The impact of communal land-use on the biodiversity of a conserved grassland at Cathedral Peak, uKhahlamba-Drakensberg Park, South Africa

Implications for sustainable utilization of montane grasslands

Component A

by

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Preface

The research described in this mini-dissertation was carried out at the Centre for Environment and Development, University of Natal, Pietermaritzburg under the supervision of Dr Michelle Hamer and Dr Terry Everson.

This mini-dissertation represents the original work of the author and has not otherwise been submitted in any form for any degree or diploma at any university. Where use has been made of the work of others it is duly acknowledged in the text.

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Acronyms

ANC	African National Congress
AU	Animal Unit (450 kg)
BATAT	Broadening Access to Agriculture Thrust
CA	Correspondence Analysis
Contralesa	Council of Traditional Leaders of South Africa
CBNRM	Community Based Natural Resource Management
CPD	Centre of Plant Diversity
CSIR	Council of Scientific and Industrial Research
DCA	Drakensberg Catchment Area
DEAT	Department of Environmental Affairs and Tourism
GEF	Global Environment Facility
ICDP	Integrated Conservation and Development Projects
IDP	Integrated Development Plan (for local municipalities)
IDH	Intermediate Disturbance Hypothesis
IUCN	International Union for the Conservation of Nature and Natural Resources
MDTP	Maloti-Drakensberg Transfrontier Park
MDTCDP	Maloti-Drakensberg Transfrontier Conservation and Development Project
NEMA	National Environmental Management Act
NGO	Non-government organization
NPB	Natal Parks Board
PCA	Principal Components Analysis
SDI	Spatial Development Initiative
SEA plan	Strategic Environmental Assessment
TFCA	Transfrontier Conservation Area
UNCED	United Nations Conference on Environment and Development
UNESCO	United Nations Educational, Scientific and Cultural Organisation
UDP	uKhahlamba-Drakensberg Park

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CHAPTER 1
Introduction

1.1 Background to the research

South Africa is the third most biologically diverse country in the world. As a signatory to the Convention on Biological Diversity, South Africa has a responsibility to conserve biodiversity as well as to use it sustainably and equitably (Glowka, Burhenne-Guilmin, Synge, McNeely and Gundling 1994). In order to achieve these objectives South Africa is required to develop national strategies, plans and programmes (Department of Environmental Affairs and Tourism 1997).

Conservation paradigms shifted in the late 20th century from protectionism to sustainable utilization (Pearce 2000). Local indigenous people now have a right to benefit from biodiversity and their support is seen as necessary for the long-term sustainability of protected areas. Another paradigm shift was the recent emphasis on management for biodiversity (Bond 1999). New approaches to reserve selection have been developed (Goodman 2001 *pers.comm.*) and conservation management emphasizes greater flexibility. However, biodiversity conservation is still evolving and conservation bodies often do not have the capacity to maintain or maximise biodiversity within reserves (Bond 1999).

Conservation planners have recognized that existing protected areas are insufficient for conserving biodiversity and that conservation also has to take place outside protected areas if representativity is to be achieved (Desmet 1999, McGeoch 2002). Communal grasslands offer opportunities as corridors and buffer zones for biodiversity conservation as they are the least likely to be transformed (O'Connor in press). At the same time, communal grasslands are perceived to be among the most degraded in South Africa (Hoffman and Todd 2000).

This study focuses on a grassland in the Cathedral Peak area of the uKhahlamba-Drakensberg Park (UDP) which is situated in the Eastern Mountain Centre of Plant Diversity (CPD). A CPD is an area with exceptionally high endemism as well as exceptionally high levels of habitat loss (Cowling and Hilton-Taylor 1994). The grassland biome is one of the most threatened vegetation types in South Africa and less than 2% of grasslands both globally and nationally are formally protected (Linden 2000, Le Roux 2002). DEAT (2002) has recognized that the poor state of grassland conservation in South Africa needs to be addressed.

There are arguments that grazing has a role to play in grassland conservation. In Europe the loss of indigenous ungulates has left a gap in grassland ecosystems (Pykälä 2000). It is likely that domestic grazers could provide a certain level of disturbance that is necessary for the maintenance of grassland biodiversity (West 1993, Niamir-Fuller 1999, O'Connor 1999, Pykälä 2000). In the Drakensberg grasslands conservation has largely meant the exclusion of grazing with fire being used to maintain grassland health. Without fire, the low grazing pressure would lead to bush encroachment (Trollope 1999).

Biodiversity is a complex concept that can never be fully measured (Purvis and Hector 2000). Surrogates for biodiversity have to be selected in order to make the assessment feasible. Grassland forbs are responsible for the bulk of grassland species richness but have been largely ignored in grassland research (Uys 2000). Invertebrates comprise up to 95% of total biodiversity (Myers, Mittermeier, Mittermeier, da Fonseca and Kent 2000) and invertebrate endemicity in South Africa is approximately 70% (Hamer 2002). Invertebrates have also been ignored in research and land monitoring programmes because of the difficulty of identification due to their huge diversity and abundance (Andersen, Hoffman, Muller and Griffiths 2002, Hamer 2002). However, there is increasing recognition of their intrinsic conservation value as well as their important role in ecosystems (Watkinson and Ormerod 2001) and concern that they are not necessarily protected by the conservation of vegetation and large mammals (Myers *et al* 2000).

The Maloti-Drakensberg Transfrontier Park (MDTP) has been initiated in the Eastern Mountain region with the goals of conserving biodiversity as well as building the capacity of local communities in South Africa and Lesotho to manage their natural resources. Strategies for the MDTP include the creation of buffer zones of appropriate land-use practices as well as linking the Park across communal lands (Lusigi and Acquay 1999, Ezemvelo KZN Wildlife 2001). Much of the Park will continue to be grazed as local populations depend upon these resources (Waddington 1999). Eroded paths will have to be rehabilitated and appropriate ways to regulate grazing and fire need to be devised and implemented (Ezemvelo KZN Wildlife 2001).

1.2 Problem statement

New models for conservation are being sought and new strategies have to be developed that will link sustainable utilization of grasslands with the conservation of grassland biodiversity. Communities adjacent to protected areas need to be included in and benefit from biodiversity conservation.

Communal grasslands adjacent to protected areas will continue to be used for grazing as this makes an important contribution to rural livelihoods. The challenge is to find ways to do this that will decrease pressure on the land, as the communal grasslands are presently perceived as degraded. In spite of this perception, communal grasslands have more potential than transformed commercial grassland for providing corridors and buffer zones for maintaining biodiversity so that protected areas are not simply "islands" in transformed landscapes.

There is political pressure for the sustainable utilization of conserved areas. It is likely that this issue will gain momentum in areas such as the MDTP where there are plans to expand the boundaries of the existing protected areas. Strategies need to be found where communities can utilize grassland products, through both grazing and collecting at the same time as allowing for biodiversity conservation.

1.3 Rationale

Biodiversity conservation is critical for the continued functioning of ecosystems which in turn support human livelihoods. An understanding of the impacts of different land-uses is necessary for the implementation of sustainable land-use.

The lease land in this study provided an unusual opportunity for examining the impact of controlled communal grazing on biodiversity. Research in this area could contribute to the management of the MDTP. This study focused on research in three areas recommended by the Convention on Biological Diversity: the identification and monitoring of the components of biological diversity, the identification of human interactions with ecosystems and species and the management of biological resources and the activities that affect them (Glowka *et al* 1994). A multi-disciplinary approach was adopted and included botanical and zoological aspects of grasslands and grassland science approaches as well as a review of the changing context of conservation and development. There is increasing recognition that the sustainable use of grasslands requires a multi-disciplinary approach to research and management (Thomas 1995, Cousins 1995).

There is value in assessing the impact of low levels of grazing on grassland biodiversity, a management approach that lies between continuous communal grazing and conventional grassland conservation. The findings of this study -may offer a way forward in the creation of low-use buffer zones and corridors outside of the core protected areas. In particular it may offer strategies for the MDTP which aims to conserve biodiversity at the same time as recognizing that local communities depend upon grazing.

This study purposefully included important but neglected components of biodiversity such as grassland forbs and invertebrates. Grassland forbs make up the bulk of grassland plant biodiversity (Scott-Shaw 1999) while invertebrates account for the bulk of biodiversity on earth. Invertebrate groups are particularly useful in biodiversity assessment and management (Andersen et al 2002) as they show rapid responses to environmental changes (Rivers-Moore and Samways 1996).

1.4 Aims and objectives

The aim of the study was to assess whether controlled communal land-use (grazing and plant harvesting) in the Cathedral Peak area had an impact on biodiversity. The study sought evidence of quantitative changes in invertebrate and plant diversity in response to controlled use of the grassland.

Objectives

1. Provide an overview of relevant literature on biodiversity conservation and assessment in relation to communal use of a conserved grassland.

2. Assess and compare the invertebrate and plant diversity and habitat condition in a low impact communally used area and the adjacent conservation land by means of a fence-line study to measure the impact of communal land-use.
3. Make recommendations regarding the opportunities and threats posed by rural communities having limited use of a conserved grassland.

Research questions

The following hypothesis was tested: There is lower biodiversity in the lease land at Cathedral Peak than in the adjacent conservation land.

The key questions which informed the study were:

1. Is there a difference in the diversity, evenness and species richness of plant communities between lease land and conservation land?
2. Is there a difference in the diversity and species richness of invertebrate communities between lease land and conservation land?
3. What is the difference between the two areas in terms of species? Have species been added or lost through different land-uses and what is the significance of these species?
4. What is the impact of communal land-use on veld condition as a measurement of ecosystem functioning in the lease land?
5. What are the implications of the findings for the conservation and sustainable use of grassland biodiversity?

1.5 Definition of key concepts

The research is a fence-line study which compares the biodiversity in the conservation land and land that is used for controlled communal grazing (the lease land) at Cathedral Peak. **The conservation land** is defined as land managed by Ezemvelo KZN Wildlife for conservation purposes. **The lease land** is an area of 535ha that has been used for controlled communal land-use by the neighbouring amaNgwane tribe since 1995 in terms of an interim "lease" agreement between the tribal authority and Ezemvelo KZN Wildlife.

Biodiversity is defined as "the variability among living organisms including, *inter alia,* terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and diversity of ecosystems" (Glowka *et al* 1994:16). For the purposes of this study only vegetation and selected invertebrate taxa were measured.

Controlled communal land-use was the use of the lease-land for grazing for 3 month blocks twice a year. Limits were set on cattle numbers in the lease land. Grazing was allowed from November to January and May to July. Harvesting of thatch grass and plants was allowed in exchange for payments in kind to Ezemvelo KZN Wildlife (Lemmer 2001 *pers.comm.*).

Key concepts and processes in this study are illustrated in Figure 1.1.

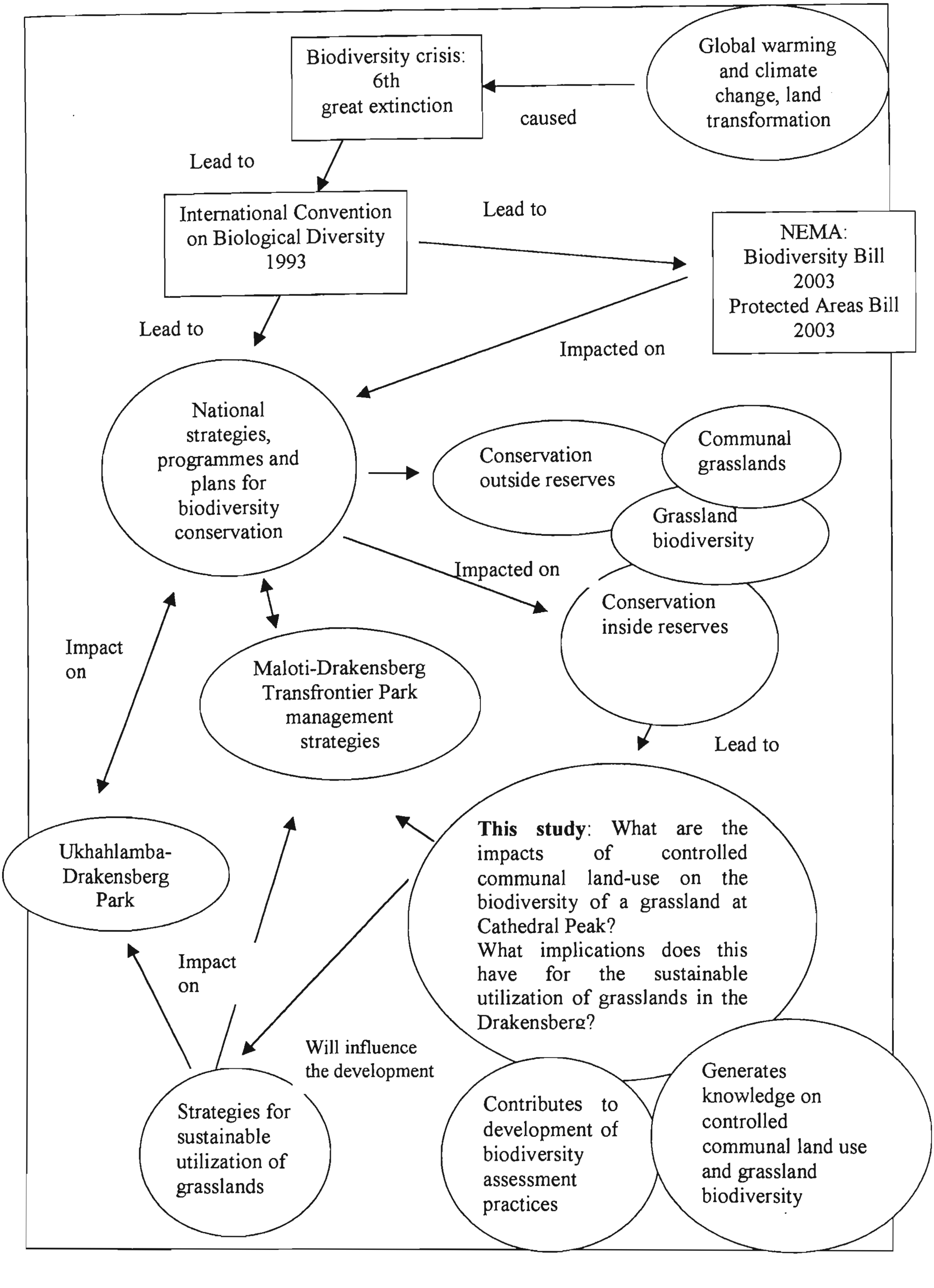

Figure 1.1: Key concepts and processes in the assessment of grassland biodiversity

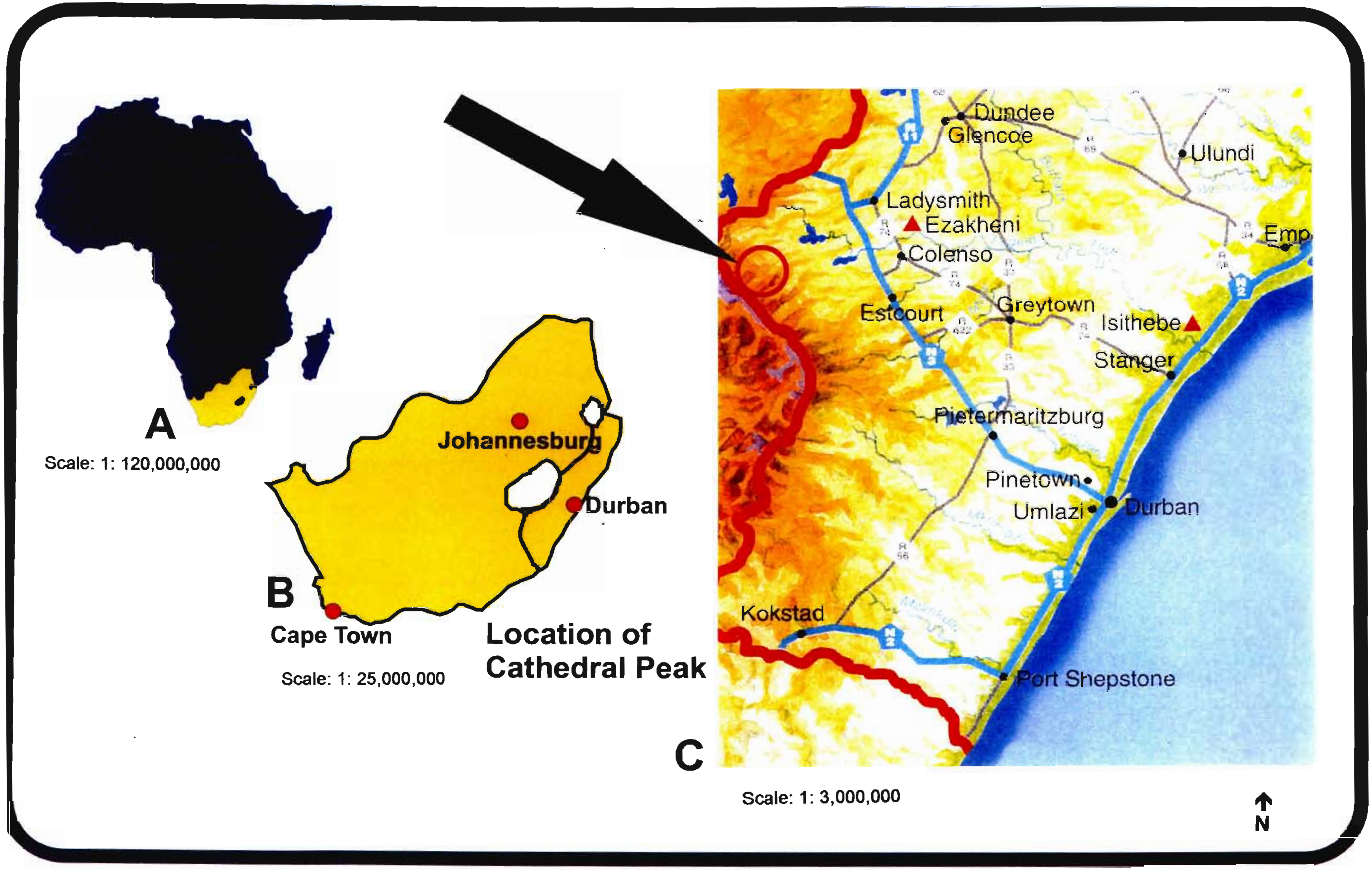

Figure 2.1 Location of Cathedral Peak, KwaZulu-Natal relative to A. Africa B. South Africa and C. Southern KwaZulu-Natal.

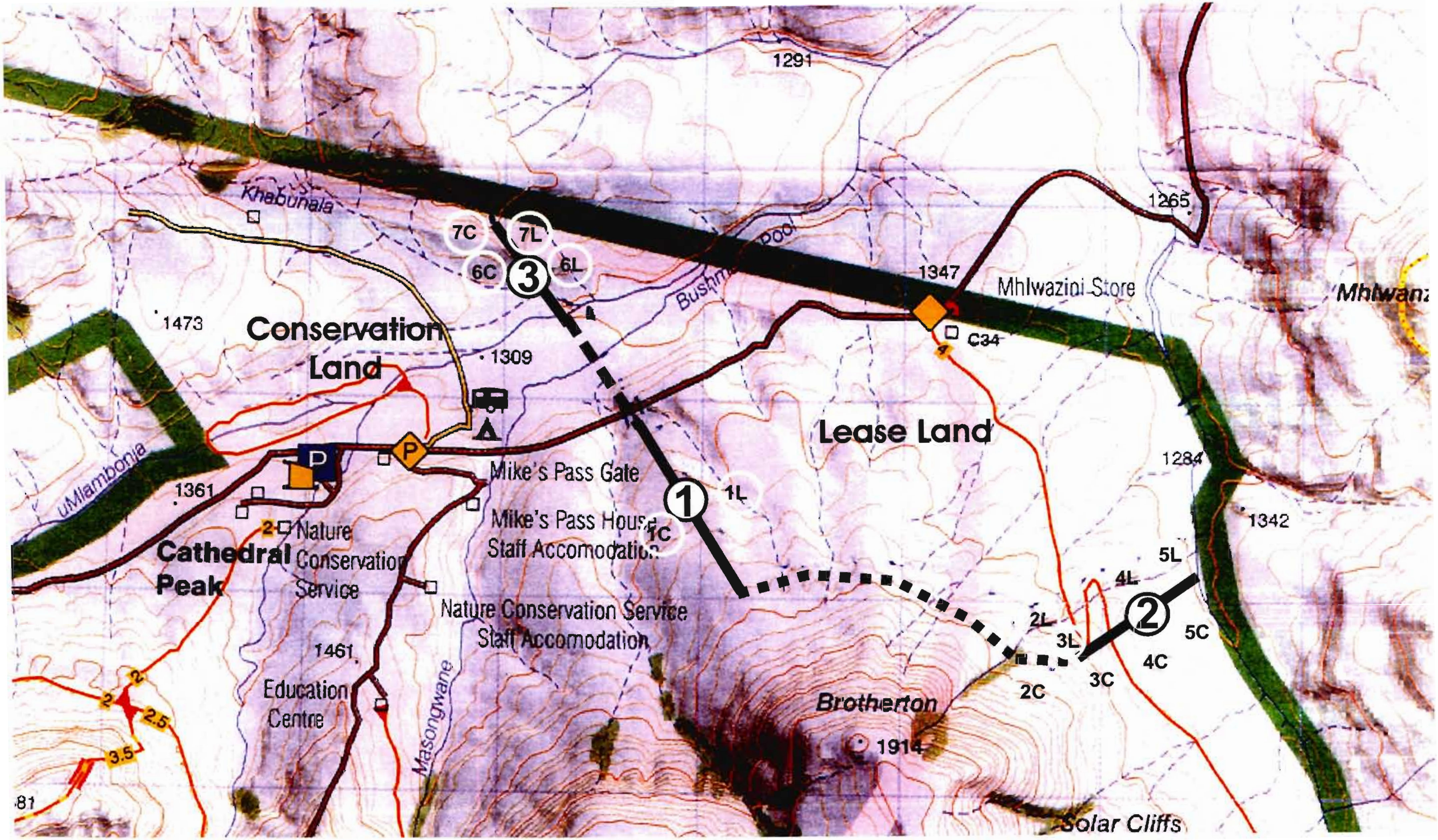

Figure 2.2 Position of the fourteen study sites at Cathedral Peak: 1 - 7C = seven sites on conservation land, 1-7L = seven sites on lease land.

N

Scale: 1:25000

CHAPTER 2
Study area

The study was conducted in the Cathedral Peak area of the Ukhahlamba-Drakensberg Park (UDP) at latitude 29° 00′ S and longitude 29° 15′ E. The altitude range of the study area is 1291m to 1377m and falls into the montane zone (1280m to 1829m) described by Killick (1963). Sampling took place during 2002 along the fence-line between the lease land and the adjacent conservation land (Figures 2.1 and 2.2).

2.1 The Drakensberg

Topography

The Drakensberg range is part of the Great Escarpment on the eastern edge of the interior plateau of Southern Africa (Killick 1961). The mountain range was formed by erosion of the Gondwanaland land surfaces over the past 120 million years. The mountain system is dynamic and in a state of continuous erosion due to heavy rainfall and steep slopes (Bainbridge 1991).

The Drakensberg range consists of two parallel escarpments, the Lesotho escarpment (above 3000m) and the Little Berg (1800 - 2400m) (Bainbridge *et al* 1986). The landscape is dissected by eastward flowing river valleys, which create a terrain of steep slopes and deep valleys resulting in a complex mosaic of vegetation (CSIR 1999).

Killick (1963) divided the Drakensberg into three distinct altitudinal zones: the alpine zone or summit (altitude: 2866 - 3353m); the subalpine zone or Little Berg (altitude: 1829 - 2865m) and the montane zone or river valley system (altitude 1280 - 1829m).

Subsequent classifications are the Afro-montane grassland biome (1700 - 2500m) roughly coinciding with Killick's subalpine zone or Little Berg (altitude: 1829 - 2865m) and the Alti-Mountain biome (above 2500m) roughly coinciding with Killick's alpine zone or summit (altitude: 2866 - 3353m) (Low and Rebelo 1996). It is in these two high altitude zones that the endemic plant biodiversity of the Drakensberg lies (Ezemvelo KZN Wildlife 2001).

Climate

The Drakensberg has one of the highest rainfalls in the summer rainfall region of South Africa, with an average of 1300mm per annum (Bainbridge 1991). The rain falls mainly between October and March, in high intensity summer storms. Peak rainfall is recorded adjacent to the escarpment, with a rain shadow on the Lesotho side. Winters are dry with snow falling in the region of the summit (Killick 1961). The area has one of the highest lightning frequencies in the country. However, as these coincide with the wet months, lightning fires are usually small and cool (Bainbridge *et al* 1986).

Soils

Drakensberg soils are ancient, shallow, highly leached and acid. This makes them infertile and leads to slow recovery of denuded areas. A good vegetation cover is essential to bind the soil and maintain soil structure. This in turn affects water storage, water control and water quality maintenance. There is significant danger of erosion accelerated by human activity and substantial erosion scars exist in the protected areas as a legacy of previous white farming and as proof of the vulnerability of the area. In some parts of the protected area, erosion has continued actively over 30 years with little sign of healing (Bainbridge *et al* 1986, 1991).

Vegetation

The Drakensberg is situated in the Eastern Mountain Centre of Plant Diversity and is the principle refuge for montane and alpine ecosystems in South Africa (Bainbridge *et al* 1986). With an altitude range of 1280m up to 3500m and an extremely broken landscape there are a wide variety of habitats. Almost 8% of plants in the uKhahlamba-Drakensberg area are endemics, with endemism particularly high amongst grassland forbs (CSIR 1999). A different suite of species is found in the north and south Drakensberg (Hilliard and Burtt 1987).

The vegetation is generally linked to the altitudinal zones. Killick (1990) proposed that the climax vegetation of the montane zone was *Podocarpus latifolius* forest, rather than the existing grassland but this has been challenged in recent years (Uys 2000). The climax vegetation of the sub-alpine zone has been described by Killick (1990) as fynbos and the climax vegetation of the alpine zone as heath.

The Drakensberg is less infested with aliens than other South African mountains. Invaders that occur include *Rubus cuneifolious,* exotic *Acacia* species, *Populus canescens* and *Pinus patula* (Bainbridge *et al* 1986).

The montane zone

The research site occurs in the montane zone which comprises mainly grasslands dominated by *Themeda triandra* and *Hyparrhenia* spp. and includes scrub communities and small patches of forest (Killick 1990). Within this zone there is a gradient from the lower altitude Southern Tall Grassland to the higher altitude Highland Sourveld (Acocks 1988). Southern Tall Grassveld is a "sourish mixed grassveld" meaning it has limited grazing value in winter. This veld type is true savanna with lower rainfall than the higher altitude areas and comprises an understorey of *T. triandra* grassland and scattered trees of *Acacia, Protea* and *Leucosidea* species (Acocks 1988:117). It has also been labelled a "False Veld Type", having been severely altered by anthropogenic factors (Scott-Shaw, Scott-Shaw, Bourquin and Porter 1996). Southern Tall Grassland lies at a lower altitude than most of the Drakensberg and has been widely used for agricultural purposes, leaving it under-represented in conservation. Eighty percent of this veld type has been transformed and only 1.56% is conserved (Low and Rebelo 1996). The only other protected Southern Tall Grassland is at Culfargie in the Cathkin area (Bainbridge 2002 *pers.comm.*).

Highland Sourveld is pure grassland although it has been proposed that it was originally forest and scrub forest (Acocks 1988). The grasses are sour (unpalatable in winter) and dominated by *T. triandra* and *Tristachya leucothrix.* Heavy continuous grazing by cattle leads to a dominance of *Eragrostis plana* (Acocks 1988), *Leucosidea sericea* (CSIR 1999), *Sporobolus africanus, Aristida* species (Tainton 1999a), *Elionurus muticus, Senecio retrorsus* and *Helichrysum argrophyllum* (Low and Rebelo 1996). Selective grazing by sheep leads to a dominance of *Acalypha schinzii* (Acocks 1988), *Elionurus muticus,, Aristida junciformis* and *Diheteropogon filifolius* (Tainton 1999a). Resting may lead to succession by *Hyparrhenia* spp. and *Cymbopogon* spp. but not back to *T. triandra* which is difficult or impossible to re-establish (Tainton 1999a). *Pteridium aquilinum,* an indigenous fern also invades *T. triandra* grassland, particularly in deep moist soil. The reasons for the invasion are unclear, but once established it shades out grasses (Killick 1990).

Where fire is excluded, *T. triandra* grassland is succeeded by *Hyparrhenia* grassland and *Miscanthidium-Cymbopogon* grassland or *Pteridium aquilinum* (bracken) veld with progression towards *Podocarpus latifolius* forest or protea savanna (Killick 1961) or fynbos grassland containing species such as *Cliffortia* spp., *Erica* spp., *Metalasia muricata* and *Anthospermum aethiopicum* (Low and Rebelo 1996). *Hyparrhenia* species also tend to dominate in wet and disturbed areas and *Miscanthidium-Cymbopogon* grasslands are found in moist areas, valley bottoms and forest margins (Killick 1961, 1990).

Highland Sourveld falls into Low and Rebelo's (1996) Moist Upland Grassland. This type of grassland has been widely degraded and poor grazing management has led to unpalatable grasses and invasion by herbaceous weeds. These grasslands are poorly conserved, with 60% transformed and only 2.52% are under protection, mainly in the Coleford and Himeville Nature Reserves (Low and Rebelo 1996).

Bioresource Group 11

The study site is also classified as Bioresource Group 11: Moist Transitional Tall Grassveld under Camp's (1997) Bioresource Groups based on edaphic and climatic data (Scott-Shaw *et al* 1996). Bioresource Group 11 is a transitional zone lying between the drier Tall Grassveld and Moist Highland Sourveld Bioresource Groups. *Themeda-Hyparrhenia* is the dominant plant association, with *Hyparrhenia hirta* dominating disturbed veld. Long-term overgrazing is indicated by the dominance of *Eragrostis curvula, Eragrostis plana* and *Sporobolus africanus*. *Elionurus muticus* occurs where there has been selective grazing (Camp 1997).

Fauna

Prior to the arrival of European settlers Southern African grasslands were used seasonally by indigenous grazers. Tainton and Hardy (1999:20) suggested that before the establishment of settled agriculture, "indigenous animal populations which roamed Southern Africa played a major role in the development of vegetation" and lead to a domination by grasses. The decimation of large grazers and browsers by early settlers substantially altered grazing patterns leading to "narrow spectrum grazing" dominated by grazers.

Recent records showed that the UDP in general and Cathedral Peak in particular have a low density of wild grazers (Van Zyl 2003 *pers.comm.*) (Table 2.1). However, in 1859 (Mann cited in Killick 1961) reported that blesbok, quagga, wildebeest and zebra were found under the Drakensberg during winter while as late as 1880, Moodie (cited in Killick 1961:131) described the landscape, a few miles below the summit as " teeming with every charming variety of wild animal in existence".

It is noteworthy that these nineteenth century writers described animals moving into the area during winter, as current understanding is that these are sour grasslands that are unpalatable during the winter months. Granger (1976) suggested that the African tribes which began to occupy the Drakensberg from the 1700s would have grazed stock in the little Berg during spring as the grass was likely to be greener than in the lower areas.

In 1961 Killick listed grazers and browsers at Cathedral Peak as eland (*Taurotragus oryx oryx*), duiker (*Sylvicapra grimmia burchellii*), rooiribbok (*Redunca fulvorufula*) vaalribbok (*Pelaea capreolus*), Cape klipspringer (*Oreotragus oreotragus oreotragus*) and Cape bushbuck (*Tragelaphusscriptus sylvaticus*). Rodents included porcupine (*Hystrix africae-australis*) dassie (*Procavia capensis*), hare (*Lepus* and *Pronolagus* spp.), ice-rat (*Myotomys sloggettii robertsii*), vlei-rat (*Otomys irroratus irroratus*), Natal mole-rat (*Cryptomys natalensis natalensis*) and the golden mole (*Chlorotalpa guillarmodii*). However, he added that it was possible to spend a whole day in the mountains without seeing a single mammal.

Table 2.1: Sightings of grazing mammals in lease land and adjacent 500m of conservation land: 1995 - 2002 (Van Zyl 2003 *pers.comm.*). Animal units calculated from weights provided by Smithers 1986). Abbreviations: AU= Animal Units (450kg)

	1995	1996	1997	1998	1999	2000	2001	2002	Approximate weight	Equivalent AU sighted in one year (Approximate weight ÷ 450kg)
Grey duiker	5	9	1			1	No data	2	20kg	0.4
Common reedbuck	8	2	2	1	9	5	No data	3	60kg	1.2
Mountain reedbuck		3	16		9	14	No data		26 kg	0.9
Grey rhebuck	14	1				5	No data	8	20 kg	0.6

History

Occupation of the Drakensberg by Middle Stone Age people dates back 22 000 years ago (Bainbridge *et al* 1986). Fires have been used as a tool to provide winter grazing ever since the climate included a dry season (Killick 1961).

The late Stone Age San probably lived in the Drakensberg about 8000 years ago (Ezemvelo KZN Wildlife 2001) and had low and seasonal population densities until Nguni pastoralists forced them higher into the mountains. To some extent the San probably kept pastoralists out of the Drakensberg. Eight hundred-year old iron age (Nguni) sites have been found in the Drakensberg foothills (Bainbridge *et al* 1986) and it is likely that cattle owners settled in the foothills about 400 years ago in the 1600s (Ezemvelo KZN Wildlife 2001).

The amaZizi, a pastoral tribe arrived in the northern Drakensberg around 1700 and occupied the river valleys while the San lived in the higher areas. The amaZizi owned large herds of stock and burned the grassland in small sections, once or twice per year, in order to have fresh grass in all seasons. In 1812 the amaNgwane tribe fled Shaka's army into the Drakensberg, but were soon uprooted by Shaka again. The Nguni population showed a dramatic decline during this period and when Retief crossed the Drakensberg in 1837 he encountered no human habitation between the Drakensberg and Port Natal (Durban) (Killick 1961).

In 1838 the Voortrekkers defeated Dingaan, Shaka's successor and declared Natal a republic. Thereafter grazing farms were awarded to trekkers in the foothills of the Drakensberg. Sheep farming was practised on a large scale and Killick (1963) described farmers from the Orange Free State sending their stock to the Natal Drakensberg for winter grazing up until the 1930s. Between 1849 and 1859 the government established five black locations along the length of the Drakensberg in order to absorb the influx of refugees from Shaka, as well as to create a buffer zone between white farmers and the San. These settlements continue to exist today (Bainbridge *et al* 1986). By 1890 the San were completely exterminated and the ecological role they played of preventing both Nguni and white pastoralists from grazing cattle in the Little Berg came to an end. In the late 19th century, the vegetation of the Little Berg was for the first time subjected to continuous selective grazing by domestic stock (Killick 1961).

2.2 Conservation in the Drakensberg

Although indigenous mammals were still found seasonally in the Drakensberg as late as 1880 (Killick 1961), by the end of the century, game was on the verge of total destruction. Between 1903 and 1967 a number of reserves were declared in order to protect game and timber (Bainbridge *et al* 1986, KwaZulu-Natal Nature Conservation Service 1999). Cathedral Peak was protected as a result of concerns about the exploitation of indigenous forests in the late 19th century. In 1927 an area of 32 246ha was declared the Cathedral Peak State Forest (KwaZulu-Natal Nature Conservation Service 1999) in order to protect it from illegal grazing, hunting and burning and in order to afforest it (Bainbridge *et al* 1986). The Drakensberg Catchment Reserve was declared in 1948 with the aim of protecting the catchment for water harvesting and this later became the Drakensberg Catchment Area (DCA). By 1963 most of the high areas of the Drakensberg from Mont-aux-Sources to Giant's Castle were protected as National Park, Forest Reserves and a Game reserve. The non-protected areas included private farms and the tribal reserves (Killick 1961).

During the first half of the 20th century all of the above-mentioned protected areas were grazed in summer by livestock both legally and illegally. In 1979 the practice of allowing "emergency grazing" for farmers in the State forests of the DCA was no longer accepted, though 1986 policy allowed for mowing of hay during emergencies (Bainbridge *et al* 1986). By the 1990s the Drakensberg was classified as Category II: National Parks and Equivalent Reserves under the IUCN system. The criteria for such classification were an outstanding natural area, provincially managed and dedicated to long-term conservation (Bainbridge 1991).

In 1993 the State Forests, Giant's Castle Game Reserve and the Natal Parks Board Reserves were consolidated under the Natal Parks Board (NPB) allowing for the establishment of the uKhahlamba-Drakensberg Park. All land within the Park was state-owned and protected under the KwaZulu-Natal Nature Conservation Management Act No.9 of 1997 and the Republic of South Africa National Forests Act No.84 of 1998. With the amalgamation of the NPB and the KwaZulu Department of Nature Conservation in 1997, the KwaZulu-Natal Conservation Service became responsible for management of the Park (KwaZulu-Natal Nature Conservation Service 1999). In 2000 the Park was listed as a mixed property World Heritage Site, meaning that it was one of 23 sites in the world with outstanding natural and cultural value (Ezemvelo KZN Wildlife 2001).

Management of the protected areas changed over the years in response to major social and scientific changes in conservation paradigms, as well as significant institutional change in the region. Reflecting a broader conservation history, the initial aim of reserves in the region was to protect game and indigenous forests. Grazing was allowed and afforestation was seen as desirable. Consciousness of the need to protect water catchments, led to the realisation that grazing ran the risk of loss of vegetative cover. The late twentieth century saw the emergence of the sustainable development concept which now had to be incorporated into management.

The San, the African tribes and the settlers burned the grassland frequently for both hunting and pastoral purposes (Granger 1976, Everson undated). With industrialisation and concerns about water supply the Soil Conservation Act of 1946 prohibited burning in the Drakensberg. This fire exclusion policy did not last very long. With wild fires and strong winds it was hard to enforce and a subsequent better understanding of the role of fire emerged. After 1948, annual or biennial burns were introduced and this appeared to create stability (Bainbridge *et al* 1986). Killick (1963) proudly described the fine sward of

T.triandra which covered the Little Berg as a result of this strategy. By the 1980s controlled block burning was practised on a 2 or 3 year cycle and alternately in early winter, mid-winter and spring. By the late 1980s this was challenged with concerns about genetic diversity and 1986 fire policy was to develop burning regimes which allowed "for the development of the full spectrum of plant and habitat diversity" (Bainbridge *et al* 1986: 64).

The first Drakensberg conservation policy was developed in 1981 (Bainbridge *et al* 1986). The primary management objectives included conservation of catchments for high quality water, soil conservation, rehabilitation of eroded areas, maintenance of mountain ecosystems and wilderness character and the promotion of research and monitoring. Land-uses that were considered incompatible with conservation, such as livestock grazing were to be eliminated. Staff were allowed to keep limited livestock but grazing in conserved areas was not permitted. Mounted forest guards controlled illegal entry, grazing, poaching and arson. There were recommendations that footpaths were appropriately constructed and maintained to avoid erosion. The Drakenberg was intended to provide an ecological baseline, for comparison with degraded catchments outside the protected area. Although the concept of sustainable development was not yet incorporated into policy, limited collection of forest products for building, medicine, utensils and food was permitted by local people as well as controlled trout fishing for tourists. Grassland products were not mentioned in the 1986 conservation policy.

By 1991 the sustainable utilisation of renewable resources was included in the mission statement of the NPB with the emphasis that utilisation needed to be controlled in order to be sustainable (Bainbridge 1991). In 1999 the mission of the KwaZulu-Natal Conservation Service was to "conserve the indigenous biodiversity of Kwa-Zulu Natal…and to ensure the sustainable use of the biosphere" (KwaZulu-Natal Conservation Service 1999:1). The management policy expanded to include neighbouring impoverished communities, who were to benefit from protected areas by having free access as well as employment opportunities (KwaZulu-Natal Conservation Service 1999).

Not only has management of the Drakensberg protected areas had to contend with changing paradigms regarding what is conserved, it has had to contend with significant changes regarding how to conserve. This has involved swings between various forms of preservationism and exploitation. And the pendulum continues to swing.

The concept of transfrontier conservation areas is the most recent development to make its mark on conservation approaches in the Drakensberg. In 1980 the IUCN recognized the Drakensberg protected areas together with Sehlabathebe National Park in Lesotho as being an ecological reserve of international significance and proposed the establishment of a single protected area (Bainbridge *et al* 1986). It had been a concern for many years that the area was inadequately protected. Lesotho had the lowest protected area coverage of any country in Africa (less than 0.4%), and the South Africa side of the mountain range had extensive unprotected areas (Maloti-Drakensberg Transfrontier Project 1999).

The Maloti-Drakensberg Transfrontier Conservation and Development Project (MDTCDP) was initiated by the South African and Lesotho governments, and incorporated the Maloti-Drakensberg range which stretches 300km along the eastern escarpment of Southern Africa. The area has global biodiversity significance (CSIR 1999, Ezemvelo KZN Wildlife 2001). Thirty percent of the plant species found in the area are endemics (Cowling and Hilton-Taylor 1994) and comprise mainly high altitude grassland forbs. There are also high levels of invertebrate endemism. The presence of palaeo-invertebrates in the high alpine tundra, possibly associated with unique plants is also of conservation interest (Ezemvelo KZN Wildlife 2001). Several endemic birds, fish and mammals are also found. The heterogeneous landscape and the isolation of high peaks are main contributors to the high endemism levels (CSIR 1999). Wetlands of the Maloti-Drakensberg play a significant role in catchment functioning for many southern African rivers (CSIR 1999, Ezemvelo KZN Wildlife 2001). Extensive degradation of wetlands in the lower-altitude montane areas has already occurred through overstocking and burning. Although there is less degradation in the higher alpine areas where most wetlands occur, wetland conservation remains an area of concern (CSIR 1999).

In 1999 the Global Environment Facility (GEF) awarded $16 million for implementation of the MDTCDP from 2002 to 2007. This was the single largest environmental grant ever received in Southern Africa. The South African and Lesotho governments signed a bilateral agreement declaring commitment to a joint future programme to conserve biodiversity in the region and to uplift the communities through nature-based tourism. This would be achieved through identifying the most important areas for biodiversity conservation, limited expansion of the protected area network, and the involvement of local communities in conservation and development (Maloti-Drakensberg Project 1999). Existing protected areas would be supported by appropriate land-use practices in the adjacent area and would be linked across communal lands (Ezemvelo KZN Wildlife 2001).

Although the thinking behind Transfrontier Conservation Areas emphasizes the non-consumptive use of resources such as tourism, most of the MDTCDP will continue to be grazed (Waddington 1999) as local populations depend on these resources for their livelihoods. Grazing associations have been proposed as strategies for improved grassland management (Maloti-Drakensberg Project 1999). The challenge lies in learning from past experience and finding effective ways to combine communal grazing with conservation.

Recognising that poverty is both a cause and result of environmental degradation, the agreement proposes community conservation programmes to promote alternative livelihoods compatible with biodiversity conservation e.g. developing local skills in conservation and tourism. The need for 'co-operative conservation" as well as comprehensive data bases is emphasised and a system of participatory monitoring of biodiversity involving local people has been proposed. The programme will have to focus on grassland management, path rehabilitation and wetland protection in communal lands outside the protected area (Ezemvelo KZN Wildlife 2001).

2.3 The lease land and the conservation land

The lease land is situated on the boundary between the Cathedral Peak protected area and the communal lands of the neighbouring amaNgwane tribe. The boundary fences have long been disputed by the amaNgwane (NPB 1989). In 1991 against a backdrop of overcrowding and overgrazing (NPB 1991), as well as a history of illegal grazing and stock impoundment in the Cathedral Peak State Forest (Thomson 1991; Bainbridge *et al* 1986), the amaNgwane Tribal Authority led by Nkosi Hlongwane made a formal request for a land swop. The request was to exchange an inaccessible piece of high lying tribal land adjacent to the Cathedral Peak State Forest for a lower lying area of 535ha within the Cathedral Peak State Forest (subsequently referred to as the lease land). This request was favourably viewed by certain members of the NPB as a way of boosting the support of the chief in the face of an ANC challenge as well as improving neighbour relations (NPB 1991). The Conservation Committee of the NPB supported the concept of a lease rather than a land swop. An annually renewable lease was proposed with a stocking rate of 4ha AU^{-1}, which amounted to 134 head of cattle in the area and no human settlement would be allowed (NPB 1991). This is a light stocking rate when contrasted with the recommended carrying capacity of 1.0 - 2.5ha AU^{-1} for fire climax grassland of potential forest areas (Tainton 1999b) and 1.4ha AU^{-1} for Bioresource Group 11 (Camp 1997). These figures would allow the community to keep between 214 and 535AU in the lease land (535ha).

The proposal was challenged from within the NPB because Cathedral Peak was a proclaimed nature reserve which precluded grazing. Communal grazing could be detrimental to water conservation, the lease could set a precedent that would strain conservation-neighbour relations in other areas and Southern Tall Grassveld was already poorly conserved (Scotcher 1991, Thomson 1991). If the lease option were pursued, it was emphasised that the lease conditions would have to be strictly controlled (Thomson 1991).

By 1994 the boundaries of the lease land were agreed upon by the NPB and the amaNgwane Tribal Authority and the Chief Conservator of the Drakensberg requested that the swop be "vigorously pursued" (NPB 1994) because the amaNgwane were extremely unhappy with the situation and were demanding immediate grazing rights (Inkatha Freedom Party 1994). In 1995 an interim agreement was signed allowing the community to bring their cattle in to graze (Lemmer 2001 *pers.comm.*).

The land is presently managed as a lease agreement between the present Ezemvelo KZN Wildlife (previously NPB) and the amaNgwane Tribal Authority. Regular meetings are held between the officer-in-charge and the cattle owners (Myeza 2003 *pers.comm.*). However, there appeared to be a discrepancy of perceptions within Ezemvelo KZN Wildlife. One staff member believed that the land swop was underway and in the hands of the Department of Land Affairs, another believed it was on hold while land claims were underway, and another believed it could never be anything other than a lease arrangement (Faure 2003 *pers.comm.*, Lemmer 2001 *pers.comm.*, Thomson 2003 *pers.comm.*).

In 2001 the lease conditions described by the second officer-in-charge included a limit of 500 head of cattle (almost four times the original figure of 134 head). No domestic grazers other than cattle were allowed in and grazing was only permitted from November to January and May to July. Cattle had to be out of the lease land by sunset each day. Cattle that grazed outside of the allowed times would be impounded (Lemmer 2001 *pers.comm.*). However, there appeared to be no standard stocking rate implemented. In 2003, the official limit for cattle numbers given by the conservation manager for the north uKhahlamba was 250 head (2.1ha AU^{-1}). The choice of grazing months was based on which months suited the community in terms of herding as well as winter feed needs, rather than on grassland science (Faure 2003 *pers.comm.*).

While the carrying capacity of fire climax grasslands has been determined as 1.0 - 2.5ha per animal unit (AU) (Tainton 1988), recommendations for Bioresource Group 11 are 1.4 ha per AU (Camp 1997). This gives the lease land of 535ha a carrying capacity of between 214 and 535AU. Proposed cattle figures for the lease land given by documentation and staff of the KZN Wildlife ranged from 134 to 500AU.

Harvesting of natural resources such as thatch grass (*Hyparrhenia hirta*), iNcema grass (*Juncus kraussii*), a building grass (*Merxmuellera*), imphephu (*Helichrysum*), wattle and gum trees was allowed in exchange for work at the Ezemvelo KZN Wildlife office or the donation of harvested material (one bundle of thatch grass had to be given to Ezemvelo KZN Wildlife for every bundle taken away). Illegal harvesting was discussed with local leaders and repeat offenders were prosecuted (Lemmer 2001 *pers.comm.*). Controlled thatch grass harvesting is also allowed within the conservation area (Faure 2003 *pers.comm.*).

It must be borne in mind that most of the conservation land along the conservation-lease fence-lines is not pristine montane grassland and not representative of the main conserved area. The conservation land here is considered marginal land that has a history of illegal grazing and disturbance through forestry activities e.g. sawmill and settlements (Everson 2003 *pers.comm.*).

The 1989 management plan for Cathedral Peak prescribed block burns for the conservation area every second year in spring (NPB 1989). In 2001, the officer-in-charge (Lemmer 2001 *pers.comm.*) reported that official burns took place every 2 to 3 years, although more frequent fires occurred due to arson. The burning objective was to maintain fire climax grasslands with vigorous and diverse vegetation (NPB 1989).

In 2001 the officer-in-charge (Lemmer 2001 *pers.comm.*) claimed that the lease land was burned every year by the community. This is in contravention of the National Forests Act No. 84 of 1998 which states that any person contributing to the threat of fire may be arrested and have their property seized (KwaZulu-Natal Nature Conservation Service 1999). It appeared that Ezemvelo KZN Wildlife had relinquished control over fire in the lease land. The Ezemvelo KZN Wildlife record of burns for conservation and lease land along the fence-lines are shown in Table 2.2. The three fence-lines are shown in Figure 2.2.

Table 2.2: Burning records for the three fence lines at Cathedral Peak: 1995-2002 (Van Zyl 2003 *pers. comm.*). Abbreviations: C= conservation land; L= lease land

	Fence-line 1	**Fence-line 2**	**Fence-line 3**
1997	No burn: L & C	Burn:L & C (except sites 5L and 5C)	Burn: L & C
1998	No burn: L & C	Burn: L & C (except sites 5L and 5C)	No burn: L & C
1999	Burn: L & C	L & C (except sites 3L and 3C)	No burn: L & C
2000	Burn: L & C *<u>But</u> the researcher observed that C had not been burned in 2000*	Burn: & C	No burn: L & C
2001	No burn: L & C	Burn: L & C	Burn: L & C
2002	No burn: L & C *<u>But</u> the researcher observed that L was burned in 2002.*	No burn: L & C *<u>But</u> the researcher observed that C was burned in 2002 and L may have had early winter burn*	No burn: L & C *<u>But</u> the researcher observed that the L & C were burned in 2002.*

Between 1997 and 2002, all the recorded burns except one were due to arson. It appeared that fence-line 1 had at least three consecutive years without burns, although this did not tally with official records. Fence-line 2 had been burned every year and the fence-line 3 had a maximum of three consecutive years without burning prior to the research. The researcher's observations in the field did not always correlate with official records (Table

2.2) nor with the OIC's claims that the lease land was burned every year. Records prior to 2001 may therefore also be inaccurate.

CHAPTER 3
Literature Review

3.1 Biodiversity

What is biodiversity?

Biodiversity emerged as a concept in 1986 during the National Forum on Biodiversity held in Washington D.C. By 1992 the concept had become one of the central concerns of scientists and politicians around the world (Wilson 1997).

Article 2 of the Convention on Biological Diversity defines biodiversity as:
"Biological diversity means the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and diversity of ecosystems" (Glowka *et al* 1994). Biodiversity or biological diversity is therefore the totality of life on earth, and the realisation that it is fast disappearing is the cause of all the attention.

Why is biodiversity important?

Biodiversity provides products such as food and raw materials that are needed for survival. However, biodiversity does more than that. It also maintains ecosystems through the maintenance of hydrological cycles, by purifying water, generating soil, pollination, maintaining balanced populations, recycling waste, and maintaining genetic libraries which hold unknown possibilities for the future (Beattie and Ehrlich 2001, Glowka *et al* 1994, Lovejoy 1997, Wilson 1992). Economists have estimated the cost of replacing the services of natural ecosystems as being in the range of US$33 trillion, although this would be impossible to achieve as many ecosystem services are irreplaceable (Costanza, d'Arge, de Groot, Farber, Grasso, Hannon, Limburg, Naeem, O'Neill, Paruelo, Raskin, Sutton and Van den Belt 1997).

With approximately 40% of the global economy based on biological products, biodiversity sustains livelihoods. Traditional societies often have the greatest reliance on biological products and services with some societies using more than 2000 plant species (Swerdlow

2000). The importance of biodiversity can be argued on the level of products and services provided. It can also be argued from the perspective of future loss. Wild plants, fungi and animals have provided the patterns for many modern medicines and are the origins of modern crops and livestock. Wild bacteria are used to clean up pollution in the process of bioremediation. Extinction means the loss of future discoveries and uses (Swerdlow 2000, Lovejoy 1997).

In addition there is the spiritual and ethical value of biodiversity. The apparently infinite array of plant and animal species, provides the fine detail in landscapes that people need for sanity, refuge, solace and re-fuelling. Although this "biophilia" or love of life and life forces has to date been underplayed by major religions and philosophies, it is here that renowned ecologist E.O. Wilson put his hope for life on this planet (Kellert and Wilson 1993:36).

The biodiversity crisis

We are in a period that has been described as the sixth great extinction which started in the middle of the 20th century (Myers 1997). The last great extinction occurred 65 million years ago, when earth was hit by an asteroid. Extinction is a natural process, but has been accelerated by human activity so that it is 1000 to 10 000 times higher than it would be without human interference (IUCN 2000). In the 19th century one bird or mammal was lost each year while the 20th century averaged three species per hour (Myers 1997).

South Africa has one of the highest populations of threatened plants for any continental area in the world (Cowling and Hilton-Taylor 1994). In addition, the overall biodiversity of South Africa is one of the most threatened in the world (Wynberg 2002). Large mammals are particularly threatened in Africa (Reid, Gardiner, Kiema, Maitima and Wilson 1999) while 14 to 37% of South Africa's plant, bird, reptile, amphibian, mammal and butterfly species are listed as threatened (DEAT 2002). It is likely that threatened plant species are under-estimated rather than over-estimated (Hilton-Taylor 1995).

Extinction is not only about the loss of large charismatic animals or highly valued plants. Extinction includes tiny unknown species, that are essential for ecosystem functioning (Ehrlich 1986). Three quarters of the identified animals on earth are insects (Armstrong undated). If all insects and arthropods were to disappear off the earth, "humanity could probably not last more than a few months" claimed renowned ecologist Wilson (1992:133). Insect species, the ultimate symbols of biodiversity, are lost by the thousands every year (Eldredge 1998).

Although it is unclear how many species we can lose before ecosystems fail to function any longer (Ehrlich 1986), global ecosystem strain is already manifest in the collapse of fisheries, deforestation and floods which have ruined economies and killed thousands of people. More than half the world's wetlands have been destroyed in the past 50 years, 80% of global grasslands are suffering from soil degradation and groundwater has been seriously depleted (Linden 2000). And unlike other environmental crises, biodiversity loss cannot be reversed (Wilson 1997).

There are two major reasons for the extinction: transformation of natural habitats and climate change. Transformation has been caused through urbanisation, industrialisation and monoculture, land-uses which do not favour diversity. Natural habitats, including protected areas, have become fragmented islands, where genetic diversity often cannot be maintained (Margules and Pressey 2000, Reid *et al* 1999, Myers 1997). It is estimated that about 16.5% of South Africa's land cover has been transformed and a further 10.1% degraded (Wynberg 2002).

Globally climates are changing due to global warming caused by the release of green-house gases. This may lead to the extinction of species, particularly sedentary species such as plants which cannot move to cooler areas. Climate change is not a new phenomenon, but climate change in combination with the fragmentation of natural habitats, where species are unable to migrate, may create the greatest biological crisis of all time (Lovejoy 1997). Maximising the size of protected areas is seen as one way of mediating the impact of climate change on biodiversity (Peters 1986).

Other threats to biodiversity include the invasion by alien plant species which colonise indigenous habitats (Lovejoy 1997). Approximately 8% of South Africa has been invaded by alien plants and most of South Africa's red data species are threatened by alien invasives (Wynberg 2002). Pollution, which damages ecosystem functioning is another serious threat (Lovejoy 1997). Scientific knowledge of biodiversity is incomplete, with only 10% of the earth's diversity identified. Species are disappearing faster than they can be identified, and with them a storehouse of wealth (Eldredge 1998).

Biodiversity conservation through protected areas is far from perfect. Reserves are often located in land that is economically unimportant e.g. too rugged or too remote, and that is not necessarily rich or representative in its biodiversity (Margules and Pressey 2000). Not only are reserves often poorly selected, many exist on paper only, while in reality they are being exploited for both subsistence and commercial purposes (Margules and Pressey 2000, Terborgh 1999). Emphasis on biodiversity conservation is fairly recent and conservation bodies often lack the knowledge and expertise to maintain or maximise biodiversity within reserves (Bond 1999). In addition many reserves are too small to maintain genetic diversity and viable populations of species (Margules and Pressey 2000).

The biodiversity of South Africa and KwaZulu-Natal

South Africa ranks as the third most biologically diverse country in the world (DEAT 2002) and is sixth in global floral diversity (Bond 1999). Although South Africa only occupies 2% of the world's surface, it contains 10% of the world's plant species and 7% of its reptiles, birds and mammals (Le Roux 2002). South African flora has exceptionally high diversity, and its species richness ranks in the top four of the 12 'megadiversity countries" of the world . It also has very high levels of endemism (Cowling and Hilton-Taylor 1994:33) and the highest known concentration of threatened plants as well as the highest plant extinction rates in the world. There has been an 80% increase in threatened plants in southern Africa between 1980 and 1995 (Wynberg 2002).

Invertebrate diversity is less well known than floral diversity although South Africa has developed a strong insect conservation research record in the last decade (McGeoch 2002). It is not known how many invertebrate species occur in South Africa and predictions are that species richness may be two or three times as much as the described number (Le Roux 2002, McGeoch 2002). Of the described species, there is a 70% level of endemism, meaning that approximately 42 000 species occur only in South Africa. Species of importance to this study include the Orders Orthoptera (grasshoppers and crickets,) which represent 10% of the global total of species; Araneae (spiders) which represent 8% of the global total of species, and are 57% endemic; Hemiptera (bugs, leafhoppers, cicadas) which represent 6% of the global total of species and are 60% endemic; Coleoptera (beetles) which represent 5% of the global total of species and are 57% endemic and Diptera (flies) which represent 4% of the global total of species and are 60% endemic (Le Roux 2002).

Floral diversity is not uniformly distributed over Southern Africa. Eight Centres of Plant Diversity (CPDs) have been described in Southern Africa. These centres have exceptionally high levels endemism as well as high levels of habitat loss (Cowling and Hilton-Taylor 1994). One of the CPDs is the Eastern Mountain region comprising the Natal Drakensberg and associated uplands. The Eastern Mountain region unit consists of temperate grasslands (CSIR 1999) and supports a number of endemic grassland forbs and low shrubs. There are no endemic trees in the region. Endemics are found in the families Asteraceae, Scrophulariaceae and Ericaceae. Major threats to the conservation of plant diversity in the Eastern Mountain region are overgrazing, agriculture, afforestation, plant harvesting and population growth. Afforestation is highlighted as one of the most significant threats in the summer rainfall CPD, and in particular afforestation of grasslands which contain most of the endemic species (Cowling and Hilton-Taylor 1994).

The flora of KwaZulu-Natal is particularly rich with endemics being mainly non-grass species. KwaZulu-Natal is home to 70% of the genera found in southern Africa and comes second after the Cape with the highest number of threatened and extinct taxa (Hilton-Taylor 1995). Approximately 16% of the flora of KwaZulu-Natal is endemic and 11% is rare and threatened (Scott-Shaw 1999).

3.2 The international and national context of biodiversity conservation

As the third most biodiverse country in the world and as a signatory to the Convention on Biological Diversity, global conservation trends impact strongly on South African biodiversity conservation. By 1992 the Convention on Biological Diversity had been accepted at the United Nations Conference on Environment and Development (Cameron 1994). It was signed by 150 states and entered into force in 1993 (Glowka *et al* 1994). The

Convention was a combination of the work of the World Conservation Union (IUCN); the 1972 United Nations Conference on the Human Environment in Stockholm, and the Our Common Future report of the World Commission on Environment and Development (Ngujuna 1994) and it brought the urgency of biodiversity conservation to the attention of the world. The main objectives of the Convention are:

- The conservation of biodiversity
- The sustainable use of biological resources
- The equitable sharing of the benefits arising from biological resources (Glowka *et al* 1994).

The Convention highlighted ideas such as the need for the in-situ conservation of ecosystems and natural habitats (Glowka *et al* 1994) as well as greater collaboration with people (Matowanyika 1994). Biodiversity conservation was recognised as a common concern of humankind (Glowka *et al* 1994).

National biodiversity conservation

Policy

As a signatory to the Convention on Biological Diversity South Africa was required to develop national strategies, plans or programmes in order to integrate conservation and the sustainable use of biodiversity (Glowka *et al* 1994).

The White Paper

South Africa began policy formulation with the 'Green Paper on the conservation and sustainable use of South Africa's biological diversity' which was published in 1996, and followed by a White Paper in 1997 (Kidd 1997). The White Paper on the Conservation and Sustainable Use of South Africa's Biodiversity (DEAT 1997) defined national policies and strategies for biodiversity conservation in South Africa. Goals listed in the White paper included:

- The conservation of South Africa's biodiversity
- The sustainable use of biological resources
- The development of human capacity for biodiversity conservation
- The creation of conditions and incentives that support the conservation and sustainable use of biodiversity (DEAT 1997).

In 1991 the Global Environment Facility (GEF) was established to assist poorer countries in implementing various conventions including the Convention on Biodiversity (WWF South Africa 2000). Between 1994 and 1999 South Africa received R128 million for environmental issues from GEF, the United Nations Development Programme and the United Nations Environment programme (Wynberg 2002).

The post apartheid South African government was keen to rid themselves of the historical baggage of conservation as the concern of white, privileged South Africans. The White Paper presented a strong development and anti-poverty approach, in line with global trends in sustainable development. However, with biodiversity primarily threatened by the side-effects of development, it is not clear how this will work. A National Biodiversity Strategy and Action Plan was announced by the Department of Environmental Affairs and Tourism in 2001 as part of consolidating policy for biodiversity conservation (DEAT 2002), but in 2003 it was still being developed. It appeared that political support for such a plan was lacking in spite of the urgent need for it (Wynberg 2002).

Protected area management

The White Paper and subsequent documentation challenged the reigning paradigms regarding protected areas. There was recognition that the current protected area system was inadequate for biodiversity conservation. A number of vegetation biomes were not adequately protected and genetic diversity was not necessarily being conserved in the protected areas (DEAT 1997).

Initiatives outlined in the White Paper for protected areas included promoting activities in adjacent areas that were compatible with conservation. This included the formation of partnerships with communities to manage resources both inside and outside protected areas and capacity building for social development. Initiatives for grassland conservation included the promotion of practices that maintain maximum species diversity.

DEAT (2002) aimed to establish a representative system of biodiversity protection and a new legal framework for protected areas. It committed itself to the expansion of protected areas from 6% to 8%, the development of World Heritage sites, the establishment of transfrontier conservation areas, the development of a network of biosphere reserves and an expansion of conservation efforts outside protected areas. The new policy on conservation endorsed sustainable utilization of biodiversity as well as the creation of low use buffer zones around protected areas.

Legal framework

A number of laws affect biodiversity conservation in South Africa. The Acts described below are relevant to the protection of an area such as the study site at Cathedral Peak. These Acts include:

- The World Heritage Convention Act (No 49 of 1999) which provides for the implementation of the World Heritage Convention in South Africa
- The National Environmental Management Act (NEMA) of 1998 which requires environmental impact assessments for various changes in land-use, with the aim of minimizing ecosystem disturbance and biodiversity loss (DEAT 1999). This Act is intended to incorporate biodiversity concerns at all levels of planning (Wynberg 2002). The Biodiversity Bill and the Protected Areas Bill (see below) fall under this Act
- The National Water Act No 36 of 1998 provides for the protection of water resources. It includes associated ecosystems and biodiversity (KZN Conservation Services 1999)
- The Conservation of Agricultural Resources Act of 1983 which provides for the conservation of soil, water and vegetation and for combating invasive plants. In terms of this Act government may grant assistance to land users for soil conservation works. Although the Act provides for severe penalties it is weakened by lack of enforcement personnel and does not apply in former homelands (Kidd 1997). This is obviously an issue of concern as there is a lack of legislation regarding land degradation in communal land
- The Mountain Catchment Areas Act of 1976 which recognizes that mountain catchments are sensitive areas and need to be conserved (DEAT 1999)
- KZN Nature Conservation Management Act No 9 of 1997 which provides for a KZN Nature Conservation Board which is responsible for ensuring effective conservation in the province. The Act provides for local protected area boards to enable community participation in decision-making as well as Community Trusts for channeling revenue into local communities (KZN Conservation Services 1999).

The Biodiversity Bill and the Protected Areas Bill.

Two new bills relating to biodiversity and protected areas were released in 2003, as part of the National Environmental Management Act of 1998. These were the National Environmental Management: Biodiversity Bill and the National Environmental Management: Protected Areas Bill.

The Biodiversity Bill provides for biodiversity conservation, the sustainable use of biodiversity, the equitable sharing of biodiversity benefits; international agreements on biodiversity; co-operative governance on biodiversity and the establishment of a National Biodiversity Institute (DEAT 2003a). Criticisms of the Bill include the inadequate participation of civil society including local communities in biodiversity conservation and use (Wynberg 2003).

The Protected Areas Bill includes the expansion of the protected area system from 6% to 8% and provides for extensive consultation prior to the declaration of new protected areas. The Bill includes the introduction of community participation in protected area management; including several transfrontier conservation areas (Grundlingh 2003). Key to implementation of these Bills will be ways for communities to benefit from conservation without threatening biodiversity.

Plans for biodiversity conservation

A variety of national plans that affect biodiversity have been developed by national government departments such as the Department of Agriculture and the DEAT. It appears that these two Departments are pulling in different directions when it comes to looking after the natural resource base. The National Strategic Plan for South African Agriculture (Department of Agriculture 2001) will impact on the way natural grasslands are viewed and managed. Although this plan includes sustainable resource management and the improvement of farmer expertise as core strategies, its main emphasis is the export market, increased production and increased inputs. Obviously, underlying this, will be pressure for increased land transformation and increased agricultural pollution. Strategies for the conservation of grasslands, soil and water are noticeably absent.

DEAT (2003b) was instrumental in the development of a National Action Plan to combat land degradation and to alleviate poverty in a response to the United Nations Convention to Combat Desertification (UNCCD) of 1994. The plan is ambitious, proposing to halve South African poverty by 2015 as well as preventing land degradation. Another undertaking by DEAT (2002) is the development of Transfrontier Conservation Areas (TCFAs) which are consolidations of existing protected areas with contiguous international borders. They are an attempt to put the sustainable utilization concept into action as the main emphasis of TCFAs is on multiple resource use and community based natural resource management rather than strict wildlife conservation (Wynberg 2002).

Significant milestones have been achieved in the regional and local planning arena.
Systematic conservation planning (Margules and Pressey 2000) has been used to develop regional conservation plans in the Western Cape (Cape Action Plan for the Environment) and in KwaZulu-Natal (SEA: Strategic Environmental Assessment plan). The aim of these plans is the conservation of a selection of habitats and species which will be both representative and able to survive in the long term (Margules and Pressey 2000). The SEA plan is intended to guide all local and regional planning initiatives. Under NEMA, national departments and provinces have to prepare environmental implementation plans and environmental management plans which should include biodiversity concerns. The plan does have limitations with a weak data base and the exclusion of non-endemic species, which may end up not being protected anywhere (Goodman 2001 *pers.comm.*).

In spite of the efforts of regional conservation bodies in developing such plans, putting them into practice has been badly neglected. It is hoped that the Integrated Development Plans (IDPs) of local municipalities as well as spatial development initiatives (SDIs) will provide an opportunity for incorporating biodiversity concerns at a local level (Wynberg 2002).

Programmes

A number of government programmes are either aimed at biodiversity conservation or have an indirect effect upon it. The Man and Biosphere Reserve programme of UNESCO was included as a strategy in DEAT's protected area plan (DEAT 2002). Its focus is co-operative management by conservation agencies and neighbours. Biospheres comprise three zones, the core protected areas, buffer zones where people may live and work, but support conservation aims and the transition zone which may contain agricultural activities and human settlements. Only the core protected area has legal status (Kidd 1997). Biosphere reserves emphasise community participation in management, as well as the role of research and education (Man and the Biosphere programme 2003). Although the idea was initiated in South Africa by the Natal Parks Board (Kidd 1997), by 2002 none of South Africa's four biosphere reserves were in KwaZulu-Natal (Man and the Biosphere Programme 2003). This programme is key to the new approach to protected areas and is in need of greater attention.

The Department of Agriculture has initiated programmes to address resource conservation and sustainable agriculture in communal areas but to date these have met with little success. The 1985 National Grazing Strategy was taken over by the Broadening Access to Agriculture Thrust (BATAT) (DEAT 1999) but with weak extension services it collapsed within a few years (Oettle, Fakir, Wentzel, Giddings and Whiteside 1998). The Landcare programme of the Department of Agriculture is presently the leading initiative for resource conservation in communal areas. It has been criticized for focusing on short term poverty relief rather than building the rights that would help create care of the land (Turner 2001) and even its own initiators have suggested it is inadequate for protecting the resource base (Department of Agriculture 2001).

The programmes described above could affect grassland biodiversity conservation but to date have had little impact. More resources are needed to strengthen both the Landcare and Man and Biosphere programme.

3.3 Conservation and development: changing paradigms

What is conservation?

Conservation is not a single static concept but rather a product of the social forces of the day. This section will examine the development of conservation and how it has changed both in South Africa and internationally in the past 300 years.

The first protected areas in both South Africa and the United States were an attempt to preserve the landscapes that were seen by the first European settlers (Carruthers 1995, Beinart and Coates 1995, Mentis 1985). This approach involved a sharp separation between conservation and people. Protected areas were set aside from people, and were places where people could go to see and experience nature (Matowanyika 1994). The irony is that the first settlers' perception of a pristine land ignored thousands of years of human influence in the landscape. This separation of conservation and people is reflected in the field of conservation biology, where human influence is often ignored or treated as a negative, intrusive factor (Nabhan *et al* 1991). There is a need for present day conservationists to be conscious of the myth of pristine landscapes as well as counter-myths that all indigenous people live in harmony with nature and have specialist knowledge of the environment (Cunningham 1994).

The present day situation is confounded by massive population growth, with greater human impacts on the natural environment than ever before. Not only was pre-settler human influence on the environment miniscule compared to the human impacts of the 21st century, but traditional African conservation practices have also been weakened over the past century by modernization, commercialisation and influxes of outsiders (Cunningham 1994).

The rise of biodiversity conservation

Conservation objectives have changed over time. Management for biodiversity is recent (Bond 1999) and led to disarray and confusion amongst late 20th century conservationists. Although it was generally accepted that the old focus on conservation of individual species was misguided, and that biodiversity should be conserved through the protection of ecosystems, habitats and species (CSIR 1999), by the late 20th century there were no general guidelines on how to achieve biodiversity conservation nor any single criterion for the selection or management of reserves (Linder 1994). Biodiversity conservation practices are still evolving with new approaches to reserve selection being developed (Goodman 2001 *pers.comm.*) and greater emphasis on flexibility in conservation management e.g. in burning regimes as well as in monitoring the effects of management on rare and endemic species (Bond 1999).

Protected areas have to date been the backbone of conservation strategies. However, there is now recognition that conservation outside protected areas is essential for maintaining ecosystem processes including the conservation of genetic diversity. High levels of endemism in South Africa mean it is not possible to proclaim enough land to conserve all biodiversity (Desmet 1999). If ecosystems are to function with species able to reproduce (Linder 1994), protected areas cannot exist as islands in a completely transformed landscape (Margules and Pressey 2000).

Sustainable development

Not only have there been profound changes in thinking about how to conserve the natural environment, there have also been profound changes in thinking around the relationship between people and conservation. At the 1972 United Nations Conference on Environment and Development (UNCED), the first "Earth Summit", preservationist thinking about the environment gave way to the need to include people, particularly the poor. In 1987 the concept of sustainable development emerged in the UNCED Brundtland report which argued that economic growth and environmental protection could co-exist (Pearce 2000).

The second Earth Summit held in 1992 led to the signing of The Convention on Biological Diversity. This document focused on the sustainable use of biological resources and the equitable sharing of benefits, highlighting the participation of people in biodiversity conservation. In spite of criticisms of the concept of sustainable development, it was still ardently espoused a decade later at the Third World Summit on Sustainable Development (WSSD) in 2002. This time round, reference to an 'Earth Summit' was dropped in favour of a 'World Summit'. More than ever, the concept had come to mean "development that is ongoing " rather than "development that is possible without destroying the environment". South Africa's conservation policy and legislation in the 21st century reflects the global acceptance of the sustainable development concept.

The concept of sustainable development has been criticized for being ambiguous, for not offering genuine solutions and for promoting the same policy of economic growth that caused environmental damage in the first place (Rist 1997). A wave of recent writings by conservationists have argued that sustainable development is impossible and that people-oriented approaches to conservation have failed to protect biological diversity. Brandon (cited in Wilshusen, Brechin, Fortwangler and West 2002: 27) called the sustainable development approach "politically expedient and intellectually appealing".

Integrated Conservation and Development Projects (ICDPs) in particular have come under attack. These are projects linked to protected areas with the purpose of providing alternative livelihoods so that local people will stop exploiting local resources. However, this has largely worked against conservation goals, with newcomers being attracted to the ICDPs and putting additional pressure on reserves (Terborgh 1999). The so-called "New Protectionists" argued that traditional African societies had never been natural conservationists and it is only their low population numbers that has protected the environment. Calls have been made for increased authoritarian protection of protected areas in developing countries (Wilshusen *et al* 2002).

Conservation and development in South Africa

South Africa's conservation history reflects the changing global conservation and development paradigms at the same time as including its own unique apartheid story. Woven throughout this history is the thread of sustainable utilization, indicating that it is not an entirely new concept nor an invention of the United Nations.

Soon after the first European settlers arrived South Africa, certain species were in danger of extinction. The introduction of legislation and the concept of sustainable utilization in the mid-nineteenth century failed to protect wildlife. The near extinction of game in the late nineteenth century (Carruthers 1995) led to the creation of the first protected area in South Africa in 1888 (DEAT 1997). There was no holistic land ethic, and the purpose of reserves was to protect timber for utilitarian use and game for sport hunting. Predators and species considered repulsive, such as crocodiles were eliminated by game rangers. The importance of habitat conservation rather than conservation of individual species was only recognised in the 1950s (Carruthers 1995). The selection of conservation areas for their representative landscapes and flora rather than for their eye-catching animals began to take place in the 1970s (Beinart and Coates 1995).

The reasons for protecting nature in South Africa have been numerous. The warden of the first national park in South Africa imagined "a national park would offer the opportunity to view wildlife as it existed in the continent previous to the arrival of the white man"

2002 protected areas were expanded by 155 000ha of land (Wynberg 2002). However the present protected area system is inadequate for the conservation of many endemic species (McGeoch 2002, Desmet 1999) particularly those found in the Eastern Mountain region CBD (Cowling and Hilton-Taylor 1994). Other concerns regarding the future of conservation is the decline in state funding for protected areas (Wynberg 2002, Bond 1999) with pressure for protected areas to commercialize and pay their own way. This leads to negative impacts from tourism, with job creation taking precedence over conservation, and deproclamation of unprofitable reserves. Although there has been a recent surge of foreign donor funding it is unlikely that these sources will be sustained. Other concerns include lack of capacity within conservation agencies to deliver on policy statements; a decline in taxonomists; the challenge of the 25 land claims that have been lodged in protected areas since 1999 and the lack of attention given to conservation in communal lands (Wynberg 2002).

There have been few incentives to encourage conservation outside of protected areas. A strong lobby from the Botanical Society is attempting to rectify this (Botha 2001) and in spite of the lack of state support, 16 million ha are already under private conservation (Wynberg 2002). Invertebrate conservation initiatives have identified that priority conservation areas fall largely outside of protected areas (McGeoch 2002). In KwaZulu-Natal 76% of landscapes are under-protected and biodiversity conservation has to include partnerships with private landowners (Goodman 2001 *pers.comm.*). Kellert and Wilson's (1993) notion of 'biophilia' plays a role here with potential for individuals to take on conservation out of their own love of biodiversity.

Other strategies for conservation outside protected areas include the establishment of corridors of semi-natural environment with less heavy use to allow the genetic flow of species between protected areas (West 1993). It is important that biodiversity outside of protected areas is used sustainably (Goodman 2001 *pers.comm.*, Cowling and Hilton-Taylor 1994, Desmet 1999). An example where the sustainable use of wild resources could be greatly enhanced is the informal trade in traditional medicinal plants. This presently supports a R60 million turnover per annum in KwaZulu-Natal alone but is largely dependent upon unsustainable harvesting practices. The future existence of many wild plants would be assisted with investment in medicinal plant cultivation programmes (Wynberg 2002).

The global nature of the biodiversity cause has increasingly played a role in South African conservation. There is a danger that the strong influence exerted by foreigners on biodiversity conservation could become an "ecological crusade" (Wynberg 2002:238) and could lead to global and national interests superseding local interests (Wilshusen *et al* 2002) and . With the DMTP being funded by GEF to the tune of $16 million it is important that the voices of local inhabitants are not drowned out.

Conservation after sustainable development

Rather than a stand-off between conservation and development, there is a need for open dialogue on the concept of sustainable utilization. A new orthodoxy, where "all protected areas must be sustainably utilized" replaces the old orthodoxy of "no utilization of protected areas" may not be the answer. The question of whether some circumstances require a protectionist approach needs to be tackled (Wynberg 2002).

Abandoning participation, argue Wilshusen *et al* (2002: 17) will be like 'reinventing a square wheel', a major step backwards. Lack of 'buy-in' from local people will lead to resistance and conflict, and does not augur well for conservation. A return to the preservationist approach will lead to lack of motivation to use natural resources sustainably or to conserve the environment (Turner 2001). Brechin, Wilshusen, Fortwangler and West (2002) and Wilshusen *et al* (2002) argued for taking the middle ground. While emphasizing the need for the protection of reserves, they proposed that this be achieved through negotiation and the creation of legitimate and enforceable agreements. They recommended 'ecologically sound, pragmatically feasible and socially just programmes' based on strong agreements with all affected parties" (Wilshusen *et al* 2002: 18). Although traditional and local institutions may not have the will or the power to enforce conservation, they need to be recognized and strengthened in order to create opportunities for conservation partnerships. Secondly, they argued for recognition of local uniqueness and the need to deal with each situation individually. Blueprints with rules about strict protection *vs.* sustainable use will not conserve biodiversity. Thirdly they highlighted the need for greater collaboration between social and natural scientists in conservation (Brechin *et al* 2002). Scientific approaches alone are not enough to save biodiversity and there is a need to incorporate socio/political aspects, local knowledge and ecological understanding into biodiversity conservation (Brechin *et al* 2002, Cunningham 1994, Njuguna 1994) as well as incorporating a love of biodiversity into prevailing religions and philosophies (Kellert and Wilson 1993).

3.4 Grasslands and biodiversity conservation

Globally grasslands are used mainly for animal production and due to better animal husbandry, grasslands are becoming over-utilized (Linden 2000, Barbier, Burgess and Folke 1994). Less than 2% of grasslands are conserved world wide with a similar figure applying in South Africa.

The Southern African grassland biome

In Southern Africa the grassland biome occurs on the high central plateau and east of the escarpment in KwaZulu-Natal and the Eastern Cape (Low and Rebelo 1996). It covers areas that are fairly cool with moderately good rainfall (Tainton 1988). The altitudinal range is sea-level up to 3300m and rainfall varies from 400 to 1200mm per annum. Temperatures range from frost-free to snow-bound in winter (O'Connor and Bredenkamp 1997).

Grass-dominated vegetation types can be classified into three categories: grasslands, savannah and tundra. Savannah has an upper layer of woody plants but true grasslands consist of a single-layered herbaceous community dominated by perennial grasses (Low and Rebelo 1996). Grasslands are cooler than savannahs during the non-growing season (O'Connor and Bredenkamp 1997).

Unlike many other vegetation types, unprotected grasslands still largely support indigenous vegetation although all have experienced some degree of ecosystem change (Bond 1999). Over the past decade there has been a decline in grazing areas in all provinces except for the Free State (Wynberg 2002). Remaining grassland ecosystems are threatened by overstocking, overgrazing, the spread of alien plants and subsistence and commercial harvesting of indigenous plant products (Bond 1999, Hilton-Taylor 1995). Grasslands in high rainfall areas are especially threatened by agricultural intensification (Watkinson and Ormerod 2001). KwaZulu-Natal has the highest level of land transformation in Southern Africa (Scott-Shaw 1999).

Grassland productivity is linked to rainfall as well as soil type (Behnke and Scoones 1993) with seasonal fluctuations in productivity most pronounced in drier areas (Tapson 1993). Moist grasslands, with rainfall over 800mm per annum tend to have sour grasses, with high plant canopy cover, high production and high fire frequency (Uys 2000).

The high species richness (3370 plant species) and endemicity of montane and highveld grasslands has been attributed to heterogeneous landscapes as well as an evolutionary history, beginning over 18 000 years ago (Le Roux 2002). The most dominant species in Drakensberg grasslands are grasses, but there is a greater richness of forbs (O'Connor in press, Tainton 1988). Most endemic plants in the Eastern Mountain region are grassland forbs and low shrubs with very few endemic grasses (Cowling and Hilton-Taylor 1994).

Grassland ecology

Equilibrium and non-equilibrium grasslands

Research in grassland ecology has focused predominantly on agricultural productivity (Uys 2000), with attention given to grass species and large herbivores. The Clementsian Succession Model based on theories of ecological succession in the climatic-climax grasslands of the North American prairies (Tainton and Hardy 1999) has guided thinking on vegetation change in grasslands for half a century (Stafford-Smith and Pickup 1993). In this model, vegetation progresses linearly from disturbed vegetation through to a single climax composition. Grazing pushes the system to the disturbed end of the continuum while resting allows progression toward the climax, which was accepted as the ideal situation. However the fire-climax Drakensberg grasslands are more complex with the exclusion of fire and grazing leading towards a climax of woody species (Tainton 1999b).

The 'state and transition' model (Westoby, Walker and Noy-Meir 1989) allowed for multi-directional changes in community composition rather than just moving in one direction. This model was proposed for non-equilibrium arid and semi-arid grasslands. It was argued that rainfall determined stock numbers, and played a greater role in vegetation change than stocking rate, which was rarely high enough to cause irreversible damage. Grasslands could move through a variety of different states, rather than progressing linearly to a single climax state, and the removal of a disturbance did not mean that linear progression would necessarily be resumed (Scoones 1995).

A number of ecologists in the early 1990s argued that a vast body of contradictory research results had been generated with a continued lack of clarity about the effect of different land-use practices (Stafford-Smith and Pickup 1993). The concepts of carrying capacity, benchmark sites and degradation were challenged (Scoones 1995, Cousins 1995) and it was argued that loss of key species did not necessarily lead to soil erosion nor did it necessarily mean long-term reduction in animal production (Tapson 1993).

The management implications of the state and transition model were more complex than the Clementsian model which guaranteed recovery if disturbance was removed. Now interventions had to be individualistic, taking into account complex interactions between soil, plants, grazing and climate (Stafford-Smith and Pickup 1993). With productivity of non-equilibrium grasslands being very variable over space and time, flexible movement or opportunistic strategies were needed for optimum utilization (Cousins 1995).

Moist grasslands, however were still accepted as equilibrium systems following the classical model where vegetation change was gradual, livestock populations were limited by available forage and stocking above carrying capacity led to degradation. However the distinction between equilibrium and non-equilibrium systems was often not clear, with non-equilibrium patterns emerging in moist grasslands during dry periods, and equilibrium pattern systems operating in wet areas within dry zones (Scoones 1995).

Disturbances

Another theory to impact on grassland ecology and management is the intermediate disturbance hypothesis (IDH) which proposes that species diversity is highest at intermediate levels of disturbance. At high levels of disturbance only species that tolerate extremes survive, while low levels of disturbance can cause a decline in diversity due to the dominance of certain species (Uys 2000). Disturbance theory has been little tested in savanna (Shackleton, Griffin, Banks, Mavrandonis and Shackleton 1994) and Uys (2000) found no support for the IDH in terms of the Cathedral Peak burning treatments. However grassland research has provided mixed results regarding the impact of disturbances such as fire and grazing. These have been shown to both increase and diminish diversity, and the intermediate level of disturbance may offer some explanations for this phenomenon.

Grazing and biodiversity

There is evidence that disturbances such as grazing can increase plant diversity (Mckenzie 1987, Shackleton *et al* 1994, Hadden and Westbrooke 1999, Niamir-Fuller 1999, Pykälä 2000) and species richness (Todd and Hoffman 1999). Grazing prevents grasses from becoming moribund and dying out (Tainton 1999a) while too little grazing can lead to a decline in abundance of palatable species in grassland or succession to woodland (Watkinson *et al* 2001, Allsopp 1999, Holden 1995, West 1993, Tainton 1999a). Ecologists rejecting the equilibrium model argued that heavy grazing in the form of constant high stocking rates in moist and mesic grasslands resulted in little change in species composition, basal cover or primary productivity and that recovery after drought or rest was rapid (Shackleton 1998). Heavily grazed areas have also shown increased resilience with more climax species and higher basal cover than moderately grazed areas (Cousins 1995).

In Europe domestic herbivores may have taken over the ecological niche of wild herbivores eliminated in the late Pleistocene, which provided disturbance and seed dispersal functions (Pykälä 2000). In Africa domestic and wild ruminants are co-existing in places such as Masai Mara Reserve in Kenya (The Natal Witness 2002) and conservancies in Namibia (Brown and Jones 1999). Proclaimed wilderness areas in USA allow for domestic grazers if they were present at the time of declaration (McClaran 2000). In South Africa, grasslands, particularly savannas and highland sourveld have had high levels of human use over centuries, and this has determined their present structure (Kotze, Twine and van Rensburg 1999).

Other studies have shown that grazing leads to a decline in diversity, especially of rare species with increasing disturbance (Bond 1999, Shackleton *et al* 1994). Grazing affects plant growth and community composition through defoliation, and results in increased competitiveness of less palatable species. It is widely accepted that productivity levels decrease when palatable climax species such as *T. triandra* are replaced with unpalatable species such as *Aristida* (Tainton 1999a, Tapson 1993).

Soil properties can influence grassland response to grazing, while uprooting, trampling, and deposition of manure can alter soil and hydrological properties (Holden 1995). Grazing on heavy soils can lead to compaction, increased water run-off, decreased infiltration and changes in vegetation while sandy soils are less vulnerable (Walker 1995, Behnke and Scoones 1993). Soil changes through nitrogen deposits in manure can affect vegetation composition in grasslands (Watkinson and Ormerod 2001) with increased nitrogen in infertile soils leading to domination by nitrogen loving plant species. Mycorrhizal activity in soil can be diminished by heavy grazing and this may also lead to changes in plant composition as different plants have different levels of dependence on mycorrhizas (Allsopp 1999).

Recent work in Drakensberg grasslands found higher vegetation diversity in a single site which was completely protected from grazers compared to grassland in communal, commercial or conservation systems all of which had been grazed. The study also found vegetation diversity in communally grazed land to be equivalent to the diversity in commercial and conservation grasslands (containing indigenous grazers). With grazing impacts reaching unprecedented levels however, forbs are more vulnerable to damage than grass species (O'Connor in press) and this is where most of the plant diversity lies.

Little is known about the effect of different grazing regimes on the composition and abundance of invertebrate species (McGeoch 2002, Hadden and Westbrooke 1999). The broader implications of land-use impacts on invertebrates is difficult to generalise without an understanding of habitat requirements of particular invertebrate groups (Bond 1999). However it is likely that micro-fauna play an important role in grassland ecosystems (West 1993). Fence line studies on grazing impacts have shown similar species richness in communal and commercial grazing (Todd and Hoffman 1999); that invertebrates are affected by intensity of trampling but with no difference between game or stock trampling (Rivers-Moore and Samways 1996) and that there can be exponential increases in the abundance of invertebrates in grazed areas with domination by a few generalist species (Todd, Seymour, Joubert, and Hoffman 1998). Species from the orders Coleoptera, Hymenoptera, Diptera and Thysanoptera as well as Curculionid beetles have been shown to decrease in grazed areas (Hadden and Westbrooke 1999).

While grazing has been shown to prevent shrub encroachment in savanna, heavy grazing can lead to increased shrub encroachment (Roques, O'Connor and Watkinson 2001). Domestic grazers in savanna can cause shrub invasion by decreasing competition from grasses, reducing fire intensity and frequency and by distributing seed through manure

(Roques *et al* 2001). Unlike domestic stock, indigenous fauna included both browsers and grazers providing "wide-spectrum grazing" which would have reduced the competitiveness of shrubs. With most browsers eliminated by early settlers, grazing patterns were significantly altered even in protected areas (Tainton 1988:21).

Fire and biodiversity

Fire is a key ecological process in grasslands and plays an important role in both grazing and conservation practices. Fire is caused naturally by lightning, and montane grasslands with high fuel loads coincide with a high level of lightning strikes (Everson 1999). There have been drastic changes in fashion around the use of fire during the course of the 20th century. Attempts were made to practice fire exclusion in the Drakensberg grasslands in the mid 20th century but subsequent approaches included regular burns to prevent succession in the 'false' grasslands. Acocks (1953) argued that the grasslands were less than 600 years old and had previously been occupied by forests. Tainton (1999c) accepted that fire had existed in this landscape for millenia. Late Stone Age people used fire to maintain grasslands 40 000 years ago but their impact was very limited. By the Holocene, 11 000 years ago human use of fire is believed to have been significant and grasslands were widespread in southern Africa (Hall 1984). Human influence in the past 300 years has aimed to create a uniform grazeable sward and increased human populations have led to significantly increased fire frequency (Uys 2000, Bainbridge, Scott and Walker 1986).

Shrub encroachment is controlled through fire, but if the fuel load is decreased by heavy grazing, fire is less effective. Global increases in bush encroachment have been explained by increased grazing and decreased burning (Roques *et al* 2001). The frequency and intensity of fire affects different species in different ways. The *T. triandra* grassland of this study has been classified as a fire subclimax grassland, a relatively stable community prevented from successional development by the presence of fire. It was proposed that fire exclusion would lead to the disappearance of *T. triandra* while annual and biennial burns would lead to domination by this species (Killick 1963). More recent research in the Cathedral Peak burning trials has shown that plant diversity was more affected by environmental gradients than by burn treatments (Uys 2000). In addition this research showed that no single burn treatment maximised plant biodiversity adding evidence to the need for a mosaic of burning practices (Short 2001, Uys 2000).

Research in invertebrates and fire has shown a decline in ant species richness with fire exclusion and a negative correlation between species richness and amount of foliage biomass (Parr, Bond and Robertson 2002). Grassland micro-organisms have also been shown to be affected by fire. Mycorrhizal fungi, which have a critical relationship with 90% of green plants can become extinct as the result of frequent fire (West 1993).

Other disturbances

Disturbances such as fertilising and mowing have led to domination by common and less specialist invertebrate species (Di Giulio, Edwards and Meister 2001). Invertebrate studies show a sensitivity of arthropods to heavy land-use, with 23% to 45% of arthropods being unique to protected areas (Bond 1999). Scarabaeidae species which co-exist with large native mammals are severely affected in communal land (Rivers-Moore and Samways 1996). Bird biodiversity in communal areas is affected by loss of nesting sites due to firewood collection (Bond 1999).

Grassland degradation

The sustainable utilization of grasslands inevitably raises the question of degradation. This section will look at definitions of degradation, the level of degradation in South African grasslands and the link with biodiversity conservation.

Grassland degradation is usually defined in terms of irreversible loss of primary and secondary production with little reference to biodiversity (Holden 1995, Behnke and Scoones 1993). The concept of degradation is controversial. A lack of long-term data means it is unclear whether changes in grasslands are caused by human activity, by the seasonal fluctuations of the non-equilibrium model or by long-term climate change (Scoones 1995, Niamir-Fuller 1999).

Conventional thinking on grassland degradation is that it begins with ground cover destruction, leading to soil and nutrient loss and secondary succession by pioneer species. The process feeds on itself with slow growth on bare soil, reduced water quality, and more grazing pressure due to poor quality forage. In fire-climax grasslands there tends to be a change in species composition, to less palatable species rather than decline in basal cover (Holden 1995, Tainton 1999d). Moist grasslands tend to stabilise with a new range of species after they have lost about 50% of their productivity (Owen-Smith 1998). Rangeland degradation was conventionally measured in terms of both vegetation changes as well as declining animal numbers (Vetter, Bond, and Trollope 1998). Ecologists espousing the non-equilibrium model proposed that livestock productivity and sustainability of the system were more important indicators (Behnke and Scoones 1993). Although ecological factors were taken into account in these measures, both approaches ignored the role of ecological diversity in the sustainability of ecosystems.

Rangeland degradation has been a concern since the mid-1800s. Cattle paths and selective grazing caused by overstocking and kraaling by white settler farmers concerned the authorities, but African peasant farmers soon followed suit when they were able to increase cattle numbers with migrant wages. The 1922/3 Drought Commission found that decreased production was not caused by drought but by land degradation. Government responses were to promote fewer better animals through stock improvement, grazing rotation and cattle sales. African peasant farmers however, argued for more land not less cattle. The betterment schemes of the late 1930s in communal areas introduced conservation practices such as camping systems which allowed for grassland resting. However, because the schemes were authoritarian with no educative role, they met with widespread resistance, leaving a legacy that conservation was impossible without coercion (Beinart 1998). In the late 20th century emphasis on soil erosion alone shifted to a more holistic, participatory land management ethic which linked conservation and production (Critchley and Netshikhovela 1998). The new ethos was incorporated into government programmes such as Landcare (Directorate Resource Conservation 1998) which was aimed predominantly at communal land-users.

The communal areas have been classified by agricultural officers as the most degraded land in South Africa (Hoffman and Todd 2000) with KwaZulu-Natal ranking second in soil and rangeland degradation (Hoffman and Ashwell undated). Surveys of grass species productivity and palatability in KwaZulu-Natal indicated that 46% of sites had low productivity and an erosion problem while 65% of sites had inadequate basal cover resulting in excessive rainfall run-off (Camp and Hardy 1999).

Management for biodiversity conservation

When the 'non-equilibrium ecologists' challenged the desirability of a single stable climax state and replaced it with the possibility of a variety of stable states, they argued that the most desirable state was purely a function of management objectives (Behnke and Scoones 1993, Tapson 1993, Mentis 1985).

With the rise of biodiversity conservation, approaches to grassland conservation had to be reconsidered. Drakensberg grasslands had been managed to maximise the palatable *T. triandra* grass (Bond 1999) and minimise forbs (Tainton 1999d), creating the ideal conditions for cattle production, in the absence of a beef herd. More complex biodiversity conservation objectives needed to be taken into account.

There is evidence that no single management approach will conserve grassland biodiversity. It is generally recognised that heterogeneity of landscapes is essential for ecosystem functioning and biodiversity is maximised with a mosaic of vegetation patches at different successional stages (Parr *et al* 2002). A variety of microclimates and grassland structures helps invertebrates to find suitable habitats and food and allows dispersal to take place, thus supporting species diversity at a regional level (Di Giulio *et al* 2001). Because relatively little is known about the impact of different management regimes on grassland plant diversity (Short 2001, Uys 2000), the application and monitoring of a mosaic of management practices including irregular point-source burns and moderate levels of grazing is most likely to create heterogeneity and offers potential for biodiversity conservation (Reid *et al* 1999).

There is evidence to suggest that limited grazing may be beneficial to biodiversity. There is also a trend in conservation to make use of a range of management practices in order to maximise biodiversity and to allow for habitats at different successional stages. With these arguments in mind, grazing cannot be ruled out in protected areas that aim to conserve biodiversity.

3.5 Communal grasslands and biodiversity conservation

Communal grasslands could play a significant role in biodiversity conservation. Although this land has been perceived as the most degraded land in South Africa, it has been under the least threat of complete transformation through monocropping (Bond 1999). The semi-natural state of communal grasslands has the potential to provide corridors for biodiversity conservation as well as buffer zones between protected areas and more intensive agriculture (West 1993). The uKhahlamba-Drakensberg Park is predominantly surrounded by communal grasslands, a situation that could provide conservation opportunities in the form of buffer zones but also conservation threats as impoverished communities put pressure on the protected area.

With multiple resource use as an aim of the Maloti-Drakensberg TCFA ways need to be found for allowing communal use of protected areas without threatening biodiversity. An understanding of the present status of communal grasslands in South Africa is helpful in developing opportunities and minimizing threats.

Communal areas and the natural resource base

The communal areas or former homelands are impoverished, overpopulated and poorly serviced (Walker 2002). Seventy five percent of South Africa's poor live in rural areas, with almost half of the poor relying on crop or livestock production (Oettle Fakir, Wentzel, Giddings and Whiteside 1998). Historically 87% of the population was allocated to 13% of the land (Hall and Williams 2000) and access to land continues to be racially skewed.

There is no 'single, pure, truly authentic and unchanging African tenure system' (Walker 2002:4) and communal tenure in South Africa continues in a state of flux as it has been for centuries. While a variety of tenure systems such as open access, communal management and private management, exist (Scoones 1995), the reality is that most South African communal land is now open access rather than being managed as common property (Ainslie 1998).

Communal land has seen a tenfold increase in human populations over the past 100 years and the replacement of subsistence farming with a cash economy (Bond 1999). The natural resource base including grasslands has been severely impacted by these trends. Although traditional societies had controlled natural resource utilization (Boonzaaier *et al* 1990) today rural communities are in transition between traditional culture and industrialization. Whereas traditional societies relied exclusively on natural resources and enforced strong controls over their use, today rural societies have urban links and a cash economy and with less dependence on natural resources, there is less commitment to natural resource management (Kotze, Twine and Van Rensburg 1999, Ainslie 1998). Local institutions have weakened and indigenous knowledge is disappearing. Market penetration has resulted in resource extraction and pressure for privatization. In KwaZulu-Natal over 400 species of medicinal plants are collected (Hutchings 1996), many illegally and local extinctions have been recorded in the Drakensberg (O'Connor in press).

Although it commonly believed that communal land is unproductive (Boonzaaier *et al* 1990), communal grasslands support multiple livelihood strategies by providing medicinal plants, wild foods, thatch and firewood in addition to grazing (Cousins 1995, 1998) and have been shown to provide yields ten times higher than commercial grasslands (Oettle *et al* 1998).

Stock systems in communal grasslands

When weighing up the opportunities and threats posed by grazing in protected areas it is useful to take into account existing approaches to grazing on communal land. Traditionally livestock in Africa are a symbol of wealth and status. In KwaZulu-Natal the local Nkosi grants grazing rights and there are no fees or restrictions on stocking rate (Thobela, Lax and Oettle 1998). Livestock plays a significant role in rural livelihoods (Beinart 1998) although it is insufficient to meet subsistence needs (Scogings, De Bruyn and Vetter 1999). Cattle ownership is generally skewed with few and often absentee owners (Scogings *et al* 1999, Vetter, Bond and Trollope 1998, Von Maltitz 1998, Cousins 1995, Tapson 1993) and it is often the wealthier people who gain the most from communal grasslands (Scogings *et al* 1999). Nearly all pastoral people are affected by limited access to land (Scoones 1995).

Offtake in subsistence pastoralism is in the form of milk, transport, bride-wealth and traction and does not require slaughter, making higher stocking rates more profitable. A high mortality rate is not altogether disastrous, as meat is used even after death. Cattle sales are low (Cousins 1995) with marketing usually occurring only during drought periods and accumulation taking place during wetter periods (Scoones 1995). The stocking rate on communal grasslands is double (Bond 1999) or treble (Cousins 1995) that of commercial farms. Stock numbers in KwaZulu-Natal and the Transkei have been stable for many years (Cousins 1995) or increasing (Von Maltitz 1998). However, if land transformation is accounted for, then stable stock numbers actually means that grazing pressure is increasing (Scogings *et al* 1999).

The current approaches to stock keeping in communal areas appear to present a scenario of limitless needs. This is enough to make concerned conservationists erect high fences, and yet there is political pressure for sustainable utilisation of protected areas. The question remains: Is it possible to marry communal grazing with biodiversity conservation?

Communal grasslands and degradation

Ever since Hardin (1968) wrote of the tragedy of the commons, reasoning that it was in the economic interest of the individual to increase the size of his herd, it has been the mainstream view that communal grazing systems have led to overgrazing and soil erosion (Todd and Hoffman 1998, Critchley and Netshikovhela 1998) and low production (Cousins 1995, Boonzaaier *et al* 1990). The 'non-equilibrium ecologists' challenged these perceptions in the 1990s, arguing that they were alarmist. They argued that hard evidence for accelerated soil loss was lacking, and that soil loss was well below accepted levels. There was no obvious decline in stock numbers during the 20th century (Bond 1999) and stock numbers in KwaZulu-Natal increased between 1974 and 1987 by 19%. This led to arguments that the original carrying capacity had been under-estimated because ecological collapse had not occurred (Tapson 1993).

Comparisons of communal vs. commercial and protected grasslands have shown mixed results. Many studies have not found evidence of degradation in communal grasslands. Communal grasslands have been found to have higher basal cover due to domination by creeping grasses (Beinart 1998, De Bruyn 1998, De Bruyn, Goqwana and van Averbeke 1998, Goqwana 1998, 1999, Shackleton 1998). The absence of selective grazing in communal grassland has resulted in more palatable grasses (De Bruyn *et al* 1998, Shackleton *et al* 1994). Some studies have shown little difference in species richness, (Todd and Hoffman 1999, Shackleton 1998, Fabricius and Burger 1997, Venter, Liggit, Tainton and Clarke 1989), soil erosion (Shackleton 1998, Venter *et al* 1989) and herbage accumulation (Venter *et al* 1989) in communal grasslands vs. commercial and protected grasslands. Resting of communal grasslands has been shown to promote rapid improvement in plant productivity with an increase in palatable species (Camp and Hardy 1999, Shackleton 1998, Mckenzie 1987) and species composition (Shackleton 1998).

Other studies have found higher levels of degradation in communal vs conservation or commercial grasslands. Comparisons of five communal grasslands with commercial and conservation grasslands found that in all cases the proportion of palatable species in communal grasslands had declined when compared to the other grasslands. Two of the five communal sites were severely degraded: one site which had previously supported livestock was now dominated by woody shrubs with significantly lower biomass and structural diversity, and another site was reduced to "unvegetated mobile sand dunes" (Palmer, Ainslie and Hoffman 1999: 1022). Other studies have shown a significant decline in palatable species such as *T. triandra,* succession towards pioneer species such as *Aristida* and *Eragrostis* in the Eastern Cape, replacement of perennial grasses by annuals or bare soil in the Limpopo province and a general invasion of communal areas by unpalatable woody shrubs as well as gully and sheet erosion (Bond 1999). A national review on land degradation, based on the perceptions of agricultural extension staff and technicians found communal grasslands to be the most physically degraded areas in South Africa (Hoffman and Todd 2000).

The study by Palmer et al (1999) found examples of good and poor condition grassland in all three categories (communal, conservation and commercial). These mixed results highlignt the need to examine grasslands on a case by case basis rather than making a blanket condemnation of communal grasslands. Overgrazing in communal grasslands has caused dramatic losses of biodiversity, especially of rarer species (Todd and Hoffman 1999, O'Connor in press). A comparison of communal and protected areas in the Eastern Cape showed substantially higher abundance of unique plant types as well as useful plants in the protected area. Conservation appeared to provide an important refuge for some species (Fabricius and Burger 1997). Medicinal plants in the Drakensberg are particularly threatened by illegal and excessive harvesting (Venter 1998, Bond 1999).

The Drakensberg communal areas in particular are prone to soil degradation with their steep slopes and erosive soils (Hoffman and Todd 2000). Heavy grazing has resulted in rill, sheet and gully erosion removing much of the soil from the mountain slopes (Bainbridge *et al* 1986).

Development in communal grasslands

International grassland assistance began in sub-Saharan Africa in the 1970s with the implementation of the ranch model using commercial farming methods such as fencing, rotation, limiting stock numbers and changing tenure from communal to group or private. The system is seen to have failed widely with little compliance in destocking (Abel 1993, Behnke and Scoones 1993), and the end result was open access replacing common property regimes (Niamir-Fuller 1999). In the 1990s the development paradigm emphasized participatory, holistic and local approaches and community based natural resource management (CBNRM) emerged. However, this has been criticized as ineffectual and often people have little incentive to become involved, or will resist common property regimes which restrict access to resources. (Ainslie 1998).

South African state agriculture policy shifted its focus from 1994 onwards towards small-scale black farmers but its achievements have been largely unsuccessful (Scogings *et al* 1999, Oettle *et al* 1998). State responses to communal grassland management have tended to be that of avoidance of an insoluble problem. In KwaZulu-Natal, attempts at management or law enforcement have been largely ineffective and field days poorly supported (Letty 2003, Mitchell 2002). There is a policy of promoting resting, rotation and the correct use of fire. A four camp system with each camp rested every four years and used for winter fodder in conjunction with licks, was proposed for the Drakensberg area. However implementation of such policies is weak with lack of co-operation from livestock owners and issues of stock theft in the furthest grazing areas (Mitchell 2002). Livestock Associations have been established but are involved in little more than managing dip-tanks. However, they are seen as a way of maintaining links between the Department and the community (Letty 2003).

Imminent changes to the legal framework of communal land may impact on communal grasslands. The Communal Land Rights Bill gazetted in 2002 aimed to provide legally secure tenure to the 15 million people living on communal land (Department of Agriculture and Land Affairs 2002). With the emphasis on privatization there are fears that it will exacerbate inequalities and poverty by removing the safety net of traditional access to land (Mkhabela 2002). There are also concerns that although the Bill provides for demarcation of communal areas within community land, they are not protected strongly enough (Walker 2002).

There are arguments that favour local common property regimes and suggest that tenure insecurity and inequalities in land ownership cause degradation, rather than communal land use *per se* (Critchley, Versveld and Mollel 1998, Oettle *et al* 1998, Hoffman and Todd 2000). Ironically the present state emphasis on privatization emanates from a European legal system, while in Europe today many countries have well functioning common property regimes and there are moves to re-instate common pastures in the Alps (Niamir-Fuller 1999).

Whether the Communal Land Rights Bill will lead to the demise of communal grasslands remains to be seen as to date the state has been notoriously ineffective in implementing changes in communal areas.

The way forward

If communal grasslands are to contribute to biodiversity conservation and if sustainable utilization of grasslands in protected areas is to take place, strategies need to be developed that take the current context of communal grasslands into account. The socio-political context needs recognition and effective, pragmatic grassland management strategies need to be identified.

Key to communal grassland management is the development of effective local institutions. Outside organizations need to assist in developing these bodies as well as re-activating traditional controls on resource use (Niamir-Fuller 1999, Scogings *et al* 1999, Critchley *et al* 1998, Cousins 1995). Outside support is particularly important for conflict resolution and dealing with tenure issues (Scogings *et al* 1999, Cousins 1995, Scoones 1995). Local Boards such as those established by the provincial conservation body in KwaZulu-Natal are an example of including the community in conservation management (Ezemvelo KZN Wildlife 2001).

Small groups as well as special interest groups such as cattle owners or traditional healers have been shown to be most effective in resource management (Walker 2002, Von Maltitz 1998). If economic returns can be clearly demonstrated, incentives for management can be created (Ainslie 1998). There is also evidence that traditional communities with strong authority structures and community participation are the most effective at creating management plans that are implemented and respected (Letty 2003).

Greatly improved service provision to communal areas would assist in marketing of livestock (Oettle *et al* 1998, Critchley *et al* 1998). There is evidence that with outside support, communities will buy inputs, sell animals (Vetter *et al* 1998), pay grazing fees, limit stock numbers and rehabilitate erosion (Thobela, Lax and Oettle 1998). Programmes such as Landcare, which support participatory conservation approaches need to be prioritized. More access to grassland is required by rural households in order to support rural livelihoods and relieve pressure on the land (Walker 2002) and the Department of Land Affairs needs to honour its promises in this regard.

It is unlikely that pragmatic grassland management strategies could include destocking. Although destocking of moist grasslands has not been challenged on ecological grounds, the reality is that destocking has not been achieved in Africa, with government extension officials shying away from dealing with this thorny issue. Destocking may be facilitated through improved access to markets but legal enforcement presents insurmountable problems.

While rotational grazing is controversial as a communal grassland practice, rotational resting seems to hold out more hope for communal grasslands. It provides direct benefits to stock owners as it provides winter fodder at the same time as preventing degradation (Scogings *et al* 1999). A successful example of this is a project in northern KwaZulu-Natal where a negotiated agreement was achieved in order to rest a grassland during summer and use the rested grassland for six months of winter grazing. The stakeholders understood the system and the technical knowledge was appropriate for the local situation. The result was intense but limited grazing with an absence of selective grazing and a grassland with good cover and palatable species (Oettle *et al* 1998). Key to the success of the project was a mix of enlightened self-interest and respect for the local institution. This experiment as well as Zimbabwean projects where formal exchanges between communal areas and private land have been used to buffer the effects of drought (McCarthy and Swallow 1999) may offer possibilities for communal grassland management in the proposed Maloti-Drakensberg TFCA.

There are no easy answers for dealing with shrub encroachment in communal grasslands. Light grazing and frequent fires are recommended (Roques *et al* 2001) but would be hard to achieve in communal grasslands. The re-introduction of indigenous grazers would increase the diversity of feeders (Scogings *et al* 1999) which may decrease the competitiveness of shrubs.

Long-term monitoring of grasslands is urgently required in order to assess the causes and effects of changes in grasslands. Remote sensing in conjunction with ground-based monitoring has been recommended as one approach (Naimir-Fuller 1999, Stafford-Smith and Pickup 1993). Participatory monitoring with communities is also possible. Recent research in the northern Drakensberg involving local communities has used simple technologies to monitoring grassland rehabilitation techniques. Further work is needed in providing incentives to the community e.g. generating income through water trading with down-stream users (Everson and Tau 2003).

CHAPTER 4
Methodology

4.1 Assessing biodiversity

Why measure biodiversity?

The high rate of habitat loss and species extinction means there is an urgent need to find out how biodiversity is distributed, how fast it is disappearing, how it is impacted by different land-uses and how it can be conserved (Purvis and Hector 2000). Long term monitoring is needed to detect gradual changes (Todd and Hoffman 1999) and to ascertain whether changes are due to human activity or long-term climate change (Niamir-Fuller 1999). To achieve any of these goals, biodiversity has to be quantified. Ideally policy makers would like a single number, allowing them to compare biodiversity changes with ease. But the concept of biodiversity is multidimensional, and wide enough to encompass the entire complexity of life and is impossible to measure or describe completely, let alone through a single number. A variety of different measures should be used and these should not be combined into a single index (Purvis and Hector 2000, Gaston 1996a).

Measurement of biodiversity

Biodiversity measurement has used numbers (species richness) and difference (ecological diversity) (Gaston 1996) as well as evenness (how abundance is distributed between the species) (Purvis and Hector 2000). Since the 1960s, diversity has commonly been measured using a combination of species richness and evenness (Gaston 1996b).

Researchers should be aware of the limitations of biodiversity assessment. Spatial and temporal scales impact on sampling (Gaston 1996b) but may be outside the control of the researcher. Seasonal fluctuations affect species diversity as well as evolutionary fluctuations which occur over hundreds of millions of years (Rosenzweig cited in Uys 2000). Sampling effort impacts on measurement; the greater the effort or the sampling area the more species will be sampled (Krebs 1989).

Species richness

Species richness is the total number of species in a site or habitat with species being considered the fundamental unit of diversity (Purvis and Hector 2000, Magurran 1988, Vane-Wright 1996). This is a common approach because it is easy to use and understand as well as describing the essence of biodiversity (Gaston 1996b).

Species richness on its own it is an inadequate measure of biodiversity (Bond 1999, Reid *et al.* 1999; West 1993) and sustainability (Palmer *et al.* 1999). Species richness can conceal patterns of dominance and evenness (Magurran 1988). Species-rich communities can be less ecologically diverse than species-poor communities (West 1993) if species comprise annual weeds rather than rare perennials, if vagrant species stray into the area and if there are exotics rather than native species. What exactly constitutes a species is also controversial. Species classification in research is not usually transparent for peer review with the result that species richness can be inflated or decreased through "splitting or lumping". The former error tends to be greater the larger the area being sampled. Classification errors also depend upon the expertise available for classification (Gaston 1996b).

There are also strong arguments in favour of using species richness as it parallels other measures of biodiversity. Examples are a positive correlation between species richness and numbers of higher taxonomic units; species richness and areas with high phylogenetic disparity; species richness and functional diversity (food webs); species richness and areas of high topographical diversity. However, species richness should not be conflated with areas of high conservation priority and should be used in conjunction with measures of dominance and evenness whenever possible (Gaston 1996b).

Diversity indices

Diversity indices capture richness and evenness in a single value and are based on the proportional abundance of species. They are non-parametric and do not assume any particular abundance distribution. Diversity indices are more informative than species richness counts on their own, and are particularly useful for environmental monitoring and allowing comparisons to be made between two habitats (Magurran 1988). Limitations of diversity indices include the difficulty of interpretation and the possibility of a site with low richness but high evenness scoring the same as a site with high richness and low evenness (Ludwig and Reynolds 1988).

Evenness indices

Species evenness is a measure of how abundance is distributed between species in a community (Ludwig and Reynolds 1988). The more evenly observations are distributed among the species, the higher the diversity (Magurran 1988, Zar 1984). In reality no community has equal species abundance. Usually there are a few very abundant species, some abundant species and the majority of species are rare (Magurran 1988), particularly in invertebrate communities (Rivers-Moore and Samways 1996). Environmental monitoring can make use of the fact that changes in species abundances can be observed in stressed communities. These communities often switch from a lognormal distribution, where moderately abundant species are common, to a geometric series where only a few species are abundant and the remainder are low in abundance (Magurran 1988). The lognormal distribution is the most commonly found model of species abundance (Gaston 1996b) particularly for many taxa of bird, moths, soil mites and trees (Howe 1994).

Species importance

When assessing diversity it is necessary to estimate species importance in the community (Krebs 1989). Although importance is commonly measured as biomass or productivity (Ludwig and Reynolds 1988) there are other importance criteria such as rarity and ecological importance (Linder 1994). Threatened species are useful indicators of ecosystem health (West 1993, Hilton-Taylor 1995).

Other measures of biodiversity

The close association of biodiversity with environmental degradation and habitat loss means that ecological processes should be taken into account as well as species richness (Gaston 1996b). Conservation at ecosystem level also has the spin-off of protecting little known species (Hilton-Taylor 1995). Measures of changes in vegetation composition are useful indicators of biodiversity loss e.g. perennial to annual grasses or the relative abundance of various growth forms (Bond 1999; Reid *et al.* 1999, Todd *et al.* 1998, Linder 1994) as well as the number of higher taxa (Linder 1994). Ecosystem functioning and functional groups (Bond 1999, Hilton-Taylor 1995) should be taken into account as a greater diversity of functions within an ecosystem leads to a greater number of species (Linder 1994).

Additional aspects of invertebrate assessment

Indicators

'Indicators' are a portion of the biota which are used to indicate larger ecosystem or biodiversity patterns. However, the use of invertebrates as indicators is largely untested with more theory than hard evidence. Assessing total invertebrate species richness is impossible due to the large number of species, many of which are unidentifiable. An alternative is to select a small number of invertebrate groups which act as surrogates or umbrellas for total diversity (Rivers-Moore and Samways 1996, Linder 1994).

Functional groups

Functional groups can be used as representatives of biodiversity and provide more information than simple species counts. Groups should be selected to represent different functions within an ecosystem e.g. groups with different feeding preferences, behaviours and habitats. The selected taxa should be ubiquitous, have substantial but not excessive numbers of species, be easy to collect and be responsive to habitat variables (Oliver, Dangerfield and York 1999).

4.2 Research description

This research is a quantitative study of the biodiversity of two sites with different land-uses in order to assess whether this has resulted in a difference in biodiversity. Quantitative data were used in order to summarize major patterns.

This study is experimental (examining a natural experiment of two different land-uses) and falls within the hypothetico-deductive and positivist paradigm. The purpose of the study was to determine the impact of controlled communal land-use on biodiversity. The findings should contribute to policy on grassland conservation in and adjacent to protected areas with particular relevance to the Maloti-Drakensberg Park.

4.3 Research design

This study focused on two neglected areas within the context of grassland biodiversity. Forbs or non-graminoid species have long been overlooked in grassland research, although they are responsible for the bulk of plant species richness. Because members of this group are more likely to be rare or threatened, it is important that they should be selected for assessment (Uys 2000, Bond 1999). Invertebrates too, have been neglected, although they comprise the bulk of biodiversity at species level and perform many key functions. In particular, little is known about the effects of grazing on invertebrates (Hadden and Westbrooke 1999).

The study focused on selected groups of invertebrates that are important to ecosystem functioning as well as identifiable to a meaningful level. The research design was a fence-line study to compare the biodiversity of two adjacent land-uses. Fence-line contrasts are a popular method for fairly rapid evaluation of the effects of different types of land tenure on soils and vegetation. (Bond 1999, Todd *et al* 1998).

The following measures were selected to assess biodiversity in the conservation and lease land:

- The diversity, evenness and richness of vegetation species
- The diversity, evenness and richness of selected invertebrate species
- Abundance of invertebrate and vegetation species exclusive to either lease or conservation sites
- The ecological condition of the grassland using the veld condition assessment technique.

Sampling units

The sample size was determined by the length of shared fence line between the two sites. A purposive sampling approach was therefore used rather than random sampling. Independent sites along the three sections of shared fence line were identified. Paired sites were required to occupy equivalent positions on the slope and have similar steepness and aspect. Obvious differences in habitat were avoided and sites were not placed closer than 100m from the fence in order to avoid firebreaks or closer than 50m to each other.

Fourteen sites (seven pairs) were identified along the three fence lines (Figure 2.2). The vegetation sampling units used in the study were 100m^2 quadrats in each pair of sites. Invertebrate sampling units were five x 50m transects, five x 20 sweep net samples and five pan traps in each of the paired sites.

Environmental assessment

Environmental data were recorded for each of the fourteen sites (Table 4.1).

Table 4.1: A list of environmental variables examined at each sample site for the purpose of indirect gradient analysis

Environmental variables	
Slope percentage	Acid saturation*
Aspect	Soil pH*
Soil density*	Zn*
P, K *	Mn*
Ca, Mg*	Organic carbon*
Exchangeable acidity*	Clay percentage*
Total cations*	Cu*

* Denotes determination by soil science laboratory, KwaZulu-Natal Department of Agriculture and Environmental Affairs, Cedara.

Vegetation assessment

Importance score method

The importance score method of Outhred (1984) was used to assess the diversity, evenness and richness of plant species. It has been found to give a more accurate measure of density than standard frequency methods, as it is less affected by dispersion of the plants i.e. whether they are clumped or evenly spread as well as choice of subquadrat size. Less-common species are sampled more quantitatively. Studies have shown that the method can detect community patterns that would not be found in presence/absence data (Morrison, Le Brocque and Clarke 1995). The importance score method entails the use of a series of nested concentric sub-quadrats of different sizes, located in the centre of the site being sampled. The subquadrats increase in size geometrically (Outhred 1984, Morrison *et al* 1995). The maximum importance score equals the total number of nested subquadrats and this is awarded to each species found in the smallest subquadrat. Species found in each consecutive subquadrat score one point lower, ending in a score of one for species in the largest subquadrat. The score can be standardized as a fraction of the total number of subquadrats giving an absolute abundance figure for each species. The score is a density estimate for each species (Morrison *et al* 1995).

Veld Condition Assessment

Veld condition assessment was used to assess the productive capacity of the grasslands. This is the standard agricultural approach to grassland assessment and is based on the Clementsian model of equilibrium grasslands where species composition reflects the state of health of a grassland (Stoddart, Smith and Box 1975). Agricultural features include the ability of a grassland to support livestock production while ecological features are the successional status of species, the long-term stability of the grassland community and the ability of the community to protect the soil. Regular veld condition assessments are useful for detecting trends and identifying the impacts of different management regimes (Tainton 1999e).

The Ecological Index method (Hardy and Hurt 1999) was selected from a range of veld condition assessment techniques as it emphasises ecological status as well as fodder production potential of species. This method also includes a category that indicates selective over-grazing unlike some other methods. Grass species are classified into four groups: Decreasers, Increaser I, Increaser II, Increaser III which have different responses to grazing and fire (Table 4.2). The study site is scored according to the abundance of species in these categories and then compared to the benchmark score, which is an example of the best possible grassland from a similar ecological zone (Du Toit 1988, Tainton 1999e, Camp and Hardy 1999).

Table 4.2:Classification of plant species for veld condition assessment (*after* Camp and Hardy 1999)

Group	Responses to fire and grazing
Decreasers	Abundant in grassland in good condition. Decline in abundance with over-utilisation (over-grazing) or under-utilisation (exclusion of fire and grazing).
Increaser I Increaser Ia Increaser Ib	Increases in abundance with under-utilisation. Indicates moderate under-utilisation and infrequent fire. Indicates minimal utilisation and fire exclusion.
Increaser II Increaser IIa Increaser IIb Increaser IIc	Increases in abundance with over-grazing. Indicates initial stages of over-grazing Indicates heavy over-grazing Indicates severe over-grazing with soil loss
Increaser III	Increases in abundance with selective over-grazing. Palatable species are out-competed

A limitation of this method and all veld condition assessments is the subjectivity of the classification and weighting of species. The inclusion of all grass species in the assessment, regardless of whether they are affected by grazing or not, is seen as a limitation by agriculturists (Hardy and Hurt 1999).

Basal cover is usually measured as part of veld condition assessment (Du Toit 1988) as it is a useful indicator of trends in primary production. When perennials are replaced by annuals, basal cover decreases leading to increased run-off and erosion (Holden 1995). Decline in basal cover can be the result of both under-utilisation due to grasses becoming moribund (Tainton 1988) as well as over-grazing. However there is evidence that with continuous heavy stocking rates basal cover increases due to a change in species composition from palatable Decreaser species to less palatable stoloniferous Increaser species as well as a tendency towards increased tillering and a low growth habit (Holden 1995). Basal cover in this study was measured using the "Distance-Diameter" method (Hardy and Tainton 1993).

Plant identification

For both the importance score method and the veld condition assessments, plant species were recorded on site whenever possible using Van Oudtshoorn (1991), Tainton, Bransby and Booysen (1976) and Pooley (1998). Unidentified plants were taken to the University of KwaZulu-Natal Herbarium for identification.

Invertebrate assessment

Sweeps, pans and transects

Invertebrates were collected using the well established methods of sweep-netting, coloured pan traps and transects which are more effective in overcoming patchiness in a disturbed area than a square site. Plastic pans are effective for collecting a large number of invertebrates. Blue and yellow pans tend to collect predators and parasites not associated with the foliage. Yellow pans also collect grass flies, non-grass foliage feeders and associated predators and parasites, grass thrips and aphids. Sweep-netting is useful for sampling invertebrates on low vegetation such as grasses (Prendini *et al* 1996, New 1998, Oliver *et al* 1999).

Limitations of the study

Biodiversity is a complex concept and can never be fully measured. A variety of surrogates of biodiversity were used (vegetation and different functional groups of invertebrates) as well as functional diversity in the form of ecological processes such as compositional changes in grass species and basal cover. However the extent to which these measures reflect overall biodiversity is unknown.

Limits to the hypothetico-deductive model

There are limits to the hypothetico-deductive model, and a broad understanding of social and political forces is required in order to fully appraise a situation (Terreblanche and Durrheim 1999). It would be useful for a study of this nature to include qualitative research focusing on the land-users and managers. Participatory Rural Appraisal techniques could be used with the local community and semi-structured interviews with conservationists from KZN Wildlife to develop a deeper understanding of the forces at play in the two sites under study (Further limitations are discussed in Component B: Discussion).

4.4 Data synthesis and analysis

Principle Components Analysis

Principle Components Analysis (PCA) was used to highlight important variables in environmental data, to reduce dimensionality (Manly 1994) and to show up ecological similarities between sites (Ludwig and Reynolds 1988). It was first used by Goodall (1954) and has been described by numerous authors such as Manly (1994) and Ludwig and Reynolds (1988). A standardized centered PCA was performed on the measured environmental data. This approach was appropriate for environmental variables which were measured on different scales.

Correspondence Analysis

Correspondence analysis is a measure of how species numbers and identities differ between communities (Magurran 1988). The technique arranges sites in relation to two axes to determine ecological relationships or similarities between species and sites (Ludwig and Reynolds 1988). The summarized community patterns can then be compared with environmental information (Gauch 1982).

Species richness, evenness and diversity

Diversity

The Shannon Wiener diversity index (H′) (Zar 1984) was used to measure the vegetation and invertebrate diversity. This index is used widely in community ecology and measures the average uncertainty of predicting the species of a randomly selected individual. As the number of species and the evenness increases, so the average uncertainty increases (Krebs 1989, Ludwig and Reynolds 1988). The index varies from a community with only one species (0) to a state of perfect evenness where all species have the same number of individuals (Ludwig and Reynolds 1988). In practice the index usually falls between 1.5 and 3.5 and is unlikely to be greater than 4.5 (Magurran 1988).

Both the Shannon index (H') and species richness (S) are useful for picking up subtle differences between similar habitats. H' is weighted towards species richness, making it more useful for detecting differences between sites than indices which emphasize dominance or evenness. H' is particularly sensitive to rare species, (Magurran 1988) which is an important characteristic in a diversity index (Peet 1974). H' should only be used to measure "indefinitely large" communities, which have not been fully censused (Pielou 1975).

J' is particularly sensitive to species richness, with its value being greatly altered by the addition of one rare species, particularly if the sample has low species richness (Ludwig and Reynolds 1988). Estimates of evenness are only accurate when total species richness is known and it is not very sensitive to evenness between similar sites (Magurran 1988).

It was possible to use J' to assess vegetation evenness as all species in the sample were included. Although the total species richness in the community was not known, J' had value for comparative purposes. However, as the invertebrate assessment included only selected invertebrate species, it was not possible to assess evenness. Thus J' was not calculated for invertebrate data.

J' is calculated as follows:

$$J' = \frac{H'}{H'_{max}}$$

$$H'_{max} = \log k$$

k = number of categories (species) per site.

(Zar 1984)

Richness (S)

Richness was calculated as the total number of plant species and selected invertebrate taxa in each site.

Comparing indices

Paired two sample t-tests (two-tailed) were conducted at the 0.05 significance level to compare Shannon's Diversity Index (H'), Shannon's evenness index (J') and species richness (S) between conservation sites and lease land sites. This test is used when there are two sets of data from related or matched samples (Howell 1995). T-tests were used to find out whether there was a true difference, attributable to the effect of the experimental treatment or the differing land use.

Veld condition assessment

The results of each 100 point survey were expressed as the percentage of each species sampled and were weighted according to the benchmark for Bioresource Group 11 (Camp

1997). Adaptations to the method involved the inclusion of additional species that were not in the benchmark list. These were allocated grazing values and classes from Camp and Hardy's (1999) table titled: Ecological grouping and the grazing value of veld species. A veld condition score was calculated for each conservation and lease site on each of the three fence-lines (six veld condition assessments in total).

The basal cover (BC) was calculated as follows:
$BC = 19.8 + 0.39\,(D) - 11.87\,(\log_e D) + 0.64\,(d) + 2.93\,(\log_e d)$
Where D= distance to the edge of the nearest tuft
And d = tuft diameter
(Hardy and Tainton 1993).

The veld condition based on basal cover was assessed using Table 4.3

Table 4.3: Veld condition based on basal cover (Camp and Hardy 1999)

Veld condition	**Basal cover (%)**
critical	1-5%
poor	6-10%
reasonable	11-15%
good to excellent	> 16%

T-tests were performed at the 0.05 significance level to compare the veld condition on the conservation land and the lease land as well as basal cover.

Vegetation: rare, endemic, threatened, useful, pioneer and weed species

T-tests were performed at the 0.05 significance level to compare abundance of all vegetation species that were present in three or more pairs of sites. Species recorded exclusively on either the conservation or the lease side were assessed in terms of characteristics such as rarity, vulnerability, endemicity, traditional uses or whether they were exotics.

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The impact of communal land-use on the biodiversity of a conserved grassland at Cathedral Peak, uKhahlamba-Drakensberg Park, South Africa

Implications for sustainable utilization of montane grasslands

Component B

by

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Abstract

South African grasslands are under-conserved and there is a need to expand conservation efforts beyond the boundaries of protected areas. While communal grasslands have conservation potential they are generally over-utilized and the impact of communal land-use on biodiversity is poorly studied. At the same time there is pressure on protected areas to allow for the sustainable utilization of biodiversity. The aim of this study was to examine the impact of communal land-use on various components of biodiversity and to make recommendations regarding communal use of protected areas.

A fence-line study was conducted to assess the impact of eight years of controlled communal land-use on biodiversity in the uKhahlamba-Drakensberg Park. The communally used land (referred to as the lease land) which was used for controlled grazing as well as plant collection was compared with land under formal conservation. Vegetation was sampled using the importance score method and veld condition assessments. Selected invertebrate taxa were sampled using sweep netting, colour pan traps and transects and were identified to morphospecies level.

Multivariate statistics revealed that sites generally grouped according to landscape position rather than land-use. No significant differences were found in diversity, evenness, richness or veld condition between the lease and conservation land. However, more than twenty-five percent of vegetation and invertebrate species were found exclusively in the lease or conservation land, suggesting that different suites of species were supported by the two land-uses. Four alien plant species were found exclusively in the lease land, while one vulnerable and one rare plant species were found only in the conservation land.

Further research is required to assess whether biodiversity was diminished by controlled communal. While the lease concept may offer potential as a low-use buffer zone, localised damage from cattle paths and weak enforcement of grazing agreements were areas of concern.

Keywords: communal grasslands, grassland flora, grassland invertebrates, transfrontier park

Introduction

South Africa is the third most biologically diverse country in the world. As a signatory to the Convention on Biological Diversity, South Africa has a responsibility to conserve biodiversity and to use it sustainably and equitably. National strategies, plans and programmes are needed to achieve this (DEAT 1997, Glowka, Burhenne-Guilmin, Synge, McNeely and Gundling 1994). Biodiversity conservation is still evolving and strategies for maintaining and maximising biodiversity need to be developed (Bond 1999) as well as strategies that allow conserved areas to be used by local communities in a way that does not threaten biodiversity. Conservation outside of protected areas needs attention if representativity of biodiversity is to be achieved (McGeoch 2002).

Grasslands in particular are under-conserved both nationally and globally. In South Africa the lowland grasslands in the foothills of the Drakensberg which form part of the Eastern Mountain Centre of Plant Diversity are most under threat (Cowling and Hilton-Taylor 1994). Communal grasslands in South Africa offer conservation opportunities because they are the least likely to be transformed (O'Connor in press), but at the same time they are perceived to be degraded and rare species are likely to be lost (Bond 1999, Hoffman and Todd 2000).

The role of grazing in grassland conservation is controversial. There is evidence that the loss of indigenous ungulates has left a gap in grassland ecosystems and that grazing disturbance is necessary for the maintenance of grassland biodiversity (West 1993, Niamir-Fuller 1999, O'Connor 1999, Pykälä 2000). While similar plant species richness has been found in heavily grazed communal and moderately grazed commercial rangelands (Bond 1999) compositional changes in vegetation have been observed (Todd and Hoffman 1999) and higher abundances of useful plants have been found in protected areas when compared to communal areas (Fabricius and Burger 1997). Some studies argue that disturbance caused by grazing threatens rarer plant species (Bond 1999, O'Connor in press). Invertebrates have been shown to increase in species richness as a result of heavy grazing with exponential increases in generalist invertebrate species (Todd, Seymour, Joubert and Hoffman 1998). Orthoptera (grasshoppers) in particular have been shown to increase in grazed areas compared to protected areas whereas a number of species from the taxa Araneae (spiders) and Formicidae (ants) have been shown to decline (Fabricius and Burger 1997). It appears that protected areas provide an important refuge to certain plant and invertebrate species (Bond 1999).

Conservation for biodiversity is relatively recent and new practices are evolving. There is evidence that biodiversity is not supported by uniform management approaches and that a mosaic of different management practices is preferable. These management practices could include a variety of burning, grazing and resting regimes (Bond 1999, Pykälä 2000, Short 2001). Strategies for the conservation and sustainable utilization of grasslands have particular importance for the Drakensberg-Maloti Transfrontier Park (DMTP) as plans for the Park include the creation of buffer zones as well as the inclusion of communal lands which will continue to be grazed (Lusigi and Acquay 1999, Ezemvelo KZN Wildlife 2001).

This study was a fence-line comparison of a grassland that has experienced controlled communal land-use for the past eight years (the lease land) with the adjacent conservation land (Figure 2).

The study was initiated to assess the impact of controlled communal land-use on biodiversity. Vegetation and selected invertebrate taxa (Araneae (Thomisidae), Diptera (Asilidae), Hemiptera (Cicadellidae), Coleoptera, Hymenoptera and Orthoptera) were chosen as surrogates for biodiversity. Invertebrates as well as grassland forbs are important but neglected components of grasslands. One of the main constraints of including invertebrates in biodiversity assessments is the inability to identify specimens to species level. Given the limits of this study, it was necessary to identify invertebrates using the morphospecies approach. Identification to morphospecies level allows for rapid assessment of invertebrates (Rivers-Moore and Samways 1996). However, it is recognised that the use of morphospecies is limited as endemism and conservation value can only be known if samples are identified to species level (Slotow and Hamer 2000). The specific objectives of the study were to compare the richness, evenness and diversity of the focus taxa on the lease and conservation land; to compare species and community composition of the selected taxa; to compare aspects of ecosystem functioning and to make recommendations regarding communal land-use in protected areas.

The study area

The study area was on the eastern boundary of the uKhahlamba-Drakensberg Park at Cathedral Peak at latitude 29°00′ S and longitude 29°15′ E (Figure 1). The Drakensberg has one of the highest rainfall figures in the summer rainfall region with annual average precipitation of 1300mm (Bainbridge 1991). Winters are cold and dry with frost recorded almost daily in winter (Killick 1963) and snow falling on the summit between April and September (Bainbridge 1991). Soils are fine-textured (Killick 1963), shallow, leached, acidic and vulnerable to erosion with denuded areas recovering slowly. With high intensity rainstorms the potential for accelerated erosion is high (Bainbridge 1991).

The altitude range of the study area was 1291m to 1377m and it fell in the montane zone or river valley system (1280m to 1829m) which lies in the foothills of the Drakensberg (Killick 1963). This zone consists of undulating grasslands dominated by *Themeda triandra* and *Hyparrhenia hirta*. The study area included lower altitude Southern Tall Grassland and higher altitude Highland Sourveld. The higher altitude grasses are sour (unpalatable in winter) (Acocks 1988).

The lease land comprised 535ha of montane grassland and is legally part of the Cathedral Peak protected area. A lease agreement for the land was negotiated in 1995 between the AmaNgwane Tribal Authority and Ezemvelo KZN Wildlife, in exchange for a piece of high-lying tribal land. The AmaNgwane Tribe has been allowed to use the lease land for controlled grazing and plant harvesting. Official limits on cattle numbers obtained from Natal Parks Board documentation (NPB 1991) and Ezemvelo KZN Wildlife staff ranged from 134 Animal Unit (AU) up to 500 AU (Lemmer 2001 *pers.comm.*, Faure 2003 *pers.comm.*). Cattle are allowed to graze from November to January and May to July each year. Harvesting of plants for thatching, building and medicinal use is allowed in exchange for work or a portion of the harvest. The area appeared to experience point source burns every year and Ezemvelo KZN Wildlife seemed to have relinquished control over burning in this area (Lemmer 2001 *pers.comm.*, Faure 2003 *pers.comm.*).

The adjacent conservation land was managed by the Ezemvelo KZN Wildlife for conservation purposes. This land has a history of illegal grazing and forestry development. Although it was not pristine grassland representative of the main conservation area (Everson 2003 *pers.comm.*3) it was able to provide a comparison between two different management regimes. Grazing in the conservation land was negligible. It was officially block burned every two years although in reality there were numerous unscheduled fires (Lemmer 2001 *pers.comm.*, Van Zyl 2003 *pers comm.*).

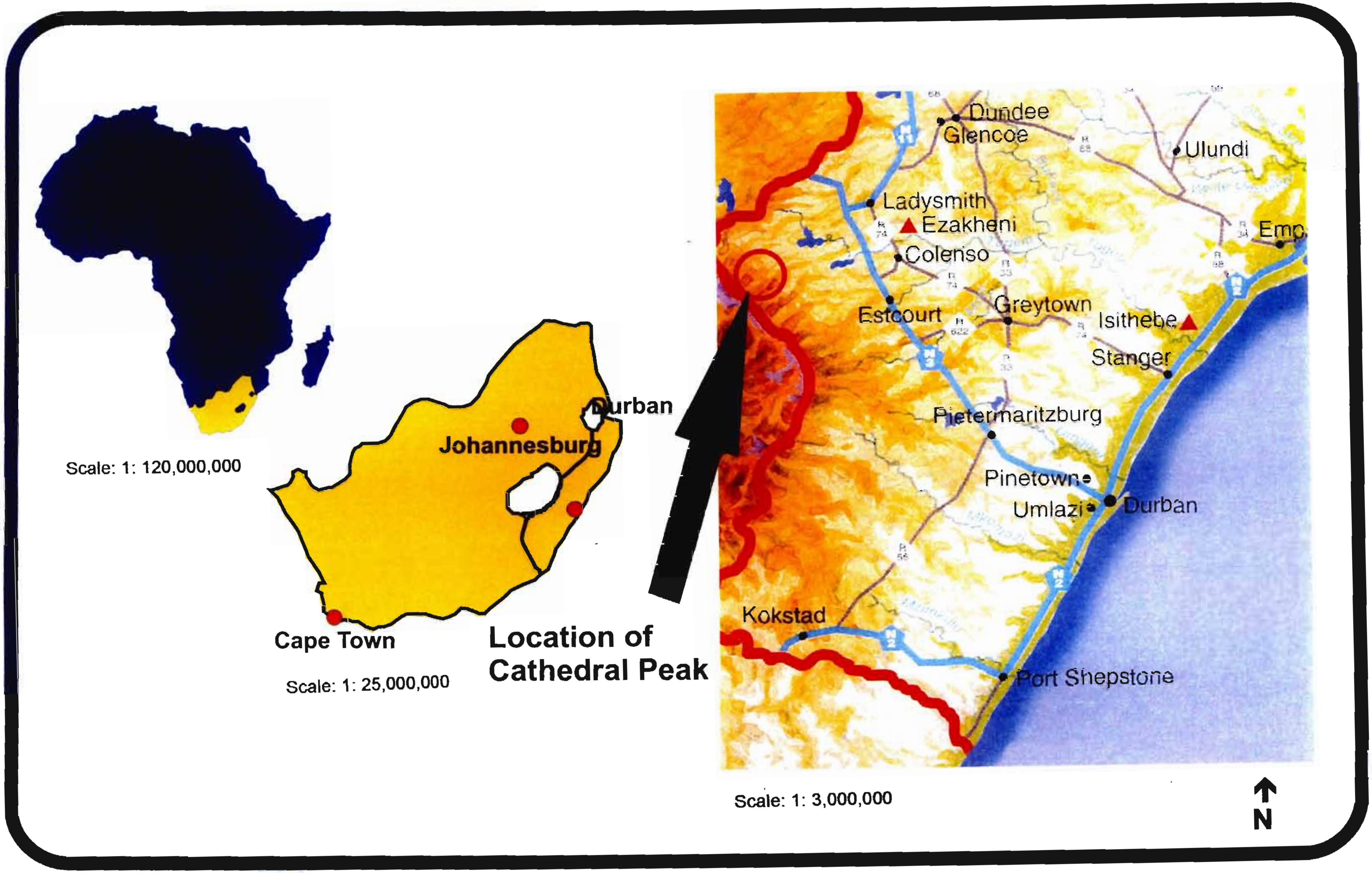
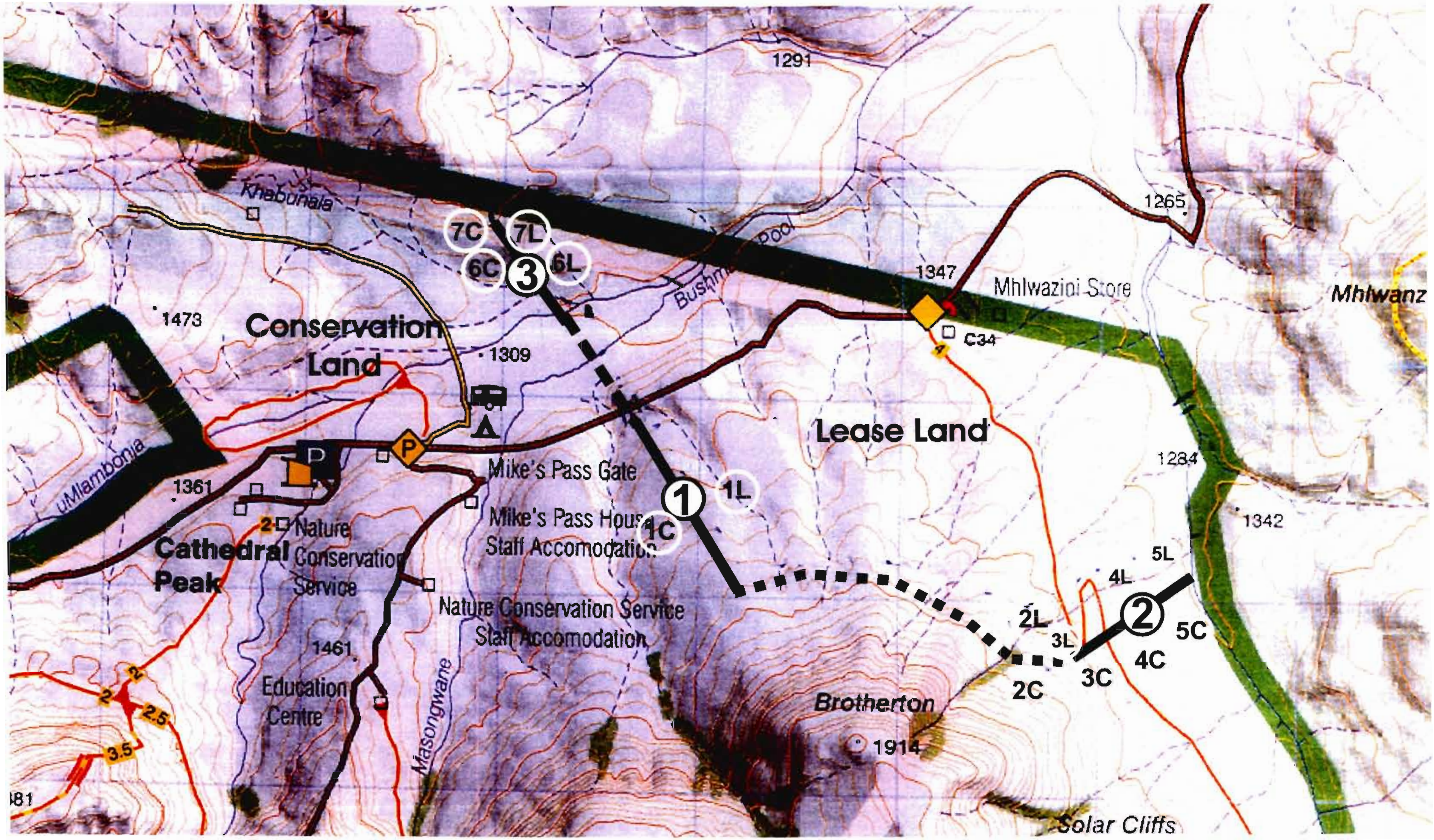


Figure 1: Location of Cathedral Peak, KwaZulu-Natal relative toA.Africa, B. South Africa and C. Southern KwaZulu-Natal



2: Position of the fourteen study sites at Cathedral Peak: 1 – 7C = seven sites on conservation land, 1 – 7L = seven sites on lease land.

N

Scale: 1:25000

Materials and methods

Sampling methods

Seven paired sites with equivalent position on the slope and similar aspect and steepness were selected along the three fence-lines (Figure 2). Sites were no closer than 100m from the fence-line in order to avoid fire-breaks and no closer than 50m to each other in order to achieve independence. Environmental variables (slope and aspect) were measured at each site and fifteen centimetre soil samples were taken to the Department of Agriculture soil laboratory for fertility analysis (Table 1). Observations were made of erosion from cattle paths in the lease land and the condition of the fences between lease and conservation land.

Table 1: Environmental variables measured at each of the fourteen sites

Soil variables	**Other variables**
Soil density*	Slope percentage
Levels of P,K, Ca, Mg, Zn, Mn, Cu *	Aspect
Exchangeable acidity*	
Total cations*	
Acid saturation*	
pH (KCl)*	
Organic carbon*	
Clay %*	

* Denotes determination by soil science laboratory, KwaZulu-Natal Department of Agriculture and Environmental Affairs, Cedara

Vegetation was sampled using the Importance Score Method (Outhred 1984, Morrison, Le Brocque and Clarke 1995). This method entailed the use of a $100m^2$ quadrats. Each quadrat contained a series of seven nested concentric sub-quadrats which increase in size geometrically. The maximum importance score is equal to the total number of nested subquadrats and this is awarded to species found in the smallest central subquadrat. Species found in in each consecutive subquadrat score one point lower, ending in a score of one for species in the largest subquadrat. The score can be standardized as a fraction of the total number of subquadrats giving an absolute abundance figure for each species. The score is a density estimate for each species. All plant species including forbs were identified in each quadrat. Vegetation sampling took place in one growing season from January to April of 2002.

Veld condition assessments using the Ecological Index Method (Hardy and Hurt 1999) were conducted in November 2002. One hundred points were identified on each side of the three fence-lines, giving a total of six assessments. Species found in the assessments that were not in the benchmark list for Bioresource Group 11 were allocated grazing values and classes from Camp and Hardy's (1999) table: Ecological grouping and the grazing value of veld species (1999). Basal cover was measured using the Distance-diameter method (Hardy and Tainton 1993). Plant species were identified using Van Oudtshoorn (1991), Tainton, Bransby and Booysen (1976) and Pooley (1998). Unidentified plants were taken to the University of Natal Herbarium for identification.

Invertebrates were sampled using sweep-nets, colour pan traps and transects (New 1998, Oliver, Dangerfield and York 1999). Collection took place in October and December 2002 on each of the seven paired sites. Sweep netting which is useful for sampling invertebrates on low vegetation was conducted on each site. Five sets of twenty sweeps were conducted on alternate sides of the sampler and approximately one meter apart. Five colour pans (blue/orange) containing water and liquid soap were set out in a line about one meter apart in each site overnight. Blue and yellow pans tend to collect predators and parasites not associated with foliage and yellow pans also collect grass-flies, non-grass foliage feeders and associated predators and parasites. (New 1998). Five sets of 50m transects were walked for each site to observe or catch flying invertebrates. Specimens were preserved in 80% alcohol and the selected groups were identified to morphospecies level using Picker, Griffiths and Weaving (2002), Leroy and Leroy (2000), Filmer (1991) and Scholtz and Holm (1985). Each morphospecies was allocated a unique code which included the taxon at order or family level. This collection will be maintained as a reference collection in the School of Botany and Zoology at the University of Natal.

Data analysis

Sixteen environmental variables were identified for each of the fourteen sites (Appendix 1). Pearson's product-moment correlations (Appendix 2) and inspection of a PCA plot of the variables were used to select a subset of eight relatively uncorrelated variables. Pearson's product-moment correlation is a standard correlation coefficient and measures the intensity of association between values (Zar 1984).

Raw data matrices were compiled to show relative abundances of invertebrates and vegetation at each site. Invertebrate sweep samples were analysed separately from pan trap samples as these two methods sampled different communities of invertebrates. The transects did not provide sufficient data for analysis. Invertebrate data were log transformed (natural logarithm of (X+1)) to reduce skewness and the influence of very high abundance values on the ordination. Rare species were down-weighted to reduce the influence of single occurrences on the ordination.

The Shannon diversity (H') and evenness indices (J') were calculated for vegetation data and the Shannon diversity index was calculated for invertebrate data (Biological Toolbox Version 0.10). Evenness was not calculated for invertebrate data as all species were not sampled. The indices and species richness (S) of both vegetation and invertebrate data were compared using paired two sample t-tests (p=0.05) (Appendix 1). The abundance of vegetation species present in three or more pairs of sites was compared using paired two sample t-tests (two tailed) (p=0.05). Plant species collected exclusively on the conservation or the lease side were assessed in terms of their conservation status, whether they are used traditionally, whether they are endemics, rare, vulnerable or exotics according to Tainton, Bransby and Booysen 1976, Hilliard and Burtt 1987, Bromilow 1995 and Scott-Shaw 1999.

The CANOCO 4 package (ter Braak and Smilauer 1998) was used to do a correspondence analysis (CA) of the vegetation and invertebrate data sets to assess how species abundance and identities differed between sites. Wilcoxon's matched pairs tests were conducted on the four axes of each CA.

Results

Environmental characteristics of study sites

Values for the sixteen topographic and soil variables measured at the fourteen sites are shown in Appendix 1. A subset of 8 relatively independent representative or surrogate variables were selected using Pearson's product-moment correlations (Table 2).

Table 2: Selected environmental variables

slope	organic carbon
total cations	clay percentage
pH(KCl)	levels of P, K and Z

A correspondence analysis (CA) of sites with the eight selected topographical and soil variables overlaid showed that sites on fence-line 1 (1L and 1C) had lower than average pH and lower than average fertility; sites on fence-line 2 (2L, 2C, 3L, 3C, 4L, 4C, 5L, 5C) had higher than average soil fertility (higher K, total cations, organic carbon and clay %) and higher than average pH and sites on fence-line 3 (6L, 6C, 7L, 7C) had higher than average slope and lower than average pH (Figure 3, Appendix 1). The CA showed that sites generally grouped according to fence-lines, with minimal within-pair variation which indicates that sites were appropriately selected for the study. Thus environmental differences between paired sites were minimal.

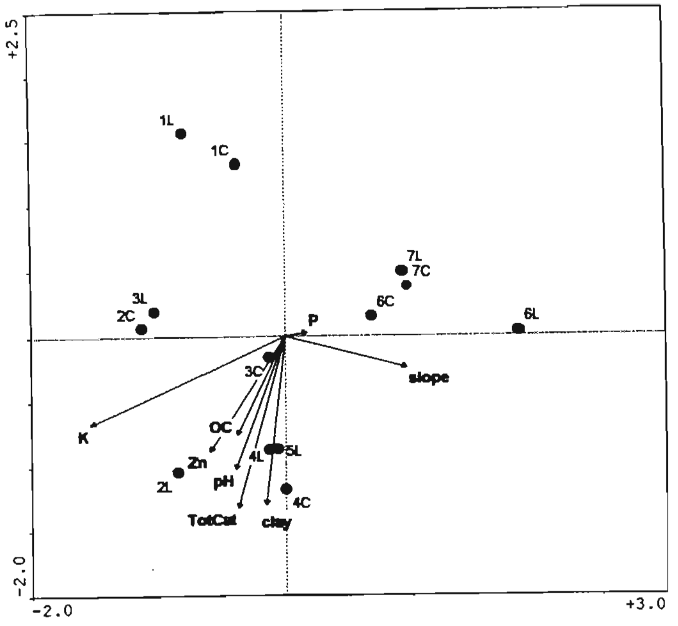


Figure 3: Ordination diagram of a Correspondence Analysis of fourteen sites at Cathedral Peak with topography and soil variables overlaid. The longest arrows indicate the variables with the greatest rate of change.
Abbrev: C= conservation sites; L= lease sites; K= potassium, OC=organic carbon; P= phosphorus, Zn=zinc, Totcat = total cations.

Comparison of species richness, diversity and evenness

Paired t-tests did not show a significant difference in species richness (S) (p=0.63, 6df), diversity (H') (p=0.48, 6df) or evenness (J') (p=0.44, 6df) for plant species between conservation and lease land. Species richness was similar in the conservation sites (range of S=34 - 59) and the lease sites (range of S=37 - 49). Diversity values for all sites (excluding the outlier site 5C) were fairly high (H' >3.5). Evenness was high in all sites (J' >0.95).

No significant difference in species richness (p=0.78, 6df) or diversity (H') (p=0.33, 6df) of invertebrates sampled by sweeping was found. The range for species richness of sweep samples was similar in the conservation sites (S=20 - 68) and the lease sites (S=16 - 54). Diversity values were generally lower than the norm and ranged from 1.56 to 3.53. No significant difference in species richness (p=0.69, 6df) or diversity (H') (p=0.31, 6df) of invertebrates sampled in the colour pan traps was found. Species richness for colour pan traps was similar in the conservation sites (range of S=4 - 31) and the lease sites (range of S=2 - 27). Diversity values were generally lower than the norm and ranged between 1.21 and 2.18.

Species level comparison of conservation and lease areas

Vegetation

A total of 237 plant species were recorded. A similar number of species was recorded in the conservation land (167 species) and in the lease land (164 species). The grass to forb ratio was similar in the conservation (33:67) and lease land (34:66). Sixty-six (27% of the total) plant species were recorded only in the lease land and 56 (24% of the total) of these occurred in only one of the seven lease sites. Sixty-eight (29%) of the plant species were recorded only in the conservation land and 51 (22%) of these occurred in only one of the seven conservation sites (Appendices 3, 4).

Species recorded only in the lease land included four exotic species and fourteen species with medicinal or spiritual use (Appendix 3). Species recorded only in the conservation land included one rare species, *Habenaria dregeana* (Hilliard and Burtt 1987) and one vulnerable species, *Eucomis autumnalis* (Scott-Shaw 1999) and nineteen species with medicinal or spiritual use (Appendix 4).

Paired t-tests on vegetation species abundances showed that the forbs *Acalypha schinzii* ($p=0.01$) and *Helichrysum micronifolium* ($p=0.02$) were significantly higher in the lease land with *Monocymbium ceresiiforme* (a Decreaser grass) ($p=0.01$) significantly higher in the conservation land. The majority of the most common species were found on both conservation and lease land. Although statistical differences were not found, the grass *Diheteropogon filifolius* was common on the conservation land and infrequent on the lease land and the grass *Brachiaria serrata* was common in the lease land and infrequent on the conservation land (Table 3).

Table 3: Most frequently occurring plant species on lease and conservation land (top 10% of all recorded species)

Conservation			**Lease**		
Spp	frequency % (cons)	mean relative abundance (cons)	Spp	frequency % (lease)	mean relative abundance (lease)
Themeda triandra	86	2.89	*Pentanisia angustifolia*	100	3.35
Aster bakerianus	71	1.25	*Themeda triandra*	100	1.73
Monocymbium ceresiiforme *	71	2.81	*Aster bakerianus*	86	2.40
Polygala gerrardii	71	1.13	*Eragrostis racemosa*	86	2.28
Senecio bupleuroides	71	2.43	*Helichrysum micronifolium*	86	1.60
Trachypogon spicatus	71	1.56	*Polygala gracilenta*	86	2.65
Tristachya leucothrix	71	2.44	*Tristachya leucothrix*	86	1.90
Vernonia natalensis	71	1.57	*Vernonia natalensis*	86	2.09
Acalypha punctata	57	2.20	*Acalypha schinzii* ♦	71	1.82
Bulbostylis humilis	57	1.84	*Bulbostylis humilis*	71	2.22
Diheteropogon filifolius *	57	1.78	*Harpochloa falx*	71	1.67
Eragrostis racemosa	57	1.93	*Oxalis obliquifolia*	71	1.03
Eulalia villosa	57	1.27	*opposite leaf forb*	71	1.47
Gladiolus crassifolius	57	0.95	*Brachiaria serrata*♦	57	1.46
Harpochloa falx	57	1.40	*Diheteropogon amplectens*	57	1.27
Helichrysum micronifolium	57	0.63	*Eulalia villosa*	57	1.63
Hyparrhenia hirta	57	1.74	*Hyparrhenia hirta*	57	1.46
Oxalis obliquifolia	57	1.59	*Polygala gerrardii*	57	1.86
Polygala gracilenta	57	0.95	*Senecio bupleuroides*	57	1.53
Sebaea sedoides	57	1.89	*Trachypogon spicatus*	57	1.39
Barleria monticola	43	0.96	*Acalypha punctata*	43	0.87
Corycium nigrescens	43	0.89	*Alectra sessiflora*	43	1.13
Diheteropogon amplectens	43	1.33	*Alloteropsis semi-alata*	43	0.78

Frequency %: Percentage of sites where species was sampled

Mean relative abundance: The sum of the relative abundances of the species in all conservation or lease sites divided by the number of sites (7).

* Low frequency or absent in lease land (frequency of ≤ 14%)

♦ Low frequency or absence in conservation land (frequency of ≤ 14%)

Invertebrates

A total of 245 invertebrate morphospecies were recorded from pan traps and sweep samples combined, with 179 morphospecies in the conservation land and 186 morphospecies in the lease land. The numbers of morphospecies recorded only in the lease land or only in the conservation land were similar. Sixty-six (27%) invertebrate morphospecies from the pan and sweep sample data combined were recorded only in the lease land and fifty-seven morphospecies (23%) were recorded only in the conservation land (Table 5, Appendix 7). There were many rare morphospecies, with 104 (42% of the total) sampled at a single site. Forty-seven of these single occurrences were found only in one of the seven conservation sites and fifty-eight were found only in one of the seven lease sites. Most invertebrate morphospecies were stenotopic with low abundances and only a few were eurytopic with high abundances (Appendices 5, 6). Those that were abundant in both lease and conservation land included two Formicidae (ant), one Coleoptera (beetle) and one Orthoptera (grasshopper) morphospecies. Morphospecies that had high local abundances in the conservation land (the top 5%) and low abundance in the lease land included three Orthoptera and one Cicadellidae (leafhopper) morphospecies. Morphospecies with high local abundances (the top 5%) in the lease land and low abundance in the conservation land included two Cicadellidae and one Coleoptera (Table 4).

Table 4: Most abundant invertebrate morphospecies (top 5% of all recorded morphospecies) in lease and conservation land from sweep netting and colour pan trap samples. (Figure following taxon refers to reference number of morphospecies).

Conservation		**Lease**	
	Abundance where found		Abundance where found
Formicidae 1	19.68	Coleoptera 8	4.80
Orthoptera 31	5.97	Orthoptera 39	4.67
Orthoptera 2 *	4.20	Coleoptera 7	2.45
Coleoptera 8	3.87	Formicidae 4	2.25
Orthoptera 39	3.40	Cicadellidae 6	1.80
Orthoptera 38 *	2.67	Formicidae 1	1.23
Cicadellidae 13	2.50	Cicadellidae 34	1.20
Formicidae 1	2.40	Cicadellidae 28 ♦	1.20
Cicadellidae 24*	2.24	Coleoptera 6♦	1.00
Orthoptera 14*	2.00	Cicadellidae 25	1.00
Formicidae 4	1.16	Cicadellidae 1 ♦	0.93
Hymenoptera 9	1.13	Orthoptera 35	0.90
Cicadellidae 57 *	0.93	Coleoptera 13	0.80

Abundance where found: sum of mean number of individuals per plot (5 samples per plot) divided by the number of plots in which the morphospecies occurred.

* low abundance or absent in lease land (abundance of ≤ 0.27)

♦ low abundance or absent in conservation land (abundance of ≤ 0.20)

A substantial proportion of the total Hymenoptera (bees and wasps) (70%), Thomisidae (67%), Cicadellidae (45%) and Coleoptera (50%) morphospecies were recorded only on the lease land or on the conservation land. Similar numbers of Thomisidae and Cicadellidae morphospecies were found on each side, with higher numbers of Coleoptera and Hymenoptera (bees and wasps) morphospecies in the conservation land and higher numbers of Orthoptera morphospecies on the lease side (Table 5).

Table 5: Invertebrates found only on lease or conservation land

	Conservation (C) No/total (%)	Lease (L) No/total (%)	Total % of morphospecies exclusive to either conservation or lease land (C + L)
Cicadellidae	13/66 (20%)	17/66 (25%)	45 %
Coleoptera	8/26 (31%)	5/26 (19%)	50 %
Asilidae	1/3 (33%)	1/3 (33%)	67 %
Formicidae	1/10 (10%)	1/10 (10%)	20%
Hymenoptera (bees and wasps)	13/30 (43%)	8/30 (27%)	70%
Orthoptera	15/88 (17%)	25/88 (28%)	45 %
Thomisidae	6/18 (33.5%)	6/18 (33.5%)	67%

Comparison of community structure

Vegetation

Correspondence analysis of vegetation showed the main axis of floristic vegetation was landscape position rather than land-use. Sites generally grouped in their pairs according to fence-lines when the outlier wetland site 5C was removed (Figure 4a - b). This was confirmed by the Wilcoxon matched pairs test which showed no significant differences between conservation and lease on Axis 1 ($p=0.75$; $z=0.31$; $n=6$), Axis 2 ($p= 0.60$; $z= 0.52$; $n=6$), Axis 3 ($p=0.17$; $z=1.36$; $n=6$) and Axis 4 ($p=0.46$; $z=0.73$; $n=6$).

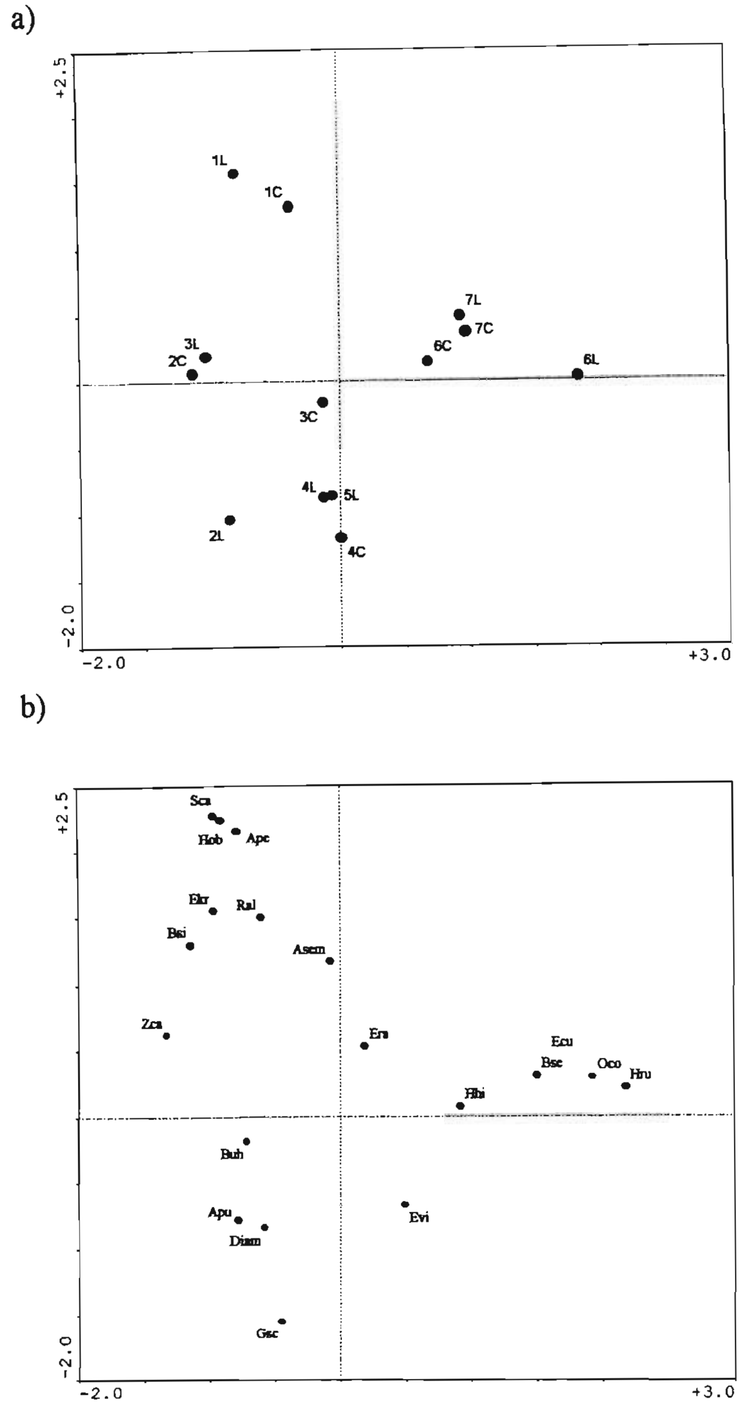


Figure 4: Correspondence Analysis (CA) ordination diagram of (a) 14 sites and (b) vegetation composition at Cathedral Peak. Only the first two axes are shown. Eigenvalues for Axis 1 and 2 are 0.539 and 0.441 respectively, cumulatively representing 25% of the total variance. Site 5C removed. Abbrev: C = conservation site; L = lease site, Ape=*Aster perfoliatus*; Apu=*Acalypha punctata;* Asem=*Alloteropsis semi-alata*; Bse=*Berkheya setifera*; Bsi=*Buchnera simplex*; Buh= *Bulbostylis humilis*; Diam=*Diheteropogon amplectens*; Ecu=*Eragrostis curvula;* Ekr=*Eriosema kraussianum*; Era=*Eragrostis racemosa*; Evi=*Eulalia villosa*; Gsc= *Graderia scabra*; Hhi= *Hyparrhenia hirta;* Hob=*Hypoxis obtusa*; Hru=*Helichrysum rugulosum*; Oco= *Orchidaceae* cf *Corycium nigrescens;* Ral=*Rendlia altera*; Sca=*Sopubia canna*; Zca= *Zornia capensis*

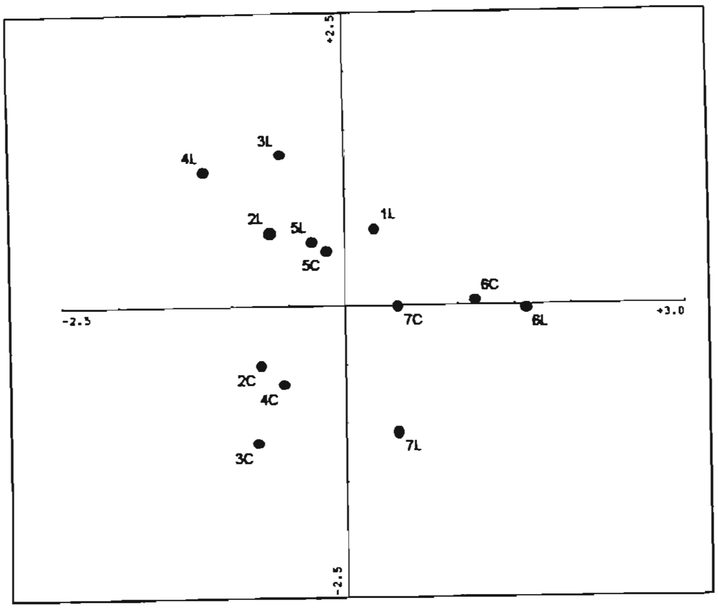
Invertebrates

Correspondence analysis of invertebrate sweep sample data showed that Site 1C was an outlier with a different set of species to the other sites. Site 1C appeared to have experienced the lowest burn frequency of all the sites with visibly denser vegetation. When site 1C was removed from the CA, the remaining sites tended to group on the first axis (Eigenvalue =0.374) according to position in the landscape rather than land-use (Figure 5a - b). This was confirmed by the Wilcoxon matched pairs test which showed no significant differences between conservation and lease land on Axis 1 ($p=0.92$; $z=0.10$; $n=7$), Axis 2 ($p=0.17$ $z=1.36$; $n=7$), Axis 3 ($p=0.46$ $z=0.37$; $n=7$) and Axis 4 ($p=0.75$ $z=0.31$; $n=7$).

However the second axis (Eigenvalue=0.275) showed some separation of sites along fence-line 2 (sites 2, 3 and 4) according to land-use. Morphospecies which contributed to the separation of sites on fence-line 2 included Orthoptera 2 which had high local abundance in the conservation land with low local abundance in the lease land overall (Table 2) as well as Hymenoptera 13, Cicadelladoidae 30, Coleoptera 20 and 23, Thomisidae 7 and Orthoptera 33 which were strongly associated with fence-line 2 and were absent in the lease land. Most of the morphospecies strongly associated with the lease plots on fence-line 2 were also recorded in the conservation plots except for Orthoptera 77 which was absent in the conservation land (Appendices 5,6).

The CA of pan trap data did not show any clear patterns (Figure 6a - b). Sites were not grouped according to fence-line nor according to land-use. Wilcoxon matched pairs test showed no significant differences between conservation and lease land on Axis 1 ($p=0.31$; $z=1.01$; $n=7$), Axis 2 ($p=0.61$ $z=0.51$; $n=7$), Axis 3 ($p=0.40$ $z=0.85$; $n=7$) and Axis 4 ($p=0.24$ $z=0.18$; $n=7$).

(a)



(b)

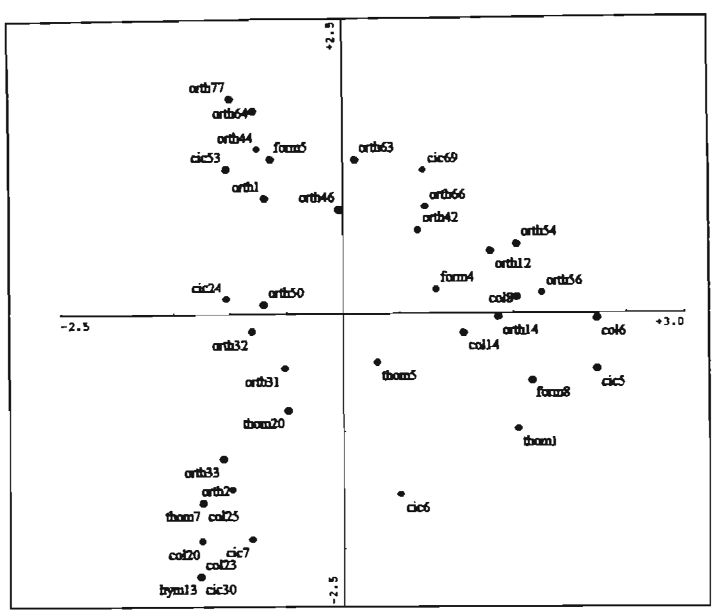
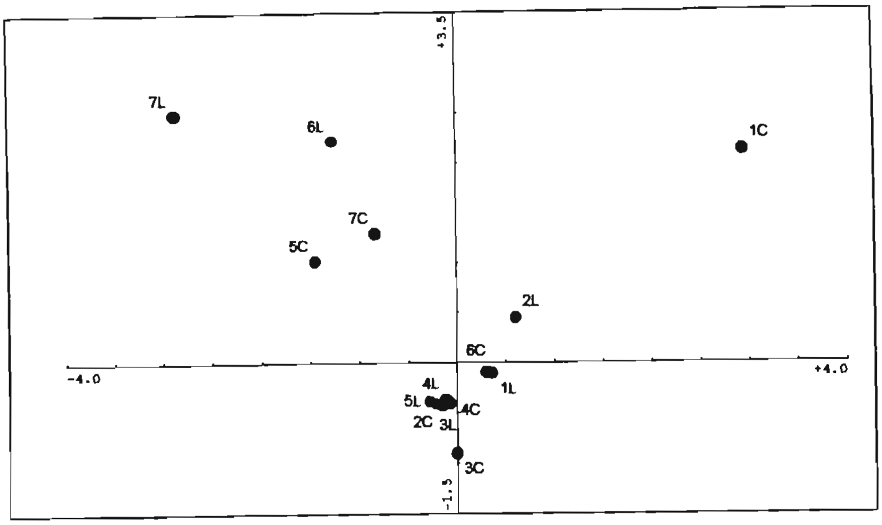


Figure 5: Correspondence Analysis (CA) ordination diagram of invertebrate sweep samples and 14 sites at Cathedral Peak. (a) Sites (b) morphospecies. Only the first two axes are shown. Only morphospecies with at least 33% of their variance accounted for are shown. Eigenvalues for Axis 1 and 2 respectively are 0.374 and 0.275 cumulatively representing 25.8% of the total variance. Site 1C is removed.

Abbreviations: C= conservation site; L = lease site; cic= Cicadellidae, col=Coleoptera, form= Formicidae, hym= Hymenoptera, orth= Orthoptera, thom= Thomisidae. Figure following taxon refers to reference number of morphospecies.

a)



(b)

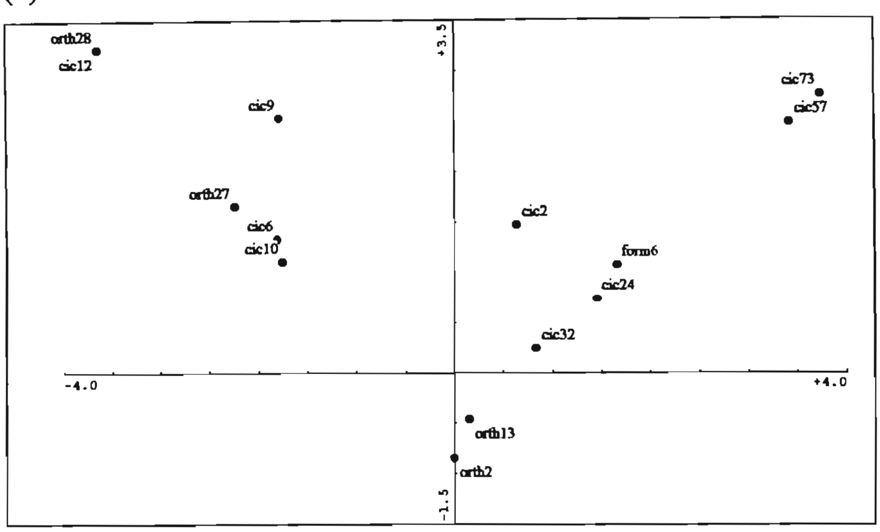


Figure 6: Correspondence Analysis (CA) ordination diagram of invertebrate pan traps and 14 sites at Cathedral Peak. (a) Sites (b) morphospecies. Only morphospecies with at least 33% of their variance accounted for are shown. Eigenvalues for Axis 1 and 2 respectively are 0.623 and 0.596 cumulatively representing 26.6% of the total variance.

Abbrev: C = conservation plot; L= lease plot; cic= Cicadellidae, col=Coleoptera, form= Formicidae, hym= Hymenoptera, orth= Orthoptera, thom= Thomisidae. Figure following taxon refers to reference number of morphospecies.

Veld condition

Veld condition of the lease land was higher (61.2% - 87.6%) when compared to the conservation land (45.25% - 55.5%). Basal cover in the lease-land was reasonable to excellent (11.72% - 16.73%) and reasonable in the conservation land (14.4% - 15.15%) (Appendices 7, 8,9). Paired t-tests on the veld condition scores ($p=0.12$) and basal cover scores (0.99) showed no significant differences between lease and conservation land. The lack of significant results for veld condition may be due to the low number of samples. The conservation land generally scored low on Decreaser species, particularly *Themeda triandra.*

On fence-line 1 the conservation land scored poorly (44%) compared to the lease land (74%) and showed signs of selective grazing with high levels of *Diheteropogon filifolius* (25%) and low levels of *T. triandra* (2%). The conservation side of fence-line 2 showed under-utilization with high levels of *Tristachya leucothrix* (36%) and low levels of *T. triandra* (1%). The lease side had high levels of *Hyparrhenia hirta* and low levels of *T. triandra* (1%) which could indicate disturbance (Camp 1997). On fence-line 3 the conservation site score (54%) compared poorly to the lease score (88%). High levels of forbs (20%) and *Aristida congesta* (20%) on the conservation land indicated over utilization (Camp 1997).

Field observations of cattle and paths

No more than 100 cattle were observed in the lease land in any one day. They tended to concentrate near the tribal/lease boundary rather than near the lease land/conservation boundary of this study. Illegal grazing was observed on fence-line 3 with cattle entering the conservation land and grazing on the lease land outside of the agreed grazing months. Fences in this area were in poor condition (Plate 1 and 2). Localised erosion damage due to cattle paths was apparent in the lease land with no path maintenance being undertaken (Plate 3 and 4). The cattle appeared to be in good condition (Plate 5).

Plate1: Broken fences (fence-line 3) December 2002

Plate 2: Cows (foreground) grazing in illegally in conservation land. December 2002

Plate 3: Erosion caused by cattle paths (lease land between fence-line 1 and 2) December 2002.

Plate 4: Cattle paths north of the uMlambonja River (near fence-line 3). December 2002.

Plate 5: Cattle in the lease land between fence-line 1 and 2. December 2002

Discussion

The impact of controlled communal land-use on biodiversity

There appeared to be little difference between the conservation land and the lease land along the shared fence-line as a result of controlled communal land-use. Position in the landscape had greater overall impact on plant and invertebrate species present than did land-use. The pairing of sites according to landscape position indicated that the majority of plant and invertebrate species were common to both lease and conservation sites. This compositional similarity was supported by the lack of statistical difference in richness, evenness and diversity of vegetation and richness and diversity of invertebrates. In addition the similar grass to forb ratio in each land-use did not indicate disturbance on the lease land.

Measurement of species richness, diversity and evenness provided an initial assessment of the biodiversity of the two land-uses. However, it is possible for sites to have similar scores in these measures but to have different degrees of ecological diversity if there are different abundances of annual or pioneer plant species or generalist or exotic plant and invertebrate species (West 1993). A closer analysis of the characteristics and abundances of individual plant and invertebrate species revealed some differences between the two land-uses.

Grazing appeared to benefit veld condition with the grazed lease land scoring higher than the ungrazed conservation land. The lease land was in good condition from an agricultural perspective with high veld condition scores that were consistently higher than the scores on the conservation land and high levels of *Themeda triandra* on fence-line 1 (30%) and fence-line 3 (37%). This is a Decreaser species which responds positively to moderate defoliation and fire. The basal cover (reasonable to excellent) showed that certain key ecological processes were maintained in the lease land. However it needs to be recognized that veld condition assessments are agricultural rather than conservation measures. Although they measure some ecological changes within vegetation communities (successional status of grass species and basal cover) (Hardy, Hurt and Bosch 1999) their focus is on the palatability of grass species. Few, if any previous studies on communal grazing have investigated a combination of veld condition and biodiversity. There is little evidence that high veld condition scores translate into high levels of biodiversity. There is evidence that under-utilization which would result in low veld condition scores may in fact have higher biodiversity, particularly in respect of rare forb species (Uys 2000, O'Connor in press). Heavy grazing in the short-term can benefit the palatable Decreaser grass species although it is likely that this type of grazing regime is damaging to forbs including rarer plants as well as rarer invertebrates (Bond 1999, O'Connor in press, Todd *et al* 1998).

In spite of the high scores for veld condition and the lack of significant difference in richness, evenness and diversity of vegetation, there were signs of disturbance and selective grazing in the lease land. The significantly higher abundance of *Acalypha schinzii* is an indication of selective over-grazing and disturbance (Acocks 1988). It is possible that the significantly higher abundance of *Helichrysum micronifolium* is an indication of disturbance as forbs increase with disturbance (Tainton 1999). The four exotic species found exclusively in the lease land with no exotics found in the conservation land also indicated disturbance. However, the high abundance of *Tristachya leucothrix* (25%) in the lease land in the veld condition assessment along fence-line 2 is an indication of under-utilization (Camp and Hardy 1999). This apparent contradiction may have been caused by the extensive rotational grazing system which caused some patches to be under-grazed and other patches to be repeatedly grazed (Hardy 1999). The presence of exotic invasive plant species in the lease land lends support to the need for long-term monitoring as exotic plant invasion poses a significant threat to grassland biodiversity (Bond 1999).

The results for the conservation land appeared contradictory. Two fence-lines appeared to be under-utilized while the third seemed to have been selectively overgrazed. The veld condition assessments showed under-utilization on fence-lines 1 and 2 and consistently lower levels of Decreaser species (2%, 1% and 15%) on the conservation land. These species tend to decline with both over and under-utilization as well as selective grazing. It is likely that low abundance of Decreasers on fence-lines 1 and 2 was a result of low utilization. The abundance of *Tristachya leucothrix* (36%) on the conservation side of fence-line 2 also indicates under-utilization (Camp and Hardy 1999). The high abundance of *Diheteropogon filifolius* (wire- grass) (25% in the veld condition assessment) on the conservation side of fence-line 1 appeared initially to contradict the trend of under-utilization. This species is classified as an Increaser II or III species and is associated with selective overgrazing (Camp and Hardy 1999, Botha 2001). However, other recent research in Highland Sourveld found that *D. filifolius* was more abundant on conservation than communal grassland and did not increase with heavy grazing in communal land (Short, O'Connor and Hurt 2003). The species can also be associated with shallow soil (Moffett 1997). Soil depth was not measured in this study and it is possible that this variable may have accounted for differences in abundance between conservation and lease land.

Other contradictions to the general trend of "under-utilization" in the conservation land was the significantly higher abundance of *Monocymbium ceresiiforme*, a palatable Decreaser grass species which declines with both over and under-utilization and also indicates acid soils (Tainton *et al* 1976). Its abundance on the conservation land may have been caused by accidental sampling inside the fire break on the conservation side where the fire break was wider than the prescribed 100m as it is possible that this species increases in areas that are burned regularly.

The sharp contrast in veld condition scores on fence-line 3 between the lease land (88%) and conservation land (54%) may be the result of selective grazing in the conservation land. Moderately low levels of *T. triandra* (15%) together with high levels of *Aristida congesta* subspecies *congesta* (20%) and forbs (20%) on the conservation side could be the result of selective grazing caused by illegal grazing, with grazing pressure low enough to allow cattle to constantly select the palatable species and leave other species. This illustrates the impact of softening the boundaries of the protected area, where the buffer zone between protected area and communal land has moved inwards and effectively shrunk the area under conservation.

The low burn frequency along the conservation side of fence-line 1 could have caused site 1C to be an outlier in terms invertebrate sweep sample data. Some differences between the composition of invertebrate sweep sample data on conservation and lease sites along fence-line 2 (Figure 5a and b) could also have been the result of lower levels of disturbance on the conservation side.

Further differences in the biodiversity of the two land-uses was characterized by the large number of plant species (27%) and invertebrate morphospecies (27%) recorded only on the lease land as well as plant species (29%) and invertebrate morphospecies (23%) recorded only on the conservation land (Appendix 2, 3 and 6).

These figures suggest that a substantial number of both plant and invertebrate species thrive in the less disturbed conservation land while a similar number thrive in the shorter grazed lease land. There is much research evidence that shows that a number of the less common invertebrate taxa are sensitive to heavy land-use. Studies have found that 23% to 45% of arthropods are unique to protected areas (Bond 1999) and that many invertebrates have narrow habitat requirements with generalist species tending to dominate in grazed areas (Todd *et al* 1998).

The Orthoptera (grasshoppers) found exclusively in the lease land may have preferred shorter grazed areas while those found exclusively in the conservation land may have preferred tall vegetation as was found in the study by Prendini, Theron, van der Merwe and Owen-Smith (1996). This study supports other evidence that many Cicadellidae (leafhoppers) are very host-specific and feed on one or a few plant species (Scholtz and Holm 1985). The conservation land may have supported rarer species of Formicidae (ants) requiring an undisturbed habitat while the lease land may have supported more common Formicidae species in line with Hadden and Westbrooke's (1999) findings that common Formicidae species decreased in abundance in ungrazed plots with a few species increasing, possibly due to the suppression of other ant species. Coleoptera (beetles) have been found to be specific to certain vegetation communities and intolerant of wide variations of environment. It is possible that the Coleoptera that were abundant in the lease land were dependent on the dung of large mammals as found by Rivers-Moore and Samways (1996). Some evidence shows that ground-dwelling Thomisidae spiders (Leroy and Leroy 2000) prefer shorter grazed vegetation (Filmer 1991, Hadden and Westbrooke 1999) while another study (Fabricius and Burger 1997) found a reduction of spiders in communal land compared to conserved land. With 57% endemicity of spiders in South Africa (Le Roux 2002) it is possible that the six morphospecies found only in conservation land were range-restricted species. Fewer Hymenoptera morphospecies (bees and wasps) (eight) were found in the lease land than the conservation land (thirteen) confirming Hadden and Westbrooke's (1999) study which showed Hymenoptera to decrease in grazed areas (Table 5).

Rare and endemic species are particularly important in biodiversity assessment. Although these species may not make a significant contribution in terms of numbers and evenness, they require special attention in terms of conservation (Uys 2000). Rare plant species are particularly important in biodiversity conservation as generalist species tend to survive heavier land-use such as communal grazing (Bond 1999). A large proportion of invertebrates (42%) and plant species (48%) in this study occurred in one site only.

One vulnerable species *Eucomis autumnalis* and one rare species *Habenaria dregeana* was found on one site on the conservation land and were not found on the lease land. *Habenaria dregeana* is used as a traditional charm (Pooley 1998) and *Eucomis autumnalis* is prized for its medicinal value (Mander, Mander, Crouch, McKean and Nichols 1995). There is evidence that forbs such as *Eucomis autumnalis* are more vulnerable than grasses to trampling (O'Connor in press) and that grassland plants with medicinal value are under threat from collection (Hilton-Taylor 1995, Bond 1999). Studies in the Drakensberg have shown that the total exclusion of grazing and fire for at least 5 years has resulted in double the number of rare plant species compared to other fire and grazing regimes (Uys 2000, O'Connor in press). These findings suggest that fire and grazing exclusion is necessary in a conserved grassland if biodiversity conservation is to be achieved.

The question that remains is whether both land-uses supported rare species, or whether all the species found in the lease land were generalist or wide-spread species? This question is key to whether grazing and biodiversity conservation can co-exist. The answer can only be found by identifying invertebrates to species level and through intensive vegetation sampling which allows for identification of all species.

Ecological processes e.g. successional status of plants, plant recruitment, presence of different functional groups of invertebrates, soil fertility maintenance and soil erosion should be included in biodiversity assessment as biodiversity is closely associated with environmental degradation (Gaston 1996). In this study the analysis of individual species of plants has given some indication of ecological trends in the two land-uses and measurement of basal cover has indicated levels of degradation. The lease land along the fence-line scored high in these aspects.

Limitations of this study

Participatory research with communal farmers is necessary if one is to achieve sustainable utilization of communal grasslands (Scogings, De Bruyn and Vetter 1999). Due to time constraints and the sensitive nature of the lease agreement it was not possible to examine the social issues in this project. The costs and benefits of the lease land to the community were not evaluated.

The veld condition assessment used in the vegetation survey is limited by the subjectivity of the system of classifying and weighting species. All grass species are included in the assessment regardless of whether or not they are affected by grazing (Hardy and Hurt 1999). In addition the Ecological Index Method gives the same weighting to Increaser I species (which increase with under-utilisation) as Increaser II species which increase with over-grazing (Camp and Hardy 1999). This means that over-grazed or under-utilized grassland could get similar scores.

The study was conducted over two seasons (one for vegetation sampling and one for invertebrate sampling) and could only assess a portion of the biodiversity. Seasonal variation was offset by the fact that all paired sites were sampled at the same time, meaning that there was unlikely to be a within-pair difference. The late season sampling of vegetation meant that a substantial portion of plant species (25%) were not identified. Pan trap data for invertebrates did not show clear compositional patterns as was the case with the vegetation and sweep-net data. Attractant pans are possibly not useful for fence-line studies as they attract mobile invertebrates from both sides of the fence.

Identification of invertebrates to morphospecies level has its limitations (Slotow and Hamer 2000). Species level identification of morphospecies is desirable to find out whether species strongly associated with the conservation or lease land consisted of rare or endemic species of conservation value or common abundant species. If one intends to assess a more representative suite of invertebrate species, several years of research may be required (Samways and Rivers-Moore 1996) and a full assessment of vegetation would require three weekly sampling over an entire season (Scott-Shaw 2001 *pers. comm.*). The figures given for both plants and invertebrates unique to one land-use may be exaggerated as a result of splitting (identifying the same species as more than one species) or lumping (classifying two species as one) which may have occurred in the identification process. Splitting inflates species richness while lumping decreases it (Gaston 1996).

A serious concern that became apparent during the study was that the sites along the fence line were unlikely to be representative of the entire lease land. The fence-line sites experienced the lowest level of grazing in the lease land as a result of daily herding of cattle. Eroded cattle paths occurred in the interior of the lease land and not along the fence-line. In addition, purposive sampling procedures meant that certain areas that may have been included in a random sampling process were omitted. These included areas invaded with *Pteridium aquinilinum* (bracken) or *Rubus cuneifolius* (bramble).

In addition, it became apparent during the course of the study that the conservation land along the fence-line was not pristine grassland representative of the main conservation area. There was also a possibility that sampling sites were placed within the firebreak which may have been greater than the recommended 100m in some areas.

There were no true replicates for this study and further research needs to be done in another study site to assess whether these results show general trends or whether they are unique.

Recommendations

The lease land at Cathedral Peak

Commitment to enforcement of the lease agreement is essential from Ezemvelo KZN Wildlife. Grazing within the agreed months as well as removing cattle from the conservation land are fundamental to the lease agreement. Fence maintenance is needed. The practice of daily herding of cattle in and out of the lease land should be reconsidered in the light of observations of eroded cattle paths in the interior of the lease land

The conservation authority appears to have foregone fire management in the lease land with burns occurring almost annually. A mixture of burning regimes is necessary in order to protect a variety of rare plants (Uys 2000). The burning of the lease land by the community is illegal in terms of the Forest Act as the land is still legally under the jurisdiction of the conservation body. If four yearly resting was undertaken, the conservation authority should take responsibility for protecting the rested grassland and burning it at the end of the rest period.

Further assessment of the impact of controlled communal grazing on the interior of the lease is recommended. Comparisons with conserved grassland would require identifying areas more representative of the conservation area. In addition, participatory research with communal land-users would provide insight into their perceptions and practices and could contribute to the development of realistic agreements around land-use practices.

Grazing in conservation land: the lease-swap concept

There are arguments to support controlled grazing within conservation areas. In this study veld condition appeared to have improved under grazing. Under-utilization can lead to the loss of palatable grasses (Watkinson and Ormerod 2001, Allsop 1999) and controlled grazing can be beneficial to biodiversity (O'Connor 1999, Todd and Hoffman 1999, West 1993). It has been proposed that the use of domestic stock to replace indigenous grazers that were eliminated, may be beneficial to grassland biodiversity (Pykälä 2000).

The lease swap concept where an agreement is negotiated for a communal grassland to be rested in exchange for limited grazing in a protected area could further the aims of biodiversity conservation. The concept may offer a strategy to extend biodiversity conservation outside of protected area boundaries in order to achieve representativity of biodiversity (Vane-Wright 1996, Goodman 2001 *pers.comm.*) and to make use of the conservation opportunities offered by communal grasslands (Bond 1999). This concept also offers an alternative to the conventional grazing associations with a focus on commercial production that have been proposed for the DMP, although they have largely failed in Africa. The concept offers ways of expanding low use buffer zones into areas that are presently under continuous grazing.

The concept of enlightened self-interest in communities should be used as the foundation for grassland conservation. If communities recognize the value of resting grasslands they will gain better grazing. Acceptable agreements need to be negotiated between conservation authorities and interest groups such as stock owners, traditional healers and plant collectors. The conservation authorities should make use of multi-disciplinary teams that include both social scientists and grassland conservationists. Agreements should define responsibilities for the maintenance of cattle paths, fences and alien invasion control. Rather than adopting a blueprint approach, these agreements need to recognize the uniqueness of different areas within the DMP, both in terms of ecology and human communities. The danger of over-harvesting of medicinal species needs to be tackled through the development of medicinal plant cultivation programmes and educational input is required on the benefits of resting grasslands. Government programmes and resources e.g. Landcare and Public Works programmes, should be included in these activities.

However it must be recognized that the lease-swap concept is not a panacea and has its drawbacks. If the lease-swap concept is pursued, fire, grazing frequency and patterns as well as the enforcement of agreements need serious consideration.

Until fairly recently the Drakensberg grasslands had low and seasonal populations of grazers. Wildlife was limited by the sour nature of the grasses and domestic stock was kept out by the presence of the San people (Bainbridge *et al* 1986). It was only in the late 19th century that continuous selective grazing began (Killick 1963). If there is a place for grazing in the Drakensberg protected areas, it is important that these grasslands are not subjected to continuous selective grazing but are grazed briefly and intermittently in a way that emulates natural patterns.

A strong commitment to enforcement of controls on land-use would have to underpin any lease agreement if it is to succeed in its objectives of biodiversity conservation. Enforcement of agreements with local communities is notoriously difficult and the enforcer has to risk being unpopular at times. It is simpler to have clear un-negotiated protected area boundaries rather than agreements which are vulnerable to different interpretations and need ongoing attention. It is essential that conservation authorities have the capacity to enforce agreements if they decide to embark on such a route. Lack of enforcement of the lease agreement at Cathedral Peak has softened the boundaries of the protected area resulting in a de facto reduction in the amount of land under conservation.

However, if one is to recognize the social and political pressures of the time, which require poverty alleviation, multiple land-use in and around conservation areas as well as the dire situation of lowland grasslands and the inadequacy of protected areas, the lease-swap concept may offer a strategy to try out, as long as there is commitment to monitoring the process and enforcing the agreements.

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Appendix 1: Measured environmental data for all fourteen sites at Cathedral Peak

	slope %	aspect	density g/ml	P mg/l	K mg/l	Ca mg/l	Mg mg/l	Exch acidity cmol/l	total cations	Acid sat %	pH (Kcl)	Zn (mg/l)	Mn (mg/l)	NIRS organic carbon	NIRS clay %	Cu (mg/l)
1C	20	5 nw	1.05	1	65	209	84	2.03	3.93	52	4.14	0.1	4	2.4	37	3
1L	20.4	5 nne	0.96	1	250	450	191	1.39	5.85	24	4.18	0.2	4	2.9	53	2.9
2L	14.2	4 e	0.87	2	491	1029	428	0.09	10	1	4.73	0.5	9	3.8	71	8.5
2C	20.3	4 e	0.84	1	427	1223	516	0.05	11.49	0	4.88	0.9	6	3.5	71	10.4
3L	15.3	3 se	0.8	2	412	242	122	2.91	6.18	47	4.06	0.2	8	3.4	61	3.3
3C	20.8	5 ne	0.82	2	452	1304	407	0.16	11.17	1	4.58	0.8	12	4.5	65	15.8
4C	23.2	5 nee	0.85	1	321	1105	431	0.54	10.42	5	4.25	0.7	10	3.4	55	3.1
4L	22.5	5 ne	0.94	1	111	703	379	0.57	7.48	8	4.41	0.2	8	3	71	13.8
5L	25	4 e	0.9	1	205	1785	679	0.1	15.12	1	4.96	0.6	12	3.7	71	10.6
5C	20.6	4 e	1.04	1	141	482	194	0.98	5.34	18	4.18	0.3	10	1.8	39	2.8
6L	30.1	3 se	0.95	1	61	959	391	0.14	8.3	2	4.66	0.1	8	2.3	60	7.6
6C	28.3	2 ssw	0.81	2	78	95	50	3.88	4.97	78	4.08	0.2	3	4.4	57	2.4
7C	12.7	2 sse	0.85	1	46	222	82	2.88	4.78	60	4.07	0.4	5	3.7	64	3.1
7L	19.1	1 s	0.88	3	124	502	292	2.74	7.97	34	4.03	0.5	9	2.9	62	3.7

Aspect was ranked on a scale of 1 to 5. 1=south; 5= north.

Abbreviations: P = phosphorus; K= potassium, Ca = calcium; Mg = magnesium, Exch acidity = exchangeable acidity; Acidsat = acid saturation, Zn= zinc, Mn = manganese;

Appendix 2: Pearson's product-moment correlations and associated p values for topographic and soil variables measured at fourteen sites in the Cathedral Peak area

	SLOPE	ASPECT	DENSITY	P	K	CA	MG	EXACID	TOTCAT	ACIDSAT	PH	ZN	MN	OC	CLAY	CU
SLOPE	1															
ASPECT	0.0683 p=.817	1														
DENSITY	0.1535 p=.600	0.3596 p=.207	1													
P	-0.2078 p=.476	-0.5643 p=.036 *	-0.4779 p=.084	1												
K	-0.367 p=.197	0.3619 p=.204	-0.4963 p=.071	0.2313 p=.426	1											
CA	0.2539 p=.381	0.3869 p=.172	-0.1632 p=.577	-0.1751 p=.549	0.4694 p=.090	1										
MG	0.2542 p=.381	0.3241 p=.258	-0.1345 p=.647	-0.1445 p=.622	0.4188 p=.136	0.969 p=.000 ***	1									
EXACID	-0.1855 p=.525	-0.6512 p=.012 *	-0.2005 p=.492	0.4153 p=.140	-0.4305 p=.124	-0.8413 p=.000 ***	-0.8306 p=.000 ***	1								
TOTCAT	0.1985 p=.496	0.2392 p=.410	-0.3401 p=.234	-0.0062 p=.983	0.5234 p=.055	0.9682 p=.000 ***	0.9567 p=.000 ***	-0.706 p=.005 **	1							
ACIDSAT	-0.1564 p=.593	-0.5368 p=.048 *	-0.0962 p=.744	0.2415 p=.406	-0.49 p=.075	-0.853 p=.000 ***	-0.8726 p=.000 ***	0.9634 p=.000 ***	-0.759 p=.002 **	1						
PH	0.2473 p=.394	0.3436 p=.229	-0.0865 p=.769	-0.2629 p=.364	0.421 p=.134	0.8857 p=.000 ***	0.8763 p=.000 ***	-0.824 p=.000 ***	0.8345 p=.000 ***	-0.771 p=.001***	1					
ZN	-0.1614 p=.582	0.1379 p=.638	-0.498 p=.070	0.1157 p=.694	0.6221 p=.018 *	0.6921 p=.006 **	0.6443 p=.013 *	-0.457 p=.100	0.7472 p=.002 **	-0.513 p=.061	0.5099 p=.063	1				
MN	0.0316 p=.915	0.1958 p=.502	-0.1053 p=.720	0.1514 p=.605	0.3939 p=.163	0.7173 p=.004 **	0.6852 p=.007 **	-0.591 p=.026 *	0.7022 p=.005 **	-0.694 p=.006 **	0.4522 p=.105	0.501 p=.068	1			
OC	-0.1002 p=.733	-0.101 p=.731	-0.8451 p=.000 ***	0.3572 p=.210	0.4488 p=.107	0.2507 p=.387	0.1628 p=.578	0.1096 p=.709	0.3789 p=.182	0.1048 p=.721	0.2106 p=.470	0.499 p=.069	0.057 p=.848	1		
CLAY	-0.0901 p=.759	-0.1517 p=.605	-0.6579 p=.011*	0.2081 p=.475	0.3989 p=.158	0.5388 p=.047*	0.6053 p=.022 *	-0.305 p=.289	0.6419 p=.013 *	-0.389 p=.170	0.5846 p=.028 *	0.478 p=.084	0.299 p=.299	0.6 p=.022 *	1	
CU	0.1391 p=.635	0.3994 p=.157	-0.1802 p=.538	-0.0415 p=.888	0.3794 p=.181	0.7033 p=.005 **	0.692 p=.006 **	-0.681 p=.007 **	0.6556 p=.011 *	-0.669 p=.009**	0.7395 p=.003 **	0.456 p=.101	0.516 p=.059	0.3 p=.223	0.6 p=.012 *	

*=p< 0,05; ** = p< 0,01; ***= p < 0,001 **Abbreviations**: P= Phosphorus, K= Potassium; Ca=Calcium, Mg= Magnesium, Exacid= Exchangeable acidity; Totcat= Total cations; Acidsat= acid saturation; Zn=Zinc, Mn=Manganese, OC=organic carbon, Cu=copper

Appendix 3: Vegetation Summary Table (lease land)

Species	Abbrev	frequency % (lease)	frequency % (all sites)	mean relative abundance (lease)	mean abundance where found (lease)	Spp. found in lease only
Pentanisia angustifolia	Pau	100.00	71.43	2.13	2.13	
Themeda triandra	Ttr	100.00	92.86	3.35	3.35	
Aster bakerianus	Aba	85.71	78.57	1.73	2.02	
Eragrostis racemosa	Era	85.71	71.43	2.40	2.80	
Helichrysum micronifolium	Hmi	85.71	71.43	2.28	2.66	
Polygala gracilenta	Pgr	85.71	71.43	1.60	1.86	
Tristachya leucothrix	Tle	85.71	78.57	2.65	3.10	
Vernonia natalensis	Vna	85.71	78.57	1.90	2.21	
Acalypha schinzii	Asc	71.43	42.86	2.09	2.92	
Bulbostylis humilis	Buh	71.43	64.29	1.82	2.55	
Harpochloa falx	Hfa	71.43	64.29	2.22	3.11	
Oxalis obliquifolia	Oxob	71.43	64.29	1.67	2.34	
opposite leaf forb	Olf	71.43	42.86	1.03	1.44	
Brachiaria serrata	Bser	57.14	35.71	1.47	2.58	
Diheteropogon amplectens	Diam	57.14	50.00	1.46	2.55	
Eulalia villosa	Evi	57.14	57.14	1.27	2.23	
Hyparrhenia hirta	Hhi	57.14	57.14	1.63	2.85	
Polygala gerrardii	Pge	57.14	64.29	1.46	2.56	
Senecio bupleuroides	Sbu	57.14	64.29	1.86	3.26	
Trachypogon spicatus	Tsp	57.14	64.29	1.53	2.68	
Acalypha punctata	Apu	42.86	50.00	1.39	3.24	
Alectra sessiflora	Ases	42.86	35.71	0.87	2.04	
Alloteropsis semi-alata	Asem	42.86	35.71	1.13	2.64	
Corycium nigrescens	Cni	42.86	42.86	0.78	1.81	
Elionurus muticus	Emu	42.86	28.57	0.94	2.19	
Hermannia gerrardii	He	42.86	35.71	0.73	1.69	
Heteropogon contortus	Hcon	42.86	35.71	0.83	1.94	
Hypericum aethiopicum	Hae	42.86	28.57	0.56	1.30	
Sebaea sedoides	Sse	42.86	50.00	0.62	1.46	
Senecio sp2	Sen2	42.86	21.43	0.20	0.46	1
Zornia capensis	Zca	42.86	28.57	1.01	2.35	
Large waxy	Lwa	42.86	28.57	1.06	2.46	
Ledebouria type	Led	42.86	35.71	0.78	1.81	
Acalypha sp.	Asp	28.57	28.57	0.91	3.18	
Anthospermum rigidum subsp *pomilum*	Arp	28.57	21.43	0.83	2.89	
Aristida junciformis	Aju	28.57	21.43	1.00	3.51	
Athrixia sp.	Atsp	28.57	28.57	0.65	2.26	
Barleria monticola	Bmo	28.57	35.71	0.84	2.93	
Becium obovatum	Bob	28.57	14.29	0.63	2.20	1 ☼
Berkheya setifera	Bse	28.57	21.43	0.69	2.41	
Berkheya sp 1	Bsp1	28.57	14.29	0.47	1.64	1
Buchnera simplex	Bsi	28.57	28.57	0.55	1.92	
Conyza pinnata	Cpi	28.57	28.57	0.58	2.03	
Cyperus sp.	Cyp	28.57	28.57	0.64	2.23	
Digitaria tricholaenoides	Dtr	28.57	28.57	0.84	2.94	
Eragrostis curvula	Ecu	28.57	35.71	1.00	3.51	
Eragrostis plana	Epla	28.57	14.29	0.80	2.80	1
Eriosema kraussianum	Ekr	28.57	21.43	0.93	3.27	

Species	Abbrev	frequency % (lease)	frequency % (all sites)	mean relative abundance (lease)	mean abundance where found (lease)	Spp. found in lease only
Euphorbia striata	Est	28.57	28.57	0.93	3.27	
Gladiolus crassifolius	Gcr	28.57	42.86	0.71	2.48	
Gnidia kraussiana	Gkr	28.57	21.43	0.37	1.30	
Graderia scabra	Gsc	28.57	28.57	0.91	3.18	
Helichrysum herbaceum	Hhe	28.57	14.29	0.30	1.04	1 ☼
Helichyrsum oreophilum	Hor	28.57	35.71	0.42	1.47	
Helichrysum rugulosum	Hru	28.57	14.29	0.32	1.12	1 ☼
Hypoxis multiceps	Hmu	28.57	14.29	0.57	1.99	1 ☼
Hypoxis sp.	Hypo	28.57	21.43	0.80	2.81	
Melinis nerviglumis	Mne	28.57	14.29	0.79	2.76	1
Paspalum sp.	Psp	28.57	21.43	0.74	2.59	
Pteridium sp.	Pte	28.57	35.71	0.84	2.94	
Rhus discolor	Rdi	28.57	21.43	0.33	1.16	
Sebaea grandis	Sgr	28.57	21.43	0.21	0.72	
Senecio sp1	Sen1	28.57	14.29	0.56	1.96	1
Setaria sp.	Set	28.57	28.57	0.46	1.61	
Sonchus oleraceae	Sol	28.57	14.29	0.24	0.83	1▲
Sporobolus africanus	Saf	28.57	14.29	1.00	3.51	1
Polygala type	Poty	28.57	28.57	0.61	2.13	
Selago type	Styp	28.57	14.29	0.60	2.11	1
sandpaper forb	Sfo	28.57	14.29	0.44	1.53	1
grass red stem	Grs	28.57	21.43	0.72	2.51	
Anthospermum herbaceum	Ahe	14.29	14.29	0.35	2.46	
Aristida congesta congesta	Acc	14.29	21.43	0.41	2.87	
Artemisia afra	Aaf	14.29	7.14	0.41	2.87	1 ☼
Aster perfoliatus	Ape	14.29	14.29	0.45	3.18	
Athrixia phyllicoides	Aph	14.29	7.14	0.07	0.50	1☼
Becium sp.	Bsp	14.29	7.14	0.13	0.91	1
Berkheya sp 2	Bsp2	14.29	7.14	0.35	2.46	1
Cephelaria oblongifolia	Cob	14.29	7.14	0.56	3.91	1
Chamaechrista sp.	Chsp	14.29	7.14	0.25	1.72	1
Commelina africana	Caf	14.29	14.29	0.27	1.91	
Crassula sp.	Crsp	14.29	7.14	0.12	0.86	1
Cucumis zeyheri	Cze	14.29	7.14	0.14	1.01	1 ☼
Cymbopogon sp.	Cyms	14.29	7.14	0.50	3.52	1 ☼
Diclis reptans	Dre	14.29	7.14	0.48	3.35	1 ☼
Digitaria flaccida	Dfl	14.29	21.43	0.45	3.18	
Digitaria sp.	Dsp	14.29	14.29	0.40	2.79	
Diheteropogon filifolius	Difil	14.29	35.71	0.32	2.27	
Eriospermum cf.*cooperi*	Eco	14.29	7.14	0.12	0.82	1
Gerbera kraussii	Gkra	14.29	21.43	0.41	2.87	
Gerbera ambigua	Gam	14.29	7.14	0.29	2.05	1☼
Gerbera sp 2	Ges2	14.29	7.14	0.07	0.48	1
Gladiolus cf *ecklonii*	Gec	14.29	7.14	0.23	1.64	
Gladilous sp1	Gsp1	14.29	21.43	0.48	3.35	
Gladiolus sp2	Gsp2	14.29	7.14	0.31	2.16	1
Gladiolus sp3	Gsp3	14.29	7.14	0.18	1.23	1
Gnidia splendens	Gsp	14.29	14.29	0.39	2.73	
Haplocarpa scaposa	Hsc	14.29	14.29	0.59	4.14	
Hebenstretia sp.	Hebs	14.29	14.29	0.32	2.23	
Hermannia sp.	Hes	14.29	14.29	0.13	0.91	

Species	Abbrev	frequency % (lease)	frequency % (all sites)	mean relative abundance (lease)	mean abundance where found (lease)	Spp. found in lease only
Helichrysum adenocarpum	Had	14.29	7.14	0.14	0.96	1
Helichrysyum aureum	Hau	14.29	28.57	0.16	1.12	
Helichrysum coriaceum	Hco	14.29	21.43	0.12	0.86	
Helichrysum glomeratum	Hgl	14.29	7.14	0.14	0.96	1
Helichrysum sutherlandii	Hsu	14.29	7.14	0.34	2.39	1☼
Helichrysum sp 1	Hsp1	14.29	14.29	0.19	1.36	
Hyparrhenia sp.	Hysp	14.29	14.29	0.29	2.01	
Hypoxis rigidula	Hri	14.29	28.57	0.32	2.27	
Hypoxis obtusa	Hob	14.29	14.29	0.39	2.73	
Ledebouria sp.	Led	14.29	14.29	0.08	0.59	
Loudetia simplex	Lsi	14.29	21.43	0.32	2.27	
Melinis sp.	Msp	14.29	7.14	0.34	2.39	1
Miscanthus sp.	Misp	14.29	7.14	0.25	1.72	1
Nidorella auriculata	Nau	14.29	7.14	0.23	1.64	1
Orchidacea cf. *Corycium nigrescens*	Oco	14.29	14.29	0.29	2.05	
Oxalis corniculata	Oxco	14.29	7.14	0.41	2.87	1▲
Paspalum dilatatum	Pdi	14.29	7.14	0.35	2.46	1▲
Peucedanum caffra	Pca	14.29	7.14	0.43	3.02	1☼
Polygala ohlendorfia	Poh	14.29	14.29	0.25	1.72	
Rendlia altera	Ral	14.29	21.43	0.45	3.18	
Rhynchosia totta	Rto	14.29	14.29	0.43	3.02	
Rhynchosia sp.	Rsp	14.29	7.14	0.12	0.86	1
Rubus cuneifolius	Rcu	14.29	7.14	0.35	2.46	1▲
Scleria sp.	Ssp	14.29	14.29	0.24	1.68	
Senecio glaberrimus	Sgl	14.29	14.29	0.51	3.55	
Setaria pallide-fusca	Spa	14.29	7.14	0.35	2.46	1
Sopubia cana	Sca	14.29	14.29	0.32	2.27	
Spermacoce natalense	Sna	14.29	7.14	0.25	1.78	1☼
Spermacoce senensis	Spse	14.29	7.14	0.07	0.50	1
Striga bilabiata	Sbil	14.29	14.29	0.14	1.01	
Thecium sp.	The	14.29	7.14	0.39	2.73	1☼
Watsonia sp.	Wsp	14.29	7.14	0.56	3.91	1
Acalypha type	Aca	14.29	14.29	0.40	2.79	
alternate leaf rough	Alt	14.29	7.14	0.13	0.91	1
black edge forb	Bef	14.29	7.14	0.26	1.82	1
bright green forb	Bgf	14.29	7.14	0.27	1.91	1
broad leaf	Brl	14.29	7.14	0.32	2.23	1
flat ground leaf	Fgl	14.29	7.14	0.16	1.12	1
grey leaf forb	Glf	14.29	7.14	0.13	0.91	1
Hairy serrated	Hse	14.29	7.14	0.29	2.05	1
Hypoxis type	Hypt	14.29	7.14	0.07	0.50	1
leathery flat	Lfl	14.29	14.29	0.34	2.37	
long leaf forb	Llf	14.29	7.14	0.14	0.96	1
longish alternate	Lal	14.29	7.14	0.34	2.40	1
monocot	Mon	14.29	7.14	0.36	2.51	1
monocot fleshy	Mfl	14.29	7.14	0.13	0.91	1
opposite leaf rough	Olr	14.29	7.14	0.13	0.91	1
opposite leaf tiny	Olt	14.29	14.29	0.50	3.52	
opposite leaves three	Olth	14.29	14.29	0.41	2.87	
Pentanisia type	Pty	14.29	14.29	0.34	2.37	

Species	Abbrev	frequency % (lease)	frequency % (all sites)	mean relative abundance (lease)	mean abundance where found (lease)	Spp. found in lease only
purple forb	Pfo	14.29	14.29	0.48	3.35	
rosette flat	Rfl	14.29	14.29	0.18	1.23	
round stem	Rst	14.29	14.29	0.16	1.12	
sedge broad leaf	Sbl	14.29	7.14	0.14	0.96	1
sedge fine tall	Sft	14.29	21.43	0.39	2.73	
Senecio type	Sent	14.29	21.43	0.59	4.14	
two leaf forb	Tlf	14.29	7.14	0.32	2.22	1
Watsonia type	Wtp	14.29	14.29	0.16	1.12	
waxy pointed	Wpo	14.29	7.14	0.37	2.59	1
grass bulbous	Gbu	14.29	14.29	0.22	1.51	
grass paspalum type	Gpa	14.29	7.14	0.48	3.35	1
grass purple stem	Gps	14.29	14.29	0.16	1.12	
grass rough leaf	Grl	14.29	7.14	0.39	2.73	1
grass stolon	Gst	14.29	7.14	0.45	3.18	1
grass white midrib	Gwr	14.29	7.14	0.34	2.39	1
Alysicarpus rugosus	Aru	0.00	7.14	0.00	0.00	
Asclepias multicaulis	Amu	0.00	7.14	0.00	0.00	
Berkheya rhapontica	Brh	0.00	14.29	0.00	0.00	
Berkheya speciosa	Bspc	0.00	7.14	0.00	0.00	
Bulbostylis sp.	Busp	0.00	14.29	0.00	0.00	
Crabbea acaulis	Cac	0.00	7.14	0.00	0.00	
Crassula vaginata	Cva	0.00	7.14	0.00	0.00	
Ctenium concinnum	Cco	0.00	7.14	0.00	0.00	
Cyphia elata	Cel	0.00	7.14	0.00	0.00	
Digitaria erianthus	Der	0.00	7.14	0.00	0.00	
Eragrostis capensis	Eca	0.00	7.14	0.00	0.00	
Eriosema salignum	Esa	0.00	14.29	0.00	0.00	
Eriosema cf *squarrosa.complex*	Esq	0.00	7.14	0.00	0.00	
Eucomis autumnalis	Eau	0.00	7.14	0.00	0.00	
Eulophia clavicornis	Ecl	0.00	7.14	0.00	0.00	
Gerbera piloselloides	Gpi	0.00	7.14	0.00	0.00	
Gerbera sp 1	Ges1	0.00	7.14	0.00	0.00	
Gladiolus sericeovillosus	Gse	0.00	7.14	0.00	0.00	
Habenaria dregeana	Hdr	0.00	7.14	0.00	0.00	
Helichrysum aureonitens	Haur	0.00	14.29	0.00	0.00	
Helichrysum ecklonis	Hec	0.00	7.14	0.00	0.00	
Helichrysum sp 2	Hsp2	0.00	7.14	0.00	0.00	
Hesperanthus sp.	Hesp	0.00	7.14	0.00	0.00	
Hyparrhenia dregeana	Hydr	0.00	7.14	0.00	0.00	
Indigofera woodii	Iwo	0.00	7.14	0.00	0.00	
Indigofera sp.	Isp	0.00	7.14	0.00	0.00	
Kohautia amatyimbica	Kam	0.00	7.14	0.00	0.00	
Loudetia sp.	Lsp	0.00	7.14	0.00	0.00	
Microchloa caffra	Mca	0.00	7.14	0.00	0.00	
Miscanthus capensis	Mcap	0.00	7.14	0.00	0.00	
Monocymbium ceresiiforme	Mcer	0.00	35.71	0.00	0.00	
Nerine sp.	Ner	0.00	14.29	0.00	0.00	
Orchidaceae cf Eulophia sp.	Oeu	0.00	7.14	0.00	0.00	
Orchidicaea sp2	Osp	0.00	7.14	0.00	0.00	
Pachycarpus sp.	Pac	0.00	7.14	0.00	0.00	
Panicum ecklonii	Pec	0.00	21.43	0.00	0.00	

Species	Abbrev	frequency % (lease)	frequency % (all sites)	mean relative abundance (lease)	mean abundance where found (lease)	Spp. found in lease only
Panicum natalense	Pna	0.00	14.29	0.00	0.00	
Rhus sp.	Rhsp	0.00	7.14	0.00	0.00	
Rhynchosia cf caribaea	Rca	0.00	14.29	0.00	0.00	
Satyrium macrophyllum	Sma	0.00	7.14	0.00	0.00	
Scilla nervosa	Sne	0.00	14.29	0.00	0.00	
Sebaea filifolius	Sfi	0.00	14.29	0.00	0.00	
Sebaea sp.	Seb	0.00	7.14	0.00	0.00	
Senecio hygrophilus	Shy	0.00	14.29	0.00	0.00	
Setaria nigrirostris	Sni	0.00	7.14	0.00	0.00	
Setaria sphacelata	Ssph	0.00	7.14	0.00	0.00	
Silene burchelli	Sibu	0.00	7.14	0.00	0.00	
Trachypogon sp.	Trsp	0.00	7.14	0.00	0.00	
Wahlenbergia montana	Wmo	0.00	7.14	0.00	0.00	
Zaluzianskya sp.	Zal	0.00	7.14	0.00	0.00	
aromatic	Aro	0.00	7.14	0.00	0.00	
dried forb	Dfo	0.00	7.14	0.00	0.00	
flat spotted	Fsp	0.00	7.14	0.00	0.00	
Helichrysum type	Heli	0.00	7.14	0.00	0.00	
heart shaped leaf	Hsl	0.00	7.14	0.00	0.00	
leafless thorny	Lth	0.00	7.14	0.00	0.00	
monocot red base	Mrb	0.00	21.43	0.00	0.00	
narrow leaf forb	Nlf	0.00	7.14	0.00	0.00	
narrow pointed	Npo	0.00	7.14	0.00	0.00	
one stem forb	Osf	0.00	7.14	0.00	0.00	
Plectranthus type	Plty	0.00	7.14	0.00	0.00	
round serrated	Rse	0.00	7.14	0.00	0.00	
Sedge long leaf	Sll	0.00	7.14	0.00	0.00	
sessile forb	Sefo	0.00	7.14	0.00	0.00	
silver	Silv	0.00	7.14	0.00	0.00	
soft fleshy	Sfle	0.00	7.14	0.00	0.00	
tear drop forb	Tdf	0.00	7.14	0.00	0.00	
tiny serrated forb	Tsf	0.00	7.14	0.00	0.00	
zigzag stem	Zzs	0.00	7.14	0.00	0.00	

Frequency % (all sites): Percentage of sites where the species was sampled
Frequency % (cons): Percentage of conservation sites where the species was sampled
Frequency % (lease): Percentage of lease sites where the species was sampled
Mean where found: The sum of the relative abundances was divided by the number of sites where the sample was found (frequency).

☼ Indicates a species with medicinal or spiritual value
▲ Indicates an exotic species (Killlick 1990, Bromilow 1995, Hutchings 1996, Pooley 1998)

Appendix 4: Vegetation Summary Table (conservation land)

Species	Abbrev	frequency % (cons)	frequency % (all sites)	mean relative abundance (cons)	mean where found (cons)	Spp.found in cons only
Themeda triandra	Ttr	86	93	2.89	3.37	
Aster bakerianus	Aba	71	79	1.25	1.74	
Monocymbium ceresiiforme	Mcer	71	36	2.81	3.93	1
Polygala gerrardii	Pge	71	64	1.13	1.58	
Senecio bupluroides	Sbu	71	64	2.43	3.40	
Trachypogon spicactus	Tsp	71	64	1.56	2.19	
Tristachya leucothrix	Tle	71	79	2.44	3.42	
Vernonia natalensis	Vna	71	79	1.57	2.20	
Acalypha punctata	Apu	57	50	2.20	3.85	
Bulbostylus humilis	Buh	57	64	1.84	3.22	
Diheteropogon filifolius	Difil	57	36	1.78	3.12	
Eragrostis racemosa	Era	57	71	1.93	3.38	
Eulalia villosa	Evi	57	57	1.27	2.22	
Gladiolus crassifolius	Gcr	57	43	0.95	1.66	
Harpochloa falx	Hfa	57	64	1.40	2.45	
Helichrysym micronifolium	Hmi	57	71	0.63	1.10	
Hyparrhenia hirta	Hhi	57	57	1.74	3.04	
Oxalis obliquifolia	Oxob	57	64	1.59	2.79	
Polygala gracilenta	Pgr	57	71	0.95	1.65	
Sebaea sedoides	Sse	57	50	1.89	3.30	
Barleria monticola	Bmo	43	36	0.96	2.24	
Corycium nigrescens	Cni	43	43	0.89	2.07	
Diheteropogon amplectens	Diam	43	50	1.33	3.10	
Eragrostis curvula	Ecu	43	36	0.48	1.11	
Helichrysym aureum	Hau	43	29	1.19	2.77	
Helichyrsum oreophilum	Hor	43	36	1.00	2.34	
Hypoxis rigidula	Hri	43	29	0.67	1.56	
Panicum ecklonii	Pec	43	21	0.92	2.15	1
Pentanisia angustifolia	Pau	43	71	1.02	2.37	
Pteridium sp.	Pte	43	36	1.04	2.42	
monocot red base	Mrb	43	21	1.02	2.38	1
Acalypha sp.	Asp	29	29	0.92	3.23	
Alectra sessiflora	Ases	29	36	0.48	1.66	
Alloteropsis semialata	Asem	29	36	1.06	3.71	
Aristida congesta congesta	Acc	29	21	0.78	2.71	
Athrixia sp.	Atsp	29	29	0.62	2.16	
Berkheya rhapontica	Brh	29	14	0.48	1.70	1☼
Buchnera simplex	Bsi	29	29	0.59	2.07	
Bulbostylus sp.	Busp	29	14	0.63	2.21	1
Conyza pinnata	Cpi	29	29	0.59	2.07	
Cyperus sp.	Cyp	29	29	0.69	2.40	
Digitaria flaccida	Dfl	29	21	0.31	1.08	
Digitaria tricholaenoides	Dtr	29	29	0.74	2.58	
Eriosema salignum	Esa	29	14	0.72	2.53	1☼
Euphorbia striata	Est	29	29	0.28	0.97	
Gerbera kraussii	Gkra	29	21	0.19	0.66	

Species	Abbrev	frequency % (cons)	freq % (all sites)	mean relative abundance (cons)	mean where found (cons)	Spp. found in cons only
Gladilous sp1	Gsp1	29	21	0.34	1.19	
Graderia scabra	Gsc	29	29	0.96	3.37	
Hermannia gerrardii	He	29	36	0.55	1.92	
Helichrysum aureonitens	Haur	29	14	0.81	2.83	1☼
Helichrysum coriaceum	Hco	29	21	0.70	2.45	
Heteropogon contortus	Hcon	29	36	0.96	3.34	
Loudetia simplex	Lsi	29	21	0.49	1.71	
Nerine sp.	Ner	29	14	0.28	0.98	
Panicum natalense	Pna	29	14	1.07	3.76	1
Rendlia altera	Ral	29	21	0.96	3.34	
Rhynchosia cf *caribaea*	Rca	29	14	0.70	2.45	1☼
Scilla nervosa	Sne	29	14	0.90	3.14	1☼
Sebaea filifolius	Sfi	29	14	0.76	2.65	1
Senecio hygrophilus	Shy	29	14	0.28	0.98	1
Setaria sp.	Set	29	29	0.84	2.93	
Ledebouria type	Led	29	36	0.79	2.77	
Polygala type	Poty	29	29	0.62	2.17	
Sedge fine tall	Sft	29	21	0.63	2.19	
Senecio type	Sent	29	21	0.87	3.05	
Acalypha schinzii	Asc	14	43	0.14	0.97	
Alysicarpus rugosus	Aru	14	7	0.38	2.64	1☼
Anthospermum herbaceum	Ahe	14	14	0.14	0.97	
Anthospermum rigidum subsp. *pomilum*	Arp	14	21	0.50	3.47	
Aristida junciformis	Aju	14	21	0.38	2.64	
Asclepias multicaulis	Amu	14	7	0.16	1.13	1
Aster perfoliatus	Ape	14	14	0.50	3.47	
Berkheya setifera	Bse	14	21	0.48	3.38	
Berkheya speciosa	Bspc	14	7	0.11	0.75	1☼
Brachiaria serrata	Bser	14	36	0.38	2.64	
Commelina africana	Caf	14	14	0.11	0.75	
Crabbea acaulis	Cac	14	7	0.11	0.75	1
Crassula vaginata	Cva	14	7	0.28	1.98	1☼
Ctenium concinnum	Cco	14	7	0.98	6.86	1
Cyphia elata	Cel	14	7	0.14	0.97	1☼
Digitaria erianthus	Der	14	7	0.41	2.90	1
Digitaria sp.	Dsp	14	14	0.38	2.64	
Elionorus muticus	Emu	14	29	0.32	2.26	
Eragrostis capensis	Eca	14	7	0.28	1.93	1
Eriosema kraussianum	Ekr	14	21	0.58	4.05	
Eriosema cf *squarrosa complex*	Esq	14	7	0.56	3.92	
Eucomis autumnalis	Eau	14	7	0.47	3.29	1☼♠
Eulophia clavicornis	Ecl	14	7	0.47	3.29	1☼
Gerbera piloselloides	Gpi	14	7	0.35	2.42	1
Gerbera sp 1	Ges1	14	7	0.41	2.90	1
Gladiolus sericeovillosus	Gse	14	7	0.28	1.96	1☼
Gnidia kraussiana	Gkr	14	21	0.50	3.47	
Gnidia splendens	Gsp	14	14	0.35	2.42	

Species	Abbrev	frequency % (cons)	freq % (all sites)	mean relative abundance (cons)	mean where found (cons)	Spp.found in cons only
Habenaria dregeana	Hdr	14	7	0.11	0.75	1☼♠
Haplocarpa scaposa	Hsc	14	14	0.25	1.73	
Hebenstretia sp.	Hebs	14	14	0.48	3.38	
Herman. sp	Hes	14	14	0.07	0.48	
Helichrysum ecklonis	Hec	14	7	0.25	1.73	1
Helichrysum sp 1	Hsp1	14	14	0.35	2.48	
Helichrysum sp 2	Hsp2	14	7	0.28	1.98	1
Hesperanthus sp.	Hesp	14	7	0.27	1.88	1
Hyparrhenia dregeana	Hydr	14	7	0.98	6.86	1
Hyparrhenia sp.	Hysp	14	14	0.27	1.89	
Hypericum aethiopicum	Hae	14	29	0.14	0.99	
Hypoxis obtusa	Hob	14	14	0.17	1.16	
Hypoxis sp.	Hypo	14	21	0.70	4.90	
Indigofera woodii	Iwo	14	7	0.21	1.45	1
Indigofera sp.	Isp	14	7	0.14	0.99	1
Kohautia amatyimbica	Kam	14	7	0.28	1.96	1☼
Ledebouria sp.	Led	14	14	0.21	1.45	
Loudetia sp.	Lsp	14	7	0.70	4.90	1
Microchloa caffra	Mca	14	7	0.05	0.38	1
Miscanthus capensis	Mcap	14	7	0.28	1.96	1
Orchidaceae cf *Eulophia* sp.	Oeu	14	7	0.98	6.86	1
Orchidacea cf. *Corycium nigrescens*	Oco	14	14	0.28	1.98	
Orchidicaea sp2	Osp	14	7	0.33	2.29	1
Pachycarpus sp.	Pac	14	7	0.07	0.48	1☼
Paspalum sp.	Psp	14	21	0.48	3.38	
Polygala ohlendorfia	Poh	14	14	0.20	1.41	
Rhus discolor	Rdi	14	21	0.48	3.38	
Rhus sp.	Rhsp	14	7	0.47	3.27	1
Rhynchosia totta	Rto	14	14	0.38	2.64	
Satyrium macrophyllum	Sma	14	7	0.11	0.75	1
Scleria sp.	Ssp	14	14	0.48	3.38	
Sebaea grandis	Sgr	14	21	0.34	2.35	
Sebaea sp.	Seb	14	7	0.84	5.88	1
Senecio glaberrimus	Sgl	14	14	0.50	3.47	
Setaria nigrirostris	Sni	14	7	0.14	0.99	1
Setaria sphacelata	Ssph	14	7	0.28	1.96	1
Silene burchelli	Sibu	14	7	0.11	0.75	1☼
Sopubia cana	Sca	14	14	0.08	0.58	
Striga bilabiata	Sbil	14	14	0.16	1.13	
Trachypogon sp.	Trsp	14	7	0.38	2.64	1
Wahlenbergia montana	Wmo	14	7	0.17	1.16	1
Zaluzianskya sp.	Zal	14	7	0.27	1.89	1☼
Zornia capensis	Zca	14	29	0.41	2.90	
Acalypha type	Aca	14	14	0.21	1.45	
aromatic	Aro	14	7	0.05	0.38	1
dried forb	Dfo	14	7	0.17	1.16	1
flat spotted	Fsp	14	7	0.16	1.13	1
Helichrysum type	Heli	14	7	0.41	2.89	1
heart shaped leaf	Hsl	14	7	0.50	3.47	1

Species	Abbrev	frequency % (cons)	freq % (all sites)	mean relative abundance (cons)	mean where found (cons)	Spp. found in cons only
Large waxy	Lwa	14	29	0.35	2.42	
leafless thorny	Lth	14	7	0.47	3.29	1
leathery flat	Lfl	14	14	0.50	3.47	
narrow leaf forb	Nlf	14	7	0.11	0.75	1
narrow pointed	Npo	14	7	0.34	2.35	1
one stem forb	Osf	14	7	0.20	1.41	1
opposite leaf forb	Olf	14	43	0.11	0.75	
opposite leaf tiny	Olt	14	14	0.98	6.86	
opposite leaves three	Olth	14	14	0.41	2.90	
Pentanisia type	Pty	14	14	0.21	1.45	
Plectranthus type	Plty	14	7	0.16	1.13	1
purple forb	Pfo	14	14	0.05	0.38	
rosette flat	Rfl	14	14	0.38	2.64	
round serrated	Rse	14	7	0.34	2.35	1
round stem	Rst	14	14	0.41	2.90	
sedge long leaf	Sll	14	7	0.58	4.05	1
sessile forb	Sefo	14	7	0.38	2.64	1
silver	Silv	14	7	0.21	1.45	1
soft fleshy	Sfle	14	7	0.35	2.42	1
tear drop forb	Tdf	14	7	0.05	0.38	1
tiny serrated forb	Tsf	14	7	0.40	2.82	1
Watsonia *type*	Wtp	14	14	0.48	3.38	
zigzag stem	Zzs	14	7	0.14	0.98	1
grass bulbous	Gbu	14	14	0.40	2.82	
grass purple stem	Gps	14	14	0.28	1.93	
grass red stem	Grs	14	21	0.35	2.42	
Artemisia afra	Aaf	0	7	0.00	0.00	
Athrixia phyllicoides	Aph	0	7	0.00	0.00	
Becium obovatum	Bob	0	14	0.00	0.00	
Becium sp.	Bsp	0	7	0.00	0.00	
Berkheya sp 1	Bsp1	0	14	0.00	0.00	
Berkheya sp 2	Bsp2	0	7	0.00	0.00	
Cephelaria oblongifolia	Cob	0	7	0.00	0.00	
Chamaechrista sp.	Chsp	0	7	0.00	0.00	
Crassula sp.	Crsp	0	7	0.00	0.00	
Cucumis zeyheri	Cze	0	7	0.00	0.00	
Cymbopogon sp.	Cyms	0	7	0.00	0.00	
Diclis reptans	Dre	0	7	0.00	0.00	
Eragrostis plana	Epla	0	14	0.00	0.00	
Eriospermum cf.*cooperi*	Eco	0	7	0.00	0.00	
Gerbera ambigua	Gam	0	7	0.00	0.00	
Gerbera sp 2	Ges2	0	7	0.00	0.00	
Gladiolus cf *ecklonii*	Gec	0	7	0.00	0.00	1☼
Gladiolus sp2	Gsp2	0	7	0.00	0.00	
Gladiolus sp3	Gsp3	0	7	0.00	0.00	
Helichrysum adenocarpum	Had	0	7	0.00	0.00	
Helichrysum glomeratum	Hgl	0	7	0.00	0.00	
Helichrysum herbaceum	Hhe	0	14	0.00	0.00	
Helichrysum rugulosum	Hru	0	14	0.00	0.00	
Helichrysum sutherlandii	Hsu	0	7	0.00	0.00	
Hypoxis multiceps	Hmu	0	14	0.00	0.00	

Species	Abbrev	frequency % (cons)	freq % (all sites)	mean relative abundance (cons)	mean where found (cons)	Spp. found in cons only
Melinis nerviglumis	Mne	0	14	0.00	0.00	
Melinis sp.	Msp	0	7	0.00	0.00	
Miscanthus sp.	Misp	0	7	0.00	0.00	
Nidorella auriculata	Nau	0	7	0.00	0.00	
Oxalis corniculata	Oxco	0	7	0.00	0.00	
Paspalum dilatatum	Pdi	0	7	0.00	0.00	
Peucedanum caffra	Pca	0	7	0.00	0.00	
Rhynchosia sp.	Rsp	0	7	0.00	0.00	
Rubus cuneifolius	Rcu	0	7	0.00	0.00	
Senecio sp1	Sen1	0	14	0.00	0.00	
Senecio sp2	Sen2	0	21	0.00	0.00	
Setaria pallide-fusca	Spa	0	7	0.00	0.00	
Sonchus oleraceae	Sol	0	14	0.00	0.00	
Spermacoce natalense	Sna	0	7	0.00	0.00	
Spermacoce senensis	Spse	0	7	0.00	0.00	
Sporobolus africanus	Saf	0	14	0.00	0.00	
Thecium sp.	The	0	7	0.00	0.00	
Watsonia sp.	Wsp	0	7	0.00	0.00	
alternate leaf rough	Alt	0	7	0.00	0.00	
black edge forb	Bef	0	7	0.00	0.00	
bright green forb	Bgf	0	7	0.00	0.00	
broad leaf	Brl	0	7	0.00	0.00	
flat ground leaf	Fgl	0	7	0.00	0.00	
grey leaf forb	Glf	0	7	0.00	0.00	
hairy serrated	Hse	0	7	0.00	0.00	
Hypoxis type	Hypt	0	7	0.00	0.00	
long leaf forb	Llf	0	7	0.00	0.00	
longish alternate	Lal	0	7	0.00	0.00	
monocot	Mon	0	7	0.00	0.00	
monocot fleshy	Mfl	0	7	0.00	0.00	
opposite leaf rough	Olr	0	7	0.00	0.00	
sedge broad leaf	Sbl	0	7	0.00	0.00	
Selago type	Styp	0	14	0.00	0.00	
sandpaper forb	Sfo	0	14	0.00	0.00	
two leaf forb	Tlf	0	7	0.00	0.00	
waxy pointed	Wpo	0	7	0.00	0.00	
grass paspalum type	Gpa	0	7	0.00	0.00	
grass rough leaf	Grl	0	7	0.00	0.00	
grass stolon	Gst	0	7	0.00	0.00	
grass white midrib	Gwr	0	7	0.00	0.00	

Frequency % (all sites): Percentage of sites where the species was sampled
Frequency % (cons): Percentage of conservation sites where the species was sampled
Frequency % (lease): Percentage of lease sites where the species was sampled
Mean where found: The sum of the relative abundances was divided by the number of sites where the sample was found (frequency).

☼ Indicates a species with medicinal or spiritual value
♠ Indicates a rare or vulnerable species
(Killlick 1990, Bromilow 1995, Hutchings 1996, Pooley 1998)

Appendix 5: Summary table: Invertebrate sweep-netting

Morpho-species	frequency (all sites) n=14	% frequency (cons) n=7	% abundance where found (cons)	frequency % (lease) n=7	abundance where found (lease) n=7
cic 1	36	29	0.20	43	0.93
cic 2	7	14	0.40	0	0.00
cic 3	36	43	0.33	29	0.20
cic 4	29	29	0.20	29	0.70
cic 5	14	0	0.00	29	0.60
cic 6	36	43	0.47	29	1.80
cic 7	29	43	0.33	14	0.20
cic 8	7	14	0.20	0	0.00
cic 9	57	43	0.53	71	0.48
cic 10	29	43	0.27	14	0.40
cic 11	50	57	0.65	43	0.80
cic 12	14	0	0.00	29	0.20
cic 13	64	86	2.50	43	0.67
cic 15	57	43	0.93	71	0.36
cic 16	7	0	0.00	14	0.20
cic 17	7	0	0.00	14	0.20
cic 18	0	0	0.00	0	0.00
cic 19	7	0	0.00	14	0.20
cic 20	0	0	0.00	0	0.00
cic 21	0	0	0.00	0	0.00
cic 22	7	14	0.20	0	0.00
cic 23	7	14	0.20	0	0.00
cic 24	57	71	2.24	43	0.27
cic 25	14	14	1.00	14	0.25
cic 26	7	14	0.20	0	0.00
cic 27	0	0	0.00	0	0.00
cic 28	7	0	0.00	14	1.20
cic 29	14	14	0.20	14	0.20
cic 30	7	14	0.20	0	0.00
cic 31	14	29	0.20	0	0.00
cic 32	36	29	0.50	43	0.27
cic 34	43	29	0.60	57	1.20
cic 35	36	0	0.00	71	0.32
cic 36	7	0	0.00	14	0.20
cic 37	7	0	0.00	14	0.20
cic 38	7	0	0.00	14	0.20
cic 39	0	0	0.00	0	0.00
cic 40	7	0	0.00	14	0.20
cic 41	14	14	0.20	14	0.20
cic 42	21	29	0.20	14	0.40
cic 43	21	14	0.20	29	0.20
cic 44	36	29	0.30	43	0.20
cic 46	0	0	0.00	0	0.00
cic 47	7	14	0.20	0	0.00
cic 49	14	14	0.20	14	0.40
cic 50	14	14	0.60	14	0.20
cic 51	36	43	0.47	29	0.40
cic 52	7	14	0.20	0	0.00
cic 53	21	14	0.20	29	0.30
cic 54	7	14	0.40	0	0.00

Morpho-species	frequency (all sites) n=14	% frequency (cons) n=7	% abundance where found (cons)	frequency % (lease) n=7	abundance where found (lease) n=7
cic 55	14	29	0.20	0	0.00
cic 56	14	14	1.00	14	0.40
cic 57	7	14	0.20	0	0.00
cic 58	14	14	0.20	14	0.20
cic 59	7	0	0.00	14	0.20
cic 60	7	0	0.00	14	0.20
cic 62	0	0	0.00	0	0.00
cic 64	7	14	0.20	0	0.00
cic 65	14	14	0.40	14	0.40
cic 67	14	14	0.20	14	0.20
cic 68	14	14	0.20	14	0.20
cic 69	21	0	0.00	43	0.20
cic 70	7	0	0.00	14	0.20
cic 71	14	14	0.20	14	0.40
cic 72	14	14	0.20	14	0.60
cic 73	7	0	0.00	14	0.20
col 1	0	0	0.00	0	0.00
col 2	14	14	0.20	14	0.20
col 3	0	0	0.00	0	0.00
col 4	0	0	0.00	0	0.00
col 5	0	0	0.00	0	0.00
col 6	14	14	0.20	14	1.00
col 7	64	71	0.72	57	2.45
col 8	43	43	3.87	43	4.80
col 9	29	57	0.40	0	0.00
col 10	29	14	0.20	43	0.33
col 11	29	29	0.20	29	0.50
col 12	21	14	0.20	29	0.20
col 13	7	0	0.00	14	0.20
col 14	36	29	0.30	43	0.80
col 15	7	14	0.20	0	0.00
col 16	21	29	0.20	14	0.40
col 17	21	0	0.00	43	0.20
col 18	0	0	0.00	0	0.00
col 19	21	14	0.20	29	0.20
col 20	14	29	0.50	0	0.00
col 21	7	14	0.20	0	0.00
col 22	7	14	0.20	0	0.00
col 23	7	14	0.20	0	0.00
col 24	0	0	0.00	0	0.00
col 25	14	29	0.20	0	0.00
col 26	7	0	0.00	14	0.20
dip 1	0	0	0.00	0	0.00
dip 2	7	0	0.00	14	0.20
dip 3	7	14	0.20	0	0.00
form 1	86	86	2.40	86	1.23
form 2	7	14	0.20	0	0.00
form 3	21	29	0.30	14	0.20
form 4	64	71	1.16	57	2.25
form 5	36	14	1.00	57	0.75
form 6	0	0	0.00	0	0.00
form 7	21	14	0.20	29	0.30

Morpho-species	frequency (all sites) n=14	% frequency (cons) n=7	% abundance where found (cons)	frequency % (lease) n=7	abundance where found (lease) n=7
form 8	21	14	0.20	29	0.30
form 9	0	0	0.00	0	0.00
form 10	7	0	0.00	14	0.20
hym 1	7	0	0.00	14	0.20
hym 2	0	0	0.00	0	0.00
hym 3	7	14	0.20	0	0.00
hym 4	0	0	0.00	0	0.00
hym 5	0	0	0.00	0	0.00
hym 6	0	0	0.00	0	0.00
hym 7	7	0	0.00	14	0.20
hym 8	7	0	0.00	14	0.20
hym 9	43	43	1.13	43	0.47
hym 10	14	29	0.20	0	0.00
hym 11	7	14	0.40	0	0.00
hym 12	7	14	0.20	0	0.00
hym 13	7	14	0.20	0	0.00
hym 14	7	14	0.20	0	0.00
hym 15	7	14	0.20	0	0.00
hym 16	7	14	0.20	0	0.00
hym 18	7	14	0.20	0	0.00
hym 19	14	0	0.00	29	0.20
hym 20	0	0	0.00	0	0.00
hym 21	7	14	0.20	0	0.00
hym 22	0	0	0.00	0	0.00
hym 23	0	0	0.00	0	0.00
hym 24	0	0	0.00	0	0.00
hym 25	0	0	0.00	0	0.00
hym 26	0	0	0.00	0	0.00
hym 27	0	0	0.00	0	0.00
hym 28	7	0	0.00	14	0.20
hym 30	7	0	0.00	14	0.20
hym 31	7	0	0.00	14	0.20
hym 32	7	0	0.00	14	0.20
orth 1	29	14	0.20	43	0.20
orth 2	36	57	4.20	14	0.20
orth 3	0	0	0.00	0	0.00
orth 4	14	14	0.20	14	0.20
orth 6	7	0	0.00	14	0.20
orth 8	14	14	0.40	14	0.20
orth 9	14	14	0.20	14	0.80
orth 11	50	57	0.30	43	0.47
orth 12	21	14	0.20	29	0.30
orth 13	0	0	0.00	0	0.00
orth 14	36	29	2.00	43	0.27
orth 15	14	14	0.20	14	0.20
orth 16	7	14	0.20	0	0.00
orth 17	7	14	0.20	0	0.00
orth 18	14	29	0.20	0	0.00
orth 19	14	29	0.20	0	0.00
orth 20	14	14	0.20	14	0.60
orth 21	7	0	0.00	14	0.20
orth 22	7	0	0.00	14	0.20

Morpho-species	frequency (all sites) n=14	% frequency (cons) n=7	% abundance where found (cons)	frequency % (lease) n=7	abundance where found (lease) n=7
orth 23	7	0	0.00	14	0.20
orth 24	7	14	0.20	0	0.00
orth 25	57	43	0.53	71	0.24
orth 26	7	14	0.20	0	0.00
orth 27	36	43	0.33	29	0.40
orth 28	0	0	0.00	0	0.00
orth 29	7	14	0.40	0	0.00
orth 30	7	14	0.40	0	0.00
orth 31	86	86	5.97	86	0.80
orth 32	79	71	0.96	86	0.53
orth 33	21	43	0.40	0	0.00
orth 34	43	71	0.44	14	0.60
orth 35	50	43	0.40	57	0.90
orth 36	36	29	0.50	43	0.20
orth 37	14	29	0.20	0	0.00
orth 38	43	14	2.67	71	0.24
orth 39	86	86	3.40	86	4.67
orth 40	29	14	0.20	43	0.60
orth 41	7	0	0.00	14	0.80
orth 42	50	29	0.40	71	0.32
orth 43	29	29	0.20	29	0.20
orth 44	29	14	0.20	43	0.33
orth 45	29	14	0.20	43	0.33
orth 46	43	29	0.50	57	0.30
orth 47	7	0	0.00	14	0.20
orth 48	7	0	0.00	14	0.20
orth 49	7	0	0.00	14	0.20
orth 50	64	71	0.68	57	0.45
orth 51	7	0	0.00	14	0.60
orth 52	36	43	0.60	29	0.20
orth 53	14	14	0.20	14	0.60
orth 54	14	0	0.00	29	0.20
orth 55	7	0	0.00	14	0.20
orth 56	21	29	0.50	14	0.40
orth 57	14	14	0.20	14	0.20
orth 58	36	29	0.40	43	0.20
orth 59	7	14	0.20	0	0.00
orth 60	7	14	0.40	0	0.00
orth 61	14	14	0.20	14	0.40
orth 62	14	14	0.20	14	0.20
orth 63	36	14	0.20	57	0.25
orth 64	21	14	0.20	29	0.30
orth 65	21	29	0.20	14	0.20
orth 66	29	14	0.20	43	0.20
orth 67	7	0	0.00	14	0.20
orth 68	14	14	0.20	14	0.20
orth 69	7	14	0.20	0	0.00
orth 70	7	0	0.00	14	0.40
orth 71	7	0	0.00	14	0.40
orth 72	21	29	0.30	14	0.20
orth 73	7	0	0.00	14	0.60
orth 74	14	14	0.40	14	0.60

Morpho-species	frequency % (all sites) n=14	frequency % (cons) n=7	abundance where found (cons)	frequency % (lease) n=7	abundance where found (lease) n=7
orth 75	7	0	0.00	14	0.20
orth 76	7	0	0.00	14	0.20
orth 77	14	0	0.00	29	0.20
orth 78	7	0	0.00	14	0.20
orth 79	29	43	0.33	14	0.20
orth 80	7	14	0.20	0	0.00
orth 81	14	14	0.80	14	0.20
orth 82	14	0	0.00	29	0.30
orth 83	14	14	0.20	14	0.20
orth 84	7	14	0.20	0	0.00
orth 85	7	14	0.20	0	0.00
orth 86	14	29	0.20	0	0.00
orth 87	7	0	0.00	14	0.20
orth 88	7	14	0.20	0	0.00
orth 89	0	0	0.00	0	0.00
orth 90	0	0	0.00	0	0.00
orth 91	0	0	0.00	0	0.00
thom 1	21	14	0.20	29	0.30
thom 2	0	0	0.00	0	0.00
thom 3	7	0	0.00	14	0.20
thom 4	0	0	0.00	0	0.00
thom 5	64	71	0.87	57	0.50
thom 6	7	14	0.20	0	0.00
thom 7	14	29	0.20	0	0.00
thom 8	50	57	0.38	43	0.40
thom 9	7	0	0.00	14	0.20
thom 10	7	0	0.00	14	0.20
thom 11	7	14	0.20	0	0.00
thom 12	21	29	0.20	14	0.40
thom 13	14	29	0.20	0	0.00
thom 14	7	0	0.00	14	0.20
thom 15	0	0	0.00	0	0.00
thom 16	7	0	0.00	14	0.20
thom 17	7	0	0.00	14	0.20
thom 19	57	43	0.80	71	0.40
thom 20	71	86	1.00	57	0.35
thom 21	7	0	0.00	14	0.40
thom 22	7	14	0.20	0	0.00
thom 23	21	29	0.20	14	0.20

Frequency % (all sites): Percentage of sites where the morphospecies was sampled
Frequency % (cons): Percentage of conservation sites where the morphospecies was sampled
Frequency % (lease): Percentage of lease sites where the morphospecies was sampled
Abundance where found: The total number of sweep samples collected in each site was divided by 5 to get the mean number of samples per set of sweeps. This figure was divided by the number of sites where the sample was found (frequency).

Appendix 6: Summary table: Invertebrate pan traps

Morpho-species	frequency % all sites n=14	frequency% (cons) n=7	Abundance where found (cons)	frequency% (lease) n=7	Abundance where found (lease)
cic1	35.71	14.29	0.47	57	0.14
cic2	28.57	14.29	0.07	43	0.10
cic3	7.14	0.00	0.00	14	0.13
cic4	21.43	14.29	0.73	29	0.17
cic5	0.00	0.00	0.00	0	0.00
cic6	35.71	57.14	0.23	14	0.40
cic7	14.29	28.57	0.27	0	0.00
cic8	0.00	0.00	0.00	0	0.00
cic9	21.43	14.29	0.20	29	0.20
cic10	14.29	14.29	0.13	14	0.10
cic11	7.14	14.29	0.73	0	0.00
cic12	7.14	0.00	0.00	14	0.20
cic13	21.43	14.29	0.27	29	0.17
cic15	7.14	0.00	0.00	14	0.07
cic16	0.00	0.00	0.00	0	0.00
cic17	0.00	0.00	0.00	0	0.00
cic18	7.14	0.00	0.00	14	0.07
cic19	7.14	0.00	0.00	14	0.07
cic20	7.14	0.00	0.00	14	0.07
cic21	7.14	0.00	0.00	14	0.20
cic22	0.00	0.00	0.00	0	0.00
cic23	0.00	0.00	0.00	0	0.00
cic24	14.29	14.29	0.13	14	0.33
cic25	21.43	28.57	0.08	14	1.00
cic26	0.00	0.00	0.00	0	0.00
cic27	7.14	0.00	0.00	14	0.07
cic28	21.43	28.57	0.15	14	0.07
cic29	0.00	0.00	0.00	0	0.00
cic30	0.00	0.00	0.00	0	0.00
cic31	0.00	0.00	0.00	0	0.00
cic32	28.57	42.86	0.09	14	0.07
cic34	0.00	0.00	0.00	0	0.00
cic35	14.29	14.29	0.13	14	0.07
cic36	0.00	0.00	0.00	0	0.00
cic37	0.00	0.00	0.00	0	0.00
cic38	0.00	0.00	0.00	0	0.00
cic39	7.14	14.29	0.07	0	0.00
cic40	0.00	0.00	0.00	0	0.00
cic41	0.00	0.00	0.00	0	0.00
cic42	0.00	0.00	0.00	0	0.00
cic43	0.00	0.00	0.00	0	0.00
cic44	0.00	0.00	0.00	0	0.00
cic46	7.14	14.29	0.07	0	0.00
cic47	14.29	14.29	0.07	14	0.10
cic49	7.14	14.29	0.07	0	0.00
cic50	0.00	0.00	0.00	0	0.00
cic51	0.00	0.00	0.00	0	0.00
cic52	0.00	0.00	0.00	0	0.00
cic53	0.00	0.00	0.00	0	0.00

Morpho-species	frequency % all sites n=14	frequency% (cons) n=7	Abundance where found (cons)	frequency% (lease) n=7	Abundance where found (lease)
cic54	0.00	0.00	0.00	0	0.00
cic55	0.00	0.00	0.00	0	0.00
cic56	0.00	0.00	0.00	0	0.00
cic57	14.29	14.29	0.93	14	0.07
cic58	0.00	0.00	0.00	0	0.00
cic59	0.00	0.00	0.00	0	0.00
cic60	0.00	0.00	0.00	0	0.00
cic62	7.14	14.29	0.07	0	0.00
cic64	0.00	0.00	0.00	0	0.00
cic65	0.00	0.00	0.00	0	0.00
cic67	0.00	0.00	0.00	0	0.00
cic68	0.00	0.00	0.00	0	0.00
cic69	0.00	0.00	0.00	0	0.00
cic70	0.00	0.00	0.00	0	0.00
cic71	0.00	0.00	0.00	0	0.00
cic72	7.14	14.29	0.13	0	0.00
cic73	7.14	14.29	0.07	0	0.00
col1	7.14	0.00	0.00	14	0.07
col2	64.29	57.14	0.42	71	0.39
col3	14.29	14.29	0.13	14	0.07
col4	14.29	14.29	0.13	14	0.07
col5	7.14	14.29	0.07	0	0.00
col6	7.14	0.00	0.00	14	0.07
col7	7.14	0.00	0.00	14	0.20
col8	0.00	0.00	0.00	0	0.00
col9	0.00	0.00	0.00	0	0.00
col10	0.00	0.00	0.00	0	0.00
col11	7.14	0.00	0.00	14	0.07
col12	0.00	0.00	0.00	0	0.00
col13	0.00	0.00	0.00	0	0.00
col14	0.00	0.00	0.00	0	0.00
col15	0.00	0.00	0.00	0	0.00
col16	0.00	0.00	0.00	0	0.00
col17	0.00	0.00	0.00	0	0.00
col18	7.14	14.29	0.10	0	0.00
col19	0.00	0.00	0.00	0	0.00
col20	0.00`	0.00	0.00	0	0.00
col21	0.00	0.00	0.00	0	0.00
col22	0.00	0.00	0.00	0	0.00
col23	0.00	0.00	0.00	0	0.00
col24	7.14	0.00	0.00	14	0.07
col25	7.14	0.00	0.00	14	0.07
col26	0.00	0.00	0.00	0	0.00
dip1	14.29	14.29	0.07	14	0.33
dip2	0.00	0.00	0.00	0	0.00
dip3	0.00	0.00	0.00	0	0.00
form1	57.14	57.14	19.68	57	0.65
form2	42.86	28.57	0.27	57	0.13
form3	28.57	14.29	0.30	43	0.29
form4	50.00	42.86	0.33	57	0.40
form5	7.14	0.00	0.00	14	0.20
form6	28.57	28.57	0.22	29	0.27
form7	0.00	0.00	0.00	0	0.00

Morpho-species	frequency % all sites n=14	frequency% (cons) n=7	Abundance where found (cons)	frequency% (lease) n=7	Abundance where found (lease)
form8	7.14	14.29	0.30	0	0.00
form9	7.14	14.29	0.07	0	0.00
form10	0.00	0.00	0.00	0	0.00
hym1	14.29	0.00	0.00	29	0.07
hym2	21.43	14.29	0.20	29	0.13
hym3	21.43	28.57	0.15	14	0.20
hym4	14.29	14.29	0.07	14	0.07
hym5	7.14	0.00	0.00	14	0.07
hym6	14.29	14.29	0.07	14	0.07
hym7	0.00	0.00	0.00	0	0.00
hym8	7.14	14.29	0.07	0	0.00
hym9	0.00	0.00	0.00	0	0.00
hym10	0.00	0.00	0.00	0	0.00
hym11	0.00	0.00	0.00	0	0.00
hym12	0.00	0.00	0.00	0	0.00
hym13	0.00	0.00	0.00	0	0.00
hym14	0.00	0.00	0.00	0	0.00
hym15	0.00	0.00	0.00	0	0.00
hym16	0.00	0.00	0.00	0	0.00
hym18	7.14	0.00	0.00	14	0.07
hym19	0.00	0.00	0.00	0	0.00
hym20	7.14	0.00	0.00	14	0.07
hym21	0.00	0.00	0.00	0	0.00
hym22	7.14	14.29	0.07	0	0.00
hym23	7.14	14.29	0.07	0	0.00
hym24	7.14	14.29	0.07	0	0.00
hym25	7.14	14.29	0.07	0	0.00
hym26	7.14	14.29	0.07	0	0.00
hym27	7.14	14.29	0.07	0	0.00
hym28	28.57	28.57	0.12	29	0.17
hym30	0.00	0.00	0.00	0	0.00
hym31	0.00	0.00	0.00	0	0.00
hym32	0.00	0.00	0.00	0	0.00
orth1	0.00	0.00	0.00	0	0.00
orth2	28.57	42.86	0.16	14	0.07
orth3	7.14	0.00	0.00	14	0.07
orth4	0.00	0.00	0.00	0	0.00
orth6	0.00	0.00	0.00	0	0.00
orth8	7.14	0.00	0.00	14	0.07
orth9	0.00	0.00	0.00	0	0.00
orth11	0.00	0.00	0.00	0	0.00
orth12	0.00	0.00	0.00	0	0.00
orth13	42.86	57.14	0.23	29	0.10
orth14	0.00	0.00	0.00	0	0.00
orth15	0.00	0.00	0.00	0	0.00
orth16	21.43	28.57	0.08	14	0.13
orth17	14.29	0.00	0.00	29	0.13
orth18	0.00	0.00	0.00	0	0.00
orth19	0.00	0.00	0.00	0	0.00
orth20	21.43	28.57	0.08	14	0.07
orth21	0.00	0.00	0.00	0	0.00
orth22	0.00	0.00	0.00	0	0.00
orth23	0.00	0.00	0.00	0	0.00

Morpho-species	frequency % all sites n=14	frequency% (cons) n=7	Abundance where found (cons)	frequency% (lease) n=7	Abundance where found (lease)
orth24	0.00	0.00	0.00	0	0.00
orth25	14.29	0.00	0.00	29	0.10
orth26	0.00	0.00	0.00	0	0.00
orth27	14.29	14.29	0.07	14	0.10
orth28	7.14	0.00	0.00	14	0.10
orth29	0.00	0.00	0.00	0	0.00
orth30	0.00	0.00	0.00	0	0.00
orth31	7.14	14.29	0.27	0	0.00
orth32	0.00	0.00	0.00	0	0.00
orth33	0.00	0.00	0.00	0	0.00
orth34	0.00	0.00	0.00	0	0.00
orth35	0.00	0.00	0.00	0	0.00
orth36	0.00	0.00	0.00	0	0.00
orth37	0.00	0.00	0.00	0	0.00
orth38	0.00	0.00	0.00	0	0.00
orth39	14.29	14.29	0.13	14	0.10
orth40	7.14	0.00	0.00	14	0.07
orth41	0.00	0.00	0.00	0	0.00
orth42	0.00	0.00	0.00	0	0.00
orth43	0.00	0.00	0.00	0	0.00
orth44	0.00	0.00	0.00	0	0.00
orth45	0.00	0.00	0.00	0	0.00
orth46	0.00	0.00	0.00	0	0.00
orth47	0.00	0.00	0.00	0	0.00
orth48	0.00	0.00	0.00	0	0.00
orth49	0.00	0.00	0.00	0	0.00
orth50	0.00	0.00	0.00	0	0.00
orth51	0.00	0.00	0.00	0	0.00
orth52	14.29	14.29	0.07	14	0.07
orth53	0.00	0.00	0.00	0	0.00
orth54	7.14	14.29	0.07	0	0.00
orth55	0.00	0.00	0.00	0	0.00
orth56	0.00	0.00	0.00	0	0.00
orth57	0.00	0.00	0.00	0	0.00
orth58	0.00	0.00	0.00	0	0.00
orth59	0.00	0.00	0.00	0	0.00
orth60	0.00	0.00	0.00	0	0.00
orth61	0.00	0.00	0.00	0	0.00
orth62	0.00	0.00	0.00	0	0.00
orth63	0.00	0.00	0.00	0	0.00
orth64	0.00	0.00	0.00	0	0.00
orth65	0.00	0.00	0.00	0	0.00
orth66	0.00	0.00	0.00	0	0.00
orth67	0.00	0.00	0.00	0	0.00
orth68	0.00	0.00	0.00	0	0.00
orth69	0.00	0.00	0.00	0	0.00
orth70	0.00	0.00	0.00	0	0.00
orth71	0.00	0.00	0.00	0	0.00
orth72	7.14	14.29	0.07	0	0.00
orth73	0.00	0.00	0.00	0	0.00
orth74	0.00	0.00	0.00	0	0.00
orth75	0.00	0.00	0.00	0	0.00
orth76	0.00	0.00	0.00	0	0.00

Morpho-species	frequency % all sites n=14	frequency% (cons) n=7	Abundance where found (cons)	frequency% (lease) n=7	Abundance where found (lease)
orth77	0.00	0.00	0.00	0	0.00
orth78	0.00	0.00	0.00	0	0.00
orth79	0.00	0.00	0.00	0	0.00
orth80	0.00	0.00	0.00	0	0.00
orth81	0.00	0.00	0.00	0	0.00
orth82	0.00	0.00	0.00	0	0.00
orth83	0.00	0.00	0.00	0	0.00
orth84	0.00	0.00	0.00	0	0.00
orth85	0.00	0.00	0.00	0	0.00
orth86	0.00	0.00	0.00	0	0.00
orth87	0.00	0.00	0.00	0	0.00
orth88	0.00	0.00	0.00	0	0.00
orth89	7.14	0.00	0.00	14	0.07
orth90	7.14	0.00	0.00	14	0.07
orth91	7.14	0.00	0.00	14	0.07
thom1	0.00	0.00	0.00	0	0.00
thom2	7.14	14.29	0.07	0	0.00
thom3	0.00	0.00	0.00	0	0.00
thom4	7.14	0.00	0.00	14	0.07
thom5	0.00	0.00	0.00	0	0.00
thom6	0.00	0.00	0.00	0	0.00
thom7	0.00	0.00	0.00	0	0.00
thom8	0.00	0.00	0.00	0	0.00
thom9	0.00	0.00	0.00	0	0.00
thom10	0.00	0.00	0.00	0	0.00
thom11	0.00	0.00	0.00	0	0.00
thom12	0.00	0.00	0.00	0	0.00
thom13	0.00	0.00	0.00	0	0.00
thom14	0.00	0.00	0.00	0	0.00
thom15	7.14	0.00	0.00	14	0.10
thom16	0.00	0.00	0.00	0	0.00
thom17	0.00	0.00	0.00	0	0.00
thom 19	0.00	0.00	0.00	0	0.00
thom20	7.14	14.29	0.07	0	0.00
thom21	0.00	0.00	0.00	0	0.00
thom22	0.00	0.00	0.00	0	0.00
thom23	0.00	0.00	0.00	0	0.00

Frequency % (all sites): Percentage of sites where the morphospecies was sampled
Frequency % (cons): Percentage of conservation sites where the morphospecies was sampled
Frequency % (lease): Percentage of lease sites where the morphospecies was sampled
Abundance where found: Sum of mean number (abundance) per site (5 samples per site), divided by the number of sites in which the morphospecies occurred (frequency).

Appendix 7: Veld Condition Scores for fenceline 1 using Ecological Index Method (BRG 11)

Species	Grazing value	Bench mark%	Bench score	L1 %	L1 score	C1 %	C1 score
Increaser 1a	7						
Alloteropsis semi-alata	7			1	7	5	35
Cymbopogon excavatus	7	1	7				
Cymbopogon plurinodis	7						
Digitaria tricholaenoides	7	1	7				
Eulalia villosa	7						
Schizachyrium sanguineum	7						
Setaria nigrirostris	7	2	14				
Trachypogon spicatus	7	2	14	7	49	3	21
Tristachya leucothrix	7	19	133	19	133	22	154
Subtotal		25	175	27	189	30	210
Decreaser	10						
Andropogon appendiculatus	10						
Andropogon schirensis	10						
Bracharia serrata	10	1	10				
Diheteropogon amplectens	10	2	20				
Melinis nerviglumis	10	1	10				
Monocymbium ceresiiforme	10			3	30		
Panicum ecklonii	2			1	2	9	18
Panicum natalense	2			1	2	7	14
Themeda triandra	10	48	480	30	300	2	20
Subtotal		52	520	35	334	18	52
Increaser II a	7						
Eragrostis capensis	7	2	14	1	7	4	28
Harpochloa falx	3			1	3	2	6
Heteropogon contortus	7	4	28	2	14	1	7
Subtotal		6	42	4	24	7	41
Increaser II b	4						
Digitaria monodactyla	4						
Eragrostis chloromelas	4						
Eragrostis curvula	4	1	4				
Eragrostis plana	4	1	4				
Eragrostis racemosa	4	2	8				
Hyparrhenia hirta	4	1	4				
Loudetia simplex	1			2	2	2	2
Setaria sphacelata var.*torta*	4	2	8				
Sporobolus africanus	4						
Sporobolus pyramidalis	4						
Sporobolus stapfianus	4						
Subtotal		7	28	2	2	2	2

Species	Grazing value	Bench mark%	Bench score	L1 %	L1 score	C1 %	C1 score
Increaser II c	1						
Aristida congesta subsp *congesta*	0					1	0
Aristida congesta subsp barbicollis	1						
Cynodon dactylon	1						
Microchloa caffra	1	1	1	1	1		
Paspalum scrobiculatum	1						
Setaria sphacelata	1						
Forbs	1	5	5	4	4	5	5
Sedges	1	2	2	7	7	1	1
Subtotal		8	8	12	12	6	6
Increaser III	1						
Diheteropogon filifolius	1	1	1	4	4	25	25
Elionurus muticus	1	1	1				
Rendlia altera	1			10	10	2	2
Subtotal		2	2	14	14	27	27
Additional species							
Digitaria flaccida	0			4	0		
Ischaemum franksiae	0					1	0
Koelaria capensis	0			2	0	9	0
				6	0	10	0
Total score			775	100	575	100	338
% of Benchmark					74.2		43.6

Abbrev: C= conservation land; L = lease land

Appendix 8: Veld Condition Scores for fenceline 2 using Ecological Index Method (BRG 11)

Species	Grazing value	Bench mark%	Bench score	L2 %	L2 score	C2 %	C2 score
Increaser 1a	7						
Alloteropsis semi-alata	7					1	
Cymbopogon excavatus	7	1	7				
Cymbopogon plurinodis	7						
Digitaria tricholaenoides	7	1	7	2	14	4	
Eulalia villosa	7					1	
Schizachyrium sanguineum	7						
Setaria nigrirostris	7	2	14	2	14		
Trachypogon spicatus	7	2	14			3	
Tristachya leucothrix	7	19	133	25	175	36	
Subtotal		25	175	29	203	45	315
Decreaser	10						
Andropogon appendiculatus	10						
Andropogon schirensis	10						
Bracharia serrata	10	1	10				
Diheteropogon amplectens	10	2	20	2	20	4	40
Melinis nerviglumis	10	1	10	4	40		
Monocymbium ceresiiforme	10						
Panicum ecklonii	2						
Panicum natalense	2					1	2
Themeda triandra	10	48	480	1	10	1	10
Subtotal		52	520	7	70		52
Increaser II a	7						
Eragrostis capensis	7	2	14	1	7	1	7
Harpochloa falx	3			2	6		
Heteropogon contortus	7	4	28				
Subtotal		6	42		13		7
Increaser II b	4						
Digitaria monodactyla	4						
Eragrostis chloromelas	4						
Eragrostis curvula	4	1	4				
Eragrostis plana	4	1	4				
Eragrostis racemosa	4	2	8	1	4	4	16
Hyparrhenia hirta	4	1	4	43	172		
Loudetia simplex	1					1	1
Setaria sphacelata var.torta	4	2	8				
Sporobolus africanus	4			1	4		
Sporobolus pyramidalis	4						
Sporobolus stapfianus	4						
Subtotal		7	28		180		17

Species	Grazing value	Bench mark%	Bench score	L2 %	L2 score	C2 %	C2 score
Increaser II c	1						
Aristida congesta subsp *congesta*	0					1	0
Aristida. congesta subsp *barbicollis*	1						
Cynodon dactylon	1						
Microchloa caffra	1	1	1				
Paspalum scrobiculatum	1						
Setaria sphacelata	1						
Forbs	1	5	5	14	14	23	23
Sedges	1	2	2	2	2	9	9
Subtotal		8	8		16		32
Increaser III	1						
Diheteropogon filifolius	1	1	1			5	5
Elionurus muticus	1	1	1				
Rendlia altera	1						
Subtotal		2	2				5
Additional species							
Digitaria flaccida	0						
Ischaemum franksiae	0						
Koelaria capensis	0			0		5	0
Total score			775		482		428
% of Benchmark					62.19		55.23

Abbrev: C= conservation land; L= lease land

Appendix 9: Veld Condition Scores for fenceline 3 using Ecological Index Method (BRG 11)

Species	Grazing value	Bench mark%	Bench score	L3 %	L3 score	C3 %	C3 score
Increaser 1a	7						
Alloteropsis semialata	7			3	21	1	7
Cymbopogon excavatus	7	1	7				
Cymbopogon plurinodis	7						
Digitaria tricholaenoides	7	1	7				
Eulalia villosa	7					1	7
Schizachyrium sanguineum	7						
Setaria nigrirostris	7	2	14				
Trachypogon spicatus	7	2	14	5	35	2	14
Tristachya leucothrix	7	19	133	21	147	7	49
Subtotal		25	175	29	203	11	77
Decreaser	10						
Andropogon appendiculatus	10						
Andropogon schirensis	10						
Bracharia serrata	10	1	10	2	20	2	20
Diheteropogon amplectens	10	2	20				
Melinis nerviglumis	10	1	10				
Monocymbium ceresiiforme	10			2	20	5	50
Panicum ecklonii	2						
Panicum natalense	2						
Themeda triandra	10	48	480	37	370	15	150
Subtotal		52	520	41	410	22	220
Increaser II a	7						
Eragrostis capensis	7	2	14	4	28	3	21
Harpochloa falx	3			6	18	2	6
Heteropogon contortus	7	4	28			5	35
Subtotal		6	42	10	46	10	62
Increaser II b	4						
Digitaria monodactyla	4						
Eragrostis chloromelas	4						
Eragrostis curvula	4	1	4	2	8	2	8
Eragrostis plana	4	1	4				
Eragrostis racemosa	4	2	8			6	24
Hyparrhenia hirta	4	1	4			1	4
Loudetia simplex	1						
Setaria sphacelata var.torta	4	2	8				
Sporobolus africanus	4						
Sporobolus pyramidalis	4						
Sporobolus stapfianus	4						
Subtotal		7	28	2	8	9	36

Species	Grazing value	Bench mark%	Bench score	L3 %	L3 score	C3 %	C3 score
Increaser II c	1						
Aristida congesta subsp *congesta*	0			2	0	20	0
Aristida. congesta subsp *barbicollis*	1						
Cynodon dactylon	1						
Microchloa caffra	1	1	1				
Paspalum scrobiculatum	1						
Setaria sphacelata	1						
Forbs	1	5	5	10	10	20	20
Sedges	1	2	2	2	2	6	6
Subtotal		8	8	14	12	46	26
Increaser III	1						
Diheteropogon filifolius	1	1	1				
Elionurus muticus	1	1	1				
Rendlia altera							
Subtotal		2	2				
Additional species							
Digitaria flaccida	0			3	0	1	0
Ischaemum franksiae	0			1	0		
Koelaria capensis	0					1	0
Subtotal				4	0	2	0
Total score			775	100	679	100	421
% of Benchmark					87.6		54.3

Abbrev: C= conservation land; L= lease land