IMPACTS OF GRAZING SYSTEMS ON NAMA KAROO PHYTODIVERSITY

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Impacts of grazing systems on Nama Karoo phytodiversity

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2. Abstract

The study was carried out on two adjacent farms on the plains of the Nama Karoo near Beaufort West. The impacts of three grazing treatments (a) zero grazing (b) non-selective grazing (c) conventional grazing, on plant diversity and certain vegetation parameters were compared.

Unpredictable and variable rainfall and major disturbance events such as droughts drive vegetation change in the Nama Karoo. Major recruitment events are rare and can determine karoo vegetation composition for many years. The diversity of plant species plays an important role in determining vegetation composition during major recruitment events and following drought or disturbance such as grazing.
Grazing can influence the composition, abundance and seed production of karoo plants and in so doing influence the future abundance of desirable and undesirable forage species. These changes may only become evident over long periods, but small changes in vegetation as a response to grazing treatment can accumulate considerably over time.

On the farm Elandsfontein, studies have shown that non-selective grazing leads to a higher plant turnover rate, resulting in more vigorous and productive plants, and improved ecosystem functioning. However there is no evidence of this grazing system promoting or reducing plant diversity. The aim of this study was to test whether the non-selective grazing system promoted or reduced plant diversity compared to no grazing and conventional grazing. The hypothesis was that there were no differences between the grazing treatments in terms of plant diversity or any of the vegetation parameters measured.

To evaluate this hypothesis, plant data were collected from the three grazing treatments using the Modified-Whittaker vegetation sampling method. The method was further modified for this study to allow for accurate abundance measurements rather than estimates, and an increased area for recording species richness.

Using various diversity indices that incorporate species richness and the proportional abundance of species, plant diversity values for each treatment were obtained.

No differences in terms of plant diversity were found between the treatments.

A significant difference between treatments was found in the density of plants, particularly in perennial grasses and shrubs. Canopy cover percentage did not differ for individual species or as total cover between the treatments.

The results indicate a shift in the non-selective grazing treatment towards a higher density of younger shrubs and perennial grasses as a result of increased plant turnover rate and possibly the greater success of plant establishment beyond the seedling stage. These results have implications for range managers.

Vegetation under the conventional grazing treatment is aging, with shrubs and perennial grasses becoming more moribund and less palatable and nutritious over time as a result of the slow plant turnover rate. Although maintaining similar levels of diversity to zero grazing controls and the non-selective grazing treatment at this stage, establishment of younger plants outside major recruitment events will be limited due to competition from the established plants.

The non-selective grazing treatment may be providing more forage to livestock in the form of younger plants over time, and is maintaining the same levels of plant diversity as both the zero grazing controls and the conventional grazing treatment.

The similarity in plant diversity across the three treatments is a positive result for both non-selective and conventional grazing system practitioners. However the differences in plant density, linked to plant size and possibly age, may influence the levels at which vegetation can support livestock under these different grazing systems.

3. Disclaimer

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

A. D. Wheeler.

January 2003

4. Introduction

This study looks at the effects of three different grazing treatments on plant diversity of the Nama Karoo plains near Beaufort West.

The research forms part of the National Botanical Institute South Africa Conservation Farming Project. The project aims to evaluate conservation farming practices in important biodiversity regions of South Africa so that these practices can be more widely applied as part of an overall conservation strategy for biodiversity conservation.

Conservation farming is defined as agricultural land use and management practices that promote sustainable economic benefits while promoting biodiversity and maintaining the structure and function of natural systems (Donaldson 1999).

The effect of grazing on vegetation structure and composition has been a major research focus of agricultural research institutes, mainly aimed at the dynamics of palatable and unpalatable species and forage productivity (Donaldson 1999). The emphasis on biodiversity conservation in rangelands is comparatively recent (Bond 1999).

4.1. Grazing and karoo vegetation.

Unpredictable and variable rainfall and major disturbance events such as droughts are the main drivers of change in the composition and abundance of karoo vegetation (O'Connor and Roux 1995). Grazing can act in combination with these environmental factors to bring about vegetation changes (Palmer *et al.* 1999), and in the longer-lived plants changes may only become evident over long periods of time (O'Connor and Roux 1995).

Nama Karoo vegetation, other than the short lived grass component, is relatively stable in communities where competition from long-lived shrubs limits recruitment (Milton 1995).

However, the consequences of livestock farming in semi arid areas are diverse and the effects of grazing on plant species richness, composition and abundance have been well documented (Nevah and Whittaker 1979; Waser and Price 1981; Noy-Meir *et al.* 1989; Westoby *et al.* 1989; Milton 1992; Olsvig-Whittaker *et al.* 1993; Milton *et al.* 1994; Milton *et al.* 1997; Milton *et al.* 1999; Todd and Hoffman 1999).

A number of mechanisms of vegetation change brought about by grazing have been considered in the karoo (Palmer *et al.* 1999). In most Nama Karoo range management systems, vegetation change due to grazing is caused by the selective removal of palatable plants or selective reduction of the reproductive potential of palatable plants (Palmer *et al.* 1999). This results in an increase in the cover of less palatable species (Hoffman *et al.* 1999). There is also evidence that prolonged grazing can reduce the fitness of palatable plant species to the advantage of unpalatable species resulting in species composition changes (O'Connor 1991). These changes include a decrease in the density of palatable plants (Noy-Meir 1982; Westoby *et al.* 1989) and an increase in the relative abundance of defended and ephemeral plants (Hoffman and Cowling 1990).

There are reports (Milton and Dean 1993; Milton 1994; Stokes 1994) that selective grazing by sheep promotes unpalatable species by reducing the seed abundance of palatable species. This determines the relative abundance of new generation seedlings by selectively reducing the sizes and reproductive success of the palatable plants (Palmer *et al.* 1999).
The probability of reduced abundance of the more long lived palatable shrubs if they are subjected to heavy grazing for prolonged periods therefore increases due to the lack of soil stored seed banks and plant species that are long-lived and palatable may gradually become locally extinct (Milton 1994).

Perennial grass cover may decrease substantially under the combined influence of drought and grazing and because of depressed seed production these are also prone to extinction (O'Conner 1991). Grazing may therefore result in the extinction of the perennial component and a transition to short lived species with seeds that have the capacity for lengthy dormancy (Palmer *et al.* 1999). Vegetation changes due to overgrazing may result in rapid transitions to annual grass domination (Milton and Hoffman 1994).
With successive years of good rainfall the cover of perennial grasses may suppress annuals that emerge during the immediate post drought period. This suppression may not occur if persistent heavy grazing restricts the growth of perennials (O'Connor 1991). The grazing pressure during and after drought periods may thus be a critical factor determining vegetation change (Milton and Hoffman 1994).

Range degradation has been defined as an effectively permanent decline in the rate at which land yields livestock products under a given system of management (Abel and Blaikie 1989) and the emphasis of rangeland degradation is on permanent change that is impractical or uneconomical to reverse (Novellie 1999).

4.2. Karoo grazing systems.

The directional, or range, succession model characterized by an equilibrium or climax state has largely dominated perceptions in range management (Kent and Coker 1994; Novellie 1999). This perception is that overgrazing causes the system to regress along a predictable pathway to earlier successional stages and a reduction in grazing pressure allows a return to the climax (Behnke and Scoones 1993).
These succession models also envisage that, in the absence of grazing, vegetation develops through succession to a single, persistent state or climax (Kent and Coker 1994).

The range succession model has proved to be inadequate in arid and semi-arid environments. It has been widely observed that neither precipitation nor withdrawal of grazing pressure invariably lead to anticipated successional stages or a reversal of range degradation (Novellie 1999) and in many cases grazing induced changes cannot be reversed by resting (Milton and Hoffman 1994).

Controversy regarding questions about the lenient, heavy or intermediate use of vegetation has given rise to a number of rotational grazing systems such as non-selective grazing, controlled selective grazing, high-utilization grazing and high-performance grazing (Tainton *et al.* 1999).

The study was carried out on two farms, Elandsfontein with zero grazing controls and a non-selective grazing system and Bleakhouse with a conventional grazing system.
The conventional grazing system applied on the farm Bleakhouse is purported to maintain species composition and abundance, with no obvious changes in vegetation over a number of years (J. Lund pers. comm. 2002). The landowner applies what is termed an open rotational grazing system (J. Lund. pers. comm. 2002; Du Toit pers. comm. 2002).

This system involves rotating stock consisting of sheep only, between camps according to a subjective evaluation of the veld condition, the requirements of the stock according to season (mating, lambing and weaning), rainfall and camp grazing history.

Sheep are highly selective grazers in terms of plant species and plant parts, tending to select the most nutritious food available (Owen-Smith 1999). If the length of time they are kept in an area allows, they will regularly return to graze the young re-growth of grazed plants (Danckwerts and Teague 1987). Du Toit (1972) found grazing by sheep alone to be very selective, and detrimental in terms of species composition.

As plant species in the karoo grow, flower and produce seed at slightly different times of the year (Roux 1968), by staggering the defoliation regime no one species or growth form is repeatedly selected. Different species are therefore provided the opportunity to flower, produce seed and establish.
The type of conventional grazing system used on Bleakhouse allows vegetation to be grazed selectively, followed by a lengthy rest period. It is expected that this will maintain species composition and abundance within acceptable limits if the recommended grazing capacity is not exceeded (Roux 1968). Diverse mixes of palatable forage grass and shrub species are predicted within this kind of grazing system (Hoffman *et al.* 1999).

Non-selective grazing has undergone controversial development since its introduction by Acocks in 1966 (Acocks 1966; Hoffman 1988; O'Reagan and Turner 1992; Beukes and Cowling 1999; Hoffman *et al.* 1999; Milton and Dean 1999; Roux 1999). There is an underlying assumption that, relative to lower intensity grazing systems, this high intensity grazing system, through severe defoliation and trampling, can improve the biomass turnover rate and result in more vigorous and productive plants with faster shifts in species composition (McNaughton *et al.* 1988).

The non-selective grazing system used on the farm Elandsfontein is aimed at forcing livestock to graze all species, including the unpalatable species. By doing this, the competitive advantage unpalatable species have under a selective grazing system should be reduced. However, Danckwerts *et al.* (1983) consider the ill effects of selective grazing being prevented by non-selective grazing to often be unfeasible, since animals continue to graze selectively even at heavy grazing pressures.

Under high intensity grazing such as non-selective grazing, trampling may cause physical damage to adult plants and seedlings through breaking and uprooting (Danckwerts *et al.* 1983; Mentis 1981; Owen-Smith 1999). This may lead to changes in the physical strength of the plants resulting in a direct loss of forage biomass (Danckwerts *et al.* 1983).

Intense grazing resulting in reduced plant canopy cover can lead to an increase in water run-off and soil erosion. Animals can alter the structure of soil by compaction or loosening, depending on the soil type and moisture content (Snyman 1999). Loosened soil can aid in water infiltration but may increase the vulnerability of soil loss to wind (Mentis 1981; Snyman 1999). Loosened soil may also bury seeds, promoting germination, or may result in dust coatings on plants, reducing grazing acceptability (Owen-Smith 1999).

Soil compaction by animals may cause loss of soil structure, increased bulk density and reduced pore space, which in turn results in reduced infiltration, aeration and water holding capacity, causing unfavorable conditions for plant growth (Owen-Smith 1999). Many of these disturbance effects on soil have been correlated with high grazing pressures (Heady 1975 in Owen-Smith 1999).

A mixture of cattle, sheep and goats are stocked on Elandsfontein. This stock mix further reduces defoliation selectivity and is potentially more productive and ecologically acceptable than stocking with single species (Nolan *et al.* 2000). Nolan *et al.* (2000) recommend that to sustain maximum animal production of high quality in arid rangelands, stocking should be mixed so that all the components of the vegetation are utilised. However, even under mixed grazing the various components of vegetation are not grazed equally and the toxic and resistant components tend to increase even under heavy grazing (Milton 2000).

The non-selective grazing system also reduces the number of times plant re-growth is removed by shortening the time stock occupies the camp, which improves plant vigour and forage production (Acocks 1966). Beukes (1999) found that the non-selective grazing system used on Elandsfontein resulted in the maintenance of fertile mound habitats with higher levels and turnover rates of soil organic carbon than in zero grazing controls. Non-selective grazing also led to improved water infiltration in soil and resistance to soil erosion. It was found that the management of livestock in this system could lead to improved ecosystem functioning and the maintenance of productivity in times of drought, but there was no evidence of this grazing system promoting or reducing plant diversity.

4.3. Plant diversity and livestock farming.

The postulate that diversity is valuable both intrinsically and functionally to humanity (Tainton *et al.* 1989; Tilman 1997) underlies all environmental conservation actions (Milton 2000). In the karoo, long term grazing experiments have shown that plant diversity is influenced by grazing pressure (Roux and Vorster 1983) and the overuse of rangelands by domestic herbivores can result in the loss of plant diversity (Milton *et al.* 1994).

Key processes driving karoo vegetation dynamics are rare recruitment events (Milton and Hoffman 1994) that can determine the composition of karoo vegetation for many years (Jeltsch *et al.* 1999). Plant diversity in the karoo is important in increasing the resilience of plant communities following drought or disturbance such as grazing. The loss of certain perennial plant species through grazing reduces resilience and lowers the capacity of the vegetation to sustain animal production (Milton and Dean 1999).
The rapid recovery of vegetation after drought or disturbance depends on the species composition and seed availability before the event, as well as the condition of the seedbed immediately thereafter (Milton and Dean 1999).

Changes in species diversity and abundance as a response to grazing can result in a reduced carrying capacity for livestock, which results in a loss of agricultural production (Milton and Hoffman 1994). A diverse species composition provides suitable forage for livestock throughout the year, as well as reducing the risk of disastrous insect damage by specialised insects (Milton and Dean 1996). Outbreaks of insects such as the karoo caterpillar (*Loxostege frustales*), and the harvester termite (*Hodotermes mossambicensis*) can result in serious shortages of forage (Vorster and Roux 1983).

The need to understand and possibly reverse detrimental grazing-induced changes in vegetation has been widely recognised (West 1993) but to rehabilitate degraded karoo rangeland may not always be practical or economical and may require active intervention by the land manager (Milton and Hoffman 1994). It is therefore important to identify grazing systems influencing plant diversity in the karoo early (Beukes 1999a).

4.4. Key questions.

Plant diversity plays an important role in vegetation dynamics where unpredictable and variable rainfall and major disturbance events such as droughts are the main driving forces of vegetation change. Changes in plant diversity can have important implications for range managers. Although detection of change in karoo vegetation is difficult due to the slow rate of population turnover (Yeaton and Esler 1990), small positive or negative responses to grazing treatment are consistent in direction and can accumulate considerable magnitude over time (Vorster 1999).

In order to establish whether non-selective grazing, which may result in more vigorous and productive plants (McNaughton *et al.* 1988), and improved ecosystem functioning (Beukes 1999), promotes or reduces plant diversity compared to zero grazing and conventional grazing, the following key questions are asked.

1. Does non-selective grazing increase or decrease plant diversity compared to no grazing and conventional grazing?
2. Does non-selective grazing result in the increased or decreased abundance of certain plant species or groups compared to no grazing and conventional grazing?
3. Does non-selective grazing result in the increased or decreased abundance of seedlings compared to no grazing and conventional grazing?

5. Study area

This study was carried out on two farms, Elandsfontein and Bleakhouse in the Nama Karoo. The Nama Karoo is the second largest biome in the region and occurs mostly on the high-lying central plateau of the western half of South Africa. The Nama Karoo is associated with high maximum temperatures and strongly seasonal summer rains (Desmet and Cowling 1999). The rainfall for this area is low and unpredictable and the vegetation is dominated by grassy dwarf shrubland. Most of the grasses are of the C_4 type (Low and Rebelo 1998).

The study area consists of a semi-arid grassy shrubland dominated by grasses, mainly species of *Stipagrostis* and *Eragrostis*, with *Pentzia* and *Rosenia* species being the dominant dwarf shrubs (Beukes and Cowling 1999). The two study site farms neighbour each other and are both situated north-east of Beaufort West. Figures 1 and 2 show the landscape of Elandsfontein and Bleakhouse.

Figure 1 Elandsfontein landscape.

Figure 2 Bleakhouse landscape

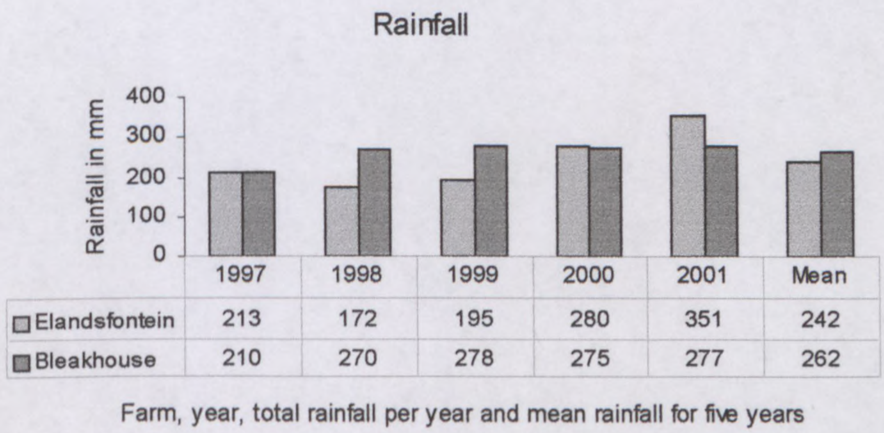
Elandsfontein is situated 32km north-east of Beaufort West (32°19'S / 22°54'E).
Bleakhouse is situated 6 kilometers to the north of Elandsfontein (32°16'S / 22°55'E).
The topography is level to near-level pediments with scattered dolerite outcrops. The mixed shale and dolerite parent materials have weathered into reddish coloured, sandy loam, duplex soils (Ellis and Lambrecht 1986).

The vegetation of this area is classified as Acocks (1975) veld type 26, Karroid Broken Veld. Low and Rebelo (1998) classify this vegetation as central lower karoo. Milton and Dean (1996) classify the vegetation of this area as falling into the mixed karoo veld type, and Bosch (1999) as falling into the upper non-succulent karoo region.

Mean annual rainfall for the region ranges between 200 and 300mm and the coefficient of variation of annual rainfall is between 35% and 40% (Schulze 1997).
The estimated average annual rainfall for the farm Elandsfontein is 220mm (A. Lund pers. comm. 2002) and for Bleakhouse 230mm (J. Lund pers. comm. 2002).

The total annual rainfall each year from 1995 to 2001 for Elandsfontein and Bleakhouse and the mean rainfall over that period is shown in Figure 3.

The area suffered a severe drought between 1990 and 1994 and the mean annual rainfall for this period was recorded as 139mm. The drought was broken in 1995 with rainfall of 238 mm for Elandsfontein (Beukes 1999). There is a trend in this area, for more rainfall, with a higher reliability, to fall in the summer months in the form of cloudbursts. Deep cold frontal systems bring light rain in the winter months (Desmet and Cowling 1999).



	1997	1998	1999	2000	2001	Mean
Elandsfontein	213	172	195	280	351	242
Bleakhouse	210	270	278	275	277	262

Figure 3 Total annual rainfall for Elandsfontein and Bleakhouse from 1995 to 2001 with mean annual rainfall over the five-year period shown in the last column.

Wild herbivores such as porcupine (*Hystrix austro-africanae*), steenbuck (*Raphicerus campestris*), hares (*Lepus* spp.) and tortoises (*Psammobates* spp.) occur on both farms (Beukes 1999; A. Lund pers. comm. 2002; J. Lund pers. comm. 2002).

Soils from each of the twelve sample plots were analysed according to colour using the Munsell soil colour chart (Munsell 1971) and texture (Ellis pers. comm. 2002).

Fifteen topsoil (top 10cm) samples were also collected randomly from each of the $2500m^2$ plots (Mills pers. comm. 2001) and analysed for acidity or alkalinity (pH) and soluble salts (resistance in ohms).

Analysis of soil samples taken from each sample plot showed soil texture across all three treatments to be sandy loam (SaLm) and colour 7.5 yellow-red, 5/6 to 5/8, strong brown. Analysis further showed a neutral $pH_{(water)}$ and low free salt content (as indicated by the soil electrical resistance analysis) across all plots (Table 6). No statistically significant differences were found between the treatments.

Table 6 Soil $pH_{(water)}$ and soil electrical resistance with standard deviation.

Soil analysis		
Zero grazing	Non-selective	Conventional
PH (water)		
Mean ± SD	Mean ± SD	Mean ± SD
7.15 ± 0.24	7.27 ± 0.33	7.2 ± 0.2
Resistance (ohms)		
Mean ± SD	Mean ± SD	Mean ± SD
1822 ± 441.8	1405 ± 273.4	1748 ± 309.9

6. Grazing systems

6.1. Elandsfontein non-selective grazing system.

Elandsfontein farm extends over an area of 7000 hectares.

Over a period of 40 years this farm has been subdivided into 139 camps with an average size of 50 hectares. The infrastructure consists of the 139 camps arranged in a wagon wheel system around 38 permanent watering points (Lund 2001).

Since 1992 a high stock density grazing system has been implemented on the farm, using mixed herds consisting of Nguni cattle, Boer goats and Merino sheep (Beukes 1999; A. Lund pers. comm. 2002).

Vorster (1999) recommends the long-term grazing capacity of the Beaufort West area, assuming the application of an acceptable management programme and an appropriate combination of animal types, to be between 30 and 36 hectares per large animal unit. Milton and Dean (1996) state that the recommended stocking rate as per the Department of Agriculture is between 31 and 40 hectares per large animal unit.

Non-selective grazing has been advocated as a grazing system that reduces selective defoliation by forcing animals to eat more species, including the less palatable species, and reduces the number of times re-growth is removed by shortening occupation time, improving plant vigour and forage production (Acocks 1966).

The Elandsfontein non-selective grazing system involves reducing the number of stock herds and concentrating the animals in camps for short periods (10 days or less). The herds are rotated through the camps to create a non-selective grazing system by concentrating a large number of animals in a small area for a short period. The average herd ratio is 14 Merino sheep: 3 Boer goats: 1 Nguni beast (A. Lund pers. comm. 2002).

Since 1995, grazing pressures of 40-60 Large Stock Unit Grazing Days per hectare (LSUGD/ha) have been applied on Elandsfontein, compared to the average recommended stocking rates used in more conventional group camp systems, of 10-20 LSUGD/ha (Beukes and Cowling 1999).

The stocking rate was kept similar across the different size camps on Elandsfontein, by manipulating the number of days stock was kept in the camp. Camps A, B, E and F, which were sampled in this study, had been subjected to this non-selective grazing treatment each year since 1996 (Table 1). The camps are rested for a period of approximately 12 months after grazing.

This grazing system consists of high intensity, short duration non-selective grazing by a mix of domestic livestock species, with long rest periods after grazing.
Atriplex nummularia (saltbush) is provided as a supplementary feed (Beukes and Cowling 1999) to help maintain rumen function and improve the intake of low quality fibre (Barnard 1986).

Table 1 Non-selective grazing history in large stock unit grazing days per hectare (LSUGD/ha) for camps A, B, E and F on farm Elandsfontein.

	Camp	A	B	E	F
	Plot	A	B	E	F
	Ha	34	87	108	32
Year	Month	LSUGD/ha			
1996	May	0	0	35-45	0
	July	35-45	35-45	0	35-45
1997	March	0	35-45	35-45	0
	April	0	0	0	35-45
	May	35-46	0	0	0
1998	June	61	52	51	54
1999	May	55	0	0	55
	June	0	50	40	0
2000	Feb	42	0	0	0
	March	0	40	41	43
	Nov	7	3	2	7
2001	Oct	0	0	17.5	35
2002	March	69	41	0	0

6.2. Elandsfontein zero grazing controls.

In April 1995, four 50m x 50m control exclosures were erected in camps A, B, E and F on the farm Elandsfontein (Beukes 1999). These exclosures were set up using a five strand stock fence to keep livestock out so that various comparisons could be made between the controls and the grazing treatments. No livestock grazing has taken place within the four control plots since 1995.

Possible herbivory may have taken place in the form of steenbok (*Raphicerus campestris*), hares (*Lepus* spp.), tortoises (*Psammobates* spp.), porcupine (*Hystrix austro-africanae*), and invertebrates (Beukes 1999).

6.3. Bleakhouse conventional grazing system.

Bleakhouse farm covers an area of approximately 8900 hectares and is divided into 43 camps. The three camps used in this study were K2 (268 hectares), K4 (156 hectares) and K5 (219 hectares). These camps are stocked with sheep only. The stock consists mainly of Merino sheep, but cross breed rams and lambs are included at certain times of the year.

The landowner uses a conventional grazing system termed an open rotational grazing system (J. Lund. pers. comm. 2002; Du Toit pers. comm. 2002). This system involves the rotation of stock between camps according to a subjective evaluation of vegetation condition, the requirements of the stock according to season (mating, lambing and weaning), rainfall and camp grazing history. This grazing system allows for the selective grazing of vegetation followed by a lengthy rest period.

The conventional rotational grazing system has been used on the farm Bleakhouse for a number of years having changed little from previous farming generations. According to J. Lund (pers. comm. 2002) slight changes to this system were implemented from 1995 on. These changes were subtle, and based on his subjective evaluation, as opposed to the subjective evaluation of the previous range manager, of range condition. This involved slight changes in stock rotation timing.

The camps are mainly used for lambing and are generally stocked for a period of approximately three to six months starting in spring (August, September, October, November) or autumn (March, April, May, June). This period may vary depending on the range managers assessment of the various factors mentioned earlier. For example, camp K2 was stocked for longer than normal, 8 months, from May to December, in 1998.

The stock composition varies between Merino and cross breed adult animals, adult animals and lambs, and lambs only during weaning periods. This composition depends on various stock farming management requirements and is decided on by the stock farmer. On average the number of sheep placed in the camps varies between 100 and 120 animals per camp, but in some cases up to 300 animals have been placed in a camp at one time.
The exact history of when these higher densities of stocking were implemented is difficult to ascertain and the stocking rate is usually based on the subjective assessment (J. Lund. pers. comm. 2002).

The recent grazing history of Bleakhouse shows that camp K2 was stocked in August 2001 to November 2001 (four months) and rested until August 2002 (rest period of approximately eight months). K4 was stocked in August 2001 to November 2001 (four months) and rested to August 2002 (rest period of approximately eight months). K5 was last stocked in January 2001 for a period of six months thereafter being rested for nine months until March 2002. The grazing history of the three camps used in this study is shown in Table 2.
Although the stock composition may vary with regards to adult or juvenile animals, an average period of approximately four months of selective grazing in these camps is accurate. Stock is never left in the camps for a full annual cycle and is rotated between camps. The three camps sampled have generally been rested for periods varying between eight and twelve months after grazing but in some instances the camps have been rested for longer periods of up to two years. It can however be accepted that the camps sampled undergo selective grazing by 100 to 120 sheep for a period of about four months after which the camp is rested for a period of between eight and twelve months.

The timing of stock rotation varies on Bleakhouse, depending on animal condition, season, rainfall, and grazing history of each camp. Beukes (pers. comm. 2002) calculated the grazing intensity on Bleakhouse to be between 11 and 17 large stock unit grazing days per hectare (LSUGD/ha) compared to that of Elandsfontein which was calculated at being between 40 and 60 LSUGD/ha.

The three camps sampled on the farm Bleakhouse therefore have a strict history of a conventional rotational grazing. No supplementary feeding is used apart from licks given to pregnant ewes approximately 6 weeks before lambing. Herbivory on Bleakhouse also takes place in the form of steenbok (*Raphicerus campestris*), hares (*Lepus* spp.), tortoises (*Psammobates* spp.), porcupine (*Hystrix austro-africanae*), and invertebrates (Beukes 1999, J. Lund pers. comm. 2002).

Table 2 Conventional grazing history in large stock unit grazing days per hectare for camps K2, K4 and K5 on the farm Bleakhouse.

	Camp	K2	K4	K5
	Plot	1 + 2	4	3
	Ha	268	156	219
Year	Month	LSUGD/ha		
1998	March-Nov	0	0	21
	April - Dec	0	12	0
	May - Dec	18	0	0
1999	Jan - Feb	0	3	0
	Aug-Dec	0	10	0
	Sept-Dec	0	0	7
	Oct - Dec	5	0	0
2000	Jan - Feb	4	0	0
	Jan-Apr	0	0	6
	Aug - Dec	14	0	10
	Oct-Dec	0	10	0
	Dec-March	0	3	0
2001	Jan - Feb	3	6	0
	Jan -Mar	0	0	6
	Mar-June	0	0	7
	Aug-Nov	8	5	0

7. Methods

7.1. Selection of sampling plots.

This study was carried out at the landscape or local ecosystem scale (Kent and Coker 1994). Four 50m x 50m replicates for each of the three grazing treatments (zero grazing, non-selective grazing and conventional grazing) (Table 3) were chosen using plots identified by Beukes (1999) and plots identified for the National Botanical Institute Conservation Farming Project (Donaldson 1999).

Table 3 Description of sample plots showing farm, treatment, camp, sample plot, size of sample plot area in m² and size of camp in hectares.

Sample plots				
Elandsfontein zero grazing control				
Camp	A	B	E	F
Sample plot	A	B	E	F
Sample plot area	2500m²	2500m²	2500m²	2500m²
Camp size ha	34 ha	87 ha	108 ha	32 ha
Elandsfontein non-selective grazing treatment				
Camp	A	B	E	F
Sample plot	AA	AB	AE	AF
Sample plot area	2500m²	2500m²	2500m²	2500m²
Camp size ha	34 ha	87 ha	108 ha	32 ha
Bleakhouse conventional grazing treatment				
Camp	K2	K2	K5	K4
Sample plot	B1	B2	B3	B4
Sample plot area	2500m²	2500m²	2500m²	2500m²
Camp size ha	268 ha	268 ha	219 ha	156 ha

Beukes (1999) assessed four non-selective grazing camps on the farm Elandsfontein as similar in terms of topography, soils and vegetation and placed a 50m x 50m exclosure in each camp. These exclosure plots were erected away from watering points and stock paths and served as the zero grazing treatment replicates for this study.
50m x 50m areas outside these exclosures served as the non-selective grazing sample plots.

On the conventional grazing treatment of Bleakhouse, a number 100m x 100m sample plots had been set out by researchers from the Conservation Farming Project. For this study four similar Conservation Farming Project plots on Bleakhouse in terms of topography, soils and vegetation were subjectively selected and a 50m x 50m plot placed in each, using alternate corner points of each of the 100m x 100m plot as starting points.
These plots were assessed as similar to the zero grazing and non-selective grazing plots used for this study on Elandsfontein in terms of topography and soils (Mills pers. comm. 2001).
The four Bleakhouse plots were situated away from watering points and stock paths. All of the selected sample plots were situated on the level to near level pediments of the landscape.

Because of the proximity of the two farms to each other (< 10km distance between sample plots on Elandsfontein and sample plots on Bleakhouse) and considering the average rainfall for the area, the climate can be considered similar across all sample plots.

7.2. Sample plot design.

Stolgren *et al.* (1995) recognised a lack of precise methodologies available to measure biological diversity and the need to standardize sampling techniques to assist in resource inventories and monitoring long-term trends in vascular plant species richness.

Diversity measurements basically take into account two factors, species richness and relative abundance (evenness or unevenness) (Magurran 1988), and patterns of plant diversity can be explained only by systematic surveys and sampling at multiple scales (Whittaker 1977; Stolgren *et al.* 1995). These allow for evaluations of the influence of spatial scale on local species richness patterns and for better comparisons of community richness than single-scale measurements.

The Whittaker plot vegetation sampling method (Shmida 1984) has been widely used for many years to collect species richness data at multiple scales ($1m^2$, $10m^2$ and $100m^2$ subplots within a $1000m^2$ plot), but Stohlgren (1994) found that this method had three distinct design flaws.

Firstly species richness is influenced by plot shape if the habitat is not strictly homogenous, circular or square plots will generally have less species than long, thin rectangular plots covering a more heterogeneous area (Bormann 1953).

Secondly, the Whittaker plot design moves from 1m x 1m and 10m x 10m squares to 2m x 5m and 20m x 50m rectangles, confusing the influence of plot shape and size.

Thirdly the smaller plots are not independent from the larger plots in terms of species richness.

Stohlgren *et al.* (1995) tested different field designs to minimize the design problems of the Whittaker plot when collecting plant diversity information. They concluded that in all habitat types of the study areas, and for all plot sizes, a long-thin plot design consistently returned higher species richness values than the original Whittaker design and more accurately reflected the total species richness recorded in a complete plant survey of the area. The Modified -Whittaker method was found to return significantly higher ($p < 0.05$) species richness values in multiple scale ($1m^2$, $10m^2$ and $100m^2$) subplots and better estimates of mean species cover for the $1000m^2$ area than the original Whittaker method.

The Stohlgren *et al.* (1995) Modified-Whittaker nested vegetation sampling method was used for this study, with further modifications to the design and sampling methods. The modifications for this study were added to allow for accurate measurements of vegetation parameters in place of estimates and to increase the sample area for the recording of species richness.

7.3. Plot layout.

The sample plots used in this study consisted of four 50m x 50m replicates on each of the three grazing treatments. Various subplots were placed within each of the 50m x 50m plots. A summary of sample plots is given in Table 4 and a diagram of the sample plot layout is shown in Figure 4.

Table 4 Subplots within each of the twelve 2500m² sample plots, showing the number of plots (No), the length and width of each subplot and the subplot area in m². The table also shows plots that overlap and those that are independent.

Sample plots				
Plot		No	Length Width	m²
R	overlapping	1	50 x 50	2500
O	overlapping	1	50 x 20	1000
P	independent	1	50 x 15	750
Q	independent	1	50 x 15	750
N	overlapping	1	20 x 5	100
M1-20	independent	20	2 x 0.5	20
A1-10	independent	10	2 x 0.5	10
G1-10	independent	10	2 x 0.5	10
B - L	independent	10	2 x 0.5	10

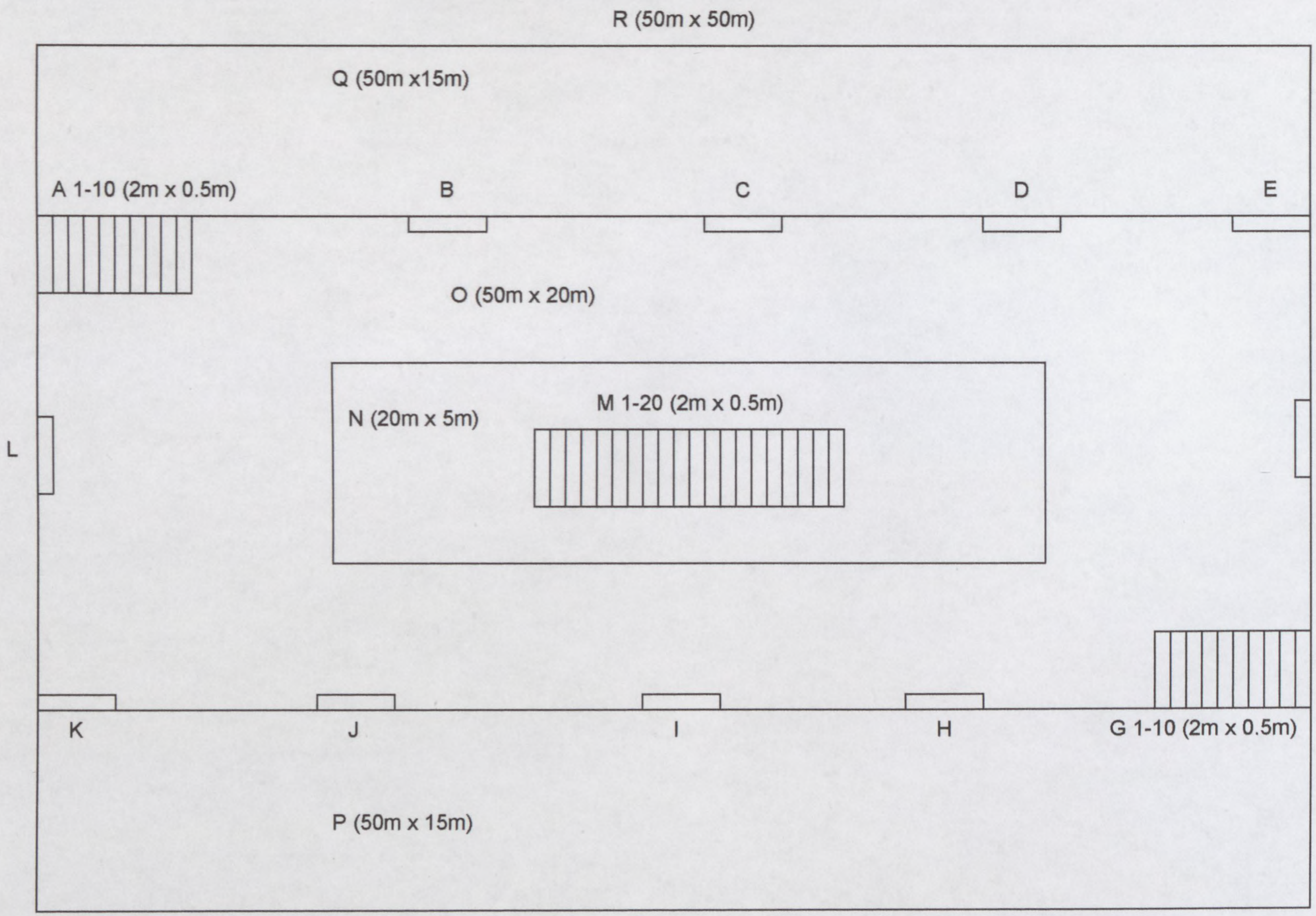


Figure 4 The Modified-Whittaker vegetation sampling plot layout (Stohlgren *et al.* 1995) with modifications for this study (Not to scale).

All plot and subplot layouts were marked using tape measures, pre-measured conspicuous yellow nylon lines marked with coloured tape and wooden marker pegs (600mm long and clearly marked with red and yellow tape). Figure 5 shows an example of the plot layout on Bleakhouse.

Figure 5 Example of sample plot layout on Bleakhouse.

Within each 50m x 50m subplot (R) a rectangular 20m x 50m subplot (O) (Modified-Whittaker method) was placed 15 meters from the corner marker of the 50m x 50m plot (running down slope to increase the coverage of possible vegetation variation).

A 20m x 5m overlapping rectangular subplot (N) (overlapping M) was centered in the 20m x 50m subplot (Modified-Whittaker method). A 2m x 10m rectangular subplot (own modification) (M) was placed in the centre of subplot (N). This was to allow for the precise measurement of various vegetation parameters in twenty 0.5m x 2m subplots (M1-M20).

Two non-overlapping 2m x 5m rectangular subplots (A and G) were placed in diagonally opposite corners of subplot O (Modified-Whittaker method). Subplots A and G were each divided into ten 0.5m x 2m subplots (A1-A10 and G1-G10) (own modification). This was also to allow for the precise measurement of various vegetation parameters in small subplots.

Ten non-overlapping 0.5m x 2m subplots (B, C, D, E, F, H, I, J, K and L) were systematically placed along the inner perimeter of subplot O (Modified-Whittaker method). Two 15 m x 50 m rectangular subplots (P and Q) were situated on either side of subplot O (own modification) to increase the sample area used to record species richness.

7.4. Vegetation sampling.

The Modified-Whittaker sampling method was adapted to include subplots in which accurate measurement of vegetation parameters could be recorded, rather than using abundance estimates. This modification was made by subdividing subplots A and G into ten 0.5m x 2m subplots each and subplot M into twenty 0.5 m x 2 m subplots. This allowed for the placing of fifty $1m^2$ (including the 10 perimeter $1m^2$ subplots) subplots within the standard Modified-Whittaker 20 m x 50 m plot in which accurate measurements could be taken. It was presumed that the small subplots would allow for more thorough scrutiny of the vegetation than larger plots, to include smaller, less conspicuous species, which would be important in terms of species richness, and seedlings (Beukes pers. comm. 2001).

Within the twelve 50m x 50m sample plots, the Modified-Whittaker method of recording species richness within 20m x 50m was modified to include subplots P and Q (Figure 4). This increased the area sampled for species richness for each sample plot from $1000m^2$ to $2500m^2$, and in total over three treatments from 12 $000m^2$ to 30 $000m^2$.

It was found during the preliminary phase of this study that by using random 1m x 1m subplots within the 50m x 50m plots, $50m^2$ of vegetation could be thoroughly and accurately measured in one day in terms of the data needed for this study. This served as the basis for the number of subplots that were used for this study.

7.5. Vegetation data collection.

A rectangular frame was used to outline the fifty 0.5m x 2m subplots in each 20m x 50m plot (Figure 6).

The frame was placed over the subplots M1-20, A1-10, G1-10 and B, C, D, E, F, H, I, J, K and L to give a series of fifty $1m^2$ subplots.

All canopy cover and height measurements were recorded using a standard one-meter steel ruler. All fieldwork was carried out over a single period in February 2002 and the same recorder made all observations with concurrence when necessary by an assistant.

Figure 6 2m x 0.5m frame used to outline the 50 x $1m^2$ subplots in sample plots.

The types of data recorded over three treatments in each of the sub-plots are summarised in Table 5.

Table 5 Data recorded in sample plots.

Species richness was recorded in $2500m^2$ per sample plot.

Other vegetation parameters were measured in 50 x $1m^2$ subplots per sample plot. Four replicate sample plots were placed in each treatment.

	Data recorded in sample plots.
1	Plant species.
2	Growth form.
3	Number of individuals of each species.
4	Canopy length of individual plants.
5	Canopy width of individual plants.
6	Height of individual plants.
7	Identified seedling species.
8	Number of identified seedlings.
9	Number of unidentified seedlings.
10	Number of seedlings occurring within the canopy area of an adult plant.
11	Number of seedlings occurring outside the canopy area of an adult plant.
12	Number of grass seedlings.
13	Percentage ground covered by prostrate dead plant material.

All plant species present as well as the number of individuals of each species per subplot were recorded.

A species list was compiled for the fifty 0.5m x 2m subplots. New species were added to this species list by noting all new species found in subplots N, O, P and Q. This was carried out by systematically scanning subplots N, O, P, and Q for species not previously recorded in the fifty 0.5m x 2m subplots.

A complete species list was compiled for each of the twelve 50m x 50m plots. A species list was then compiled for each treatment.

Measurements of vegetation parameters were taken in $50m^2$ within each 2 $500m^2$ plot.

The canopy diameter of each individual plant was measured along two axes (length and width). A measurement of canopy cover spread provides a realistic indication of species composition in karoo vegetation, and provides more useful information than does basal cover measurements (Roux 1963; Vorster1982; DuToit 1995 and Westfall *et al.* 1994).
Length was measured along the longest part of the canopy, followed by a width measurement at a right angle to the length measurement. A plant was judged to fall within the 0.5m x 2m subplot if half or more of that plants canopy cover fell within the subplot. Canopy cover area per plant in cm^2 was calculated using the formula $\pi LW/4$. (Esler and Cowling 1993).

Plant height was measured in centimeters from ground level to the highest point of the plant. All seedlings in each subplot were counted and grouped by species or as unidentified seedlings (a seedling was defined as a plant of a height less than 3cm). The proximity of each seedling to an adult plant of any species was noted. Seedlings were subjectively judged to be within or outside the canopy cover area of an adult plant in terms of the perpendicular projection of the canopy cover onto the ground surface.

Grass seedlings were recorded separately (a grass seedling was defined as an unidentifiable grass plant of a height less than 10cm). Only grass seedlings occurring away from main grass culms were recorded in order to reduce the possibility of including young plants stemming from the stolons or rhizomes of adult grasses.
Litter was estimated as the percentage of prostrate dead plant material present within the 0.5m x 2m subplot area.

Species were allocated to one of the following broad categories based on major growth form and life history: annual herbs; perennial herbs; annual grasses; perennial grasses; perennial shrubs; perennial spiny shrubs; perennial woody shrubs; leaf succulent woody shrubs; succulents and plant size. Plant size was judged according to canopy spread as an indicator of above ground biomass (Du Toit 2001).

Many studies (see Friedel *et al.* 1988; Noy-Meir *et al.* 1989; Friedel 1997; McIntyre *et al.* 1999 and Weiher *et al.* 1999) have tried to identify functional groups related to grazing and these have been confined to recognizing changes in morphology such as heavy grazing favoring small, prostrate plants over erect, tall plants.

In most of these studies classifications have been undertaken from more or less extensive lists of traits, but the greatest effect appears to be on life form as this trait is correlated with other functionally important traits such as plant size. (Landsburg *et al.* 1999).

Because some traits are related to many aspects of plant function, the minimal list suggested by Weiher *et al.* (1999) consisted of leaf area, seed mass and above ground biomass.

Du Toit (2001) found that the measurement of canopy cover could be used successfully to non-destructively estimate the available above ground biomass of the dominant bushes in the Nama Karoo.

Plants were grouped for this study according to canopy cover size as an indication of plant size and above ground biomass. Grass species were excluded from the plant size grouping as they were found to be unsuitable for correlating canopy cover size to above ground biomass (Du Toit 2001).

Also taken into consideration for the grouping of plants for this study, were the core plant functional traits described by Weiher *et al.* (1999) that are useful for predicting vegetation responses to disturbance such as grazing. These core traits include plant life span that can be approximated by allocating species as either annual or perennial, and stem density as a measure of woodiness.

McIntyre *et al.* (1999) state that one of many plant grazing response traits is the avoidance of damage by defense traits that act as deterrents, such as spines, thus the grouping of obviously spiny plants into one category for this study. Leaf succulent woody shrubs were separated from woody shrubs, as they may be more sensitive to overgrazing (Cowling *et al.* 1994; Todd and Hoffman 1999).

It is accepted that the broad growth form categories used for this study may conceal the more subtle differences caused by grazing to the finer plant functional groups, (McIntyre *et al.* 1999). However the categories chosen are deemed suitable for evaluating the effect of different grazing systems on the abundance of any one broad group of plants.

The allocation of species to these broad categories was based on growth form and life history descriptions in literature (Bond and Goldblatt 1984; Milton 1992; Van Oudtshoorn 1992; Shearing 1994; Le Roux *et al.* 1994; Dean and Milton 1999; Du Toit 2001a) and on personal observations made in the field.

An importance value was calculated for each species based on the relative contribution of that species to the total plant composition of each treatment. The importance value was expressed as a total of the values for relative density, relative frequency and relative cover (Smith 1990). This value was expressed as a percentage.

Plant specimens were collected in accordance with the Western Cape Nature Conservation Scientific Services guidelines for the collection of plants and the preparation of herbarium specimens for identification and reference purposes (Cape Nature Conservation 1994). Identification was carried out where possible in the field and unidentified specimens were identified at the Compton Herbarium, National Botanical Institute, Kirstenbosch.

7.6. Species diversity and diversity indices.

Diversity is one of the central themes of ecology, but there is disagreement about how it should be measured (Magurran 1988). There is also confusion over the meaning, the methods of assessing, and the ecological interpretation of diversity (Kent and Coker 1994).

Species diversity includes the number of species, the abundance of the species and the apportioning of abundance among the species (Green 1979; Magurran 1988; Smith 1990). Fischer *et al.* (1943) first introduced the idea of species diversity in connection with log-series distribution (Pielou 1969; Green 1979). The work of Margalef (1958) popularized this concept among ecologists (Green 1979) and created strong interest in diversity indices as a means of simplifying and explaining communities. Species diversity measures have been divided into three main categories (Magurran 1988). These are species richness indices, species abundance models that describe the distribution of species abundances, and indices based on the proportional abundances of species.

The diversity indices are attractive, as they appear to reduce the information of large amounts of data to single numbers (Kent and Coker 1994), but the application of single-figure diversity indices to characterize complex community structure can be criticised, as much of the original species information is lost (Green 1979).

A widely used approach (Clarke and Warwick 1994; Kent and Coker 1994) to measuring species diversity, is the Shannon-Wiener diversity index method (Shannon and Wiener 1963) which is based on the proportional abundances of species. Here a single index value is used to express the species diversity of a sample area (Kent and Coker 1994).
The formula is sensitive to changes in the number of species and to the distribution of individuals among the species, and is sometimes referred to as a heterogeneity index, as it is based on the proportional abundance of species taking into account both equitability and species richness (Magurran 1988).

The Shannon-Wiener formula requires species richness data and the relative abundance of each species, for the comparison of diversity in sample areas of equal size (Green 1979; Magurran 1988; Smith 1990; Kent and Coker 1994). This method is considered useful for comparing diversity when using a number of replicate samples, and the indices are suitable for the use of parametric statistics such as analysis of variance.

The species and abundance data collected for this study provides a full description of diversity (Magurran 1988). Therefore the data collected in terms of species richness and abundance (cover and density) per species was deemed suitable for application to the Shannon-Wiener formula.

However, there is much controversy regarding the use of indices as an efficient way of summarizing and interpreting biological data (see Hurlburt 1971; Goodman 1975; Green 1979; Magurran 1988; Smith 1990; Kent and Coker 1994; Clarke and Warwick 1994). When a measure of species diversity is required for comparative purposes, simple, meaningful indices, such as species richness (S) and the Margalef species richness index (D_{mg}) may be less ambiguous, but as informative, as the more complex indices such as the Shannon-Wiener diversity index (Green 1979).

In a simulation study Green (1979) found that species richness (S) was a better indicator of biological change than the Shannon-Wiener index (*H'*). However Kempton (1979 in Magurran 1988) observed that the distribution of species abundances is often a more sensitive measure of environmental disturbance than species richness alone. Also, although as a heterogeneity measure the Shannon-Wiener index does take into account evenness of species abundances (Peet 1975), a separate, additional measure of evenness, the Pielou evenness index (*E*) (Pielou 1969) can be calculated. High evenness, which occurs when species are equal or virtually equal in abundance, is conventionally equated with high diversity (Magurran 1988).

Using data from this study, four of the most commonly used methods of expressing diversity (Clarke and Warwick 1994) in terms of species richness, equitability, or both, were compared across the three grazing treatments.
These methods of expressing diversity are species richness (S) (Magurran 1988), Margalef's index (D_{mg}) (Margalef 1958), the Shannon-Wiener diversity index (*H'*) (Shannon and Wiener 1963) and Pielou's evenness index (*E)* (Pielou 1969).

Species richness (S) provides an instantly comprehensible expression of diversity (Magurran 1988) and comprises of the number of species recorded in any given sample area.

The Margalef diversity index (D_{mg}) incorporates both species richness and evenness and is a measure of the number of species present (*S*) for a given number of individuals (*N*).
The Margalef diversity index formula in a general form is:
$D_{mg} = (S - 1) / \log N$

The Shannon-Wiener diversity index (*H'*) is based on the proportional abundance of species, taking into account both equitability and richness. The index value usually falls between 1.5 and 3.5.
The Shannon-Wiener formula in a general form is:
$H' = - \sum p_i \log p_i$

where p_i is the proportion of the total abundance arising from the ith species.

The Pielou evenness index (E) gives a value of the ratio of observed diversity to maximum diversity. Maximum diversity would be found where all species are equally abundant. This index is constrained between 0 and 1, 1 representing all species being equally abundant.

The Pielou evenness index (E) formula in a general form is:

$E = H' / \log S$

where H' is the diversity value calculated using the Shannon-Wiener formula.

7.7. Statistical analysis.

The null hypothesis tested for this study was that at the time of the survey and under the same environmental conditions, plant diversity and certain vegetation parameters measured across three different grazing treatments, would not differ significantly.

It was presumed that should the vegetation differ more than what could be expected on the basis of chance alone, and accepting that the only known difference between the sample areas was treatment, it would be reasonable to conclude that treatment was responsible for the observed differences.

Results were analysed using STATISTICA (2003) Data Analysis Software System (version 6) and Microsoft Excel. The mean was used as the measure of central tendency with standard deviation as the measure of variability. Parametric one-way analysis of variance (ANOVA) and a non-parametric alternative, the Kruskal-Wallace test, were used to test for statistically significant differences between the means of measured vegetation parameters across the three treatments.

The post hoc pairwise Bonferroni test and t tests were used to determine differences between two groups. The critical alpha level (α) for rejection of the null hypothesis was 0.05.

8. Results.

8.1. Phytodiversity.

71 plant species were recorded in the 12 x 2500m² sample plots across three treatments (Table 7). 58 species were recorded in the zero grazing control, 52 species were recorded in the non-selective grazing treatment and 53 species were recorded in the conventional grazing treatment. Taxonomic nomenclature follows Arnold and De Wet (1993).

Table 7 Plant species recorded across three treatments.

Species	Classification	Family	Growth form
Aizoaceae 81		Aizoaceae	Perennial woody shrub.
Aptosimum procumbens (var. procumbens)	(Lehm.) Steud.	Scrophulariaceae	Perennial shrub.
Aptosimum spinescens	(Thunb.) Weber.	Scrophulariaceae	Perennial spiny shrub.
Arctotis leiocarpa	Harv.	Asteraceae	Annual herb.
Aridaria sp. 98	N.E. Br.	Mesembryanthemaceae	Succulent.
Aristida congesta (subsp. congesta)	Roem & Schult.	Poaceae	Perennial grass (occasionally annual).
Aristida diffusa (subsp. diffusa)	Trin.	Poaceae	Perennial grass.
Asparagus cf. lignosus	(Burm. F.)	Asparagaceae	Perennial shrub.
Asparagus recurvispinus	(Oberm.) Fellingham.	Asparagaceae	Perennial spiny shrub.
Asparagus suaveolens	Sensu Jessop, non Burch.	Asparagaceae	Perennial spiny shrub.
Chenopodium cf. phillipsianum	Aell.	Chenopodiaceae	Annual herb.
Chenopodium mucronatum	Thunb.	Chenopodiaceae	Annual herb.
Chrysocoma ciliata	L.	Asteraceae	Perennial shrub.
Crassula muscosa	L.	Crassulaceae	Perennial woody shrub. (leaf succulent).
Dicoma capensis	Less.	Asteraceae	Perennial shrub.
Drosanthemum cf. lique	(N. E. Br.) Schwant.	Mesembryanthemaceae	Perennial woody shrub. (leaf succulent).
Duvalia sp. 51	Haw.	Asclepiadaceae	Succulent.

Table 7 *Continued*

Species	Classification	Family	Growth form
Eragrostis bergiana	(Kunth) Trin.	Poaceae	Perennial grass.
Eragrostis curvula	(Schrad.) Nees.	Poaceae	Perennial grass.
Eragrostis lehmanniana (var. lehmanniana)	Nees.	Poaceae	Perennial grass.
Eragrostis obtusa	Munro ex Fical. & Hiern.	Poaceae	Perennial grass.
Eriocephalus ericoides	(L. f.) Druce.	Asteraceae	Perennial woody shrub.
Eriocephalus spinescens	Burch.	Asteraceae	Perennial woody shrub.
Felicia muricata	(Thunb.) Nees.	Asteraceae	Perennial shrub.
Fingerhuthia africana	Lehm.	Poaceae	Perennial grass.
Galenia cf. secunda	(L. f.) Sond.	Aizoaceae	Perennial herb.
Gazania krebsiana	Less.	Asteraceae	Perennial herb.
Helichrysum sp. 47	Mill.	Asteraceae	Annual herb.
Hermannia vestita	Thunb.	Sterculiaceae	Perennial shrub.
Ifloga glomerata	(Harv.) Schltr.	Asteraceae	Annual herb.
Indigofera sp. 53	L.	Fabaceae	Annual herb.
Indigofera sp. 65	L.	Fabaceae	Annual herb.
Jamesbrittenia atropurpurea	(Benth.) Hilliard.	Scrophulariaceae	Perennial shrub.
Kedrostis cf. africana	(L.) Cogn.	Cucurbitaceae	Perennial shrub.
Lasiopogon glomerulatus	(Harv.) Hilliard.	Asteraceae	Annual herb.
Lebeckia spinescens	Harv.	Fabaceae	Perennial woody shrub.
Ledebouria sp. 54	Roth.	Hyacinthaceae	Succulent.
Lepidium africanum	(Burm.f.)	Brassicaceae	Annual herb.
Lessertia sp. 49	DC.	Fabaceae	Perennial herb.
Lessertia sp. 92	DC.	Fabaceae	Perennial herb.
Limeum aethiopicum	Burm.	Aizoaceae	Perennial woody shrub.
Lycium cinereum	Thunb. (Sens. Lat.)	Solanaceae	Perennial spiny shrub.
Melolobium sp. *Genus under revision.* 11	Eckl. & Zeyh.	Fabaceae	Perennial woody shrub.
Melolobium sp. *Genus under revision.* 57	Eckl. & Zeyh.	Fabaceae	Perennial woody shrub.

Table 7 *Continued*

Species	Classification	Family	Growth form
Mesembryanthemum guerichianum	Pax.	Mesembryanthemaceae	Succulent.
Microloma armatum	(Thunb.) Schltr.	Asclepiadaceae	Perennial woody shrub.
Monechma incanum	(Nees) C.B. Cl.	Acanthaceae	Perennial woody shrub.
Osteospermum calendulaceum	L. f.	Asteraceae	Annual herb.
Peliostomum leucorrhizum	E. May. ex Benth.	Scrophulariaceae	Perennial woody shrub.
Pentzia incana	(Thunb.) Kuntze.	Asteraceae	Perennial woody shrub.
Pentzia spinescens	Less.	Asteraceae	Perennial woody shrub.
Plinthus karooicus	Verdoorn.	Aizoaceae	Perennial woody shrub.
Polygala leptophylla	Burch.	Polygalaceae	Perennial shrub.
Psilocaulon coriarium	(Burch.) N.E. Br.	Mesembryanthemaceae	Succulent.
Psilocaulon junceum	(Haw.) Schwant.	Mesembryanthemaceae	Succulent.
Pteronia glauca	Thunb.	Asteraceae	Perennial woody shrub.
Rosenia humilis	(Less.) Bremer.	Asteraceae	Perennial woody shrub.
Rosenia oppositifolia	(D.C.) Bremer.	Asteraceae	Perennial woody shrub.
Ruschia ferox	(L. Bol.) L. Bol.	Mesembryanthemaceae	Perennial woody shrub. (leaf succulent).
Salsola cf. calluna	Fenzl. ex C.H. Wr.	Chenopodiaceae	Perennial woody shrub.
Salvia sp. 40	Codd.	Lamiaceae	Annual herb.
Senecio leptophyllus	DC.	Asteraceae	Perennial shrub.
Solanum tomentosum	L.	Solanaceae	Perennial shrub.
Stipagrostis ciliata. (var. capensis)	(Desf.) De Winter.	Poaceae	Perennial grass.
Stipagrostis obtusa	(Del.) Nees.	Poaceae	Perennial grass.
Tetragonia sarcophylla	Fenzl.	Aizoaceae	Perennial woody shrub.
Thesium hystrix	A. W. Hill.	Santalaceae	Perennial woody shrub.
Tragus koelerioides	Aschers.	Poaceae	Perennial grass.
Ursinia nana	D.C.	Asteraceae	Annual herb.
Zygophyllum lichtensteinianum	Cham. & Schlechtd.	Zygophyllaceae	Perennial woody shrub.
Zygophyllum microcarpum	Licht. ex Cham. & Schlechtd.	Zygophyllaceae	Perennial woody shrub.

Of the 71 species recorded for the three treatments, 40 species were common to all three treatments (Table 8).

Table 8 Species common to all three treatments. (* indicates species for which vegetation parameters were measured).

Species common to three treatments	
Aptosimum procumbens	*Lepidium africanum*
*Aptosimum spinescens**	*Lessertia sp. 49**
*Aristida congesta**	*Lycium cinereum**
*Aristida diffusa**	*Melolobium sp. 11**
Asparagus recurvispinus	*Osteospermum calendulaceum*
*Asparagus suaveolens**	*Peliostomum leucorrhizum*
*Chenopodium cf. phillipsianum**	*Pentzia incana**
*Chenopodium mucronatum**	*Pentzia spinescens*
*Chrysocoma ciliata**	*Plinthus karooicus*
*Crassula muscosa**	*Polygala leptophylla*
*Drosanthemum cf. lique**	*Psilocaulon coriarium*
Eragrostis bergiana	*Rosenia humilis**
*Eragrostis lehmanniana**	*Rosenia oppositifolia*
Eragrostis obtusa	*Ruschia ferox*
Eriocephalus ericoides	*Stipagrostis ciliata**
Fingerhuthia africana	*Stipagrostis obtusa**
*Galenia cf. secunda**	*Tetragonia sarcophylla*
Helichrysum sp. 47*	*Thesium hystrix**
*Ifloga glomerata**	*Zygophyllum lichtensteinianum*
*Lebeckia spinescens**	*Zygophyllum microcarpum*

Of the 31 species not common to all three treatments, 8 species were recorded only in the zero grazing treatment, 2 species were recorded only in the non-selective grazing treatment and 9 species were recorded only in the conventional grazing treatment.

Eight species occurred in both the zero grazing and non-selective grazing treatment, 2 species occurred in both the non-selective and conventional grazing treatments and 2 species occurred in both the zero grazing and conventional grazing treatments.
Thirteen species were not recorded in the zero grazing treatment, 19 species were not recorded in the non-selective grazing treatment and 18 species were not recorded in the conventional grazing treatment (Table 9).

Table 9 Species not common to all treatments, showing species shared between two treatments and species unique to a treatment. The letter indicates species occurring in those treatments only (Z = zero grazing, N = non-selective grazing, C = conventional grazing, ZN = Z and N, NC = N and C, ZC = Z and C).

Species unique to a treatment or occurring in two treatments					
N	*Jamesbrittenia atropurpurea*	Z	*Aizoaceae* sp. *81*	ZN	*Arctotis leiocarpa*
N	*Kedrostis cf. africana*	Z	*Dicoma capensis*	ZN	*Eragrostis curvula*
C	*Aridaria* sp. *98*	Z	*Eriocephalus spinescens*	ZN	*Indigofera* sp. *65*
C	*Asparagus cf. lignosus*	Z	*Gazania krebsiana*	ZN	*Lasiopogon glomerulatus*
C	*Duvalia sp. 51*	Z	*Lessertia* sp. *92*	ZN	*Microloma armatum*
C	*Felicia muricata*	Z	*Limeum aethiopicum*	ZN	*Salsola cf. calluna*
C	*Hermannia vestita*	Z	*Pteronia glauca*	ZN	*Salvia* sp. *40*
C	*Indigofera* sp. *53*	Z	*Solanum tomentosum*	ZN	*Tragus koelerioides*
C	*Ledebouria* sp. *54*	NC	*Mesembryanthemum guerichianum*	ZC	*Monechma incanum*
C	*Melolobium* sp. *57*	NC	*Senecio leptophyllus*	ZC	*Psilocaulon junceum*
C	*Ursinia nana*				

The sample plot layout used in this study consisted of an area in which species richness alone was recorded and an area in which species richness and other vegetation parameters were measured.

Fifty species occurred in the subplots in which vegetation parameters were measured (within $200m^2$ of each treatment). Of these 50 species, 22 species were common to all three treatments. Excluding *Chenopodium mucronatum* for which only seedlings were found, vegetation parameter measurements were recorded for 21 species and these were used to compare the impacts of different grazing systems on individual species in terms of canopy cover, density and height.

Mean canopy cover percentage and mean species density per hectare with standard deviation was calculated for species recorded in all three treatments (Table 10).
Mean canopy cover percentage and mean species density per hectare with standard deviation was calculated for species recorded in two of the three treatments (Table 11).
Mean plant height and standard deviation per species over the three treatments was also calculated (Table 12).
One way analysis of variance was conducted on species for which suitable data was available for statistical analysis.

Table 10 Mean plant canopy cover percentage and mean plant density per hectare with standard deviation (SD) for 21 species common to three treatments for which vegetation parameters were measured. The mean total canopy cover percentage and density indicated at the bottom of the table were calculated using all plants measured per treatment. One-way analysis of variance (ANOVA) was conducted on four species with suitable data available for statistical analysis and on the mean totals and this is shown in the (Sig) column. (NS = not significant, * = $p < 0.05$, ** = $p < 0.01$). Means with similar superscript letters do not differ significantly

Canopy cover percentage				
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aptosimum spinescens	0.188 ± 0.281	0.076 ± 0.152	0.939 ± 1.878	
Aristida congesta	0.073 ± 0.085	0.27 ± 0.318	0.004 ± 0.007	
Aristida diffusa	0.067 ± 0.135	0.094 ± 0.173	0.393 ± 0.643	
Asparagus suaveolens	0.025 ± 0.05	0.007 ± 0.014	0.583 ± 0.5	
Chenopodium cf. phillipsianum	0.02 ± 0.034	0.0003 ± 0.001	0.051 ± 0.102	
Chrysocoma ciliata	0.262 ± 0.524	0.165 ± 0.195	0.16 ± 0.278	
Crassula muscosa	0.005 ± 0.007	0.0001 ± 0.0002	0.014 ± 0.029	
Drosanthemum cf. lique	0.53 ± 0.97	0.011 ± 0.023	0.037 ± 0.075	
Eragrostis lehmanniana	15.455 ± 2.144	14.183 ± 3.619	14.716 ± 8.336	NS
Galenia cf. secunda	0.042 ± 0.083	0.025 ± 0.033	0.023 ± 0.027	
Helichrysum sp.47	0.005 ± 0.007	0.013 ± 0.022	0.003 ± 0.005	
Ifloga glomerata	0.003 ± 0.004	0.002 ± 0.001	0.005 ± 0.005	
Lebeckia spinescens	0.025 ± 0.051	0.223 ± 0.39	0.402 ± 0.51	
Lessertia sp.49	0.004 ± 0.007	0.02 ± 0.04	0.015 ± 0.031	
Lycium cinereum	0.74 ± 1.37	0.587 ± 1.175	0.741 ± 0.696	
Melolobium sp.11	0.002 ± 0.005	0.052 ± 0.055	0.055 ± 0.101	
Pentzia incana	18.664 ± 4.303	15.497 ± 4.52	10.723 ± 7.025	NS
Rosenia humilis	0.118 ± 0.157	0.789 ± 1.259	0.019 ± 0.038	
Stipagrostis cilita	1.144 ± 0.885	0.648 ± 0.558	1.639 ± 1.528	NS
Stipagrostis obtusa	0.758 ± 1.003	0.265 ± 0.447	1.162 ± 1.194	NS
Thesium hystrix	0.053 ± 0.106	0.153 ± 0.281	0.327 ± 0.404	
Mean total	38.182 ± 4.749	33.079 ± 5.732	32.011 ± 5.833	NS

Table 10 *continued*

	Density per hectare			
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aptosimum spinescens	250 ± 252	100 ± 200	900 ± 1800	
Aristida congesta	1300 ± 1793	5200 ± 6915	750 ± 1038	
Aristida diffusa	100 ± 200	150 ± 191	300 ± 346	
Asparagus suaveolens	100 ± 200	100 ± 200	950 ± 854	
Chenopodium cf. phillipsianum	250 ± 300	150 ± 300	150 ± 300	
Chrysocoma ciliata	200 ± 400	200 ± 231	250 ± 379	
Crassula muscosa	1950 ± 2323	100 ± 200	300 ± 600	
Drosanthemum cf. lique	2200 ± 3626	50 ± 100	150 ± 300	
Eragrostis lehmanniana	48400 ± 7605	56900 ± 14994	37550 ± 14578	NS
Galenia cf. secunda	650 ± 1300	400 ± 490	300 ± 346	
Helichrysum sp.47	600 ± 693	1850 ± 2500	1500 ± 2868	
Ifloga glomerata	3150 ± 5639	1950 ± 1436	5200 ± 6976	
Lebeckia spinescens	50 ± 100	200 ± 283	800 ± 966	
Lessertia sp.49	50 ± 100	200 ± 400	450 ± 900	
Lycium cinereum	200 ± 283	200 ± 400	250 ± 252	
Melolobium sp.11	50 ± 100	300 ± 258	250 ± 300	
Pentzia incana	20300[a] ± 4353	20500[a] ± 2392	10550[b] ± 6434	*
Rosenia humilis	150 ± 191	700 ± 808	50 ± 100	
Stipagrostis cilita	3900 ± 3380	2700 ± 2812	6150 ± 4509	NS
Stipagrostis obtusa	3600 ± 4977	2500 ± 4337	4650 ± 3272	NS
Thesium hystrix	250 ± 500	550 ± 971	200 ± 283	
Mean total	87700[a] ± 7070	95000[a] ± 9917	71650[b] ± 5981	**

Table 11 Mean canopy cover percentage per species and mean plant density per hectare with standard deviation (SD) for species for which measurements were taken in two treatments. (Species may have been recorded in three treatments but were only measured in two). Analysis of variance (ANOVA) was carried out on one species with suitable data and is shown in the (Sig) column. (NS = not significant).

	Canopy cover percentage			Density/ha		
	Non-selective	Conventional	Sig	Non-selective	Conventional	Sig
Species	Mean ± SD	Mean ± SD		Mean ± SD	Mean ± SD	
Aptosimum procumbens	0.001 ± 0.001	0.001 ± 0.002		100 ± 115	250 ± 500	
Asparagus recurvispinus	0.138 ± 0.275	0.124 ± 0.249		50 ± 100	50 ± 100	
Eriocephalus ericoides	0.061 ± 0.121	0.004 ± 0.009		50 ± 100	50 ± 100	
	Zero grazing	Conventional		Zero grazing	Conventional	
Species	Mean ± SD	Mean ± SD		Mean ± SD	Mean ± SD	
Lepidium africanum	0.0001 ± 0.0002	0.020 ± 0.039		50 ± 100	50 ± 100	
Plinthus karooicus	0.040 ± 0.080	0.037 ± 0.073		50 ± 100	50 ± 100	
Psilocaulon junceum	0.014 ± 0.028	0.010 ± 0.020		100 ± 200	150 ± 300	
	Zero grazing	Non-selective		Zero grazing	Non-selective	
Species	Mean ± SD	Mean ± SD		Mean ± SD	Mean ± SD	
Eragrostis obtusa	0.079 ± 0.102	0.079 ± 0.158		150 ± 191	400 ± 800	
Fingerhuthia africana	0.108 ± 0.092	0.289 ± 0.492		250 ± 252	1150 ± 1921	
Indigofera sp. 65	0.027 ± 0.053	0.011 ± 0.021		100 ± 200	100 ± 200	
Pentzia spinescens	0.315 ± 0.525	0.197 ± 0.394		450 ± 661	100 ± 200	
Ruschia ferox	0.028 ± 0.055	0.019 ± 0.023		50 ± 100	100 ± 115	
Salvia sp. 40	0.012 ± 0.016	0.003 ± 0.005	NS	2500 ± 2661	1550 ± 2715	NS
Tragus koelerioides	0.026 ± 0.051	0.160 ± 0.271		50 ± 100	650 ± 915	

Table 12 Mean plant height in centimeters and standard deviation (SD) per species. One-way analysis of variance (ANOVA) was conducted on four species with suitable data available for statistical analysis and is shown in the (Sig) column. (NS = not significant, * = $p < 0.05$). Means with similar superscript letters do not differ significantly.

	Plant height			
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aptosimum spinescens	7.2 ± 7.5	5.3 ± 10.5	3.5 ± 7.1	
Aristida congesta	10.0 ± 11.6	8.5 ± 9.9	2.1 ± 2.7	
Aristida diffusa	14.0 ± 28.0	27.1 ± 32.9	38.3 ± 28.2	
Asparagus suaveolens	6.3 ± 12.5	4.6 ± 9.3	19.4 ± 13.0	
Chenopodium cf. phillipsianum	8.5 ± 9.8	7.8 ± 15.7	7.7 ± 15.3	
Chrysocoma ciliata	6.8 ± 13.5	12.9 ± 15.4	12.4 ± 15.4	
Crassula muscosa	2.2 ± 3.4	0.6 ± 1.3	2.0 ± 3.9	
Drosanthemum cf. lique	14.5 ± 9.8	2.0 ± 4.0	4.9 ± 9.8	
Eragrostis lehmanniana	43.6 ± 1.3	39.2 ± 3.8	43.0 ± 5.2	NS
Galenia cf. secunda	0.8 ± 1.6	1.8 ± 2.3	2.0 ± 2.3	
Helichrysum sp. 47	0.9 ± 0.6	0.9 ± 0.6	0.5 ± 0.6	
Ifloga glomerata	0.8 ± 0.5	1.0 ± 0.0	1.1 ± 1.0	
Lebeckia spinescens	4.5 ± 9.0	9.1 ± 10.6	14.4 ± 9.8	
Lessertia sp. 49	4.3 ± 8.5	3.6 ± 7.3	3.3 ± 6.5	
Lycium cinereum	15.1 ± 20.5	11.4 ± 22.9	29.8 ± 24.6	
Melolobium sp. 11	2.5 ± 5.0	13.8 ± 10.0	7.5 ± 9.2	
Pentzia incana	22.1^{a} ± 2.1	18.3^{b} ± 2.3	22.2^{a} ± 1.6	*
Rosenia humilis	15.0 ± 17.4	16.4 ± 11.2	4.0 ± 8.0	
Stipagrostis cilita	39.5 ± 26.7	37.1 ± 26.1	39.2 ± 5.7	NS
Stipagrostis obtusa	29.4 ± 3.6	25.5 ± 17.0	26.6 ± 1.0	NS
Thesium hystrix	4.3 ± 8.5	5.5 ± 6.4	9.4 ± 11.0	

Table 13 Mean plant canopy cover percentage and mean plant density per hectare with standard deviation. Results of analysis of variance are indicated in the (Sig) column. NS = not significant. ** = $p < 0.01$. (The totals differ from the values in table 10, as species not common to all three treatments are included here). Means with similar superscript letters do not differ significantly.

Canopy cover percentage and density per hectare				
	Zero grazing Mean ± SD	Non-selective Mean ± SD	Conventional Mean ± SD	Sig
Total cover %	38.91 ± 4.71	34.04 ± 6.19	33.03 ± 4.75	NS
Total density/ha	91600 ^{a} ± 8104	99400 ^{a} ± 12616	72900 ^{b} ± 5821	**

Seedlings were recorded either by species or as unidentified seedlings and were grouped by proximity to an adult plant, either occurring within close proximity to, or away from an adult plant. Grass seedlings were recorded separately.

The mean seedling density per m² for each treatment, seedling proximity to adult plant and standard deviation is shown in Table 14. The total mean seedling density per m² shown in this table excludes grass seedlings.

Table 14 Mean seedling density per m² and standard deviation, showing seedlings recorded close to adult plants (SC) seedlings recorded away from adult plants (SA), grass seedlings (GS), unidentified seedlings (US), *Galenia cf. secunda* seedlings (G), *Chenopodium mucronatum* seedlings (C) and total seedlings excluding grass seedlings (T). Results of analysis of variance are indicated in the (Sig) column. NS = not significant.

	Seedlings			
	Zero grazing	Non-selective	Conventional	
	Mean ± SD	Mean ± SD	Mean ± SD	Sig
SC	2.07 ± 2.72	1.29 ± 1.96	1.43 ± 0.68	NS
SA	2.53 ± 3.65	1.69 ± 2.17	1.61 ± 1.37	NS
GS	4.07 ± 3.60	5.34 ± 4.04	8.86 ± 7.28	NS
US	0.63 ± 0.57	0.33 ± 0.16	0.60 ± 0.60	NS
G	3.06 ± 4.49	1.93 ± 3.03	1.88 ± 2.30	NS
C	0.91 ± 1.33	0.73 ± 1.04	0.56 ± 0.19	NS
T	4.60 ± 6.36	2.98 ± 4.12	3.04 ± 2.04	NS

Table 15 Grass species, mean percentage canopy cover, mean density per hectare and grass proportion of total canopy cover and density (Grass % of total) with standard deviations. Analysis of variance (ANOVA) was conducted on three species and on the total grass proportion of total plant cover and density and this is shown in the (Sig) column. NS = not significant, * = $p < 0.05$. Means with similar superscript letters do not differ significantly.

	Canopy cover percentage			
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aristida congesta	0.073 ± 0.085	0.27 ± 0.318	0.004 ± 0.007	
Aristida diffusa	0.067 ± 0.135	0.094 ± 0.173	0.393 ± 0.643	
Eragrostis bergiana	0.043 ± 0.086	0 ± 0	0 ± 0	
Eragrostis lehmanniana	15.455 ± 2.144	14.183 ± 3.619	14.716 ± 8.336	NS
Eragrostis obtusa	0.079 ± 0.102	0.079 ± 0.158	0 ± 0	
Fingerhuthia africana	0.108 ± 0.092	0.288 ± 0.492	0 ± 0	
Stipagrostis cilita	1.144 ± 0.885	0.648 ± 0.558	1.639 ± 1.528	NS
Stipagrostis obtusa	0.758 ± 1.003	0.265 ± 0.447	1.162 ± 1.194	NS
Tragus koelerioides	0.026 ± 0.051	0.16 ± 0.271	0 ± 0	
Total mean	17.754 ± 3.11	15.987 ± 4.217	17.914 ± 7.639	NS
Grass % of total	45.62 ± 6.84	46.96 8.6	54.24 ± 22.95	NS
	Density per hectare			
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aristida congesta	1300 ± 1793	5200 ± 6915	750 ± 1038	
Aristida diffusa	100 ± 200	150 ± 191	300 ± 346	
Eragrostis bergiana	50 ± 100	0 ± 0	0 ± 0	
Eragrostis lehmanniana	48400 ± 7605	56900 ± 14994	37550 ± 14578	NS
Eragrostis obtusa	150 ± 191	400 ± 800	0 ± 0	
Fingerhuthia africana	250 ± 252	1150 ± 1921	0 ± 0	
Stipagrostis cilita	3900 ± 3380	2700 ± 2812	6150 ± 4509	NS
Stipagrostis obtusa	3600 ± 4977	2500 ± 4337	4650 ± 3272	NS
Tragus koelerioides	50 ± 100	650 ± 915	0 ± 0	
Total mean	57800 ^{a} ± 7231	69650 ^{a} ± 9725	49400 ^{b} ± 11493	*
Grass % of total	63.1 ± 7.18	70.07 ± 1.97	67.76 ± 11.66	NS

The relative contribution of species to the total plant composition of each treatment was evaluated by calculating an importance value for each species. The mean importance value with standard deviation for species common to all three treatments or occurring in two treatments is shown in Table 16.

The importance value was expressed as a total of the values for relative density, frequency and cover for each species (Smith 1990). This value is expressed as a percentage.
Frequency was calculated in terms of species occurrence in the 50 x $1m^2$ subplots per replicate. This made it possible to calculate the mean importance value per species for each replicate in order to conduct analysis of variance (ANOVA).
No significant differences were found for the importance values of individual species between grazing treatments.

Table 16 Mean importance value percentages with standard deviations. Results of analysis of variance are indicated in the (Sig) column. NS = not significant.

	Importance value percentage		
	Zero grazing	Non-selective	Conventional	
Species	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Aptosimum spinescens	3.24 ± 3.02	1.32 ± 2.64	11.64 ± 23.28	NS
Aristida congesta	10.67 ± 15.33	24.65 ± 32.62	6.38 ± 9.04	NS
Aristida diffusa	0.79 ± 1.57	1.88 ± 2.49	4.73 ± 6.26	NS
Asparagus suaveolens	1.17 ± 2.34	0.61 ± 1.22	9.11 ± 7.64	NS
Chenopodium cf. phillipsianum	2.33 ± 3.21	1.66 ± 3.31	1.33 ± 2.67	NS
Chrysocoma ciliata	2.31 ± 4.61	2.13 ± 2.61	2.61 ± 3.73	NS
Crassula muscosa	4.71 ± 5.56	0.59 ± 1.18	2.65 ± 5.30	NS
Drosanthemum cf. lique	18.12 ± 28.75	0.59 ± 1.18	1.66 ± 3.32	NS
Eragrostis lehmanniana	190.30 ± 7.35	198.31 ± 13.77	180.95 ± 54.36	NS
Galenia cf. secunda	4.27 ± 8.54	4.03 ± 5.35	2.81 ± 3.41	NS
Helichrysum sp.	5.68 ± 5.79	17.75 ± 23.61	9.66 ± 17.85	NS
Ifloga glomerata	8.70 ± 10.39	9.58 ± 6.49	23.22 ± 20.05	NS
Lebeckia spinescens	0.62 ± 1.24	2.84 ± 4.20	9.19 ± 10.35	NS
Lessertia sp.49	0.56 ± 1.13	2.23 ± 4.46	4.20 ± 8.41	NS

Table 16 *continued*

Species	Zero grazing Mean ± SD	Non-selective Mean ± SD	Conventional Mean ± SD	Sig
Lycium cinereum	4.26 ± 6.82	3.86 ± 7.73	4.57 ± 3.06	NS
Melolobium sp 11	0.56 ± 1.12	3.43 ± 2.96	2.88 ± 3.57	NS
Pentzia incana	156.75 ± 18.74	154.61 ± 7.57	96.70 ± 59.24	NS
Rosenia humilis	1.97 ± 2.54	10.17 ± 12.45	0.63 ± 1.25	NS
Stipagrostis cilita	36.66 ± 30.40	23.74 ± 22.62	50.48 ± 33.63	NS
Stipagrostis obtusa	25.27 ± 27.33	21.19 ± 36.20	35.38 ± 21.43	NS
Thesium hystrix	2.97 ± 5.93	5.51 ± 9.53	3.02 ± 4.05	NS
	Zero grazing	Non-selective		
Eragrostis obtusa	1.87 ± 2.37	3.56 ± 7.12		NS
Fingerhuthia africana	3.06 ± 2.88	10.32 ± 17.42		NS
Indigofera sp.65	0.20 ± 0.40	1.12 ± 2.24		NS
Pentzia spinescens	3.78 ± 5.76	1.59 ± 3.17		NS
Ruschia ferox	0.62 ± 1.24	1.15 ± 1.33		NS
Salvia sp.	16.67 ± 17.14	11.92 ± 20.91		NS
Tragus koelerioides	0.62 ± 1.23	6.72 ± 9.09		NS
	Zero grazing	Conventional		
Lepidium africanum	0.55 ± 1.1	0.64 ± 1.28		NS
Plinthus karooicus	0.18 ± 0.36	0.7 ± 1.4		NS
Psilocaulon junceum	1.14 ± 2.28	1.73 ± 3.45		NS
	Non-selective	Conventional		
Aptosimum procumbens	2.61 ± 2.49	2.60 ± 5.19		NS
Asparagus recurvispinus	0.94 ± 1.88	0.98 ± 1.95		NS
Eriocephalus ericoides	0.72 ± 1.45	0.59 ± 1.17		NS

Species were allocated to one of the following broad categories based on major growth form and life history: annual herbs; perennial herbs; annual grasses; perennial grasses; perennial shrubs; perennial spiny shrubs; perennial woody shrubs; leaf succulent woody shrubs; succulents (all other succulents not falling into the previous group) and plant size (based on canopy cover spread as an indication of above-ground biomass).

Plants grouped by mean canopy size were allocated to one of four categories based on the calculated canopy cover area using the formula $\pi LW/4$. Mean canopy cover area greater than 1000cm²; between 500cm² and 1000cm²; between 100cm² and 500cm²; less than 100cm². These size groups exclude grasses (Table 17).

Table 17 Major plant growth form categories with abundance in terms of mean canopy cover percentage and density per hectare with standard deviations. Size = mean plant canopy cover area in cm² (excluding grasses) calculated using the formula $\pi LW/4$. Results of analysis of variance are indicated in the (Sig) column. * = $p < 0.05$. NS = not significant. Means with similar superscript letters do not differ significantly.

Growth form categories				
	Canopy cover percentage			
	Zero grazing	Non-selective	Conventional	
	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Annual herbs	0.07 ± 0.057	0.03 ± 0.031	0.09 ± 0.087	NS
Perennial herbs	0.05 ± 0.081	0.05 ± 0.037	0.04 ± 0.027	NS
Perennial grasses	17.76 ± 3.111	15.99 ± 4.217	17.92 ± 7.639	NS
Perennial shrubs	0.26 ± 0.524	0.17 ± 0.195	0.18 ± 0.268	NS
Perennial spiny shrubs	0.95 ± 1.315	0.81 ± 1.597	2.39 ± 1.428	NS
Perennial woody shrubs	19.26 ± 4.406	16.98 ± 4.153	11.88 ± 7.656	NS
Perennial woody shrubs (succulent leaves)	0.56 ± 1.032	0.03 ± 0.036	0.05 ± 0.103	NS
Other succulents	0.02 ± 0.028	0.00 ± 0.000	0.49 ± 0.825	NS
Size > 1000cm²	0.74 ± 1.370	0.00 ± 0.000	1.17 ± 1.340	NS
Size = 501 - 1000cm²	18.85 ± 4.467	17.01 ± 3.755	11.49 ± 7.241	NS
Size = 100 - 500cm²	1.36 ± 1.345	0.93 ± 1.076	2.17 ± 1.460	NS
Size < 100cm²	0.21 ± 0.206	0.12 ± 0.105	0.29 ± 0.112	NS

Table 17 *continued*

	Density per hectare			
	Zero grazing	Non-selective	Conventional	
	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Annual herbs	6650 ± 7496	5700 ± 3789	7100 ± 7326	NS
Perennial herbs	700 ± 1270	600 ± 432	750 ± 755	NS
Perennial grasses	57800 [a] ± 7231	69650 [a] ± 9725	49400 [b] ± 11493	*
Perennial shrubs	200 ± 400	200 ± 231	450 ± 252	NS
Perennial spiny shrubs	550 ± 412	450 ± 661	2150 ± 1464	NS
Perennial woody shrubs	21400 ± 5177	22550 ± 3070	12300 ± 7377	NS
Perennial woody shrubs (succulent leaves)	4200 ± 5650	250 ± 300	450 ± 900	NS
Other succulents	100 ± 200	0 ± 0	300 ± 258	NS
Size > $1000cm^2$	200 ± 283	0 ± 0	300 ± 258	NS
Size = 501 - $1000cm^2$	20550[a] ± 4506	21450[a] ± 2125	10900[b] ± 6574	*
Size = 100 - $500cm^2$	3250 ± 4110	1500 ± 1747	3000 ± 1532	NS
Size < $100cm^2$	9800 ± 7230	6800 ± 4421	9300 ± 6734	NS

	Mean plant size per group (canopy cover area in cm^2)						
	Zero grazing		Non-selective		Conventional		
	n	Mean ± SD	n	Mean ± SD	n	Mean ± SD	Sig
Size > $1000cm^2$	1	1375 ± 2221	0	0 ± 0	2	2792 ± 935	NS
Size = 501 - $1000cm^2$	2	879 ± 98	4	713 ± 36	4	810 ± 142	NS
Size = 100 - $500cm^2$	9	461 ± 102	7	296 ± 158	6	307 ± 114	NS
Size < $100cm^2$	13	24 ± 23	15	19 ± 26	19	36 ± 35	NS

n = number of species per size group.

8.2. Diversity indices.

Mean species richness over the increasing sample plot area in each treatment was calculated using the species richness recorded in the $1m^2$, $10m^2$, $20m^2$, $100m^2$, $850m^2$, $1750m^2$ and $2500m^2$ subplots of each replicate.

No significant difference was found between the treatments in terms of species richness over increased sample area (Table 18).

Table 18 Mean species richness for increasing sample plot size over three treatments with standard deviation and analysis of variance (ANOVA) shown in (Sig) column. NS = not significant.

	Species richness for increasing sample area			
	Zero grazing	Non-selective	Conventional	
	Mean ± SD	Mean ± SD	Mean ± SD	Sig
1 m^2	4 ± 1.63	3.5 ± 1.29	4.5 ± 1.29	NS
10 m^2	9.25 ± 1.26	8.25 ± 1.26	9.75 ± 1.50	NS
20 m^2	12.25 ± 0.96	15 ± 3.37	15 ± 0.82	NS
100 m^2	15 ± 0.82	16 ± 2.71	17 ± 0.00	NS
850 m^2	29.25 ± 4.43	25.5 ± 4.12	23 ± 0.82	NS
1750 m^2	30.75 ± 5.50	28.75 ± 4.57	26.5 ± 1.29	NS
2500 m^2	32.75 ± 4.27	31.75 ± 5.32	29.25 ± 1.71	NS

The various methods of determining the diversity indices listed below (Table 19) have been discussed in the chapter on methods.

The results show no significant statistical difference between any of the diversity indices determined for each treatment.

Table 19 Species richness (SR); Margalef species richness indices (*M*); Shannon-Wiener diversity indices (*H'*) and Pielou evenness indices (*E*) with cover and density as the measures of abundance showing means and standard deviations for each treatment. Results of analysis of variance are indicated in the (Sig) column. NS = not significant.

Diversity indices				
	Zero grazing	Non-selective	Conventional	
Index	Mean ± SD	Mean ± SD	Mean ± SD	Sig
SR	32.75 ± 4.272	31.75 ± 5.315	29.25 ± 1.708	NS
M	2.45 ± 0.409	2.53 ± 0.713	2.59 ± 0.465	NS
H' cover	1.174 ± 0.209	1.184 ± 0.163	1.332 ± 0.292	NS
H' density	1.453 ± 0.266	1.383 ± 0.304	1.568 ± 0.404	NS
E cover	0.341 ± 0.068	0.346 ± 0.037	0.399 ± 0.087	NS
E density	0.422 ± 0.085	0.406 ± 0.091	0.469 ± 0.117	NS

A summary of statistically significant results is shown in Table 20.

Table 20 Summary of significant results. Results of analysis of variance (ANOVA) are indicated in the (Sig) column.

$* = p < 0.05$. $** = p < 0.01$.

Means with similar superscript letters do not differ significantly.

Summary of significant analysis of variance (ANOVA) results				
	Zero grazing	Non-selective	Conventional	
	Mean ± SD	Mean ± SD	Mean ± SD	Sig
Pentzia incana height	22.1^{a} ± 2.1	18.3^{b} ± 2.3	22.2^{a} ± 1.6	*
Pentzia incana density/ha	20300^{a} ± 4353	20500^{a} ± 2392	10550^{b} ± 6434	*
Mean total plant density/ha	87700^{a} ± 7070	95000^{a} ± 9917	71650^{b} ± 5981	**
Plant size 500-1000cm² density/ha	20550^{a} ± 4506	21450^{a} ± 2125	10900^{b} ± 6574	*
Mean total grass density/ha	57800^{a} ± 7231	69650^{a} ± 9725	49400^{b} ± 11493	*
% ground covered by prostrate dead plant material	18.5^{a} ± 8	21^{a} ± 7.44	34^{b} ± 4.42	*

9. Discussion.

The Shannon-Wiener method, which is based on species richness and the proportional abundances of species, is sensitive to changes in the number of species and to the distribution of the number of individuals among the species. The incorporation of the proportional abundances of species into the index is important for understanding the heterogeneity of species abundances in the karoo. The diversity of species and the abundance (linked to seed availability) of karoo species, plays an important role in the response of vegetation to major driving forces influencing change, through increasing resilience and allowing for more rapid recovery after disturbance such as drought or grazing (Milton and Dean 1999a). This is important as regards the capacity of karoo vegetation to sustain animal production.

The results obtained from accurate abundance data input into the formula, such as the data collected for this study, provide a valuable indication of species and abundance diversity in a single value for sample areas.

The Modified-Whittaker sampling method is designed for the recording of species richness per sample area, with an estimate of abundance. The sampling method used in the study further modified the Modified-Whittaker method (Stohlgren *et al.* 1995) by increasing the area sampled for species richness and by using small 0.5m x 2m subplots in which to record various vegetation parameters. The increased area, as expected, resulted in an increase in the number of species recorded per sample area.

As the emphasis of this study was on the effects of grazing on plant diversity, the use of the small subplots allowed the recorder to scrutinize each subplot and obtain accurate measurements of vegetation parameters needed. Focus on the small subplot area also ensured that the chances of not recording smaller and hidden species, were minimized.

The trade-off in obtaining these accurate species, density and cover measurements, is that it is more time consuming than abundance estimate methods, and fewer plots can be sampled over the same time period.

The results showed that in terms of species richness, the evenness of species abundances and combinations of both richness and evenness, there were no significant differences between the zero grazing controls, the non-selective grazing treatment and the conventional grazing treatment.

This can be expected considering that unpredictable and variable rainfall and major disturbance events such as droughts are the main drivers of change in the composition and abundance of karoo vegetation (O'Connor and Roux 1995). Pulses of mortality and population renewal occur at intervals of years, decades or centuries (Milton *et al.* 1999) and the influence of grazing on vegetation may only become evident in long-lived plants over a long period of time (Palmer *et al.* 1999). The grassy Nama Karoo vegetation of the study area also has a high resistance to grazing disturbances (Beukes and Cowling 1999), possibly as a result of the evolutionary history under which it developed through severe but irregular defoliation by large herds of migrating wild ungulates (Skead 1982).

Although changes in karoo vegetation composition and abundance are difficult to predict (Jeltsch *et al.* 1999) the results from this study are positive for range managers using the non-selective and conventional grazing systems. Neither of the grazing systems is shown to significantly influence phytodiversity or canopy cover percentage adversely compared to the other. Results showing comparatively lower plant diversity due to a particular grazing system could have negative implications in terms of vegetation resilience to future major disturbance events such as droughts and in terms of the availability of suitable livestock forage.

The fact that no difference was found in canopy cover percentage between the treatments is also positive for range managers in terms of protection against water run-off and soil erosion (Snyman 1999).

Plant diversity and canopy cover percentages do not differ significantly between the grazing treatments at this stage, but there may be differences occurring in the turnover rate of plants, influencing the demographic structure of the vegetation in terms of plant age and density. Results show that the mean total density of plants per hectare differs significantly between the treatments, with fewer plants per hectare in the conventional grazing treatment compared to the zero and non-selective grazing treatments.

Specific groups of plants, and one species, *Pentzia incana* also differ significantly in terms of density but no individual species or groups of plants, show significant canopy cover differences between the three treatments.

This finding indicates a difference, although not significant, in the canopy size of grasses and shrubs between the treatments. There seems to be a shift in the vegetation of the conventional grazing treatment towards larger (in terms of canopy cover size) but fewer (in terms of density per hectare) grasses and shrubs, with the zero grazing and non-selective grazing treatments consisting of smaller, but more, grasses and shrubs. More specifically, the density of the group of shrubs with canopy cover sizes of between $500cm^2$ and $1000cm^2$ and the perennial grass group, differed significantly between the conventional and both the zero and non-selective grazing treatments, with fewer perennial grasses and shrubs being found on the conventional grazing treatment. The density of perennial grasses and the shrub group was not significantly different between the zero and non-selective grazing treatments.

The difference in densities of the shrub size group between treatments was probably influenced by the fact that many of the *Pentzia incana* individuals recorded, fell into this size category across all three treatments. *Pentzia incana* was found to be the dominant shrub species in terms of density per hectare, across all three treatments, with the perennial grass *Eragrostis lehmanniana* being the dominant grass. *Pentzia incana* was the only individual species that showed a significant difference in density between the treatments.

Pentzia incana was also the only species that showed a significant difference in plant height across the treatments, being shorter in the non-selective grazing treatment than in the zero and conventional grazing treatments.

Pentzia incana is a vital part of domestic stock diet in the karoo (Le Roux *et al.* 1994), and makes up the staple feed on most farms (Shearing 1994).

The difference in *Pentzia incana* height can possibly be explained by the fact that there are more young plants in the non-selective grazing treatment and the compensatory growth associated with the higher intensity grazing system is influencing canopy size (Beukes and Cowling 1999) but not the plant height.

Although statistically significant, *Pentzia incana* individuals were on average only four centimeters shorter in the non-selective grazing treatment than in the zero grazing and conventional grazing treatments.

Differences in the density of certain key species and plant groups, although not significant, were observed between the treatments and these were more pronounced than differences in canopy cover spread. This may be indicating shifts in the density of these species and groups.

For example, the density of plants in the perennial spiny shrub group is higher in the conventional grazing treatment than in the zero and non-selective grazing treatment.

Although analysis of the leaf succulent group of plants across the three treatments showed no significant differences in density or canopy cover percentage, it is noticeable that the densities of the leaf succulents *Crassula muscosa* and *Drosanthemum cf. lique* are lower in the non-selective treatment and the conventional grazing treatments than in the zero grazing controls. Because leaf succulents are generally shorter-lived than other woody shrubs they may be more sensitive to grazing (Cowling *et al.* 1994) and therefore good indicators of the effects of grazing on vegetation. There is an absence of plants of the succulent group in the non-selective grazing treatment.

In terms of importance value, expressed as a total of the percentages of relative density, frequency and cover, the value for *Aristida congesta*, with a low grazing index value (Du Toit 1995), is higher than the values for *Stipagrostis ciliata* and *Stipagrostis obtusa* (both important in terms of grazing index values) in the non-selective grazing treatment. *Aristida congesta* also has a higher importance value in the non-selective grazing treatment than in both the zero grazing controls and the conventional grazing treatment.

Eragrostis obtusa (also a low grazing index value grass) shows a higher importance value in the non-selective grazing treatment than in the zero grazing treatment and is not present in the conventional grazing treatment.

Due to the difficulty in predicting event driven vegetation change in the karoo, and because of the long time scales involved and the lack of significant results from this study, it is not possible to determine clearly at this stage whether spiny plants, succulent leafed plants, succulent plants, or less favorable grasses are in fact increasing or decreasing in abundance across the three treatments.

Shifts towards the increase or decrease in density of these plants would however influence species diversity in terms of local extinctions and reduced abundances, and range management in terms of forage availability for livestock.

The shifts may result in an increase in the cover of less palatable species (Hoffman *et al.* 1999), reduce the fitness of palatable plant species to the advantage of unpalatable species (O'Connor 1991) and increase the relative abundance of defended and ephemeral plants (Hoffman and Cowling 1990).

Although not significant, the differences in certain grass densities may be indicating an increase in the abundance of less favorable (in terms of grazing value index) grass species in the more disturbed non-selective grazing treatment. An increase in abundance of *Aristida congesta* and *Eragrostis obtusa* could be indicative of overgrazed or disturbed rangeland (Van Oudtshoorn 1992).

Species that were found only in a particular treatment did not indicate much in terms of the effects of grazing on plant diversity. Most of these species were recorded as present in the sample plots only with no abundance data recorded. It was not possible to identify any particular group of plants amongst these species, based on growth form and life history, which were present in any one grazing treatment.

As previously stated, the difference in density and the similarity in canopy cover of the three grazing treatments, indicates that the plants in the conventional grazing treatment are larger and less than in the zero grazing controls and the non-selective grazing treatment.

Plant growth is mainly determined by common factors such as age or climate (Wiegand *et al.* 2000) and if individual site factors such as nutrient availability or competition are less important, the characteristic shape of plant size distribution should be maintained. The larger canopy sizes of the shrubs and grasses of the conventional grazing system may therefore indicate that these plants are older than the plants in the zero grazing controls and the non-selective grazing treatment.

The significantly higher percentage of prostrate dead plant material in the conventional grazing treatment may also be an indication of older vegetation in that treatment, with the older shrubs with larger canopy sizes having higher above-ground biomass (Du Toit 2001). Very little information is available on the longevity and growth rates of even the most common shrub species in karoo plant communities (Wiegand *et al.* 2000). However, if canopy size were linked to plant age for the purpose of understanding the age structure of plants across the three treatments, results from this study would indicate that the non-selective grazing treatment has a higher percentage of younger plants than the zero and conventional grazing treatments.
To understand the size-age distribution of plants across the three treatments better, the percentage each canopy size group of *Pentzia incana* (from less than 100cm² to greater than 6000cm²) contributed to the total density of *Pentzia incana* was calculated (Figure 7).

Pentzia incana was used as an example of the size-age distribution of shrubs in each treatment for three reasons. The high density of *Pentzia incana* recorded across all three treatments, the significant difference in *Pentzia incana* density found between the treatments and because *Pentzia incana* is a relatively long-lived perennial shrub (Esler 1999) that is well utilised by domestic livestock (Le Roux *et al.* 1994).

Figure 7 indicates that the age distribution in the non-selective grazing treatment is shifted in favour of younger plants compared to the conventional grazing treatment and to a lesser degree the zero grazing controls.

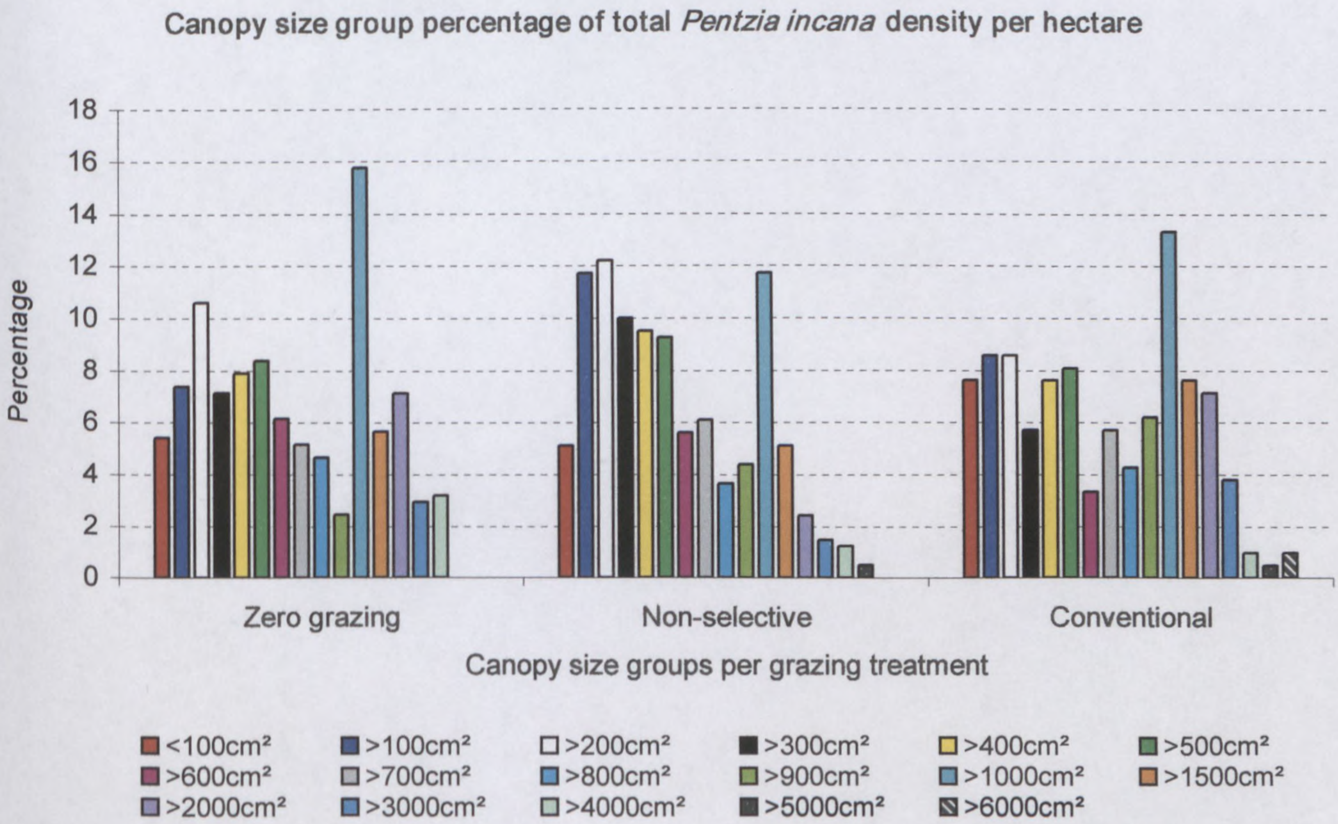


Figure 7 Canopy size groups of *Pentzia incana* (canopy area in cm²) as a percentage of the total *Pentzia incana* density per hectare for each grazing treatment.

Although the turnover of plant populations in the karoo is not well understood it is critical in understanding the regeneration phase of karoo vegetation (Esler 1999). The shift towards younger plants in the non-selective grazing treatments indicates a faster turnover or death-life cycle of plants and this can result in a vegetation shift towards a more desirable state, depending on seed availability and rainfall (Milton and Hoffman 1994).

For example, in communities with longer-lived plants there is a lag phase before shrubs die and make space for other species (Milton *et al.* 1995). During this lag phase or slow plant turnover, the amount of available crude protein in younger forage plants decreases as the plant becomes older, resulting in less nutritional forage available to domestic livestock over time (Meissner *et al.* 1999). Young forage plants may also be more productive than older, moribund plants (Milton 1992).

This would mean that more palatable and nutritious forage would be available over time in the non-selective grazing treatment than in the conventional grazing treatment with its slower plant turnover rate and greater percentage of older plants. In the karoo, plant life span, competitive interactions between species in the establishment phase, and between these species and established shrubs, and gap availability are major determinants of community dynamics (Midgley and van der Heyden 1999).

It is interesting to find that the zero grazing controls, from which grazing has been excluded since 1995, a period of approximately six years of resting before this study, do not significantly differ from the non-selective grazing treatment in terms of canopy cover percentage and plant density. Before grazing was removed the exclosures were subjected to a non-selective grazing system. Considering the long time scales involved in vegetation change and the persistence of grazing tolerant perennial plants in the karoo (Beukes 1999), the relatively short time scale involved here may not allow for measurable shifts in diversity.

Canopy cover percentage in the zero grazing controls is slightly higher than in the non-selective grazing treatment. Compensatory growth of plants such as *Pentzia incana* and *Eragrostis lehmanniana* after non-selective grazing (Beukes and Cowling 1999) can probably explain why there is no significant difference in canopy cover percentage between the two treatments.

Under the non-selective grazing system stock has been able to utilize forage without effecting vegetation composition and abundance differently to the zero grazing controls.

This indicates that the vegetation is resilient to grazing and such long rest periods for vegetation recovery may not be necessary. The loss of grazing by resting for such long period (six years) is not justified by any benefits for plant diversity.

In the longer term, the exclusion of grazing from the controls will probably result in the vegetation becoming more moribund, loosing nutritional value and palatability. Although seed production may not be reduced in the zero grazing treatment, the competition from older established plants would limit the development of plants beyond the seedling stage.

The percentage of prostrate dead plant material was found to be significantly higher in the conventional grazing treatment than in the zero grazing and non-selective grazing treatments. This is probably a result of the larger, older plants with higher aboveground biomass found in this treatment. The litter protects against topsoil erosion and serves as microsite refuges for seedling recruitment. It also plays an important role in soil microbial activity (Jackson and Caldwell 1993) and water infiltration (Tongway 1994).

These positive consequences for ecosystem processes on the conventional grazing treatment due to the high percentage of litter may influences the establishment of seedlings. However, high levels of competition from large established plants, with few deaths occurring and few gaps created, will limit plant development past the seedling stage.
Milton *et al.* (1999) found no increase in plant density five years after the addition of plant litter to undisturbed vegetation, but seedling survival was improved by the death or removal of established neighbouring plants, which prolonged the soil moisture content (Milton 1995).

No significant differences were found in the density of seedlings per m^2 between the treatments. The shift in vegetation demographics of the non-selective grazing treatment towards more young plants, is probably the result of a greater establishment success of plants beyond the seedling stage compared to the conventional grazing treatment. Beukes (1999) found that non-selective grazing increased the germination and emergence of seedlings.

Key processes driving the dynamics of the karoo vegetation are the rare recruitment events (Milton and Hoffman 1994) that can determine the composition of karoo vegetation for many years (Jeltsch *et al.* 1999). The timing and amount of rainfall which influences seed production and seedling recruitment and establishment (Esler 1999) drive these major establishment events.

The most crucial stage in the life cycle of plants in arid areas is establishment, during which stage mortality is high because of the harsh environment (Esler 1999). For seeds to germinate and the seedlings to develop past the seedling stage, a succession of environmental factors is necessary (van Rooyen 1999). The most limiting of these factors is moisture availability, directly related to the timing and the amount of rainfall (Esler 1999). The availability of suitable establishment microsites is also an important factor influencing the successful establishment of seedlings (Esler 1999) with changes in establishment sites by herbivores being a mechanism for change in the abundance and structure of karoo vegetation (Milton and Dean 1990). Dean and Milton (1991) found seedling survival to be greater in disturbed or nutrient-enriched microsites and Milton (1992) found that grazing by domestic and wild animals reduced the size but not the survival of palatable perennial seedlings.

Competition from other plants is another major determinant of seedling development (Midgley and van der Heyden 1999) with seedling survival and growth being reduced by competition from neighbouring plants (Milton 1992; Esler 1993).

In the conventional grazing treatment, competition for moisture from neighbouring plants may be greater than in the non-selective and zero grazing treatments because of the larger root systems of the older plants, especially dominant non-succulent shrubs such as *Pentzia incana*. These plants tend to have well-developed and deep root systems, but also retain roots near the soil surface (Midgley and van der Heyden 1999). Midgley and Van der Heyden (1999) have suggested that *Pentzia incana* may also use high water loss rates as a means to reduce water availability to competitors.

Loosened soil microsites are created for seedling establishment in both the non-selective and conventional grazing treatments through livestock hoof-action, with higher intensity disturbance on the non-selective grazing treatment. Although it has been found that seedling germination and emergence (Beukes 1999) and survival (Milton and Dean 1990) is greater on disturbed soil and seedling recruitment more frequent on overstocked rangeland compared to well vegetated rangeland (Milton 1995), in this study seedling density did not differ significantly between the treatments.

It is known that seedlings do benefit from protection by established plants (Milton *et al.* 1999) which provide refuge from abiotic stress (Dean and Milton 1991) and grazing (Milton 1992) but no significant differences between seedling density close to adult plants or in open areas were found between the treatments.

The density of grass seedling in the conventional grazing treatment, although not significant, tended to be higher in the conventional grazing treatment than in the zero and non-selective grazing treatments.

This is probably as a result of the high seed production of the larger grass plants in that treatment, especially the dominant grasses *Eragrostis lehmanniana* and *Stipagrostis ciliata* that produce large amounts of seed (McClaren and Anable 1992; Esler 1999).

The influence of the suitability of microsites between the grazing systems for seedling establishment will probably be more evident in the density of seedlings after major recruitment events. However, recruitment outside the major events will be limited by competition for resources from the established mature shrubs and perennial grasses (Milton *et al.* 1999).

The aging vegetation in the conventional grazing treatment is becoming more moribund, and less palatable and nutritious for livestock, as crude protein content declines in forage species with increasing age (Meissner *et al.* 1999). It is probable that few young plants will develop beyond the seedling stage due to competition from established plants, outside major recruitment events.

No one grazing treatment is promoting or reducing plant diversity at this stage, but the non-selective grazing treatment may be providing more suitable forage to livestock over time because of the higher density of smaller (canopy size) and probably younger shrubs and perennial grasses.

10. Conclusions

The zero grazing controls, the non-selective grazing treatment and the conventional grazing treatment showed no significant differences in plant diversity, canopy cover percentage or seedling abundance. This may be due to the time it takes for grazing induced changes to become evident, the high resistance to grazing disturbance and the event driven dynamic behaviour of karoo vegetation.
More data on individual species and specific plant groups that may be indicating possible future shifts in vegetation composition and abundance due to grazing, need to be collected.

The conventional grazing treatment consists of larger and probably older shrubs and perennial grasses than the zero grazing controls (with their legacy of non-selective grazing) and the non-selective grazing treatment. These will become more moribund over time and limit the emergence of younger plants through competition for space and resources.
During the lag phase, before the longer-lived shrubs die and create gaps for new plants, less suitable forage will be available to livestock.
The gaps available in the conventional grazing treatment will allow short-lived species to establish, but competition will be great for new long-lived plants. A major disturbance event in the conventional treatment would be necessary to kill the older shrubs and grasses and create gaps for younger plants to emerge.
Plant size-age relationships under different grazing treatments and the longevity of karoo plants would be interesting future research areas to better understand how plant age structure influences diversity.

The significant difference observed in plant density between the treatments indicates a shift towards a higher density of younger shrubs and perennial grasses in the non-selective grazing treatment, as a result of increased plant turnover rate and possibly the greater success of plant establishment beyond the seedling stage.

The non-selective grazing treatment probably has more palatable forage in terms of younger plants over time than the conventional grazing system, and is maintaining the same levels of plant diversity as both the zero grazing controls and the conventional grazing treatment.

These factors will influence the levels at which the vegetation can support livestock on the non-selective and conventional grazing treatments.

Because of the time scales involved in karoo vegetation dynamics, it would be important to monitor the vegetation for changes in diversity (species richness and the abundance of species), as a result of grazing treatment over longer periods of time, especially after the occurrence of major disturbance, rainfall and recruitment events.

11. References

ABEL N. O. J. AND BLAIKIE P. 1989. Land degradation, stocking rates and conservation policies for the communal rangelands of Botswana. *Land degradation and Rehabilitation 1: 101-123.*

ACOCKS J. P. H. 1966. Non-selective grazing as a means of veld reclamation. *Proceedings of the Grassland Society of Southern Africa 1: 33-39.*

ACOCKS J. P. H. 1975. Veld types of South Africa (2nd edition). *Memoirs of the Botanical Survey of South Africa 40.*

ARNOLD T. H. AND DE WET B. C. (eds). 1993. Plants of southern Africa: names and distribution. *Memoirs of the Botanical Survey of South Africa 62.*

BARNARD S. A. 1986. Oumanssoutbos in die winterrëenstreek. *Boerdery in Suid-Afrika, Weiding Nr. 140.*

BEHNKE R. H. AND SCOONES I. 1993. Rethinking range ecology: implications for rangeland management in Africa. In: *Range ecology at disequilibrium: new models of natural variability and pastoral adaptation in African savannas. Overseas Development Institute, London.*

BEUKES P. C. 1999. Livestock impact assessment and restoration strategies in the semi-arid Karoo. *Ph.D. Thesis, Department of Botany, University of Cape Town.*

BEUKES P. C. 1999a. Aspects of plant ecology of South African biomes: Karoo. *B.Tech module. Department of Nature Conservation and Oceanography, Cape Technikon, Cape Town.*

BEUKES P. C. 2001. Faculty of Applied Sciences, Department of Nature Conservation and Oceanography Cape Technikon. *Personal communication.*

BEUKES P. C. AND COWLING R. M. 1999. Impacts of non-selective grazing on cover, composition and productivity of Nama Karoo grassy shrubland. *African Journal of Range and Forage Science 17: 27-35.*

BOND W. J. 1999. Rangelands and biodiversity in southern Africa - impacts of contrasting management systems. *Proceedings of the VI International Rangeland Congress, VI International Rangeland Congress, Inc.*

BOND P. AND GOLDBLATT P. 1984. Plants of the Cape Flora: a descriptive catalogue. *Trustees of the National Botanical Gardens of South Africa, Kirstenbosch.*

BORMANN F. H. 1953. The statistical efficiency of sample plot size and shape in forest ecology. *Ecology 34: 474-487.*

BOSCH O. J. H. 1999. The Karoo Biome. In: *Tainton N. M. (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

CAPE NATURE CONSERVATION. 1994. The collection of plants and the preparation of herbarium specimens for identification and reference. *Unpublished document. Cape Nature Conservation Scientific Services Jonkershoek, Stellenbosch.*

CLARKE K. R. AND WARWICK R. M. 1994. Change in marine communities: an approach to statistical analysis and interpretation - Plymouth Routines in Multivariate Ecological Research. *Natural Environment Research Council, U.K. Hitchings and Mason Ltd. Plymouth U.K.*

COWLING R. M.; ESLER K. J.; MIDGLEY G. F. AND HONIG M. A. 1994. Plant functional diversity, species diversity, and climate in arid and semi-arid southern Africa. *Journal of Arid Environments 27: 141-158.*

DANCKWERTS J. E. 1987. The animal plant interaction. In: *J. E. Danckwerts and W. H. Teague (eds). Veld management in the Eastern Cape. Pasture Research Section Stutterheim, South Africa.*

DANCKWERTS J. E.; AUCUMP A. J. AND BARNARD H. J. 1983. Herbaceous species preference by cattle in the False Thornveld of the Eastern Cape. *Proceedings of the Grassland Society of southern Africa 18: 89-94.*

DANCKWERTS J. E. AND TEAGUE W. H. 1987. Veld management in the Eastern Cape. *Pasture Research Section, Stutterheim, South Africa.*

DEAN W. R. J. AND MILTON S. J. 1991. Patch disturbance in arid grassy dunes: antelope, rodents and annual plants. *Journal of Arid Environments 20: 231-237.*

DEAN W. R. J. AND MILTON S. J. 1999 (eds). The Karoo, ecological patterns and processes. *Cambridge University Press, UK.*

DESMET P. G. AND COWLING R. M. 1999. The climate of the karoo, a functional approach. In: *W. R. J. Dean and S. J. Milton (eds). The Karoo, ecological patterns and processes. Cambridge University Press, UK.*

DONALDSON J. S. 1999. Conservation of globally significant biodiversity in agricultural landscapes in South Africa through conservation farming. *Global Environmental Facility Targeted Research Proposal, National Botanical Institute South Africa, Kirstenbosch.*

DU TOIT P. F. 1972. The goat in a bush-grass community. *Proceedings of the Grassland Society of southern Africa 7: 44-50.*

DU TOIT P. C. V. 1995. Grazing index method of range condition assessment and comparison of techniques. *African Journal of Range and Forage Science 12: 61-67.*

DU TOIT P. C. V. 2001. The relation between canopy spread cover and the above ground available phytomass of Nama Karoo subshrubs and grasses. *African Journal of Range and Forage Science 18: 143-146.*

DU TOIT P. C. V. 2001a. Objektiewe weidingsindekswaardes van Nama Karooplantegroei: grasse en bossies van die karoo. *Grootfontein Agric 1: 8-19. Grootfontein Agricultural Development Institute, Middleburg.*

DU TOIT P. C. V. 2002. Grootfontein Agricultural Development Institute, South Africa. *Personal communication.*

ELLIS F. 2002. Faculty of Agriculture, University of Stellenbosch. South Africa. *Personal communication.*

ELLIS F. AND LAMBRECHTS J. J. N. 1986. Soils. In: *Cowling R. M; Roux P. W. and Pieterse A. J. H. (eds). The Karoo biome: a preliminary synthesis Part 1 - physical environment. South African National Scientific programmes Report 124, CSIR, Pretoria.*

ESLER K. J. 1993. Vegetation patterns and plant reproductive processes in the Succulent Karoo. *Ph.D. Thesis. University of Cape Town, Cape Town.*

ESLER K. J. 1999. Plant reproductive ecology. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

ESLER K. J. AND COWLING R. M. 1993. Edaphic factors and competition as determinants of pattern in South African karoo vegetation. *South African Journal of Botany 59: 287-295.*

FISCHER R. A.; CORBETT A. G. AND WILLIAMS C. B. 1943. The relation between number of species and the number of individuals in a random sample of an animal population. *Journal of Animal Ecology 12: 42-58.*

FRIEDEL M. H. 1997. Discontinuous change in arid woodland and grassland vegetation along gradients of cattle grazing in central Australia. *Journal of Arid Environments 31: 145-164.*

FRIEDEL M. H.; BASTIN G. N. AND GRIFFIN G. F. 1988. Range assessment and monitoring in arid lands: the derivation of functional groups to simplify vegetation data. *Journal of Environmental Management 27: 85-97.*

GOODMAN D. 1975. The theory of diversity-stability relationships in ecology. *Biology 50: 237-266.*

GREEN R. H. 1979. Sampling design and statistical methods for environmental biologists. *University of Western Ontario, Wiley-Interscience Publications, New York.*

HOFFMAN M. T. 1988. The rationale for karoo grazing systems: criticisms and research implications. *South African Journal of Science 84: 556-559.*

HOFFMAN M. T. AND COWLING R. M. 1990. Desertification in the lower Sundays River Valley. *South African Journal of Arid Environments 19: 105-117.*

HOFFMAN M. T.; COUSINS B.; MEYER T.; PIETERSON A. AND HENDRICKS H. 1999. Historical and contemporary land use and the desertification of the karoo. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

HURLBURT S. H. 1971. The non-concept of species diversity: a critique and alternative parameters. *Ecology 52: 577-586.*

JACKSON R. B. AND CALDWELL M. M. 1993. Geostatistical patterns of soil heterogeneity around individual perennial plants. *Journal of Ecology 81: 683-692.*

JELTSCH F.; WIEGAND T. AND WISSEL C. 1999. Spatially explicit computer simulation models: tools for understanding vegetation dynamics and supporting rangeland management. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

KENT M. AND COKER P. 1994. Vegetation description and analysis: A practical approach. *John Wiley and Sons CRC Press, UK.*

LANDSBURG J.; LAVOREL S. AND STOL J. 1999. Grazing response groups among understorey plants in arid rangelands. *Journal of Vegetation Science 10: 683-696.*

LE ROUX P. M.; KOTZE C. D.; NEL G. P. AND GLEN H. F. 1994. Bossieveld: Grazing plants of the karoo and karoo-like areas. *Department of Agriculture, Pretoria, South Africa.*

LOW A. B. AND REBELO A. G. 1998. Vegetation of South Africa, Lesotho and Swaziland. *Department of Environmental Affairs and Tourism, Pretoria.*

LUND A. 2001. Karoo grazing experiment proposal. *Elandsfontein farm, Beaufort West. Unpublished document.*

LUND A. 2002. Elandsfontein farm, Beaufort West. *Personal communication.*

LUND J. 2002. Bleakhouse farm, Beaufort West. *Personal communication.*

MAGURRAN A. E. 1988. Ecological Diversity and its measurement. *Princeton University Press, New Jersey.*

MARGALEF R. 1958. Information theory in Ecology. *General Systematics 3: 36 -71.*

McCLARAN M. P. AND ANABLE M. E. 1992. Spread of introduced Lehmann lovegrass along a grazing intensity gradient. *Journal of Applied Ecology 20: 92-98.*

McINTYRE S.; LAVOREL S.; LANDSBERG J. AND FORBES T. D. A. 1999. Disturbance response to vegetation - towards a global perspective on functional traits. *Journal of Vegetation Science 10: 621-630.*

McNAUGHTON S. J.; RUESS R. W. AND SEAGLE S. W. 1988. Large mammals and process dynamics in African ecosystems. *BioScience 38: 784-800.*

MEISSNER H. H.; ZACHARIAS P. J. K. AND O'REAGAIN P. J. 1999. Forage quality. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

MENTIS M. T. 1981. The animal factor in pasture and veld management in South Africa. *N. M. Tainton (ed). Shuter and Shooter, Pietermaritzburg.*

MILLS A. 2001. Conservation Farming Project, National Botanical Institute, South Africa. *Personal communication.*

MILTON S. J. 1992. Studies of herbivory and vegetation change in karoo shrublands *Ph.D. Thesis, University of Cape Town, South Africa.*

MILTON S. J. 1994. Growth, flowering and recruitment of shrubs in grazed and in protected rangeland in the arid karoo, South Africa. *Vegetatio 111: 17-27.*

MILTON S. J. 1995. Spatial and temporal patterns in the emergence and survival of seedlings in arid karoo shrubland. *Journal of Applied Ecology 32: 145-156.*

MILTON S. J. 2000. Interactions between diversity and animal production in natural rangelands. *African Journal of Range and Forage Science 17: 7-9.*

MILTON J. S. AND DEAN W. R. 1990. Seed production in rangelands of the southern karoo. *South African Journal of Science 86: 231-233.*

MILTON S. J. AND DEAN W. R. 1993. Selection of seeds by harvester ants (*Messor capensis*) in relation to condition of arid rangeland. *Journal of Arid Environments 24: 63-74.*

MILTON S. J. AND HOFFMAN M. T. 1994. The application of state and transition models to rangeland research and management in arid succulent and semi-arid grassy karoo, South Africa. *African Journal of Range and Forage Science 11: 18-26.*

MILTON J. S.; DEAN W. R.; DU PLESSIS M. A. AND SIEGFRIED W. R. 1994. A conceptual model of arid rangeland degradation: the escalating cost of declining productivity. *Bioscience 44: 70-76.*

MILTON J. S.; DEAN W. R.; MARINCOWITZ C. P. AND KERLEY G. I. H. 1995. Effects of the 1990/91 drought on rangeland in Steytlerville karoo. *South African Journal of Science 91: 78-84.*

MILTON S. J. AND DEAN W. R. J. 1996. Karoo veld: Ecology and management. *ARC Range and Forage Institute, South Africa.*

MILTON S. J.; YEATON R. I.; DEAN W. R. J. AND VLOK J. H. J. 1997. Succulent karoo. In: *R. M. Cowling.; D. M. Richardson and S. M. Pierce (eds). Vegetation of Southern Africa. Cambridge University Press.*

MILTON S. J. AND DEAN W. R. J. 1999. (eds). The karoo: Ecological patterns and processes. *Cambridge University Press, UK.*

MILTON S. J. AND DEAN W. R. J. 1999a. The karoo: past and future. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

MILTON S. J.; DAVIES R. A. G. AND KEARLY G. I. H. 1999. Population level dynamics. In: *W.R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

MIDGLEY G. F. AND VAN DER HEYDEN F. 1999. Form and function of perennial plants. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

MUNSELL 1971. Munsell Soil Color Charts 1971 Edition. *Munsell Color Company, Inc. Baltimore, Maryland U.S.A.*

NEVAH Z. AND WHITTAKER R. H. 1979. Structural and floristic diversity of shrublands and woodlands in Northern Israel and other mediterranean areas. *Vegetatio 41: 171-190.*

NOLAN T.; CONNOLLY J.; SALL C. AND CESAR J. 2000. Mixed livestock grazing in diverse temperate and semi-arid environments. *African Journal of Range and Forage Science 17: 10-21.*

NOVELLIE P. A. 1999. Dynamics. In: *W. R. Dean and S. J. Milton (eds) The karoo, ecological patterns and processes. Cambridge University Press, UK.*

NOY-MEIR I. 1982. Stability of plant-herbivore models and possible applications to savanna. In: *B. J. Huntley and B.H. Walker (eds). Ecology of tropical savannas. Springer-Verlag, Berlin.*

NOY-MEIR I.; GUTMAN M. AND KAPLAN Y. 1989. Responses of Mediterranean grassland plants to grazing and protection. *Journal of Ecology. 77: 290-310.*

O'CONNOR T.G. 1991. Local extinction in perennial grasslands, a life history approach. *American Naturalist 137: 753-773.*

O'CONNOR T. G. AND ROUX P. W. 1995. Vegetation changes (1949-1971) in a semi-arid, grassy dwarf shrubland in the karoo, South Africa. Influence of rainfall and grazing by sheep. *Journal of Applied Ecology 32: 612-626.*

OLSVIG-WHITTAKER L. S.; HOSTEN P. E.; MARCUS I. AND SHOCHAT E. 1993. Influence of grazing on sand field vegetation in the Negev Desert. *Journal of Arid Environment 24: 81-93.*

O'REAGAN P. J. AND TURNER J. R. 1992. An evaluation of the empirical basis for grazing management recommendations for rangeland in Southern Africa. *Journal of the Grassland Society of Southern Africa 9: 38-49.*

OWEN-SMITH N. 1999. The animal factor in veld management. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

PALMER A. R.; NOVELLIE P. A. AND LLOYD J. W. 1999. Community patterns and dynamics. In: *W. R. J. Dean and S. J. Milton (eds). The karoo: Ecological patterns and processes. Cambridge University Press, UK.*

PEET R. K. 1975. Relative diversity indices. *Ecology 56: 496-498.*

PIELOU E. C. 1969. An introduction to mathematical ecology. *Wiley, New York.*

ROUX P. W. 1963. The descending point method of vegetation survey. A point sampling method for the measurement of semi-open grasslands and karoo vegetation in South Africa. *South African Journal of Agricultural Science 5: 273-288.*

ROUX P. W. 1968. Principles of veld management in the karoo and adjacent dry sweet-grass veld. In *W. J. Hugo (ed). The small stock industry, 318-340. Government Printer, Pretoria.*

ROUX P. W. 1999. Forward. In: *Dean W. R. and Milton S . J. (eds). The karoo, ecological patterns and processes. Cambridge University Press, UK.*

ROUX P. W. AND VORSTER M. 1983. Vegetation change in the karoo. *Proceedings of the Grassland Society of South Africa 18: 25-29.*

SCHULZE R. E. 1997. South African atlas of agrohydrology and climatology. *Report TT82/96, Water Research Commission, Pretoria.*

SHANNON C. E. AND WIENER W. 1963. The mathematical theory of communication. *University of Illinois Press, Urbana.*

SHEARING D. 1994. Karoo: South African wild flower guide No. 6. *Botanical Society of South Africa, Kirstenbosch, South Africa.*

SHMIDA A. 1984. Whittaker's plant diversity sampling method. *Israel Journal of Botany 33: 41-46.*

SKEAD C. J. 1982. Historical mammal incidence in the Cape Province. Volume 1: The western and northern Cape. *Department of Nature and Environmental Conservation, Cape Town.*

SMITH R. L. 1990. Ecology and field biology. *Harper Collins Publishers, New York.*

SNYMAN H. A. 1999. Soil erosion and conservation. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

STOKES C. J. 1994. Degradation and dynamics of succulent karoo vegetation. *M.Sc. Dissertation, Botany, University of Natal, Pietermaritzburg, South Africa.*

STOLGREN T. J. 1994. Planning long-term vegetation studies at landscape scales. In: *T. M. Powell and J. H. Steele (eds). Ecological Time Series. Chapman and Hall, New York.*

STOLGREN T. J.; FALKNER M. B. AND SCHELL L. D. 1995. A Modified-Whittaker nested vegetation-sampling method. *Vegetatio 117: 113-121.*

TAINTON N. M.; ZACHARIAS P. J. K. AND HARDY M. B. 1989. The contribution of veld diversity to the agricultural economy. In: *B. J. Huntley (ed). Biotic diversity in southern Africa: concepts and conservation. Oxford University Press, Cape Town.*

TAINTON N. M. AND DANCKWERTS J. E. 1999. Resting. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

TILMAN D. 1997. Biodiversity and ecosystem functioning. In: *G.C. Daily (ed). Nature's service: Societal dependence on natural ecosystems. Island Press, Washington DC.*

TODD S. W. AND HOFFMAN M. T. 1999. A fence-line contrast reveals effects of heavy grazing on plant diversity and community composition in Namaqualand, South Africa. *Plant Ecology 142: 169-178.*

TONGWAY D. 1994. Rangeland soil condition assessment manual. *CSIRO, Australia.*

VAN DER HEYDEN F. AND STOCK W. D. 1999. The response of forage plants to defoliation; Karoo shrubs. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

VAN OUDTSHOORN F. P. 1992. Guide to grasses of South Africa. *Briza Publications, Arcadia, South Africa.*

VAN ROOYEN M. W. 1999. Functional aspects of short-lived plants. In: *Dean W. R. and Milton S. J. (eds). The karoo, ecological patterns and processes. Cambridge University Press, UK.*

VORSTER M. 1982. The development of the ecological index method for assessing veld condition in the karoo. *Proceedings of the Grassland Society of Southern Africa 17: 84-89.*

VORSTER M. 1999. Karoo and associated sweet grassveld. In: *N. M. Tainton (ed). Veld Management in South Africa. University of Natal Press, Pietermaritzburg.*

VORSTER M. AND ROUX P. W. 1983. Veld of the karoo areas. *Proceedings of the Grassland Society of Southern Africa 18: 35-39.*

WASER N. M. AND PRICE M. V. 1981. Effects of grazing on diversity of annual plants in the Sonoran Desert. *Oecologia 50: 407-411.*

WEIHER E.; VAN DER WERF A.; THOMPSON K.; RODERICK M.; GARNIER E. AND ERIKSON O. 1999. Challenging Theophrastus: A common core list of plant traits for functional ecology. *Journal of Vegetation Science 10: 609-620.*

WEST N. E. 1993. Biodiversity of rangelands. *Journal of Range Management 2-13.*

WESTFALL R. H.; VAN STADEN J. M.; PANAGOS M. D.; BREYTENBACH P. J. J. AND GREEF A. 1994. Scale related vegetation sampling. *Unpublished report, ARC-Range and Forage Institute, Pretoria.*

WESTOBY M.; WALKER B. AND NOY-MEIR I. 1989. Opportunistic management for rangelands not in equilibrium. *Journal of Range Management. 42: 266-274.*

WHITTAKER R. H. 1977. Evolution of species on land communities. *Evolutionary Biology 10: 1-67.*

WIEGAND T.; MILTON S. J.; ESLER K. J. AND MIDGLEY G. F. 2000. Live fast, die young: estimating size-age relations and mortality patterns in the semi-arid karoo, South Africa. *Plant Ecology 150: 115-131.*

YEATON R. I. AND ESLER K. J. 1990. The dynamics of succulent karoo vegetation: a study of species association and recruitment. *Vegetatio 88: 103-113.*

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