \tag{3}$$

The recovery period (R_p, years) for a simulated hectare depends on C_t and C_n such that:

$$R_p = \mathrm{ROUND}\left[(C_n - 1)C_t \Big/ 365 \right]$$

R_p determines when it is a graze year (G_y, 0 or 1) for the hectare. Thus:

$$G_y = \text{If } R_p \le 1 \text{ or } R_p = \text{COUNTER } (1, 1 + R_p) \text{ then 1 else 0} \tag{4}$$

If it is a graze year for the simulated hectare, the amount of grass eaten (G_e, kg/ha) and shrub eaten (S_e, kg/ha) is calculated as follows:

$$G_e = \text{If } G_y = 1 \text{ then } (C_t * C_{dens} * I_{LSU} + C_t * S_{dens} * I_{LSU} * 0.5) \Big/ dt \text{ else 0} \tag{5}$$

$$S_e = \text{If } G_y = 1 \text{ then } (C_t * G_{dens} * I_{LSU} + C_t * S_{dens} * I_{LSU} * 0.5) \Big/ dt \text{ else 0} \tag{6}$$

The vegetation sub-model assumes that veld remains in a relatively good condition, and does not consider possible influences of annual rainfall fluctuations nor the impacts of the NSG system on vegetation composition, and therefore primary production and acceptability to livestock.

PRODUCTION SUB-MODEL

In this sub-model the grass and shrub forage ingested per hectare over the time that the animals stay on the hectare (C_t) is converted into marketable meat (M_p, kg) and wool production (W_p, kg) for the whole year by incorporating the area grazed per year (A_g, ha). A_g can be calculated from F_s and R_p as follows:

$$A_g = \text{If } R_p \geq 1 \text{ then } F_s / R_p \text{ else } F_s / 1 \tag{7}$$

It is further assumed that livestock require 15 kg dry matter intake to gain 1 kg in lean body mass (Meissner & Roux 1984) and sheep require 90 kg dry matter intake to produce 1 kg wool (Boyazoglu 1997). All the wool and 15 % of the meat produced is marketable (A. Lund pers. comm.). It then follows that:

$$M_p = \text{If } G_y = 1 \text{ then}$$
$$\left((C_t * C_{dens} * I_{LSU} + C_t * G_{dens} * I_{LSU} + C_t * S_{dens} * I_{LSU}) / 15 \right) * A_g * 0.15 \text{ else } 0 \tag{8}$$

and

$$W_p = \text{If } G_y = 1 \text{ then } \left(C_t * S_{dens} * I_{LSU} / 90 \right) * A_g \text{ else } 0 \tag{9}$$

FINANCIAL SUB-MODEL

Annual gross income (I_{gross}, Rands) is calculated by converting M_p and W_p into monetary terms using the current meat (P_m, Rands and cents) and wool prices (P_w, Rands and cents).

$$I_{gross} = M_p * P_m + W_p * P_w \tag{10}$$

Four items comprise the expenses (E_x, Rands) on an annual basis, namely, stocking the farm, setting up the infrastructure, maintaining the infrastructure, and the costs of rumen stimulants (if provided).

The agriculturally recommended stocking rate for Elandsfontein is 300 LSU (24 ha/LSU for a 7000 ha unit) (A. Lund pers. comm.). The animal increment (A_{incr}, LSU) is calculated if the manager wants to increase total animals stocked (A_{tot}, LSU) above the recommended:

$A_{incr} = A_{tot} - 300$

For the purpose of the model it is assumed that money will have to be borrowed to acquire these animals at the average value per LSU (A_{val}) of R2 000 (P. Van Zyl, Dept. Agric. pers. comm.). If 10 camps are assumed to be the minimum for an acceptable rotational grazing system (Hobson 1987), then the set-up costs (SU_{cost}, Rands) for additional camps ($C_{incr} = C_n - 10$) can be calculated using the marginal cost per camp (C_{cost}, Rands)

$SU_{cost} = C_{incr} * C_{cost}$

where the average costs of fencing material (J. Strümpher from Progressive Wire pers. comm.) and watering points (A. Lund pers. comm.) was used to construct an exponential relationship between C_{incr} and C_{cost}. With the marginal cost per camp declining exponentially from R26 000 to an average of ca. R4 000 (Figure 2).

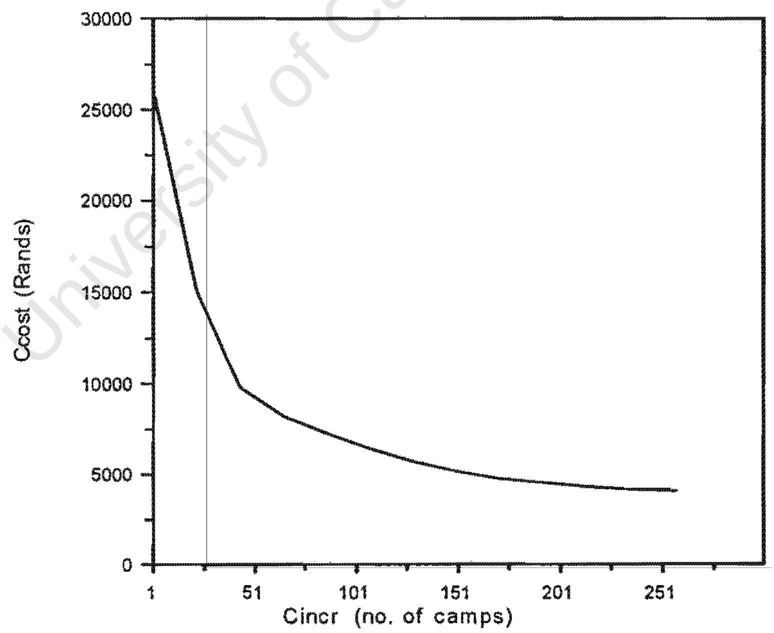

Figure 2 The relationship between additional camps (C_{incr}) and the marginal cost per camp (C_{cost}; Rands).

According to the landowner, maintenance costs (M_{cost}, Rands) amount to approximately 2 % of SU_{cost} per annum;

$M_{cost} = SU_{cost} * 0.02$

If rumen stimulants are provided then rumen stimulant costs (R_{cost}, Rands) amount to R32 per LSU (A. Lund pers. comm.) which gives;

R_{cost} = If $R_s = 1$ then $A_{tot} * 32$ else 0

With the assumption that the funds for extra camps and stocking the farm above the recommended number of animals have to be borrowed at an interest rate of 20 % per year and inflation amounts to 10 % per year then E_x can be calculated as follows:

$$E_x = ((A_{incr} * A_{val} + SU_{cost}) * 0.1) + R_{cost} + M_{cost} \quad (11)$$

This leaves annual profit (I_{profit}, Rands) to be calculated as follows:

$$I_{profit} = I_{gross} - E_x \quad (12)$$

MODEL VALIDATION

Here I compared observed annual wool and meat production figures (kg) from Elandsfontein (A. Lund pers. comm) with figures predicted by the model in order to test the validity of the model output (Table 3). For each of the years in Table 3 the model was adjusted according to the rainfall and actual stock compliment figures (cattle, sheep and goats) provided by the landowner (pers. comm.), and 10 simulations were run over 25 years. From this output the average predicted annual wool and meat production was calculated.

Table 3 Comparison between observed annual wool and meat production figures (kg) from Elandsfontein and figures predicted by the model.

Year	Rainfall (mm)	Annual wool production (kg)		Marketable annual meat production (kg)*	
		Predicted	Observed	Predicted	Observed
1987	200	6170	5365	-	-
1988	290	6745	7354	-	-
1989	310	6612	9586	-	-
1994	230	7077	6121	-	-
1995	230	7231	7437	10087	9547
1996	370	7680	10146	10906	12028

*regarding 15 % of total meat production as marketable.

In general it appears the model predicts realistic figures in terms of annual meat and wool production (Table 3). The lowest observed and predicted wool production values were obtained during 1987 when the annual rainfall was relatively low at 200 mm. There is, however, a substantial difference between the observed and predicted values for 1996, which was a very wet year. This could be because of the high quality of available forage during that year, with food quality being an important determinant of wool production (Boyazoglu 1997). The relationship between rainfall and forage quality is not considered by the model. The predicted annual meat production figures are also promising (Table 3) although the figure of 15 % of total annual meat production which is marketable is a figure which could change considerably between years as demands for cash flow, and market prices varies (A. Lund pers. comm.).

SCENARIO ANALYSIS

Scenario definition

The scenario analysis was used in an attempt to answer four questions;

a) Does an increase in camp numbers result in maintaining a forage buffer over time?

b) What is the optimal number of camps in terms of annual profit for a given set of conditions?

c) How can annual profit be optimized with the optimal number of camps in place?

d) Under which conditions are fewer, or more camps than the optimal number economically viable?

Addressing the first question three rainfall regimes (150, 200, 250 mm) were simulated for grass and shrub biomass respectively. For each rainfall regime the simulation was run for three camp numbers (20, 60, 150 camps) over 50 years. Rainfall figures were selected to vary from low to the maximum per definition for arid and semi-arid areas with coefficient of variation of rainfall (0.6, 0.5, 0.4) varying accordingly (Desmet & Cowling 1999). Camp numbers were selected to vary from average (ca. 20 camps) to exceptionally high (150 camps). With all these scenarios the initial grass biomass was assumed to be 1000 kg/ha and that of shrubs 1200 kg/ha (see Table 1). A further assumption was that veld condition, as influenced by species composition, remained unchanged throughout i.e. values of G_p, S_p, G_d and S_d did not change over time. Although this could be seen as a weakness there is as yet no quantitative data to either support or reject any relationships between camp numbers and veld condition, or between veld condition and productivity figures (see Chapter 4). For all these simulations the 7000 ha farm was stocked with 350 LSU (sheep only), and no rumen stimulants were fed.

In order to address the second question an analysis was used to motivate the elimination of exceptionally high camp numbers (C_n = 150 and 300). In this scenario analysis annual profit (I_{profit}, Rands) with standard error margins was simulated over 50 years for 20, 60, 150 and 300 camps while rainfall and other management inputs (stocking rate, and rumen stimulants) were held constant. Meat and wool prices were assumed to be R11.95 and R12.76 per kg, respectively. A further analysis was performed using 20, 40, 60 and 80 camps under different rainfall regimes (P_{av} = 150, 200, 250 mm; CV = 0.6, 0.5, 0.4), maintaining the stock compliment at 350 LSU (sheep only) with no rumen stimulants. For this analysis an amount of R150 000 was subtracted from I_{profit} in order to plot relative mean annual profit against time.

An attempt was made to answer the third question by simulating a 7000 ha unit with 60 camps stocked with 350 LSU sheep under a rainfall regime of 250 mm (CV = 0.5). Management strategies were employed to optimize the relative mean annual profit of this scenario. Rumen stimulants were added as well as different levels of goats and cattle to the stock compliment.

In an attempt to answer the fourth question, P_{av} was raised to 300 mm (CV = 0.4) while camp numbers (C_n = 60, 150), rumen stimulants and stocking rates were altered.

Scenario results

Changes in aboveground grass and shrub biomass over time for different camp numbers and for three rainfall regimes are shown in Figures 3 and 4. With a low mean annual rainfall (P_{av} = 150) and high coefficient of variation (CV = 0.6) there is virtually no biomass accumulation irrespective of camp numbers. Under these conditions a stocking rate of 350 LSU sheep might be too high and grass and shrub resources will be depleted within 10 to 20 years. Under the rainfall regime where P_{av} = 200, and CV = 0.5, the 60, and 150 camp systems do maintain the grass and shrub biomass slightly longer than the 20 camp system, but again it appears as if 350 LSU sheep might be too many in the long run. However, it is already apparent at this rainfall level that increasing camp numbers does allow for forage buffers to be maintained for longer compared to a system with fewer camps (e.g. 20). The real value of increasing camp numbers in building and maintaining a forage buffer is shown in the 250 mm rainfall regime. Both grass and shrub biomass is maintained under the 60 and 150 camp systems compared to the depletion of biomass under the 20 camp system. The higher rainfall accentuates the "carry over" effect generated by the longer recovery periods of the 60 and 150 camp systems. With the slower decompositon rate of shrubs, the build up of shrub biomass under long recovery-period (R_p) systems (e.g. 150 camps) might even become problematic in a sense that shrubs might outcompete the herbaceous layer in the long run (Figure 4). Figures 3 and 4, therefore, show that the benefits of longer recovery periods (R_p) created by increasing camp numbers only applies to higher rainfall regions. In lower rainfall regions (≤ 200 mm) stock numbers will have to be manipulated in order to maintain a forage buffer for dry years.

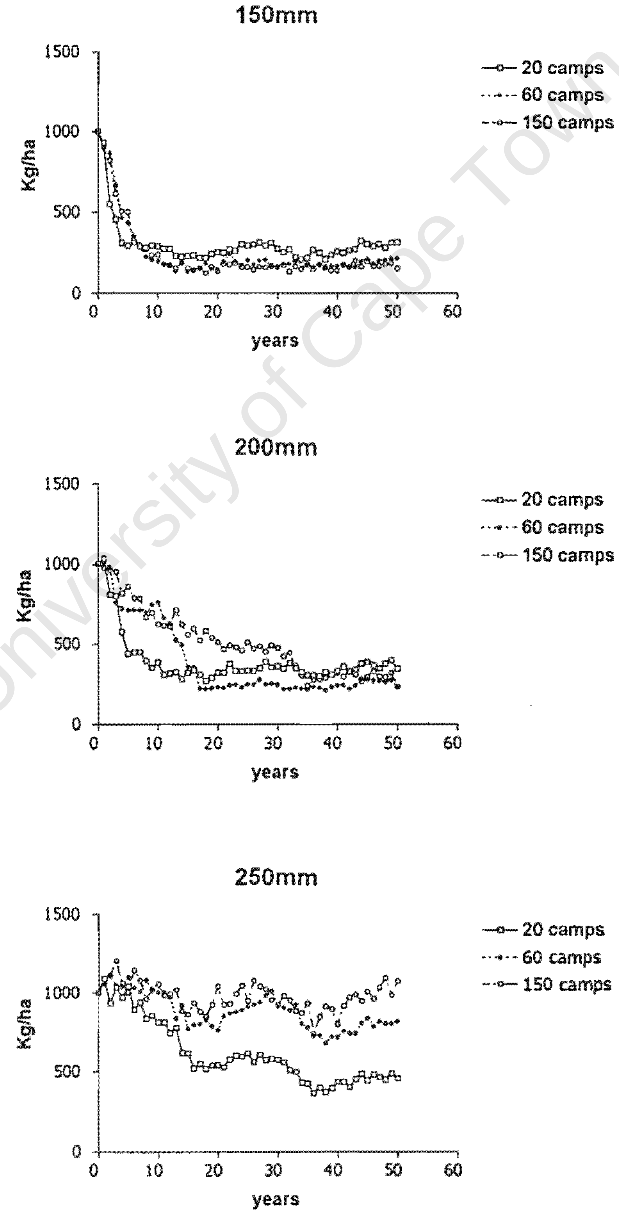

Figure 3 Changes in aboveground grass biomass with time for three rainfall regimes (P_{av} = 150, 200, 250 mm; CV = 0.6, 0.5, 0.4) for a 1 ha patch of Nama karoo veld under different management systems (C_n = 20, 60, 150 camps)

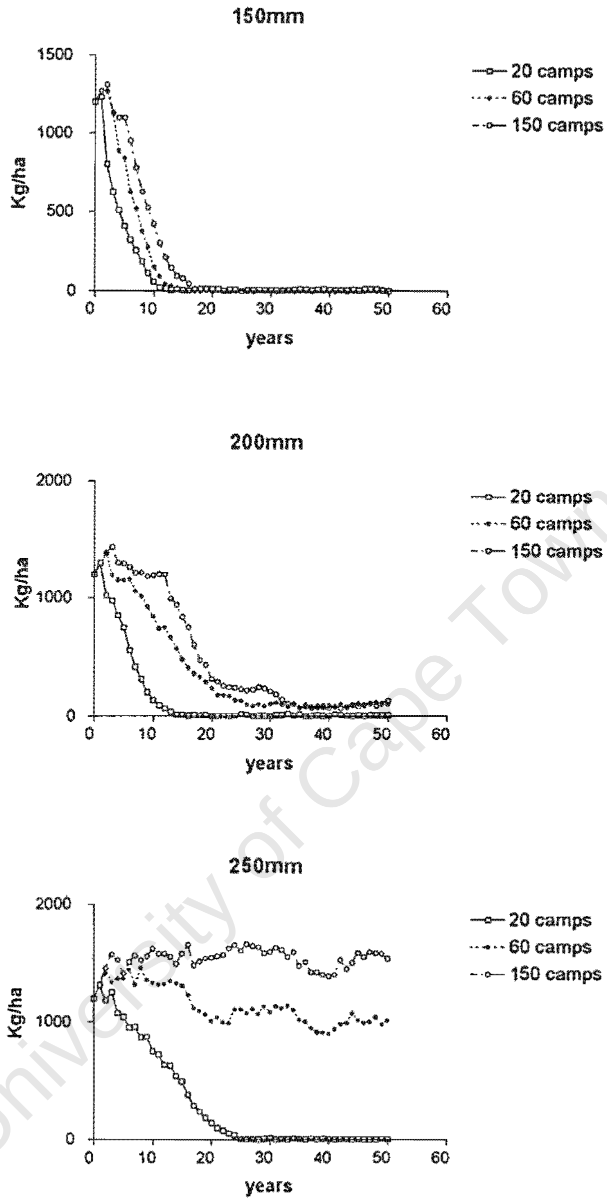

Figure 4 Changes in aboveground shrub biomass with time for three rainfall regimes (P_{av} = 150, 200, 250 mm; CV = 0.6, 0.5, 0.4) for a 1 ha patch of Nama karoo veld under different management systems (C_n = 20, 60, 150 camps)

According to Figure 5, setting up and maintaining an infrastructure of 300 camps on a 7000 ha unit is too costly and can never be compared in terms of profit with a 20 camp system. Although the 20 camp system has a very limited buffering capacity and experiences a drastic decline in profit in the early years, the 300 camp system is too costly to set up and maintain. With their buffering capacities, the 60 and 150 camp systems are less susceptable to an unpredictable series of dry years, and, therefore, better maintain biomass, which converts into profit, than the 20 camp system. The last mentioned system is initially superior because of the lower set-up costs and the starting figures for grass and shrub biomass (G_b = 1000 kg/ha; S_b = 1200 kg/ha), but the 60 and 150 camp systems exceed the profit from the 20 camp system within 10 to 15 years, and thereafter proves superior most of the time. The marginally better buffering capacity of the 150 camp system (Figures 3 and 4) does not convert into superior profits compared to the 60 camp system (Figure 5) and 150 camps is therefore rejected in the process of optimizing profit. It is, however, important to realize that this model only measures the benefits of a farming system in terms of Rands, whereas there might be very valuable ecological benefits accrued from the short duration, high density grazing and long recovery periods of the 150 camp system (Chapters 3 and 4). Under a low rainfall regime no management system succeeds in building a forage buffer and biomass and, therefore, profit fluctuates with rainfall (Figure 6). However, as rainfall increases the buffering capacity of the 40 – 80 camp systems come into play and these systems then generally perform better than the 20 camp system. Although the 80 camp system shows excellent buffering capacity under the 250 mm regime (Figure 6) it is very difficult to choose between 40, 60 and 80 camp systems in an attempt to optimize profit. In this regard it is useful to look at the results presented in Table 4. The 40 camp system has the highest average and lowest CV over a 50 year period because the lower set-up costs results in higher profits initially. However, in the long run forage resources start to fluctuate more in the 40 camp system, and the 60 camp system with its higher average and lower CV over the last 25 years of a 50 year simulation, seems to be a better option. Although the 80 camp system has a lower CV in the long run, the average profit is substantially lower compared to the 60 camp system.

Table 4 Mean annual profit (Rands) over 50 years for three management systems (C_n = 40, 60, 80 camps). Rainfall regime and stocking rate remain constant. (P_{av} = 250; CV = 0.5; 350 LSU sheep; R_s = 0).

	Management System		
	40 camps	**60 camps**	**80 camps**
Average after 50 years	255656	248043	230194
Standard deviation	51003	54569	52337
Coefficient of variation (%) (CV)	20	22	23
Average for the last 25 years of the 50 year simulation runs	279134	282667	266948
Standard deviation	32474	17142	13054
Coefficient of variation (%) (CV)	12	6	5

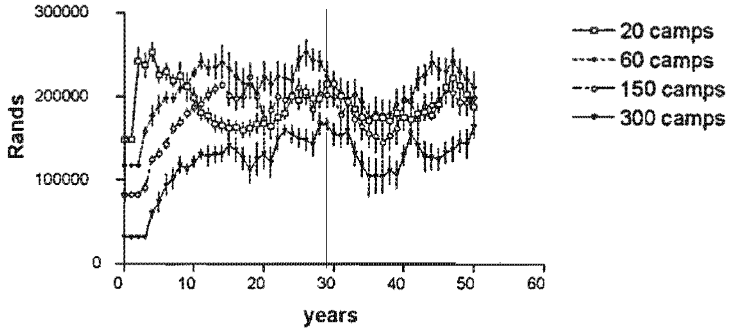

Figure 5 Mean annual profit (with standard error margins) under a constant rainfall regime for four management systems (C_n = 20, 60, 150, 300 camps). 350 LSU sheep stocked and no rumen stimulants fed.

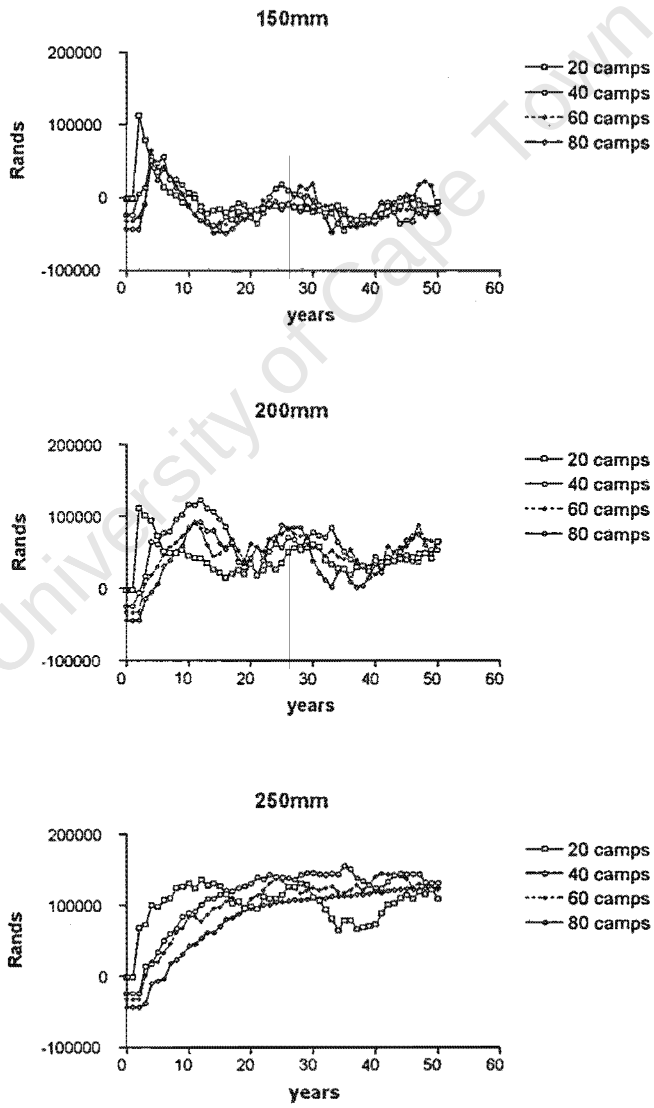

Figure 6 Relative mean annual profit under three rainfall regimes (P_{av} = 150, 200, 250 mm; CV = 0.6, 0.5, 0.4) for four management systems (C_n = 20, 40, 60, 80 camps). 350 LSU sheep stocked and no rumen stimulants fed.

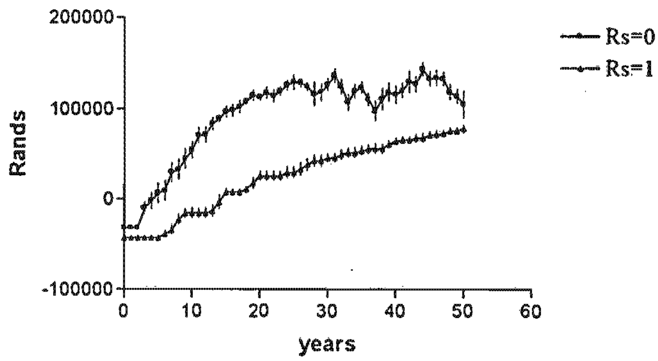

Figure 7 Relative mean annual profit (with standard error margins) from a 7000 ha unit with 60 camps stocked with 350 LSU sheep and subjected to different management strategies. R_s = rumen stimulants.

In an attempt to optimize annual profit under a specific rainfall regime (P_{av} = 250, CV = 0.5) and with the 60 camp system in place various management systems were simulated. According to Figure 7 the feeding of rumen stimulants greatly enhances the buffering capacity of the 60 camp system, but at such a cost that it does not appear to be an economically viable strategy. The value of feeding rumen stimulants may be realised in the long term when the manager is forced to maintain high animal numbers through unfavourable periods as a result of, for instance, low market prices. Whereas with high animal numbers recovery periods would have been too short for a forage buffer to build up, animals can stay slightly longer in a camp when supplied with rumen stimulants, which then increases recovery period and allows for a forage buffer to accumulate. This argument is supported by the relatively high average annual profit and low CV of profit for the last 25 years of management system 5 (Table 5).

keep the animals for a few days longer in a camp. This scenario is likely to get a farmer through an exceptionally severe one-in-fifty-year drought without having to destock drastically and sacrifice income.

Fewer camps, e.g. 20 to 40, appears to be more economically viable in drier regions (≤ 200 mm) (see Figure 6). The low rainfall and high variability thereof overrides any buffering effects that the camp system might have on forage availability. In very dry areas (≤ 100 mm) the approach would be to have a highly flexible stocking rate, and to keep overheads, such as interest on infrastructure investments and maintenance costs, as low as possible.

Higher rainfall enhances the buffering capacity of many camps (e.g. 150) and with the feeding of rumen stimulants to maintain animal condition under high stocking densities, more animals can be maintained successfully with an increase in gross annual income (Table 6, MS_8). However, this system only makes economic sense if overheads are not as high as simulated in the model i.e. the infrastructure of a 150 camp system has been set up over time without borrowed money, stock are allowed to breed up over time, maintenance costs are kept low by using quality materials and workmanship initially, and rumen stimulant costs are kept low by feeding for instance chopped fodder growing naturally on the farm e.g. *Atriplex nummularia* (Old man saltbush) (A. Lund pers. comm.).

Table 6 Average annual gross income (Rands) from a 7000 ha unit under two different management systems with a rainfall regime where P_{av} = 300mm and with CV = 0.4

	Management Systems	
	60 Camps **350 LSU Sheep** **20 LSU Goat** **$R_s = 0$** **MS_7**	**150 Camps** **400 LSU Sheep** **20 LSU Goat** **80 LSU Cattle** **$R_s = 1$** **MS_8**
Average after 50 years	310344	312911
Standard deviation	57405	83384
CV (%)	18	27
Average for the last 25 years of the 50 year simulation runs	351454	364997
Standard deviation	17231	51233
CV (%)	5	14

DISCUSSION

The interactions between ecological and economic systems are many and strong. Analysis and modelling of linked ecological economic systems can provide valuable insights, which will hopefully change the behaviour of resource managers towards a sustainable pattern of resource use that works in synergy with the life-support ecosystems on which it depends (Costanza *et al.* 1993). The fundamental purpose of this model, which was to evaluate cost effectiveness of various management tactics for extensive pastoral enterprises in the Nama Karoo of South Africa, warranted the sacrafice of simulating system behaviour in a qualitatively realistic (realism), and in a quantitatively precise (precision) way.

Realism and to a lesser extent precision were compromised in order to accurately determine the overall magnitude and direction of change of an enterprise's profit over time (Costanza *et al.* 1993). Three simplifications and omissions contributed to reducing the realism and precision of the model, namely (1) the complex relationships between rainfall, grazing intensity, veld condition (species composition and cover), and forage production (Hatch *et al.* 1996); (2) the relationships between veld condition, forage quality, animal density, and animal intake (Noy-Meir 1978; Owen-Smith 1991); and (3) the complexities of converting animal intake into meat and wool production (Meissner & Roux 1984).

Despite these weaknesses the model's value lies in articulating the economic risks of management tactics which strive for optimizing an ecological process i.e. building a forage buffer for dry periods (Pickup & Stafford Smith 1993). It has been shown that there is no point in providing ecological information unless this is presented in the necessary economic context, that is, the financial circumstances at the scale of the farming unit (Stafford Smith & Foran 1990).

Coping with variability – the forage buffer

In uncertain environments forage availability fluctuates widely over time and space (Scoones 1994) and managers need to apply farming systems and techniques that can reduce the impact of seasonal and periodic droughts (Vorster 1994). In this regard a manager has basically two options: to apply an opportunistic, highly flexible management approach in order to track the variable forage resource (Scoones 1994; Milton *et al* 1995), or to use forage and financial buffers to minimize the impact of this variability (Pickup & Stafford Smith 1993; Vorster 1994). One way of achieving tracking, which is matching available food with animal numbers, is to destock in the face of drought and restock when forage is available after the drought (Riechers *et al.* 1989; Stafford Smith & Foran 1992; Scoones 1994; Hatch *et al.* 1996). However, the decision of when and to what extent to destock is not an easy one (Stafford Smith & Foran 1992; Pickup and Stafford Smith 1993) and stock prices may be low because many managers are taking the same action. Furthermore, high animal prices after drought-breaking rain may result in a strong incentive to retain stock during a dry time, which may be damaging to the rangeland (Pickup & Stafford Smith 1993; Vorster 1994).

The second option of building forage and/or financial buffers for droughts involves making long-term changes to management that enhance the capability of the property to cope with drought (Foran & Stafford Smith 1991). One such a strategy is to invest in waterpoints and fencing in order to make more, smaller camps and to implement a non-selective rotational grazing system. This permits a high degree of control over the impact of the animals and capability to reduce the impact (Pickup & Stafford Smith 1993). It also enables the manager to separate vegetation units of differing characteristics in order to promote uniform veld utilisation (Edwards 1988). This strategy obviously entails increased capital and operating costs and the decision whether to invest, and thereby increase management intensity, or not, is fundamental in terms of the sustainability of pastoralism as a land use in these variable environments (Pickup & Stafford Smith 1993).

The objectives of camping veld and rotating animals are (1) to limit the period of stay (C_t, camp time) so as to avoid grazing of regrowth, (2) to increase grazing intensity and thereby reduce selectivity (Acocks 1966), and (3) to allow for long term rests for seeding, vigour and forage accumulation (Booysen *et al.* 1974). Many issues, such as whether animals actually graze regrowth, whether selectivity can be reduced by increasing grazing intensity, and whether extended periods of absence would result in an accumulation of forage, remain unresolved (see O'Reagain & Turner 1992; Hoffman 1988). There is, however, general agreement that leaving as large a proportion as possible of the property unoccupied at any one time allows for a large forage buffer left from the last good rains, and whatever rain does fall during the drought can be optimally converted into forage over a large proportion of the farm.

It is this forage buffer that enables farmers, like the Lunds, to reduce uncertainty and, therefore, endure droughts without destocking at times of unfavourable stock prices. This also enables the manager, who can comfortably maintain stock numbers through droughts, to sell animals after drought-breaking rains at relatively high prices. Our model shows that there is no point in opting for the forage buffering strategy in lower rainfall (≤ 200 mm) areas; and under these environmental conditions tracking (Scoones 1994) is probably a better option. However, under higher rainfall (> 200 mm) conditions, investing in infrastructure, and therefore better ecological control, does make sense. The model shows that effective buffering is achieved with multi-camp systems of 40 to 80 camps on a 7000 ha farming unit, but the highest, most reliable economic return is realized in the order of 60 camps. This optimum is obviously affected by the current prices of meat, wool, fencing material, and maintenance. This relationship between rainfall and camp size corroborates the findings of Acocks (1966) who suggested a stocking intensity of "two sheep per inch per morgen" on the basis of veld improvements achieved by karoo farmers with smaller camps.

<u>**Acceptability of forage buffering strategies**</u>

Weighing the costs of intensification against ecological and economic benefits is one aspect. The social, political, and cultural forces against intensification is another. Scoones (1994) argues that the blueprint ranch model aiming at higher control and boosting single outputs (e.g. meat and wool) is unlikely to work in most of Africa's pastoral areas, and that an emphasis on flexibility and mobility is the key to success in these areas. O'Reagain & Turner (1992) present an argument for simple grazing systems using adaptive and opportunistic management in Southern African rangelands. Technological requirements e.g. for water distribution across the property, management expertise for implementing complex rotational grazing systems (Hoffman 1988), and the long term vision required for farm development might be beyond the scope of most African pastoralists. It is possible that in coping with environmental variability there is room for both the concepts of tracking and buffering, and they do not need to be mutually exclusive. Whatever strategy is opted for, it is of paramount importance to maintain the ability of the biological system to respond to rainfall events and produce forage (Acocks 1966; Savory 1991; Pickup & Stafford Smith 1993).

Stocking rate, animal mix, and rumen stimulants

Stocking rate is generally regarded as the most important management tool under control of a rangeland manager (Danckwerts 1987), and several models have shown the relationships between farm outputs and stocking rates (Noy-Meir 1978; Riechers *et al.* 1989; Stafford Smith & Foran 1992; Hatch & Tainton 1995; Hatch *et al.* 1996). Stocking rate generally has to be somewhere between maximum production per animal and maximum production per hectare (Danckwerts 1987), and should not exceed the long term carrying capacity for a particular vegetation type (O'Reagain & Turner 1992). However, under drought conditions severe destocking might be necessary (see previous section) while under intensive systems very high stocking rates (2 x recommended) might be sustainable and even desirable (Acocks 1966; Savory 1978; Chapters 3 and 4). The model supports the findings of Danckwerts (1987) that conservative stocking rates increase the farmer's resilience towards drought without adversely affecting profitability. Because camp time (C_t) and recovery period (R_p) in the model are based on grass biomass, even moderate increases in the stocking rate of grazers reduced the buffering capacity of the multi-camp system.

Selecting the correct grazer : browser combination is necessary for efficient and ecologically balanced use of mixed herbaceous and woody vegetation (Nolan 1996). Animal performance can be improved (O'Reagain & Turner 1992; Nolan 1996), carrying capacity increased (Botha *et al.* 1983), potential for veld degradation decreased (O'Reagain & Turner 1992), and diversification of income improved (Nolan 1996) when implementing a suitable mix of animal types. In order to select the correct mix, the manager needs information on dietary overlap, which in turn depends on vegetation type, animal breed, and season (Botha *et al.* 1983; Du Toit *et al.* 1995). Overlap on semi-arid rangeland can be generalized by assessing sheep diets as intermediate between cattle (graze-only) and goat (browse-only) diets (Nolan 1996). The model incorporates this overlap, but only in calculating the total amount of grass and shrub biomass removed by each animal type (sheep, cattle and goats). This ignores the very important complimentary effect of these animals by feeding on different plant species and plant parts (A. Lund pers. comm.). Although the model shows that adding browsers (goats) to the animal mix can increase annual profit, the real value of goats lies in their ability to utilize forage that is largely unacceptable or unavailable to other animals (e.g. *Lycium* shrubs). In a similar way the model ignores the value of cattle in removing the coarser grass material and thereby improving sheep performance (Danckwerts 1987), and their value as "engineers" for maintaining ecosystem health (Chapters 3 and 4). Product demand, and meat and wool price fluctuations are other important considerations when deciding on a suitable animal mix.

The input costs of supplementary feeding can be excessively high, making it uneconomical at first glance (Vorster 1994). However, there are situations when the limited use of rumen stimulants, which improve rumen fermentation and roughage intake, can be advantageous. Through its effect on rumen microbial activities, these stimulants help ruminants to ingest more fibrous, less palatable forage. Without rumen stimulants animal condition will drop dramatically when they are forced to remain in a camp in order to remove a larger proportion of the available forage reserve. It is towards the end of an occupation period (camp time) that rumen stimulants help animals to ingest lower quality forage while still maintaining an acceptable level of performance (A. Lund pers. comm.). Especially during dry periods, rumen stimulants can be used to force animals to remove up to 80 % of the aboveground grass biomass. An *ad libitum* supply of clean, cool water is absolutely essential to achieve this (A. Lund pers. comm.). With more forage per hectare that can be removed, more animals can be accommodated and, because camp time is lengthened, the overall buffering capacity of the camp system is enhanced. This enables the manager to maintain higher animal numbers for longer periods before forage deficits set in. The model shows that it is possible, within limits, to maintain high animal numbers through dry periods using rumen stimulants. However, this practice is not economically viable. The model assumes that rumen stimulants are supplied to all animals throughout every year, which is obviously unnecessarily costly. It also disregards the use of cheaper, readily available supplements e.g. *Atriplex nummularia* (salt bush).

Future research needs

The realism of the model can be improved by expanding the plant and animal production sub-models. In the plant production sub-model different herbage pools, e.g. green, standing dead, and litter (McKeon *et al.* 1982), as well as flows between these pools can be incorporated. The feedback effects of rainfall and patterns of herbivory on plant population dynamics, soils, and therefore productivity needs to be considered (Owen-Smith 1991). Herd structure (including age classes) could be more accurately reflected in the conversion of forage into animal products (see the *HerdEcon* model of Stafford Smith & Foran 1988).

More importantly, it is necessary to apply our model to a wider array of environmental conditions by providing more accurate production and decomposition figures for the Karoo region. Precision of our model can be improved by refining the relationship between camp size and maximum proportion of biomass that can be removed by animals. The ecological merits and demerits of concentrating large herbivores for a short period has important consequences for sustainable farming strategies and restoring degraded land. A further challenge is to incorporate spatial heterogeneity on a farm scale, and seasonal distribution of rainfall. Reducing the time frame of the model might not be as difficult as coping with spatial heterogeneity. Existing models like *Paddock* (Stafford Smith & Foran 1990) have taken up this challenge.

A further useful development would be to link this and similar models, to an integrated decision support system (Ludwig *et al* 1992) for managers, land use planners, and extension officers.

CONCLUSIONS

It is likely that every livestock farmer in the Karoo will have to cope with drought at some stage in their farming career. There are basically two strategies to remain in business during such an unpredictable event i.e. tracking the varying forage availability by, for example, applying a flexible stocking system, or opting for a more capital-intensive farming system which allows for a forage buffer to accumulate during good years.

Most conservation orientated commercial farmers want to maximize annual profit, minimize variability thereof, while maintaining the health of their biological systems. The more capital intensive a farming operation becomes, the more important predictability of financial returns becomes. I modelled the optimum level of infrastructure development, stocking rate, animal mix, and supplementation which will maximize mean annual profit, and minimize variability thereof over a period of 50 years. Rainfall is a crucial element of the model, and determines the optimum strategy. In lower rainfall regions ($\leq$ 200 mm) tracking forage resources would be a better option, whereas under higher rainfall (> 200 mm) investing in a 60 – 80 camp system on a 7000 ha unit and stocking at the agriculturally recommended stocking rate appears to be the better option. The consistent income of this farming system can be further increased by manipulating the grazer : browser ratio with a suitable mix of cattle, sheep and goats, and by providing rumen stimulants during dry seasons in order to improve forage use efficiency and maintain animal performance levels.

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Appendix A Parameter names, units, symbols and estimates for the model

Parameter Name (units)	Symbol	Estimates
Vegetation sub-model		
Annual rainfall (mm)	P	Eq. (1)
Mean rainfall (mm/yr)	P_{av}	Adjustable
Coefficient of variation of rainfall	CV	Adjustable
Annual grass production (kg/ha)	G_p	1.9 * P
Annual shrub production (kg/ha)	S_p	0.9 * P
Grass decomposition (kg/ha)	G_d	0.4 * G_b
Shrub decomposition (kg/ha)	S_d	0.08 * S_b
Camp numbers	C_n	Adjustable
Farm size (ha)	F_s	7000 or adjustable
Camp size (ha)	C_s	F_s / C_n
Proportion of grass biomass which can be removed	G_{bprop}	Eq. (2)
Rumen stimulants	R_s	1 = yes; 0 = no
Cattle density (LSU/ha)	C_{dens}	Adjustable
Sheep density (LSU/ha)	S_{dens}	Adjustable
Goat density (LSU/ha)	G_{dens}	Adjustable
Intake per Large Stock Unit per day (kg/LSU/day)	I_{LSU}	10
Camp time (days)	C_t	Eq. (3)
Recovery period (years)	R_p	ROUND $[(C_n - 1)C_t/365]$
Graze year	G_y	Eq. (4)
Amount of grass eaten (kg/ha)	G_e	Eq. (5)
Amount of shrub eaten (kg/ha)	S_e	Eq. (6)

Appendix A (continued)

Parameter Name (units)	Symbol	Estimates
Production sub-model		
Area grazed per year (ha)	A_g	Eq. (7)
Meat production (kg/year)	M_p	Eq. (8)
Wool production (kg/year)	W_p	Eq. (9)
Financial sub-model		
Meat price (Rands and cents)	P_m	11.94
Wool price (Rands and cents)	P_w	12.76
Annual gross income (Rands)	I_{gross}	Eq. (10)
Average value per Large Stock Unit (Rands)	A_{val}	2000
Total animals stocked (LSU)	A_{Tot}	Adjustable
Animal increment (LSU)	A_{incr}	$A_{Tot} - 300$
Additional camps	C_{incr}	$C_n - 10$
Marginal cost per camp (Rands)	C_{cost}	Figure 2
Set-up costs (Rands)	SU_{cost}	$C_{incr} * C_{cost}$
Maintenance costs (Rands)	M_{cost}	$SU_{cost} * 0.02$
Rumen stimulant costs (Rands)	R_{cost}	If $R_s = 1$ then $A_{Tot} * 32$ else 0
Expenses (Rands)	E_x	Eq. (11)
Annual profit (Rands)	I_{profit}	Eq. (12)

GENERAL CONCLUSIONS

The study on the farm Rietfontein in the Succulent Karoo showed that small scale spatial changes in plant community structure are highly correlated with soil physical and chemical properties. Physical factors such as soil depth, stoniness, surface cover and texture are all related to the hydrological characteristics of the soil and are therefore important for plant establishment and survival. Microtopography and water movement within and on top of the soil further creates soil habitats which differ in topsoil structure and nutrient status.

The long history of large grazer activities around the central watering point at Rietfontein resulted in another pattern superimposed upon the physically determined one. The years of frequent, selective defoliation by livestock created a low-palatable, species-poor and largely ephemeral community within a 1 to 2 km radius around the watering point. Even after approximately 20 to 25 years of fences and some form of rotational grazing, this high utilization zone is still scattered with numerous bare areas and is clearly in a degraded state compared to the rest of the farm. The proposed mechanism of degradation might help to explain why recovery is slow, if at all possible.

The results of the study strongly support the theory that long-term overgrazing, which is not necessarily equal to overstocking, can lead to the loss of resources, namely water, topsoil, and nutrients. The impoverished soil environment feeds back into the plant community by allowing only xerophytic and ephemeral species to persist. This is an undesirable situation from both an ecological and economic point of view, and should serve as a motivation for karoo rangeland managers to monitor their range condition with the aim of timeous identification of retrogression. In this regard it is useful to concentrate on the abundances and dynamics of low-growing, succulent shrubs like *Malephora crassa* and *Rhinephyllum macradenium*. These shrubs create fertile patches, capturing seeds, and facilitating the establishment of other species. A decline in their numbers could serve as an early warning sign to either adjust stock numbers or, more likely, to introduce infrastructural changes which will improve control over animal movements. Further research needs to find similarities between this and other Succulent karoo grazing gradients in order to strengthen the generalities regarding soil changes and key species. More importantly there is a need to find cost effective farming systems which will reduce the sacrifice zones around watering points to a minimum and in so doing conserve biotic diversity and soil resources.

The results presented in Chapter 2 lends further support to the fact that soils ultimately change as a result of karoo rangeland degradation and that these changes act as an effective threshold to autogenic veld improvement. Although there was an abundance of seeds from the surrounding community, very few of these, or of the hand-sown seeds, germinated in the untreated bare soils. The significantly better germination and establishment of naturally-seeded and hand-sown seeds on the gypsum and/or mulch treated bare soils, supports the hypothesis that restoration has to start with improving the topsoil environment by addressing physical, chemical, hydrological and biological properties. Gypsum ($CaSO_4$) improved the soil surface structure, reduced crusting, and therefore improved infiltration, whereas the mulch reduced raindrop impact, improved infiltration, and created sheltered microsites. The seed source does not appear to be limiting although palatable, preferred species were virtually absent from the seed bank. Rainfall emerged as a strong determinant of germination, establishment and survival of seedlings. Autumn and spring appears to be good seasons for reseeding in the Succulent Karoo, but follow-up rainfall is crucial for the success of the venture.

The practical implication is that reseeding denuded karoo veld without some form of soil treatment is likely to fail. Although costly, gypsum application does improve the success rate. A more affordable approach is to improve the resource retention capacity of the soil with an organic mulch. The mulch may not be as effective as the gypsum in improving infiltration, but contributed to capturing wind-blown sediments and seeds. Cost effective methods of harvesting locally abundant plant material for this purpose, and techniques for holding it down against wind movement can be investigated. Herding livestock across these mulches could result in some of the litter being incorporated into the topsoil, with the consequent improvement of structure, and possibly the stimulation of the very important microbial activities. Large-seeded, wind-dispersed species should be selected for reseeding when a mulch is used, since these species show better results compared to small, water-dispersed seeds adapted to exposed soils. Harvesting less-palatable shrubs like

Pteronia spp., *Ruschia* spp. and *Lycium* spp. as mulches could serve a dual purpose since results show that brushcutting these woody shrubs releases resources for seedlings and young growth. This could improve the acceptability of moribund veld to livestock. However, cognisance should be taken of the increased possibility of accelerated erosion when plant cover is reduced. Restoration research in the Succulent Karoo should concentrate on affordable techniques of stimulating soil microbial activities, including mycorrhizas. The soil microbes, and the processes they mediate, is a possible key to restoring the resource conservation capacity of the Karoo. A valuable contribution would be to show that large grazers have a vital role to play, and that total destocking for restoration purposes is in fact counter-productive.

Concentrating large, mixed herds of cattle, sheep, and goats for relatively short periods of time (ca. 14 days) in small paddocks (ca. 50 ha) with long rests (> 1 year) in between occupations resulted in certain significant impacts on the Nama karoo soils. Soil microbial respiration rates were higher in the open, inter-mound soils of the non-selectively grazed (NSG) compared to the zero-grazed controls. It is postulated that an increase in litter flux into the open areas is generated by the concentrated defoliation, trampling, and dunging by large herbivores. Hoof action fragments and thoroughly mixes the organic matter with the topsoil. Lateral crusts are broken and the improved aeration, together with the organic food source, stimulates microbial activities.. This in turn promotes the formation and maintenance of stable soil aggregates, which is reflected in the increased infiltrability, and reduced erodability of the NSG soils compared to the controls. Although there are signs of increased plant productivity in the non-selectively grazed areas which might be related to the improved structure and hydrology of these topsoils, a number of questions remain unanswered. The relationship between soil microbial populations, their activities, and processes like mineralization and immobilization of nutrients (mainly N and P) needs clarification. Unless more is known about microbes and their ability to bind and release vital nutrients in these soils, it will be very difficult to evaluate results such as the significantly lower total N measurements in the grazed compared to the control soils. Another aspect which became apparent during the study period was the gradual deterioration of the soil surface structure over time, mainly due to raindrop impact. A worthwhile research avenue would be the relationship between time since grazing, amount and type of rainfall, and soil physical properties. Soil type will certainly play a role but it is hypothesized that rests between grazings of much longer than a year will not be beneficial to ecosystem functioning.

The impact of NSG on Nama karoo plant cover dynamics and species composition was less important than the effects of annual and shorter term (quarterly) rainfall. Periodic droughts create gaps in this grassy shrubland, which are occupied by opportunistic species e.g. *Eragrostis lehmanniana* as soon as moisture conditions improve. This may result in temporary shifts in growth form mix and species composition, with slower reacting grasses like *Stipagrostis* spp. being ousted. When moisture conditions turn favourable the recovery strategy of long-lived karoo shrubs like *Pentzia incana* is more based on an increase in plant size than on recruiting seedlings. After three occasions of severe defoliation (more than 50 % of aboveground biomass) of supposedly most plants in the grazed areas, there is no evidence to support the claim that NSG reduces the competitiveness of less-preferred species in favour of a higher diversity, including more preferred species. The long-term monitoring of cover and composition in this and similar study areas is essential in order to evaluate NSG as a management tool for promoting desirable species shifts. However, NSG does appear to stimulate aboveground biomass accumulation, especially in the grass guild. This is important in that it underlines the resilience of this vegetation to severe defoliation over short periods of time. It also implies that there are feedbacks between forage removal, soil impacts, and forage production which can be wisely exploited through manipulating the mouths and hooves of grazing animals. There is a need for detailed investigations into biomass changes over time in order to improve our understanding of compensatory growth in karoo plants. NSG also succeeded in moving standing inactive organic matter to the ground where it contributed to soil quality and hence a more favourable seedbed.

It is still unresolved whether NSG neutralizes the negative effects of selective defoliation. However, the generally positive impacts of NSG on physical soil properties, organic turnover, and forage production warrants further research on this grazing system as a means of restoring ecosystem functioning in the Nama Karoo. The approach of using the animal factor as a restoration tool is unlikely to apply in the more arid Succulent Karoo where the ecosystem is more fragile and the plants are more sensitive. Many of the Succulent karoo species have deterministic growth forms and do not sprout after defoliation – hence their vulnerability to grazing and trampling. In Chapter 5 I also show that NSG, with its costly multi-camp infrastructure, is unlikely to be an economically viable option in low-rainfall areas (< 200 mm), mainly because of slower forage build-up over time. However, in the higher-rainfall parts of the Nama Karoo, with its more resilient grass and karoo shrub components, NSG is a farming system which allows for a more efficient utilization of forage and an accumulation of forage reserves for the dry years. This greater ecological control reduces uncertainty by enabling the manager to maintain stock numbers through droughts. In an attempt to optimize mean annual profit with NSG, the overall picture which emerged is that camp numbers should be such that grazing intensities of 40 – 50 LSUGD/ha can be achieved within the 14-day maximum occupation time, with rests of a year and longer between grazings. Furthermore, stocking rate should be conservative by balancing production per hectare with production per animal. Conservative stocking with mixed herds of sheep, cattle and goats reduces variability of profit, and therefore realises higher averages in the longer term. Rumen stimulants could be useful, but then only during dry periods in order to maintain animal condition.

There is clearly a need for refining the model presented in Chapter 5. Research should focus on the feedbacks between grazing, rainfall and veld condition, and also between veld condition (mainly cover and composition) and biomass production. The rate of biomass loss to litter and the relationship between stocking density and proportion standing crop removed are further aspects where there is a dearth of knowledge.