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Assessment of the effects of fire and associated grazing on the recovery of *Merxmuellera drakensbergensis* in the Sani Pass region, Lesotho

Submitted in fulfillment of the academic requirements for the degree of Doctor of Philosophy

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Declaration

I declare that this thesis is my own, unaided work, except where otherwise acknowledged. It is being submitted for the degree of PhD to the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.



Signature

27 March 2014

Date

Abstract

The Lesotho highlands are unique in many ways yet are at risk from severe degradation, primarily as a result of grazing and burning. This study aimed to determine whether recent (last 10-15 years) burning and grazing practices in the Sani region of eastern Lesotho are sustainable for the long-term survival of tussock grasses in the alpine belt. It was hypothesized that the alpine tussock grasses of Lesotho are being burnt and grazed at a rate that does not enable them to adequately recover between burns. A GIS based burn frequency and extent map was produced and indicated many small patchy burns (<100m^2) were occurring within the Lesotho border, yet large scale burning (>10 000m^2) occasionally breached the escarpment edge. Selected sites were monitored to determine the rate of growth and recovery of *Merxmuellera drakensbergensis* tussocks for a three year period (January 2009- December 2011). Increases were observed in basal circumference, living basal area and standing biomass yet the average height decreased across all sites and was attributed to grazing. A burn frequency of 4 burns per decade was found to be the optimum for retaining the greatest biomass, while a lack of burning for more than 9 years was suggested to result in a decrease in biomass. The leaves observed at a third of the diameter required the least average time of 6.3 years to recover to pre-burn heights as they are sheltered from environmental impacts and grazing. The middle tillers generally required the most time to recover to pre-burn heights (7.1 years) as the greatest competition for resources may be found in the centers as well as high possibilities of fire damage. The current burning activities are considered sustainable under the current levels of grazing occurring. The current levels of grazing are not advisable, especially as livestock numbers are increasing and grazing of the Sani region is becoming continuous throughout the year. A shortening of the average height of individuals within *M. drakensbergensis* communities is found, while increases were observed in invasive species cover. This research will add to the knowledge of environmental managers and hopefully actions may then be taken to better conserve the Lesotho highlands for future generations.

Preface

The high altitude alpine regions of Lesotho are seen as being 'relatively intact,' while the lowlands are threatened and degraded (McGinley, 2007). Yet, increasing numbers of livestock and rapidly encroaching invasive karoid species are changing this. In 2009 the Drakensberg montane grasslands were described as being critically endangered (Gibson, 2009). Yet despite 29.3% of the species being endemic to this region (Pomela *et al.*, 2000), very little is protected. The deterioration in vegetation cover has been recognized as being the result of a lack of knowledge and information used in the formation of management policies among other causes (Motsamai, 1995). This research aims to generate knowledge to assist in conservation of this unique landscape.

Tussock grasses form a significant component of the Lesotho highlands and play a significant role in preventing erosion. They are able to resprout after defoliation allowing them to be generally long lived. Wetlands form a significant area of the Lesotho highlands and have their own unique vegetation. Thus a species needed to be found which would highlight the impact of over-burning and associated overgrazing in both grasslands and wetlands. *Merxmuellera drakensbergensis* (Scweick.) Conert is the ideal species for this study and is common to the study region. *Merxmuellera* species are only able to meet the nutritional needs of sheep and goats as well as being a relatively unpalatable species (Morris *et al.*, 1989). However, their palatability significantly increases after burning. The burning events cannot be described as 'prescribed' nor as wildfires, as they are not planned burns, but are usually ignited by herdsmen. In the past lightning was the most likely cause of fires as thunderstorms are frequent (Morris *et al.*, 1989). Today most fires are anthropogenic. Burning occurs for many reasons: green growth for grazing, livestock theft prevention, animosity and may be accidental (MDTP, 2007).

Lesotho fires tend to be patchy (generally less than 10 ha), with some areas being burnt more than once a year, whilst others have been excluded from fire for many years. Historical records of burning are minimal yet satellite imagery is assisting in changing this, at least in recent history. This project uses satellite imagery to determine burn frequencies over the last ten years and aims to determine the effect these fires have had on the tussocks. Despite recent studies on Lesotho vegetation, the effect of fire on high altitude (>2500m.a.s.l.) tussock grasses has not yet been examined. However, several studies examining high altitude tussocks in New Zealand (e.g.

Payton and Mark, 1979; Rose and Platt, 1990; Gitay *et al.*, 1992; Mark, 1994; Payton and Pearce, 2001) have found that such grasslands are destroyed by unsuitable burning regimes. It has been suggested that burning and associated grazing increase the mortality of tussock grasses (Morris *et al.*, 1989) leaving soil exposed to erosion and colonization by invasive species. Thus altering the landscape in a negative manner.

The threat of alpine grazing resources being lost highlights the need for an improved understanding of the vegetation-fire relationship in order to design management strategies that are able to help sustain biodiversity and ecosystem structure (Bond, 1997). The proposed study assesses the effects of burning (and grazing) on *M. drakensbergensis* tussock grasses in the Sani Pass region of Lesotho. This project determines what impacts ad hoc burning and associated grazing have on alpine tussock grasses in eastern Lesotho. The study primarily focuses on the condition of *M. drakensbergensis* tussocks, given that they form the basis of the plant community in the study region and are a vital resource for livestock production and associated human livelihoods.

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Thank you to my grandparents who are no longer with us.
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List of Abbreviations

ANOVA	-	Analysis of Variance statistical test
DAC	-	Drakensberg Alpine Center
DAR	-	Drakensberg Alpine Region
GA's	-	Grazing Associations
GIS	-	Geographical Information Systems
LBA	-	Living basal area
MDTP	-	Maloti-Drakensberg Transfrontier Project
NES	-	National Environmental Secretariat
RMAs	-	Range Management Areas
USAID	-	United States Agency for International Development

Site Abbreviations

A.P.	-	Apostles Plateau
H.E.	-	Hodgson's East
H.W.	-	Hodgson's West
K.P.	-	Kotisephola Pass
M.F.	-	Mshweshwe's Finger
S.E.	-	Sani East
S.S.	-	Sani South
S.Str.	-	Saddle Stream
V.F.	-	Valley Floor
V.S.E.	-	Valley Slope East
V.S.N.	-	Valley Slope North

Chapter 1:

Introduction



CHAPTER 1:

Introduction

1.1 Alpine Grasslands

One of the most endangered ecoregions of southern Africa is the grassland (Cowling *et al*., 1997), in which fire is a natural and necessary phenomenon, and thus also a valuable tool for management and conservation. Fire and its role in grasslands have been studied in African grasslands since the early twentieth century (De Ronde *et al*., 2004). The flora and its associated relationships with fauna and abiotic factors of South African grasslands is generally well known, yet knowledge of Lesotho vegetation and related abiotic and biotic processes have remained a relatively 'blank space' (Guillarmod, 1972; van Zinderen Bakker, 1981) until recently (e.g. Carbutt and Edwards, 2004; Carbutt and Edwards, 2006; Mucina *et al*., 2006; Boughton *et al*., 2013). Our lack of knowledge on the ecology of the montane regions of Lesotho has been highlighted in the Drakensberg/Maluti Catchment Conservation Program report (Quinlan, 1995a). However, the quest to understand more is being thwarted as the montane grasslands are loosing their cultural heritage, biodiversity and ecosystem processes, primarily as a result of landuse change and degradation (Uys and Lechmere-Oertel, 2006). The southern African montane grasslands form a significant bioregion, as well as a valuable grazing resource and water catchment; consequently any loss of these grasslands impacts on livelihoods.

Afro-alpine areas, such as those of Lesotho, represent isolated 'islands' or 'refuges' of geographical and ecological importance (Hedberg, 1970). Lesotho is a small mountainous kingdom, varying in altitude between 1500m and 3482m a.s.l. (Sene *et al.*,1998; Marake *et al.*, 1998). Among many factors which make this area unique from the areas surrounding it, are the stronger wind conditions, high humidity and intense insolation (van Zinderen Bakker, 1981). The harsh physical conditions of the alpine environment have yielded a high level of specialization amongst alpine vegetation (Morris *et al.*, 1989; Körner, 1999), and thus the grasslands and wetlands of Lesotho are somewhat unique (Marneweck and Grundling, 2000). This forms the basis for the widespread role this region plays in ecotourism, for both experts and tourists. Rangelands represent 80% of the natural vegetation in Lesotho (Uys and Lechmere-Oertel, 2006). Based on surviving vegetation and early travelers' accounts, it would seem that vegetation changes are a relatively recent phenomenon (i.e. last ca 150 years) (Marake *et al.*, 1998). Reasons given for such vegetation changes include landuse change, overgrazing and climate

change (Guillarmod, 1972; Morris *et al.*, 1989; Marake *et al.*, 1998; Molumeli *et al.*, 2010). Landscape degradation is extensive at lower altitudes and thus livestock are increasingly being forced further into the highlands for prolonged periods of time (Quinlan, 1995a), perhaps forcing herdsmen to use fire more frequently to stimulate new grass for grazing.

Studies on the relationships between fire and grasses tend to be both species and site specific; consequently only a small body of such literature is relevant to Lesotho and even less is relevant to the highland regions. Despite recent studies on the Lesotho vegetation, the effect of fire on high altitude (>2 500m.a.s.l.) tussock grasses has not yet been examined. However, several studies examining high altitude tussocks in New Zealand (e.g. Payton and Mark, 1979; Rose and Platt, 1990; Gitay *et al.*, 1992; Mark, 1994; Payton and Pearce, 2001) have found that such grasslands are being destroyed by unsuitable burning regimes. Yet, it has also been demonstrated that the exclusion of fires is proving to be as destructive to both individual grasses and the landscape as a whole (Gibson, 2009). Although grasses form a valuable grazing resource (O'Connor and Bredenkamp, 1997) and play a vital role in preventing erosion (Breebaart *et al.*, 2006), they are potentially being lost through uncontrolled *ad hoc* burning in eastern Lesotho, as was the case in New Zealand. The grasses of Lesotho are under heavy livestock and grazing pressure, leading many species towards extinction and changing the vegetative community (Guillarmod, 1972; Showers, 1989; Carbutt and Edwards, 2006). Even though much of eastern Lesotho is located within the Maloti-Drakensberg Transfrontier Conservation Park and also within the parks' Development Program, little is known on the potential fire-grazing impact on the alpine grasslands. Such information should be of value to the Lesotho government, environmental managers, conservationists and local livestock farmers, as it would offer better insight into sustainable burning practices. This in turn has the potential to prevent further destruction of the unique grasslands and wetlands of the Lesotho highlands.

The montane grasslands and wetlands of the Drakensberg have been listed as 'critically endangered' (Gibson, 2009). The environmental resources of Lesotho are essential to sustain rural communities, yet the impact from such communities is having a negative impact on these resources (NES, 2000). The inaccessibility of much of the highlands means that regular monitoring of the grasslands is not feasible. The threat of alpine grazing resources being lost highlights the need for an improved understanding of the vegetation-fire relationship in order to design management strategies that are able to help sustain biodiversity and ecosystem structure (Bond, 1997). Burning generally increases vegetative productivity, with grasses becoming more

palatable and accessible to livestock (Bond, 1997). Yet, desirable vegetation can be significantly reduced or even eliminated if an area is grazed too soon after burning (Brown, 2000). This PhD study aims to assess the effects of burning (and to a lesser extent grazing) on tussock grasses in the Sani Pass region of Lesotho. The term 'tussock' is most internationally recognized and used to describe densely tufted grasses on persistent old leaf bases with setaceous (bristly) or convolute (inrolled) leaves (Bews, 1918), and is thus used in this study. The term 'alpine' traditionally refers to the vegetated region above the treeline (Körner, 2004), however, in the case of the study region, no true treeline exists. Much debate has arisen as to the correct term to describe the unique flora of the Lesotho highlands, and can be referred to in Carbutt and Edwards (2001). As per recommendation by Carbutt and Edwards (2001, 2004), the term 'Drakensberg Alpine Center (DAC)' is used to distinguish this region from east African alpine regions. For ease of expression, the terms 'alpine' and 'DAC' are used in this thesis. The aim of this project is to determine what impacts ad hoc burning and associated grazing have on alpine tussock grasses in eastern Lesotho. The study primarily focuses on the condition of *Merxmuellera drakensbergensis* tussocks, given that they form the basis of the plant community in the study region and are an essential resource for livestock production and associated human livelihoods. It is in understanding the processes and characteristics of the recovery after disturbances of mountainous regions that we can begin to understand how to conserve and manage alpine ecosystems (Körner, 1999; Scherrer and Pickering, 2005).

1.2 Aims and Hypotheses

A study conducted by Schwabe (1995) in the Lesotho highlands demonstrated the deterioration of alpine wetlands through the use of aerial photography. Gullies were observed (pers. obs.), and consistent with the literature, found to be deeper and wider than in previous years. Similarly, marginal wetland vegetation was meager and in many places entire communities of *Merxmuellera drakensbergensis* had apparently vanished. Notably, no research has since been undertaken to verify this. Many research questions were formulated from these observations, yet the focus was narrowed down to defoliation as a result of burning and grazing, as these are thought to be the main contributors towards landscape degradation within this region. This study aimed to determine the spatial and temporal patterns of tussock burning over an 18 year period (1993-2011) in the Sani study region ($25km^2$) and assesses relative recovery rates after burns in various spatial contexts (aspect, altitude). The long-term conditions of *M. drakensbergensis* tussocks under various burning frequencies were determined for the last 18 years. This was aided by the construction of a burn frequency map using satellite images within a GIS. Ultimately, this

study determines whether recent (last 18 years) burning and grazing practices in the Sani region of eastern Lesotho are sustainable for the long-term survival of tussock grasses in the alpine belt. To assist this aim two hypotheses were investigated.

Hypotheses

Firstly, it was hypothesized that **the alpine tussock grasses of Lesotho are being burnt and grazed at a rate that does not enable them to adequately recover between burns**. Under the harsh alpine climatic conditions and the short growing season of alpine grasses, the herdsmen are required to turn unpalatable tussock grasses into valuable forage for growing livestock numbers and associated grazing requirements through burning, at a rate that is not sustainable over the long-term (Quinlan, 1995b).

Secondly, it was hypothesized that as a consequence of frequent burns, **biomass is being reduced over time and inter-tussock plant communities are being degraded through loss of protective cover**. If mortality increases, and sprouting and the reproduction of new plants is slowed or repressed (Mark, 1994), the ground cover is likely to decrease.

1.3 Thesis Structure

The introductory chapter is followed by Chapter 2 which discusses the relevant literature concerning the topics of burning, grazing and their relation to tussock grasses. This chapter also discusses the use of GIS (Geographical Information Systems) and satellite imagery with reference to burning and associated vegetation cover change. Chapter 3 presents the setting of the study within Lesotho, with particular reference to the climate, geology and vegetation of the study region. Chapter 4 outlines the climate and soil characteristics of the Sani region and their relationship to the tussock grasses. A study on the grazing patterns observed in the study region follows in Chapter 5. Chapter 6 involves the formation and analysis of the burn frequency maps and includes the selection of study sites. Chapters 7, 8 and 9 report the results of the studies regarding tussock shape, characteristics, relationships to burning history and monitoring of growth rates between burning events and after recent burning. The results are then linked together into a broad discussion so as to address the project aims and hypotheses (Chapter 10). Ultimately, it is hoped that the project will contribute in some way towards the development of better management decisions and plans when conserving this unique and valuable region for future generations through sustainable use and appropriate burning strategies.

Chapter 2:

Literature Review



CHAPTER 2:

Literature Review

2.1 Introduction

Management of grasslands and rangelands in terms of their biodiversity and functioning has for many years been based on the handling of disturbances such as grazing and burning (Uys and Lechmere-Oertel, 2006). Fire has an important role in determining vegetation structure and composition (Hudak *et al*., 2004). It is one of the few natural disturbances that alone, or in relation to another disturbance, can kill individual plants on a regular basis (Bond and Van Wilgen, 1996). The use of fire has been debated for many years. Pastoralists state that fire increases palatable grazing, allows for better access to forage and removes woody species from pasture lands (Payton and Pearce, 2001). Fire may also help maintain species richness (Cremer, 2004) as competitive exclusion may be prevented by the correct burning regime (Brown *et al*., 1991). Other effects of fire include an increase in the use of coarse grasses, increased forage, increased landscape diversity, increased decomposition, control of invasive species (Wright and Bailey, 1982) and control of diseases (Wright, 1974). Fire also plays a role in determining soil microbal populations and regulating carbon and nitrogen within an ecosystem (Harris *et al.*, 2008). However, many environmental advocates strongly believe fire decreases biodiversity, increases soil erosion and increases opportunities for invasive species (Payton and Pearce, 2001). As fuel for fires increases, plant species and community structures are altered when an area is not regularly burned, as observed in Kansas (Briggs and Knapp, 1995). In the past, much of the grassland research centered around dominant species or those with commercial value. However, the recognition of a need to protect biodiversity as well as ecosystem processes has shifted the research focus, as well as management practices, to those that enable the best conservation measures as well as grassland use (Uys and Lechmere-Oertel, 2006). This was assisted by increased knowledge in the ecological importance of fire and the vital role it plays in ecosystem processes, together with a subsequent radical change in fire stewardship programs (Krüger, 2006).

The literature on fire characteristics such as behaviour, temperature, intensity and chemistry will not be discussed in this literature review. This would increase the complexity and length of the literature review beyond the scope of this project. The nature of fire at each burn event is distinctive and varies with soil type, topography, vegetation type, moisture, fuel load and

weather conditions at the time of burning (Whelan, 1995). Thus, despite being relevant, the focus here is rather on the recovery of vegetation after burning and the burning regimes.

2.2 Southern African grassland studies and the effects of burning

The role played by fire in plant ecology was not recognized until after 1947 (Daubenmire, 1968), but since then scientific outputs have steadily increased (Table 2.1). Until 1971, the focus of fire research in relation to forage had focused on the season, frequency of burning and the shrub to grass ratio (Trollope *et al.*, 2002). However, there remains a general lack of knowledge on African grassland and savanna fire ecology (Trollope *et al.*, 2002). Our knowledge of African grasslands and their relationship to fire is mainly from studies undertaken in southern Africa (Trollop *et al*., 2002), although there have also been some studies from eastern Africa (e.g. Van de Vijver et al., 1999; Wesche, 2000). Consequently, much theory concerning burning of grassland fires originates from Australian based studies, such as that by Cremer (2004). However, many principles learnt from international studies apply globally and provide a valuable foundation for southern African studies.

Table 2.1: The number of publications which have examined burning and grasses in South Africa and Lesotho per decade.

	1930-1940	1940-1950	1950-1960	1960-1970	1970-1980	1980-1990	1990-2000	2000-2012
Number of Publications	2	0	1	2	6	18	12	47

Despite much of the literature being observational articles or reviews, many studies have taken place in southern Africa (Table 2.2). Considerable work has been undertaken on the relationship between fire and the fynbos of the Cape region, and the relationship between fire and savanna biomes (e.g. Trollop and Tainton, 1986; Bond, 1997; Scholes and Archer, 1997; Higgins *et al*., 2000; Hudak *et al*., 2004). In addition, the impact of fire on plant productivity has also been examined in semi-arid climates of southern Africa (e.g. Novellie, 1990; Snyman 2004a, 2004b; Hudak *et al*., 2004). Although many southern African studies have focused on selected species used as forage for grazing livestock (Uys *et al.*, 2000), little literature is available on such work from the Drakensberg and Lesotho highlands. However, Short *et al.* (2003) examined medium-term changes in grassland diversity as a response to grazing and fire in the southern Drakensberg foothills. Three groups of species were identified as being either 'increaser' or 'decreaser'

Table 2.2: Publications which have examined burning and grasses in South Africa and Lesotho

Lesotho and South African Research Studies	**Location**	**Feature Studied**
Staples (1930)	KwaZulu Natal grasslands	Effects of a summer burn
Cook (1939)	South African grasslands	Fire management
Mes (1958)	South African grasslands	Influence of burning on nutrient content
Edwards (1969)	Drakensberg mountains	Fire effects on vegetation
Nänni (1969)	Drakensberg mountains	Management
Scott (1970)	KwaZulu Natal grasslands	Eliminating burning
Scott (1971)	KwaZulu Natal grasslands	Veld burning
Herbst and Roberts (1974)	Lesotho	Ecological relationships
Mentis *et al.* (1974)	Drakensberg mountains	Effects of burning on vegetation
Le Roux and Morris (1977)	Pretoria	Burning, cutting and rainfall
Tainton *et al.* (1977)	KwaZulu Natal grasslands	Effects of mowing and burning
Dillon (1980)	KwaZulu Natal grasslands	Fire effects on vegetation
Scotcher and Clarke (1981)	Drakensberg mountains	Effects of burning on vegetation
Trollop (1982)	South African grasslands	Ecological effects of fire
Savage and Vermeulen (1983)	KwaZulu Natal grasslands	Spring burning modification of microclimate
Edwards (1984)	South African grasslands	Fire regimes
Everson and Tainton (1984)	Drakensberg mountains	Effects of burning regime
Martin (1984)	Lesotho	Range inventory
Trollop (1984)	South African grasslands	Regime and Effects
Everson *et al.* (1985a)	KwaZulu Natal grasslands	*Themeda triandra*and burning
Everson *et al.* (1985b)	KwaZulu Natal grasslands	Fire behavior
Trollope and Potgieter (1985)	Kruger National Park	Fire behaviour
Manry and Knight (1986)	South African grasslands	Relationship between ightening and burning
Trollope and Tainton (1986)	Eastern Cape	Effect of fire intensity
Everson *et al.* (1987)	KwaZulu Natal grasslands	Grassland regrowth after fire
Novellie (1987)	Bontebok National Park	Relationships between fire, grazing and cover
Everson *et al.* (1988a)	Drakensberg mountains	Effect of shading on tillering
Everson *et al.* (1988b)	Drakensberg mountains	Curing rates
Morris *et al.* (1989)	Lesotho	Fire, grazing and effects
Novellie (1990)	Mountain Zebra Nat. Park	Effect of burning on tussock grasses
Brown *et al.* (1991)	Cederberg mountains	Prescribed burning and conservation
Morris *et al.* (1992)	KwaZulu Natal grasslands	Management of tall grasslands
Baxter *et al.* (1994)	Drakensberg mountains	Fire and gemination of *Themeda triandra*
Quinlan (1995b)	Lesotho	Grassland degradation
Shea *et al.* (1996)	South Africa and Zambia	Fuel and flammability
Stocks *et al.* (1996)	Kruger National Park	Fuel and fire behaviour
Scholes and Archer (1997)	South African grasslands	Tree-grass interaction
De Ronde (1999)	South African grasslands	The 1998 fires
Hudak (1999)	South African grasslands	Human-fire interactions
Jansen *et al.* (1999)	Mpumalanga	Burning and bird conservation

Table 2.2 cont.: Publications which have examined burning and grasses in South Africa and Lesotho

Lesotho and South African Research Studies	**Location**	**Feature Studied**
Zacharias and Danckwerts (1999)	South African grasslands	Management of humid grasslands
Cwielong *et al.* (2000)	Lesotho, Madagascar	Fire and culture
Higgins *et al.* (2000)	Southern African grasslands	Fire, rsprouting; tree-grass relationship
Titshall *et al.* (2000)	KwaZulu Natal grasslands	Effects of fire and grazing exclusion
Bijker *et al.* (2001)	Drakensberg mountains	Burning effect on vegetation and soil
Brockett *et al.* (2001)	Pilanesberg and Kruger N.P.	Regime and grass biomass
Morris (2002)	KwaZulu Natal grasslands	Burning and grazing by cattle
Morris and Tainton (2002)	KwaZulu Natal grasslands	Effect of fire, grazing and rest on diversity
Trollop and Trollop (2002)	Southern African grasslands	Fire behavior
Verboom *et al.* (2002)	Paardenberg, Western Cape	Post-fire flowering of *Ehrharta capensis*
Bond *et al.* (2003)	South African grasslands	Fire, climate and ecosystem type
Letsela *et al.* (2003)	Lesotho	Plant uses
Parr and Chown (2003)	Southern African grasslands	Critique of burning research
Short *et al.* (2003)	Drakensberg mountains	Fire and species composition
Snyman (2003)	Bloemfontein	Fire and plant survival
Van Langevelde *et al*. (2003)	Southern African grasslands	Modeling fire and grazing effects
Archibald and Bond (2004)	Hluhluwe iMfolozi Park	Grazer resposes to burning
Carbutt and Edwards (2004)	Drakensberg mountains	Flora of the Drakensberg
Fynn *et al.* (2004)	KwaZulu Natal grasslands	Long-term burning and mowing effects
Hudak *et al.* (2004)	South African grasslands	Trends in fire patters and landuse
Mdee (2004)	Lesotho	Resource management
O'Connor *et al.* (2004)	KwaZulu Natal grasslands	Effect of burning fire breaks
Snyman (2004a)	Zastron, South Africa	Productivity responses after burning
Snyman (2004b)	Bloemfontein	Productivity responses after burning
Uys *et al.* (2004)	KwaZulu Natal grasslands	Fire and species diversity
Archibald *et al.* (2005)	Hluhluwe iMfolozi Park	Fire-grazer interactions
Fynn *et al.* (2005)	KwaZulu Natal grasslands	Long-term burning and mowing effects
Kepe (2005)	Pondoland, South Africa	Burning by local people
Snyman (2005a)	Bloemfontein	Effect of fire on productivity
Snyman (2005b)	Bloemfontein	Influence of fire on grass roots
Carbutt and Edwards (2006)	Drakensberg mountains	Endemism of angiosperms
Ferwerda *et al.* (2006)	Kruger National Park	Fire as a driver of foliage quality
Govender *et al.* (2006)	Kruger National Park	Factors influencing fire intensity
Kraaij and Ward (2006)	Northern Cape	Tree-grass interaction
Krüger (2006)	South African grasslands	Fire management
Snyman (2006)	Bloemfontein	Effect of fire on defoliation
Uys *et al.* (2006)	Drakensberg mountains	Effect of burning on invertebrates
Manson *et al.* (2007)	Drakensberg mountains	Effects of burning on soil and systems
Archibald (2008)	Hluhluwe iMfolozi Park	Development of grazing lawns
Oluwole *et al.* (2008)	Alice, Eastern Cape	Effects of different burning frequencies
Mbatha and Ward (2010)	Barkly West, Northern Cape	Nutritional quality of grass
Novellie and Kraaij (2010)	Bontebok National Park	Monitoring indicator of grazing/burning

Table 2.2 cont.: Publications which have examined burning and grasses in South Africa and Lesotho

Lesotho and South African Research Studies	Location	Feature Studied
O'Connor *et al*. (2010)	KwaZulu Natal grasslands	Grazing management and conservation
Zimmerman *et al.* (2010)	Etosha National Park	Grass mortality
Mulqueeny *et al.* (2011)	Mkuzi Game Reserve	Determinants of variation in burnt area
O'Connor *et al.* (2011)	KwaZulu Natal grasslands	Grazing/burning management
Trollop (2011)	Southern and Eastern Africa	Perspectives on rangeland management
Breman *et al.* (2012)	northern South Africa	Historical fire regimes
Ghebrehiwot *et al.* (2012)	KwaZulu Natal grasslands	Fire and gemination of *Themeda triandra*
Knapp *et al.* (2012)	Kruger National Park	Grazing, burning and productivity

species, and a general trend towards more typical highland sourveld was observed. A host of other studies dealing with the effects of fire on vegetation have been completed in the grasslands and mountainous regions of KwaZulu-Natal (e.g. Bews, 1918; Everson *et al.*, 1985a; Everson *et al.*, 1990; Bowie and Frank, 2001; Manson *et al*., 2007). Although not at a high altitude, and possibly not directly relevant to Lesotho, these studies are still valuable as they provide useful insight into the nature of tussock grasses as well as an insight into southern African burning policies and recommendations. There is almost no literature concerning burning in Lesotho, and is only briefly discussed in reviews or in assessments of grassland conditions. The most relevant study undertaken within the study region is that by Morris *et al.* (1989), who examined burning and grazing regimes and associated conservation requirements. Morris *et al.* (1989) found that the area was overstocked and that burning was thus unnecessary, and concluded that given the current degradation, immediate conservation efforts were required.

2.3 Burning regimes in tussock grasslands

The natural fire regime has been influenced over time through anthropogenic causes (Whelan, 1995), often resulting in a substantial change in the frequencies of fire. Fire has been used for hundreds of years by indigenous people, and together with lightening, are the most common causes of grassland fire (Manry and Knight, 1986; Everson, 1999; Krüger, 2006). Both of these causes result in patchy fires and are seldom documented in historical scripts. This leads to difficulty in determining past burning regimes of an area. A burning regime is described as being the frequency, intensity, and type of fire, which ultimately determines the biological impacts of a fire (Whelan, 1995; Bond, 1997). Additionally, the vegetation type may also influence the

burning regime through differences in flammability (Leonard *et al*., 2010). Research into the different effects of varied fire regimes has been undertaken since at least the 1920s (Bond, 1997), and findings indicate that grass composition alters under different fire frequencies (Bond, 1997). Thus, it is not only important to know the natural fire regime, but also whether it is beneficial within the current setting, which has been altered by anthropogenic forces (Krüger, 2006).

Burn events occurring outside the natural fire season are a concern, as it is believed that this could lead to the extinction of some local plant species (Brown *et al*., 1991). The observed decrease in species richness and grassland heterogeneity after a fire was observed in the North American prairies, support this (Veen *et al*., 2008). Within the natural fire season, different fire frequencies have varied impacts on different plant species (for example, fire eliminates more sensitive species and favours recovery through sprouting of more resilient species) (Cremer, 2004). The interval-dependent hypothesis argues that the time between burning events is essential to establish population and community stability (Bond and Van Wilgen, 1996). The hypothesis is frequently depicted as burning to control the invasion of other species into grasslands, or burning to prevent unplanned destructive fires (Bond, 1997). Thus, variable regimes may positively influence plant diversity (Hudak *et al*., 2004), with frequent fires and extended unburnt periods resulting in a decrease in plant diversity and an optimum found in moderate burning at site specific intervals (Morgan,1999).

Frequent burning degrades the physical and chemical characteristics of soil over time, as observed in Australia (Cremer, 2004), which should cause concern for the future of soils and vegetation in areas subject to frequent burning. This is supported by Díaz-Delgado *et al.* (2003), who believe that high burning frequency may be linked to erosion and surface vegetation loss. Frequent burning also retains organic matter with a lower nitrogen content than those burnt less often. As observed in the Drakensberg, this results in a higher C:N ratio, suggesting nitrogen is a limiting factor for growth at frequently burnt sites (Manson *et al*., 2007). However, Hudak *et al.* (2004) argue that an increased frequency of burning promotes landscape diversity, as observed in Madikwe Game Reserve. Supporting this, frequent burning increased forb diversity through rapid colonization of inter-tussock spaces on the Flooding Pampa, Buenos Aires (Laterra *et al*., 2003). Yet, this also applies to increased colonization by invasive species (Masocha *et al*., 2011). Ultimately, frequent burning under low stored carbohydrate levels increases the negative impacts of burning (Miller, 2000; Zimmermann *et al*., 2010). Aerial cover is the percentage of soil covered or overshadowed by plants in an area (Breebaart *et al*., 2006). Tussock aerial cover is

known to decrease with longer intervals between burning in some regions, such as in the Mlambonja Wilderness area, near Cathedral Peak (Uys *et al.*, 2004). This is due to the inability of tussock tillers of some species to tolerate shading from dead plant matter (Everson *et al.*, 1988a; O'Conner and Bredenkamp, 1997; Morris and Tainton, 2002; Zimmermann *et al*., 2010). Similarly, many plants rely on fire for germination, and thus when fire is excluded, species diversity may decrease (Trabaud, 1987). Grassland fires assist the expansion and maintenance of grasslands through preventing the spread of woody species (Vogl, 1974; Bijker *et al*., 2001; Bond *et al*., 2003). Without burning, the woody species are able to expand their range into grasslands and thus grasslands progress into shrublands. The findings of studies (e.g. Everson *et al.*, 1988a; O'Conner and Bredenkamp, 1997) imply that tussock grasses need to be burnt regularly to provide maximum grazing for livestock. However, a balance must be achieved between providing maximum grazing and maximum conservation. Similarly, a balance needs to be achieved concerning safety and natural fire frequencies. In the recent past, burning has been designed to mimic natural regimes as much as possible. Three burning alternatives designed to mimic the natural burning regime were proposed for the Kruger National Park: patch mosaic burning, burning based on vegetation type and lighting dependant (Bond and Archibald, 2004).

2.4 Seasonality of burning

The season in which burning takes place is often dictated by the climate, which controls the occurrence of natural ignition sources such as lightening (Manry and Knight, 1986; Whelan, 1995). Additionally, the climate also regulates fuel moisture and hence, flammability (Bond and Keeley, 2005). Fuel moisture influences the spread of fires, with drier fuels generally producing more wide spread burns (Brockett *et al*., 2005; Mulqueeny *et al*., 2011). Fuel moisture may have a greater impact on the grasses being burnt than the frequency of the burning events, while influencing the intensity of the burn (Govender *et al*., 2006; Knapp *et al*., 2009; Mulqueeny *et al*., 2011). Seasonality of a burn event can have an impact on the vegetation, and the most appropriate season in which to burn has been debated for many years. It not only affects the productivity of the following growing season, but also influences the species composition of an area (De Ronde *et al.*, 2004; Knapp *et al*., 2009). Yet, burning during the growing season can cause severe damage to grass cover and populations, as shown through *Themeda triandra* populations in KwaZulu-Natal (Everson *et al.*, 1985a). Many farmers burn just before the onset of spring to provide early forage for livestock, as new growth at this time of the year has a high protein and mineral content (Hardy, 1999). Burning in early spring essentially extends the growing season (Vogl, 1974). However, this growth uses reserves saved for spring growth. If a

frost event or defoliation occurs, the grasses may not have the reserves stored for regrowth (Hardy, 1999; Knapp *et al*., 2009). Spring burning of alpine grasslands in New Zealand does not significantly decrease total diversity, nor the diversity of indigenous vegetation (Espie and Barratt, 2006). The same results were observed in a more recent study in New Zealand and attributed to the increased moisture of the soil in spring burns, compared to winter burns (Payton and Pearce, 2009). In an earlier New Zealand study, Rowley (1969) demonstrated that summer burning produces fewer tillers than spring burning and summer burning had a higher mortality than spring burning. Similarly, burning in summer may be less productive than in winter, which may be noted up to two growing seasons later, due to slower regrowth following the summer burn, as observed in Texas, U.S.A. (Ansley *et al.*, 2008). Summer burning may also alter the species composition (Scotcher and Clarke, 1981; Limb *et al*., 2011). Trabaud (1987) adds that fire damage to plant organs is often greatest during summer, and explains why repeated summer burning resulted in increased bare ground in Texas, U.S.A. (Ansley *et al*., 2010). Summer burning may thus also promote woody vegetation cover (Boughton *et al.*, 2013). Despite these findings, summer burning was observed to increase grass composition in Florida, U.S.A., suggesting the season most beneficial for grass cover is region dependent (Boughton *et al*., 2013). Burning of new growth can permanently damage new tillers, hence only 8% of *Themeda triandra* individuals survived a summer burn in the montane grasslands of KwaZulu-Natal reported by Everson *et al*. (1985a). This may be due to a lack of auxillary tillers being produced after burning (see Rowley, 1969). Similarly, in Natal highland sourveld, burning after the start of the growing season resulted in increased tiller mortality (Everson and Everson, 1987).Thus, the closer the burn is to the onset of the growing season, the more rapid the new growth rate is, but burning after the start of the growing season will be detrimental (Everson and Everson, 1987).

It has been demonstrated in a grassland of Oklahoma, U.S.A., that autumn burning may increase soil erosion for a longer period of time than spring burns (Vermeire *et al*., 2005). However, an autumn or late summer burn is the least harmful towards tussock grasses, as observed in southern Idaho (Wright and Klemmedson, 1965). A late summer or autumn burn may be useful to farmers after a dry summer where little forage has been available, yet this uses resources permitting essential services to continue throughout the winter months (Hardy, 1999; Bruhjell and Tegart, 2001). Defoliation during winter dormancy has little effect on the tussocks (Everson, 1999; Hurt, 1999; Trlica, 1999; Morris and Tainton, 2002; Ansley *et al*., 2010; Trollope, 2011). However, defoliation just after the onset of dormancy can have a negative effect as crowns are exposed to harsh winter conditions (Schmitz, 1984). Changes in structure can be harmful if they

occur within the growing season. The structure of tussock grasses allows the crown, dead leaf matter and tillers to insulate the central regions (5^{o}-8^{o}C difference) against cold winter conditions (Nagy and Grabhorr, 2009). Burning in winter can thus leave the crowns exposed to harsh winter conditions. Despite this, mid-winter burning may allow adequate recovery to prevent erosion during spring and summer storms, as observed in the Underberg district, South Africa (O'Conner *et al.*, 2004). Winter burns, such as those observed in the DAC, are also likely to stimulate a greater number of new plants than biannual spring burns (Everson *et al.*, 1985a). Winter burning may increase grass cover as observed in Florida (Boughton *et al*., 2013). More specifically, late winter or spring burning may favour C_4 grasses or tall grasses and not C_3 grasses or shrubs, and this has potential implications for grassland management (Fynn *et al.*, 2005; Ansley *et al*., 2010).

Burning should occur when the shortest recovery time is possible, usually in early spring, just after the first spring rains have occurred but should take place before dormancy ceases (Hardy, 1999; Hurt, 1999). Thus, burning should not be permitted from October to the end of the growing season within the DAC (Everson and Everson, 1987; Everson *et al.*, 2004) and the optimum time to burn is late August to September within southern African grasslands (Everson *et al.*, 2004). In Lesotho, the optimum time to burn is October, as growth would have begun and snow is less likely (Morris *et al*., 1989). However, this is impractical when burning occurs in the form of fire breaks as runaway fires mostly occur during the dry winter months. However, different species respond differently to burning in different seasons (De Ronde *et al.*, 2004), thus the issue may rather have to do with which species is being studied, rather than which season is preferable for burning. The time of burning should be based rather on the physical characteristics of the plants at the time of burning, and the location of the start of the growing season for that specific plant to prevent erosion (Everson *et al.*, 2004). Additionally, the purpose of burning (e.g. for bush encroachment, grazing forage, fuel removal) may dictate the season in which burning occurs to provide the desired outcome (Trollope, 2011). Burning within the historical burning season may also result in the greatest species richness and recovery (Knapp *et al*., 2009).

2.5 How often should grasses be burnt?

An important consideration concerning the correct management of fire in an area must include understanding of floral evolution (Brown *et al*., 1991). It has been suggested that if we knew more about plant processes, relationships and recovery abilities, management would be more able to devise regimes most suited to preserving the maximum diversity of an area (Cremer, 2004). Disturbance is a necessary part of grassland ecosystems; lack of disturbance limits

biomass, increases plant stresses through shading, decreased soil nutrients and increases competition stress (Smith, 1985). Exclusion of fire from an area results in the area burning at significantly higher temperatures due to accumulation of fuels than those burned regularly (Mistry, 1998). Higher fire temperatures have a greater impact in altering the physical and chemical characteristics of the soil, and increase plant and micro-organism mortality. This is largely due to the build-up of litter, yet litter plays a crucial role in shielding the soil against extreme temperatures (Wahren *et al.*, 2001a). Morgan (1999) adds that the difference between annual fires and those burned less frequently is the duration of surface temperatures above 100 ^{o}C (< 1min in annually burnt grasslands to >2-3min for 4-7 year intervals). However, burning alone can alter the physical and biological characteristics of the soil. Total Carbon concentrations were 14% less in a burnt site than in an unburnt site in Otago, New Zealand (Ross *et al.*, 1997). Loss of nutrients may occur through released gases and to a lesser extent through windblown ash, as observed in New Zealand (O'Connor *et al.*, 1997). Similarly, microbial biomass as well as microbial Nitrogen and Phosphorus were lower in a burnt site than an unburnt site on the slopes of Mt Benger, New Zealand (Ross *et al.*, 1997). Thus, the frequency of burning depends on fire dependent and interval dependant factors (Bond and van Wilgen, 1996; De Ronde *et al.*, 2004). Dependent factors occur at the time of burning and alter the intensity and duration of the fire through climatic and vegetation conditions at the time of burning. Interval dependent factors are the influence of the conditions between burns. Major short term changes in vegetation structure and composition are the outcome of burning tall-grasslands, as observed near Mt Benger, New Zealand (Yeates and Lee, 1997). Similarly, frequent burning may alter the species diversity through rapid inter-tussock space colonization by other species, as observed in the pampa grasslands, near Buenos Aires (Laterra *et al*., 2003).

According to De Ronde (1999), most grasslands, such as those in the Drakensburg, can be burned after one dry season. However, at least two years and at most eight years between burning events is suggested by Cremer (2004) for Australian grasslands. Yet, if the Drakensberg grasslands are left unburnt for more than three years, fuel builds up to levels which promote fires of high intensity and long duration (De Ronde, 1999). Hence, in order to maintain vegetation cover and diversity in the tall grasslands of KwaZulu-Natal, a burn frequency of 2 to 3 years is recommended (Morris and Tainton, 2002). Supporting this, three years was required for soils and vegetation to recover in the Eastern Cape (Oluwole *et al*., 2008). Similarly, a burn frequency of 1-2 years is suggested to maintain diversity in areas with a history of frequent burning in south-eastern Australia (Morgan, 1999). This is supported by a study that recorded a larger

compositional change with infrequent burning (every 6 years) than annual burning in the southern Drakensberg foothills (Short *et al*., 2003). Yet, other studies have indicated tussock grasses may become less resilient to the effects of burning at this frequency (e.g. Prober *et al.*, 2007). However, due to the slow growth rate of alpine plants, 2½ years is less recovery time than required for the conservation of alpine soil (Wahren *et al*., 2001a), and may only be adequate for 25% recovery as observed in New Zealand (Yeates and Lee, 1997). Despite this, Australian alpine grasslands require 3-5 years to complete regeneration and restore diversity (Williams *et al.*, 2006). A study of *Themeda triandra* and the post-burn age also indicated that tussocks should be burned at most every five years to preserve its productivity and existence, and if left for longer periods, the aerial organs of the tussocks 'collapse,' covering the ground in a thick layer of plant litter (as seen in a temperate Australian grassland after 11 years without fire and in South African grasslands after seven years) (Morgan and Lunt, 1999; Zimmermann *et al*., 2010). This is supported by observations of grasses dying due to self-shading, in African grasslands (Trollope, 2011). Yet, the return to pre-burn conditions was estimated to take 20-25 years in grasslands near Otago, New Zealand (Gitay *et al.*, 1992). The optimum burning regime for the Drakensberg foothills was confidently suggested to be every two years according to Everson *et al.* (1985b) and Everson (1999). Despite such studies, it is not yet clear what the optimum burn intervals should be at higher altitudes where plant growing seasons are shorter.

2.6 Tussock characteristics

2.6.1 Structure

The most important attributes of grasses are that they prevent erosion and provide food for grazers (Acocks, 1971). Tussock grasses are defined as "grasses producing tillers, but not stolons or rhizomes, and having an erect and clumped growth form" (Allen *et al*., 2011). Grass plants are formed by collections of units called tillers, each with its own leaves and roots, sharing carbohydrates and nutrients (Trlica, 1999; Wolfson and Tainton, 1999; Oliva *et al*., 2005; Derner *et al*., 2012). The inner leaf (leaf 8) of a tiller is the youngest while the outer leaf (leaf 1) is generally the oldest (Figure 2.1). Their size is dependant on the birth and death rates of the individual tillers (Oliva *et al.*, 2005). Tussock grasses or tussocks are grasses whose structure forms dense bunches with 'a tight mass of buds at 1-3cm below the surface' often holding soil between the stems (the crown), a formation that is likely to protect them from death by fire (Cremer, 2004, p17) and freezing (Monteiro *et al*., 2011). The crown contains meristematic buds which form the sources of roots and new tillers. Essentially the crown is vital for regrowth of the

tussock after defoliation, as well as for maintaining its size and productivity (Hannaway *et al.*, 2000). The inner core may develop a resistant dead section as the tussock ages (Figure 2.2). The inner core or basal tuft and their protective leaf sheaths act to shield the tiller initials or growth buds (Bond and Van Wilgen, 1996; Zedler, 2007). Thus, the compactness of the crown ensures the persistence of tussock grasses (Brzosko, 2001).

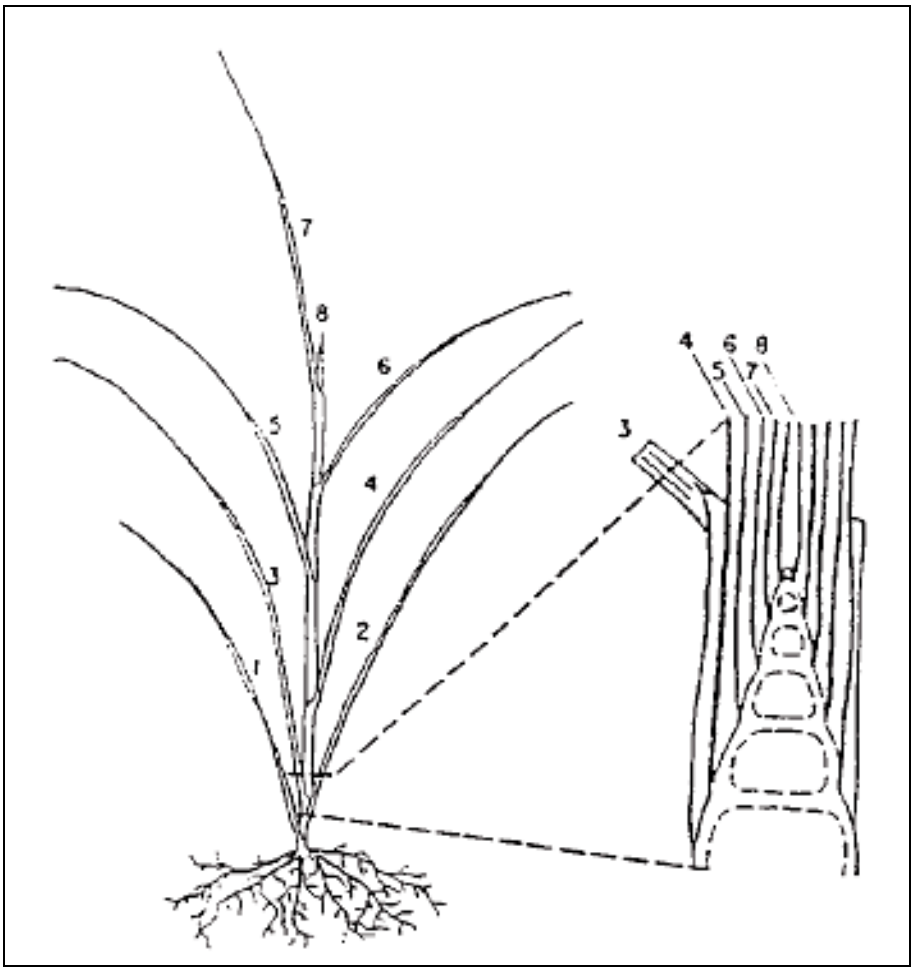


Figure 2.1: Diagram showing the composition of a grass tussock base (after Trlica, 1999).

2.6.2 Structural progression with age

As tussock grasses age they expand radially and die back in the center, a process observed and attributed to different causes by many (e.g. Danin and Orshan, 1995; Lewis *et al*., 2001; Feldman *et al*., 2007). Differences in tussock structure were first noticed by Cole and Wilkins (1958), who distinguished three different forms: normal, hollow center and clump edge (Figure 2.3). Hollows are often called "Monk's tonsure-like gaps" (e.g. Lewis *et al*., 2001) or even rings (e.g. Wikberg and Mucina, 2003). Grass ring or hollow formations have mostly been observed in ecosystems lacking in resources, especially water (Sheffer *et al.*, 2007). Consequently these formations have

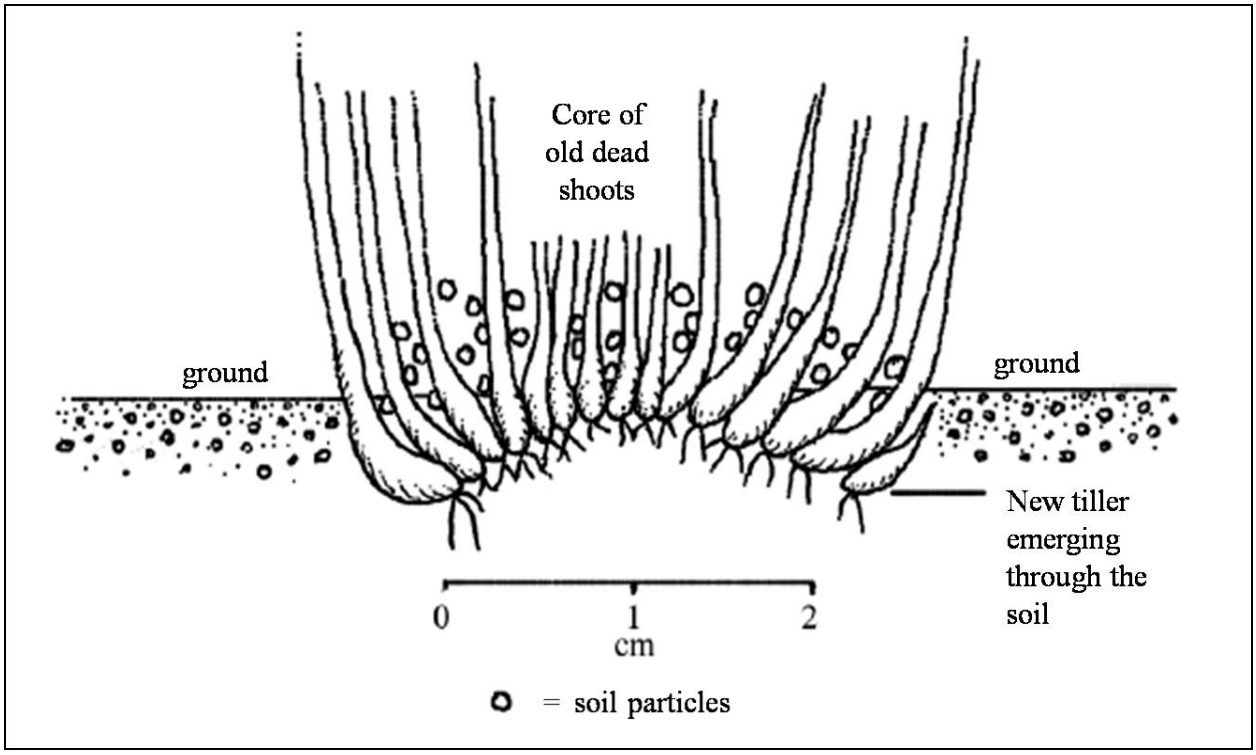


Figure 2.2: Diagram showing the composition of a grass tussock base (after Cremer, 2004).

mostly been observed and studied in species producing ramets, such as *Carex humilis* (Wikberg and Mucina, 2002; Wikberg and Svensson, 2003) and *Poa bulbosa* (Sheffer *et al.*, 2003). It was commonly believed the hollow center and clump edge depicted deterioration, with the normal form representing the initial structure. Later, studies on many different species confirmed the presence of age stages and associated forms. Tussock grasses are able to grow and change shape due to the emergence and death of tillers (Gibson, 2009). Eight stages were identified by Gatsuk *et al*. (1980): seed, seedling, juvenile, immature, virginile, reproductive (young, mature and old), subsenile and senile. A series of qualitative features characterizes each age stage such as nutrition, leaf form, presence of shoots and seeds, type and pattern of growth, living and dead structural balance and fully grown structures (Gatsuk *et al*., 1980) (Figure 2.4).

The tussock begins to take on its tussock-like structure at the virginile stage. Here an empty space appears around the tussock due to competition, shading and in some cases the phytotoxic nature of the plant (Gatsuk *et al*., 1980). During the young reproductive stage, the typically round tussock produces its first inflorescences and has no dead areas. The mature reproductive stage produces the highest number of inflorescences. It is also at this stage that necrosis begins, usually in the center (Gatsuk *et al*., 1980). A reduction in inflorescences and increase to almost half of the center of dead material characterizes the old reproductive stage.

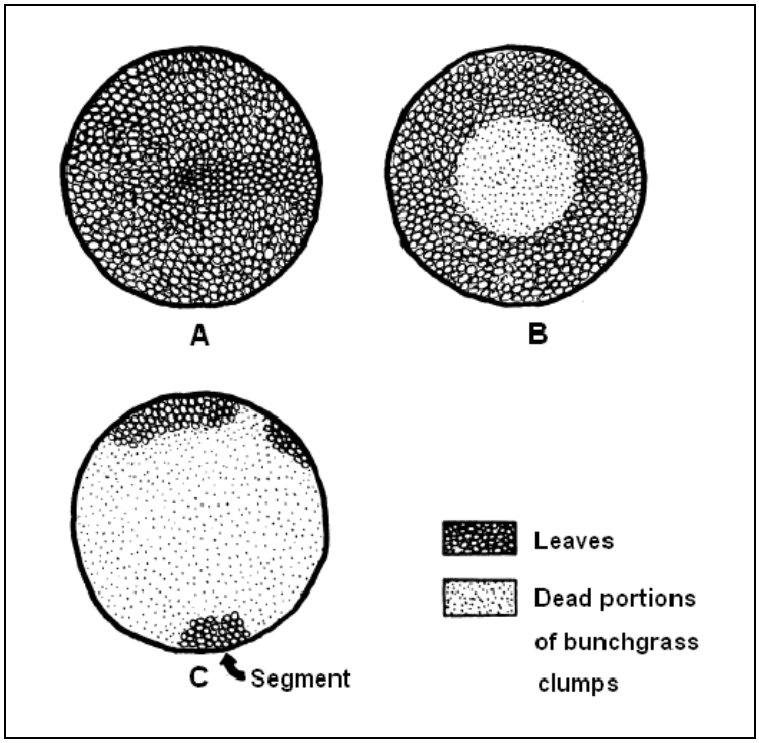


Figure 2.3: Diagram showing tussock form classes; A: Normal, B: Hollow center, C: Clump edge (after Cole and Wilkins, 1958).

Invasion of neighbouring plants begins as the rate of new shoots reduces (Gatsuk *et al*., 1980). The subsenile stage is characterized by a further increase in the dead area and invasion by neighbouring plants. Lastly, the senile stage sees the fragmentation of the original tussock (Gatsuk *et al*., 1980). The smaller clumps of a degraded tussock individual have the potential to develop into new individuals, thus propagating vegetatively (Tsuyuzaki and Tsujii, 1992; Danin and Orshan, 1995; Lewis *et al*., 2001; Feldman *et al.*, 2007).

Rose and Platt (1990) identify four structural age states: seedlings, juvenile tussocks, mature tussocks and senescent tussocks. This was based on the percentage dead crown, basal diameter, height and number of tillers. Sanchez and Puigdefabregas (1994) added to Gatsuk *et al*.'s (1980) theory by using it as a computational model which showed good correspondence with *in situ* tussock, and found that at a high plant growth rate the tussock expands outwards as a means to escape suffocation from dead matter in the center. This finding was supported by Wan and Sosebee (2000). Through a series of experiments, tiller emergence (tillering) increased significantly in crowns after litter had been removed. Defoliation in the form of clipping (to a height of 15cm) decreases tiller birth when leaf litter remained within the crown (Wan and Sosebee, 2000). A low growth rate causes the tussock to collapse as the leaves were not able to

escape through the dead leaf litter (Sanchez and Puigdefabregas, 1994). At an intermediate level, the tussocks divide into segments after a shorter period of time than under vigorous growth. Despite this, the conclusion reached by Wan and Sosebee (2000) was that tiller emergence patterns were not solely responsible, but that resource depletion was also a factor which should not be overlooked. However, even with these theories, it is not certain whether the change in structure is due to environmental factors or merely the natural growth pattern of tussock grasses.

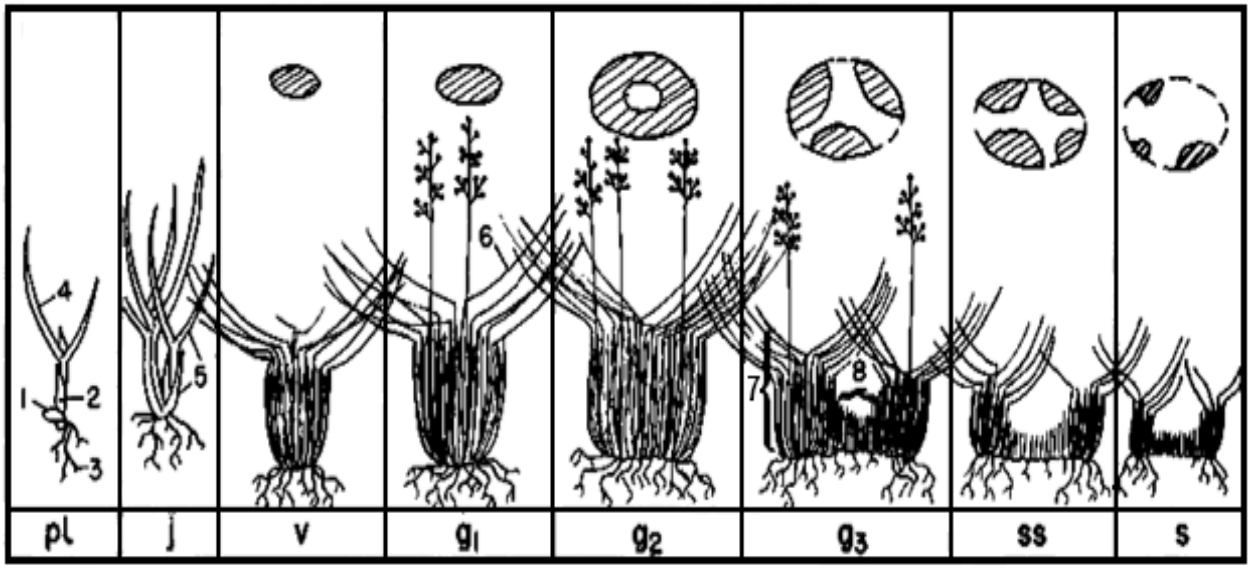


Figure 2.4: Diagrammatic representation of *Deschampsia cespitosa* Beauv.

pl = seedling, j = juvenile, v = virginile, g1 = young reproductive, g2 = mature reproductive, g3 = old reproductive, ss = subsenile, s = senile, 1 = caryopsis, 2 = coleoptile, 3 = adventitious roots, 4 = assimilating leaves, 5 = rosette shoot, 6 = reproductive semi-rosette shoot, 7 = living parts of a tussock, 8 = the dead parts of a tussock (After Gatsuk *et al*., 1980).

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Research has shown these forms generally occurred in either resource limited or semi-arid environments (Ravi *et al*., 2008) such as those found in Lesotho. Interactions between biotic and abiotic factors are believed to be the cause of alterations in growth form, and changes in structure impact abiotic features and processes (e.g. soil moisture, nutrients and erosion) (Ravi *et al*., 2008). Individuals grown from seed are believed to show an early death of the core, resulting in the hollow form (Lewis *et al*., 2001). The death of central tissues may also be the result of limitations to central meristematic tissues which limit the formation of new tillers (Derner *et al*., 2012). Research in the grasslands of the Sevilleta National Wildlife Refuge, Mexico, has identified that sediment deposition and a consequential decrease in growth in the center of clumps was responsible for the change in form (Ravi *et al*., 2008). Defoliation removes protective vegetation cover, exposing sediment to movement by aeolian and hydrological

processes, thus enabling sediment to become trapped in tussocks as they sift runoff and wind (Ravi *et al*., 2008). This finding supports the earlier model used to show tussock shape development with time, growth and sediment in the Filabres Range, Spain (Sanchez and Puigdefabregas, 1994). Thus, it is likely that overgrazing and burning assist in this process and may accelerate it. Overgrazing has previously been believed to accelerate this deterioration towards the death of the tussock (Cole and Wilkins, 1958). Regrowth after burning tends to be more erect, reducing the plant cover between tussocks (Daubenmire, 1968). The result would be an increase in erosion of inter-tussock spaces, which in turn would negatively influence the outer edges of the tussocks. However, total plant density may be increased as tussock grasses fragment under grazing pressure, as observed in Texas (Butler and Briske, 1988). In the Drakensberg, burning increased secondary tiller numbers with annual spring burning producing more tillers than biannual burning (Everson *et al.*, 1985b). Tillers emerging from the outer edges of a tussock were taller than those emerging from the center, despite similar grazing intensities, and would further increase the rate of structural change, as observed near El Reno, Oklahoma (Volesky, 1994). However, others believe that defoliation merely allows the gap to be seen when long leaves covering the gap are removed (Lewis *et al*., 2001; Feldman *et al*., 2007).

The local environment can influence the shape of tussock grasses, as was found for *Festuca varia* (Haenke.) in the northwest Caucasus, Russia (Pokarzhevskaya, 1998). Round unordered tussocks were located on gentle slopes and oblong shaped tussockswere observed in rows on steeper slopes. This finding reinforced those of an earlier study on the effects of slope on tussock shape in the Filabres Range, Spain (Sanchez and Puigdefabregas, 1994). This study suggested that the mound of eroded sediment which builds up against individual tussocks on the upslope side prevents upslope growth, thus resulting in an oblong or kidney shape.

2.6.3 Lifespan

Fragmentation of tussock grasses with age makes it difficult to determine the lifespan of an individual tussock, yet estimates can provide a relative indication of the time it takes a new individual to grow, reproduce and fragment. The lifespan of the study species is unknown, but other studies may provide some insight. Tussock grasses may live for 20 years or more as observed in New Zealand (Moore, 1955), and while the maximum age is unknown, *Eriophorum vaginatum* found in Alaska is known to live for between 120 and 190 years (Mark *et al.*, 1985). *Carex stricta* was observed to have a lifespan of approximately 50 years, in Wisconsin, U.S.A.

(Lawrence and Zedler, 2013). The long lifespan was supported by slow decomposition rates of leaf tissue found on the tussock and in inter-tussock spaces (Lawrence and Zedler, 2013). *Deschampsia cespitosa*, which grows in wet alpine grassland meadows of Europe, Asia, Africa, Australia and North and South America, has a lifespan of between 25 to 60 years (Gatsuk *et al*., 1980). The reproductive phase for this species lasts for between 15 and 30 years, while the post-reproductive phase lasted for between 15 and 25 years (Gatsuk *et al*., 1980). Thus, age may be an indicator of shape due to a natural growth structural degradation. Research has been undertaken in the restored Blacklands, Texas, to determine whether time played a role in the number of hollow forms of the tussock *Schizachyrium scoparium* (Derner *et al*., 2004). The results indicated that basal area grew with time by 80% and 163% over a period of 17 and 23 years respectively since restoration. Similarly, the number of hollow crowns increased within the same older age sites (Derner *et al*., 2004). At the same time, tiller density had decreased with age. The long life span of tussock grasses, through long term observation, was also highlighted as being over 23 years for their studied species. Thus, size may be an indicator of age, and age may be a good predictor of survival ability, as demonstrated in Kansas, U.S.A. (Laurenroth and Adler, 2008) and in Oklahoma, U.S.A. (Limb *et al*., 2011).

2.6.4 Thermal properties

Due to their shape, tussocks frequently have their own form of temperature control. The centers of tussock grasses have their own microclimate, which, in one study, averaged 5.5^{o}C higher than the surrounding air and soil temperatures within an alpine environment, Venezuela (Hedberg and Hedberg, 1979). This was confirmed more recently in the alpine areas of Bolivia, where the temperature inside the tussock was recorded to be consistently warmer than the minimum air temperature (Monteiro *et al.*, 2011). This will act to insulate the central tissues and new shoots from potential damage by frost (Hedberg and Hedberg, 1979). Similarly, the temperatures within the central regions were consistently cooler than the maximum air temperature in western Bolivia (Monteiro *et al.*, 2011), protecting the tussock from heat damage and reducing water loss. In alpine or tundra regions, the thermal properties of tussock grasses as a result of their shape, may assist with the longer lifespan than that of the surrounding vegetation. Soils within tussock grasses in the Alaskan tundra have temperatures 6 to 8^{o}C warmer than surrounding soils (Chapin III *et al.*, 1979). The soils reached these temperatures earlier as they became snow free sooner than the inter-tussock spaces. This would allow tussock grasses a longer growing season than plants within the inter-tussock spaces. Similarly, other results of this study suggest that the root structure and growth form allow for maximum absorption and increased nutrient cycling

(Chapin III *et al.*, 1979). Yet, another strategy has also been voiced concerning the thermal properties and ability to tolerate freezing alpine temperatures. Tolerance to freezing temperatures through supercooling capacity and low leaf injury temperatures are also considered important, and together with structural adaptations, have ensured the survival of grasses in the Venezuelan Andes (Márques *et al.*, 2006). Together, the structure and chemical adaptations account for the comparatively high productivity and success of tussock species within cold climates.

2.7 Tussock growth rates

Very little is known about the growth rate of alpine tussock grasses, and no previous mention has been made about the growth rates for *Merxmuellera drakensbergensis*. Growth rates are likely to be different for different species, topographies, climates, aspects and altitudes. Previously, growth has been measured through differences in height (e.g. Cole and Wilkins, 1958; Lee *et al.*, 1993), leaf elongation (e.g. Cole and Wilkins, 1958; Everson and Everson, 1987; Busso and Richards, 1992; Volesky, 1994; Snyman, 2004) and basal diameter or circumference (e.g. Morgan and Lunt, 1999; Bond and Van Wilgen, 1996). Growth rate can also be measured in terms of tiller birth and death rates. Birth rates may be seasonal while death rates are continuous (Bullock *et al*., 1994). An imbalance in birth and death rates leads to growth of a tussock or a decrease in its density. The tissues responsible for regrowth and general growth are the intercalary meristems (base of existing leaves), primordial meristems and through axillary buds (Olson and Richards, 1988). Growth is generally most rapid from the intercalary meristematic tissues and slowest from the axillary buds (Olson and Richards, 1988). Expansion of cells in the intercalary meristem is responsible for the majority of growth, especially the elongation of leaf blades (Trlica, 1999; Hannaway *et al*., 2000). Elongation of leaves positively influences photosynthesis, providing the means for further elongation to take place. However, if the tip of the leaf blade is removed, it will not regrow but growth will continue (Hannaway *et al.*, 2000).

Growth is influenced by species type, such that two species burnt at the same time may have different growth rates (Defossé and Robberecht, 1996). Additionally, different growth rates may occur for two similar species as seedlings, but their difference decreases with age in a greenhouse setting (Eissenstat and Caldwell, 1987). A difference in growth rate between two species is often due to varying competitive abilities (Eissenstat and Caldwell, 1987; Defossé and Robberecht, 1996). Growth may also be influenced by soil richness and climate, with drought years slowing growth (Guàrdia *et al*., 2000). Yet, even under unfavourable conditions, such as those experienced during a drought, minimal growth continues in order to maintain the tussock

grasses. The return to more favourable conditions for growth, such as snow melts and associated soil temperature increases, may result in significant elongation of leaves as observed near Logan, Utah (Eissenstat and Caldwell, 1987). Ambient temperature is able to influence the metabolic processes occurring within a plant, thus influencing the growth rate. The lower the temperature, the less able plants are to double their cells as a function of growth (Körner, 2006). Thus, grasses found in alpine environments have adapted in order to be capable of maintaining their physiological processes at low temperatures, and extend the growing season (Billings, 1974). For example, three arctic graminoid species had a growth rate of between 1 and 2mm per day at 0^{o}C (Billings, 1974). However, a limit to growth processes seems to occur between 0^{o}C and 2^{o}C in all plant types, with growth only really becoming significant at around 6^{o}C (Körner, 2006). Hence, temperature may either accelerate or decelerate growth daily. Increases in growth rate were observed to occur from dawn to midday, accelerate to early evening and declining until sunrise when temperatures decrease below a threshold level in northern Utah (Busso and Richards, 1992). Temperature seems more important in controlling growth during the night than during the day. Yet, it has also been suggested that light availability is the most important factor for tussock growth (Busso and Richards, 1992). Additionally, growth is likely to be influenced by altitude, and will be discussed further in section 2.10 of this chapter.

Grazing may impact the growth rates of tussock grasses in several ways, depending on the grazing intensity and frequency. Growth rates in the form of tiller elongation are generally higher on grazed tussock grasses than ungrazed populations in the Tointic Valley, Utah (Olson and Richards, 1988). However, growth rates and auxillary tiller production were reduced when grazing takes place after internode elongation (Olson and Richards, 1988). Thus, grazing may have a positive effect on growth rates, depending on when tussock grasses are grazed. Defoliation has also been observed to stimulate the production of ancillary tillers (Mark, 1965; Olson and Richards, 1988). A study in Otago, New Zealand, found that the number of tillers decreases after burning but increases to 21% more tillers 30 months after burning (Yeates and Lee, 1997). Dead organic matter from the tussock grasses builds up within and around the crown leading to shading of the new tillers, and if this is not removed, growth may be severely slowed (Daubenmire, 1968; Caldwell *et al*., 1983; Hardy, 1999; Zimmermann *et al*., 2010). Rowley (1969) measured a growth rate of 34.1cm per leaf in untreated plots, 41.3cm in grazed plots (single defoliation event) and 56.2cm in burnt plots (single burn event), with the same time since defoliation for both burning and grazing in the Rock and Pillar Range, Otago. However, growth was counteracted by an increase in die-back from the tips (Mark, 1965; Rowley, 1969).

2.8 Characteristics permitting the survival of tussock grasses through burning

The severity and intensity of a fire, plant characteristics at the time of burning, and the ability of the plant to resist injury and recover, determine the response of a plant to burning (Miller, 2000). The recovery is determined by the extent of injury sustained by plant organs from fire behaviour, heat duration, extent of soil heating and the pattern of fuel consumed (Miller, 2000). The intensity of the fire and the phenologic characteristics of the plants determine the extent of damage sustained by the plants (Daubenmire, 1968; Díaz-Delgado *et al.*, 2003). Topography, aspect and slope also have an influence on both the recovery and damage to plant organs, as observed using remote sensing in Barcelona province, Spain (Díaz-Delgado *et al.*, 2003). Some of these characteristics determine the adaptations of plants for the survival and resistance to burning. Essentially, survival depends on the endurance of meristamatic tissues in the form of buds and the adaptations which permit their survival (Trabaud, 1987). Due to differences in characteristics of individual species, the persistence of different species faced with burning varies with species and individual size (Bond and Van Wilgen, 1996).

Survival can be divided into survival of the direct effects of fire and survival of post-fire conditions (Whelan, 1995). The direct effects mainly concern the destruction of aerial plant organs while the post-fire conditions include nutrient loss through runoff and alteration or loss of surface characteristics (Díaz-Delgado *et al.*, 2003). The direct effects of fire depend on many factors such as moisture, fuel load, type of fire (e.g. back burn or head fire) and weather conditions at the time of the burn. Back fires tend to burn slower as they move against the direction of the wind, yet damage to plants may be greater than in head fires which move fast over the landscape, however, in terms of die-back the difference may be small as observed in Bloemfontein (Snyman, 2006). Death of plant tissues is the result of high temperatures reached by the fire, duration of high temperatures and moisture content, and may be direct (protein denaturation) or indirect (metabolic changes) (Whelan, 1995). The death of an entire plant is dependent on the extent of injuries to its organs and the type of tissues harmed (Whelan, 1995). Possible causes of injury are indicated in Figure 2.5. Plant mortality is most often the result of damage to several organs and tissues (Miller, 2000). Resistance to fire is determined by the number of growth points or buds protected from damage by fire (Bond, 1997) and the location of the meristematic tissue (Peláez *et al*., 1997; Zedler, 2007). Meristematic tissue is required for the growth of new tissues (Whelan, 1995) and is located differently within various species, thus

accounting for variations in growth responses (Bond and Van Wilgen, 1996). Grass species are generally considered adapted to fire as their growth points and associated meristematic tissues are located in protective crowns or at varying depths below the soil surface where soil warming is reduced (Sarmiento, 1992; Whelan, 1995; Miller, 2000; Bruhjell and Tegart, 2001; Cremer, 2004; Zedler, 2007; Gittins *et al.*, 2011). The dormancy of grasses during dry, cold winters in the Drakensberg region can be termed an adaptation to fire as growth points are more resistant to fire, and less susceptible to increased soil temperatures (Hungerford *et al*., 1991).

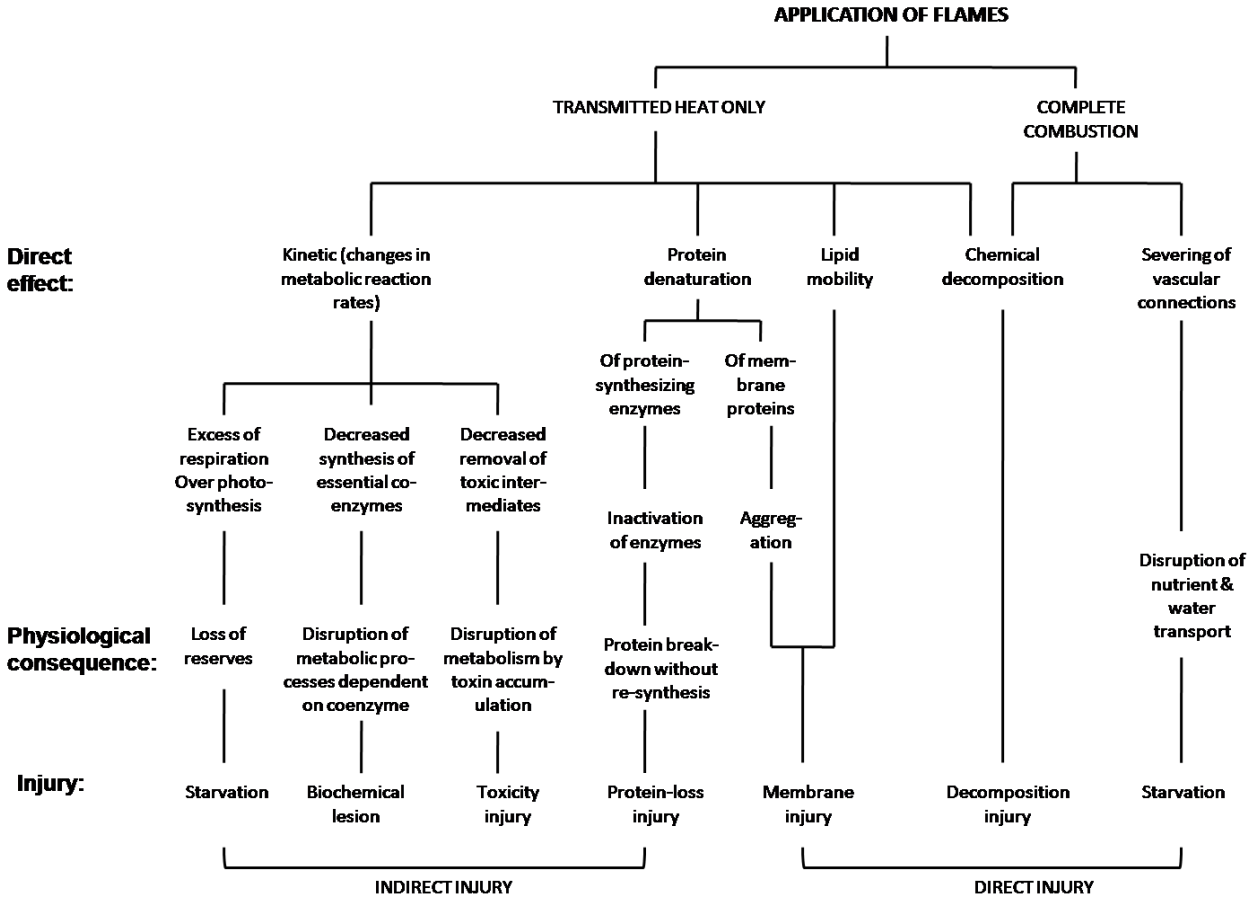


Figure 2.5: The range of possible means of indirect and direct damage to plants (after Whelan, 1995).

Tussock or bunchgrasses are grasses whose structure forms dense bunches with ‘a tight mass of buds at 1-3cm below the surface’ often holding soil between the stems, a formation that protects them from death by fire (Cremer, 2004, p17). The trapped soil acts to reduce the temperature of the crown due to its poor conductive and insulating properties (Trabaud, 1987), thus starving the fire of oxygen and so reduces the effects of fire intensity (Trollope *et al.*, 2002). The inner core is also protected by the density and moisture content of the crown (Traubaud, 1987), making ignition difficult even under dry crown conditions (Paysen *et al*., 2000). This explains why only the lower 20-30mm of *Temeda triandra* tillers were observed to remain after burning in the

Derrimut Grassland Reserve, Australia (Lunt, 1990). The inner core may develop a resistant dead section as the tussock ages (Figures 2.3 and 2.4), and their protective leaf sheaths act to shield the tiller initials or growth buds (Bond and Van Wilgen, 1996). However, central tissues are known to be killed first (Wright, 1971; Peláez *et al*., 1997), which is attributed to heat being able to dissipate at the edges while greater transmission of heat to the centre from accumulation of dead plant matter occurs. The impact of burning is determined by the extent to which crowns are damaged and by the survival of growth tissues (Bond, 1997).

Different tussock species vary in their vulnerability to damage by fire (Paysen *et al.*, 2000). Among grass species, larger tufted species such as *Themeda triandra* and *Cymbopogon plurinodis* lose the greatest amount of biomass to burning, as observed in South African rangelands (Snyman, 2004a). Species with accumulated leaf tissue within the core tend to burn for longer at a slower rate (Paysen *et al.*, 2000), resulting in a slower recovery (Bruhjell and Tegart, 2001). Similarly, species and individuals with large tuft diameters burn slower than those with small diameters, resulting in increased mortality in tufts of larger diameter and finer leaves (Bruhjell and Tegart, 2001). Yet, large tussock grasses have been observed to sustain less damage than smaller tussock grasses when fuel levels are high, in southeastern Louisiana, U.S.A. (Gagnon *et al*., 2012). Survival favours plant tissues which burn rapidly as this exposes the vital organs to high temperatures for shorter periods, resulting in less damage (Gagnon *et al.*, 2010). The high surface area to volume ratio of tussock grasses and low moisture content associated with frost and dry winters make these plants excellent fuel for fires (Bond, 1997). Consequently, a characteristic of plants indicating their susceptibility to fire damage is the ability of individuals to retain dead matter (Bond and Van Wilgen, 1996). If the dead grass matter is not removed, it can limit growth of new shoots, as observed in KwaZulu-Natal grasslands (Hardy, 1999), and affect the growth of individuals and the community, as well as retard many ecosystem processes through associated effects on microbes (McNaughton, 1991). The need for defoliation by fire is supported by findings that fire is necessary to maintain a tussock community in the grasslands of the U.S.A. (Paysen *et al.*, 2000). Conversely, sprouting may decrease as increasing amounts of dead grass matter are removed, with shoots only appearing on outer edges of the crown when most litter is removed (Miller, 2000).

Additional factors of tussocks characterizing their ability to survive fire are: low moisture content, organ arrangement, surface area to volume ratio and fine-leaved structure (Bond and Van Wilgen, 1996). The heating of the soil surface may also result in significant damage to

crown tissues, and this is dependent on the duration of burning and the soil to fuel distance (Gagnon *et al.*, 2010). The more fuel, the more damage may be experienced by tussocks and hence, the greater the mortality (Gagnon *et al*., 2012). Fire burns quickly through coarse-stemmed tussock grasses as fuel is not concentrated around the crown (Wright, 1971). Similarly, fine-stemmed grasses in dense clumps burn slower with an increased likelihood of the crown igniting (Miller, 2000) as passing flames ignite the outer dead litter and move inwards (Wright, 1974). Once ignited, dense clumps can burn for 2 or 3 hours and reach maximum temperatures of up to 538 ^{o}C as observed for squirreltail and needle-and-thread tussocks near Boise, Idaho (Wright, 1971; Wright and Bailey, 1982). This occurs when the clump centre is dry, generally killing many or all growth buds (Miller, 2000). Thus, tussocks with a large accumulation of dead plant material burn for longer and at higher temperatures (Wright and Bailey, 1982), as has also been observed by Peláez *et al.* (2009) in central Argentina. These adaptations not only enable survival during fire, but are associated with survival under a range of different stresses and disturbances to which the plant may be exposed (Trabaud, 1987).

2.9 Recovery after fire

Fire severity was found to have little impact on Australian alpine vegetation composition and diversity due to the strong ability of the vegetation to reseed and resprout (Camac *et al*., 2012). The resilient nature of tussock structure to fire often means that much of the early recovery of a landscape is a result of the rapid regrowth of tussocks, as in North Slope, Alaska (Jones *et al.*, 2009). Plant response to fire has been widely classified into those which sprout new growth and non-sprouters (Whelan, 1995; Bond and Van Wilgen, 1996; Vesk and Westoby, 2004). Sprouting allows species to persist for long periods of time under a wide range of disturbances and is especially true where seedling establishment occurs under harsh conditions (Vesk and Westoby, 2004), such as in Lesotho, and is also the most observable reaction to fire by most species (Wahren *et al.*, 2001a; O'Dea and Guertin, 2003). Resprouting after fire has been studied in detail in relation to woody vegetation, yet little is known about tussock responses to fire, especially those located at high altitudes. In grasses, resprouting is often a function of elongation of protected meristematic tissues at the base of the leaf blade (Zedler, 2007). Sprouting ability is size dependent with the weakest ability found in smaller grasses, so the ability of new clumps to resprout and survive is reduced as the frequency of burns increases (Bond and Van Wilgen, 1996). The size dependence correlates to root system size and stored resources (Fynn *et al*., 2005; Monteiro *et al*., 2011), and their ability to compete for below ground resources (Gittins *et al*., 2011). Many burning events are compared through intensity and vegetation damage rather

than resprouting ability (Vesk and Westoby, 2004). The location of growth buds (Bruhjell and Tegart, 2001), species type, age, intensity, season and regime of burning all influence the ability of plants to re-sprout (Cremer, 2004), while fire severity determines how many growth points survive to resprout (Bruhjell and Tegart, 2001). High intensity fires destroy stems and force regrowth to originate from basal buds within the crown (Vesk and Westoby, 2004). Regrowth is enabled by basal meristems and their ability to draw on carbohydrate reserves found within the crowns (Mucina *et al.*, 2006), leaving carbohydrate stores low for the year following burning (Payton and Mark, 1979).

Burning may result in declines in productivity and be harmful to some species (Vogl, 1974, Trollope, 1999). This is often due to soil nitrogen content and availability, which to a large extent (and in addition to light and water) determine grassland productivity (Gibson, 2009). Nitrogen may be lost through burning resulting in lower concentrations in frequently burnt soils (Gibson, 2009). Observed increased productivity following a fire is partially explained through increased nutrient availability, removal of dead plant litter, increased soil temperature, increased microbial activity, the removal of competitive vegetation and in some cases an early start to the growing season (Whelan, 1995; Limb *et al*., 2011). Resprouting has been described as "almost instantaneous" by O'Dea and Guertin (2003). Plants re-growing from rhizomes or sprouting from the roots may be larger after burning, as observed in dry forests of Australia (Cremer, 2004). The roots play an important role in re-growth as they provide the nutrients needed to re-grow (Monteiro *et al.*, 2011). Increased nutrient content of the soil subsequent to burning is well documented in relevant literature (Bond and Van Wilgen, 1996; Morgan, 1999; Mutanga and Skidmore, 2004), and is a result of rapid release of nutrients, distribution of nutrients among less biomass, rejuvenation of vegetation and changes to the leaf stem ratio (Mutanga and Skidmore, 2004). Increased nutrient content may also be explained by an increase in root activity resulting in an increase in nutrient uptake (Daubenmire, 1968). However, Van de Vijver *et al.* (1999) argue that the cause is due to an increased leaf to stem ratio with the same nutrient content spread across less above-ground plant cover, as they observed in northern Tanzania. Increases in potassium availability could increase the tolerance of new seedlings and shoots to cold temperatures and frost (Blank *et al.*, 2007). The reduced production by the crown following burning is believed to be the result of soil water availability, rather than damage to meristematic tissues, as observed near Moscow, Idaho (Defossé and Robberecht, 1996). Additionally, a lack of available water may reduce the ability of grasses to regain lost tissues (Knapp *et al*., 2012). Nevertheless, an increase in tissue moisture has been widely observed on burnt areas as opposed

to that in unburnt areas (Daubenmire, 1968). The increased water content of the tissues decreases the resistance of the grasses to frost (as water swells as it freezes resulting in tissue damage) (Daubenmire, 1968) and thus reduces the ability of grasses to survive in areas prone to frequent frosts, such as is the case in the Sani study region.

A plants' life cycle determines its response to burning; such as for example resprouting species need to complete a juvenile stage post-burning before obtaining maturity again (Trabaud, 1987). A short secondary juvenile period of less than 11 months is common to most grass species and enables a complete phenological cycle to occur under annual burning (Morgan, 1999). In terms of both above and below ground biomass, litter and rain-use efficiency, South African grasslands require a minimum of two years to recover (Snyman, 2004a). Consequently, a single burning event can have a significant impact on the vegetation during the following growing season, depending on post-fire management (Snyman, 2004a, 2004b, 2005b). Snyman (2004a) found that after two years, three South African grass species (*C. plurinodis*, *T. triandra* and *Harpochloa falx*) did not recover to an equal biomass as opposed to those which were left unburnt. However, rejuvenation of smaller species was greater as the loss of biomass from taller surrounding species resulted in increased light availability. The capacity of two north American tussock species (*Festuca idahoensis* and *Agropyron spicatum*) to survive fire is directly related to the characteristics of the post-burn micro-climate and their ability to compete for vital resources (Defossé and Robberecht, 1996). This possibly explains why 14 years were required for *Chionochloa rigida*, a mountainous New Zealand tussock grass, to recover its biomass to a level equivalent to unburnt individuals (Payton and Mark, 1979). Thus, the recovery of a community from fire involves survival by adult plants, dispersal of seeds, grazing, competition, invasion of other species and the post-fire conditions (Whelan, 1995).

2.10 Effect of altitude on vegetation recovery

For centuries, the slowing effect of altitude on growth rates has been documented (Bews, 1918). Little available literature exists on the effect of altitude on the recovery of tussock grasses after burning. However, it is known that increasing altitude affects climatic variables such as orographic influences on rainfall, decreases temperature, increases frost events and influences insolation (Mucina *et al.*, 2006), thus impacting on the growth rates of vegetation and post-fire conditions. Yet, soil pH and soluble salt content are not observed to be influenced by altitude in the Lesotho highlands (Herbst and Roberts, 1974). The effects of environmental change resulting from altitudinal changes can be divided into two categories: factors linked to height above sea

level (e.g. temperature and atmospheric pressure) and factors not necessarily altitude specific (e.g. wind, pedology, geology, landuse and moisture availability) (Körner, 2007). These factors all impact the growth pattern and phenotype of the vegetation growing within the alpine and afro-alpine zones.

Increasing altitude is marked by a decrease in atmospheric pressure, which corresponds to a decrease in atmospheric temperature (Körner, 2007; Nagy and Grabherr, 2009). The lower temperatures experienced with increasing elevation correspond on a broad scale to increasing vegetation stress (Smith, 1985). Plant temperatures vary with the surrounding air temperatures (i.e. they are poikilothermic) and are unable to move away from unsuitable temperatures except through reproduction, thus have to adapt or perish in unsuitable temperatures (Körner, 2006). Temperature is able to influence the metabolic processes occurring within a plant, thus influencing a plants growth rate. The lower the temperature, the less able plants become to double their cells as a function of growth (Körner, 2006). Lower atmospheric pressure has implications for humidity and associated rainfall, as a lower temperature reduces the capacity of air to hold moisture. The erosive power associated with summer thunderstorms decreases with increasing altitude along the eastern Lesotho escarpment, and is more connected to droplet size and fall impact than storm duration (Nel and Sumner, 2007). Frost damage to plants increases with altitude, as do the number of frost risk days (Smith, 1985). Under clear skies, increasing radiation as a result of reduced atmospheric turbidity is observed. Similarly, increases in UV-B radiation are observed at increasing altitudes at any given total solar radiation, despite being negated or reversed by fog and frequent cloud cover observed at high elevations, and may pose a threat of heat damage to plants despite cooler atmospheric temperatures (Körner, 1999; 2007). A shortening of growing seasons with altitude can be observed where temperature is the driver of seasons, rather than moisture (Keller and Körner, 2003; Körner, 2007). Additionally, increasing snow weight with increasing altitude and its associated seasonality may also influence growth (Keller and Körner, 2003). Essentially, the recovery of grasses at high altitudes, such as observed in the Lesotho highlands, is slower than low altitudes (Morris *et al.*, 1989). Productivity declines with increasing altitude resulting in decreased fuel loads (Morris *et al.*, 1989).

Land available for biota decreases with increasing altitude, leading to ecological ‘islands’ of high altitude communities; this narrows biodiversity with increasing altitude at a mean loss of 40 angiosperm species per 100m (Körner, 2007; McGinley, 2007). Thus, both the distribution and the growth within alpine grasslands are strongly influenced by altitude and, to a lesser extent,

aspect and gradient (Morris *et al*., 1989). Altitude is able to influence vegetation in many ways (e.g. influencing the ambient temperature, number of frost days, snow cover and length of the growing season). This results in a catena of differing vegetation types with altitude. The higher the altitude the more adapted the species. Higher altitudes are generally characterized by shorter vegetation, which is adapted to extreme temperatures and arid conditions. C_3 grasses and sedges increase in number with increasing altitude and similarly, C_4 grasses are able to tolerate higher altitudes than C_4 sedges (Sieben *et al*., 2009).

2.11 Influence of aspect on plant growth

The length of the growing season is influenced by the solar angle and so the equator experiences growth all year round and the poles almost no growing season (Nagy and Grabhorr, 2009). The amount of solar radiation received by an area depends on the aspect, slope angle, albedo, angle of the sun, altitude and vegetative cover (Nagy and Grabhorr, 2009). Thus, aspect exerts a direct influence on plant growth through differences in temperature generated by the amount of solar radiation received (Morris *et al.*, 1993). Temperature in turn affects the movement of water, and both together are able to influence plant growth, species composition and structure (Holland and Steyn, 1975). Thus, in the southern-hemisphere, north-facing slopes are more prone to drought and as a result the vegetation found on these slopes tends to be more resilient to dry conditions (Bennie *et al*., 2006). The vegetation may be both positively and negatively affected by the amount of solar radiation it receives. The difference between north and south-facing slopes is more evident during winter months as the solar angle is low (Granger and Bredenkamp, 1996; Nagy and Grabhorr, 2009).

The varying micro-climatic characteristics observed on different aspects may cause considerable contrast in vegetation types. The altitudinal range over which a species occurs may also vary with aspect; in the Lesotho highlands subtropical or C_4 grasses are found up to 2950m a.s.l. on north-facing slopes, but only up to 2750m a.s.l. on south-facing slopes (Morris *et al.*, 1993). *Merxmuellera macawanii* (Stapf.), a species very similar to *Merxmuellera drakensbergensis* occurs mainly on east-facing slopes (Granger and Schulze, 1977). Aspect may also play a role in the colonization of invasive species, yet is dependant on the location and invasive species (Granger and Schulze, 1977). Similarly, a study in east England demonstrated a greater abundance of grasses (both individuals and species) on north- and west-facing slopes than south- and east-facing slopes (Hutchings, 1983). Aspect also influences the gradient of slopes through its influence on weathering and the movement of water (Morris *et al*., 1993). Plant distribution

may be influenced by gradient with fewer plants found on steep slopes and greater numbers on gentle slopes. According to Holland and Steyn (1975), slopes steeper than 15-25^{o} have very shallow soils, which reduces the water retaining capacity on such slopes, thus also reducing plant growth. The growing season on north-facing slopes is believed to be longer than that on south-facing slopes in the Drakensberg (Morris *et al.*, 1989). Aspect, along with slope, is able to influence the exposure to wind, and in turn, the amount of windblown snow and snow drifts (Billings, 1974), thus also impacting on plant growing seasons.

South-facing slopes in Lesotho generally have fair to good vegetation cover while north-facing slopes have both good and poor vegetation cover (Morris *et al.*, 1989). This difference stems from the north-facing slopes being generally drier and warmer, while the south-facing slopes tend to be wetter and cooler, and thus have a longer snow duration (Nüsser, 2002b). Particularly noteworthy is that the composition and percentage cover has apparently changed on warmer slopes in the mountainous regions of Lesotho due to overgrazing (Martin, 1984; Morris *et al.*, 1989). When given the freedom of choice, sheep prefer to graze warm slopes despite increased abundances of forage on cooler slopes (Hercus, 1964). This may be due to the differing palatabilities of species, species structure and the ability of neighbouring plants to influence forage selection (Meissner *et al.*, 1999). Aspect is also able to influence the fuel load available to fire. In Lesotho, south-facing slopes have a higher fuel load than north-facing slopes during the burning season, which is attributed to higher grazing pressure on the warmer north-facing slopes (Morris *et al*., 1989). In KwaZulu-Natal grasslands, burning exclusion was able to alter the vegetation on the south-facing slopes through increasing the number of forbs and shrubs, while the north-facing slopes were maintained as grasslands (Titshall *et al*., 2000).

2.12 Grazing concerns

Grazing has been recognized as a maintaining force in many ecosystems through its role in controlling invasive species and preventing dominant species taking over (Crawford, 2008). Similarly, grazing of alpine grasslands may aid in maintaining high plant diversities, as observed in the Alps, Germany (Hoiss *et al*., 2013). However, the effects of grazers are not always advantageous. The selectivity of livestock for favourable species can result in grassland deterioration, even when only a few livestock are present, such as observed at the Burgess Spring Experimental Range, California (Hormay, 1956). Livestock are more likely to graze the same favourable forage each season, even re-graze the same individuals within a season, rather than grazing less favourable ungrazed plants (Hormay, 1956). The inter-tussock vegetation (forbs and

shrubs) provides more valuable protein filled forage than that found on tussock grasses (Körner, 2004). Insect herbivores and their effects have not been studied in detail in the literature and their impact remains unknown (Whelan, 1995). Fire and grazing are frequently viewed as equals as both act to remove plant tissue (McNaughton, 1991), stimulate resprouting (Trabaud, 1987) and select for plant traits promoting survival (Bond and Keeley, 2005). However, their timing is often different within a natural setting as fire tends to occur in the dry season when grasses are dormant and thus survival is high. On the other hand, grazing tends to occur throughout the year, but especially when actively growing leaves are available (McNaughton, 1991), yet the net effect may be positive or negative (Hannaway *et al.*, 2000).

Grazing has two main effects on grass, as outlined by McNaughton (1991); firstly, it acts to reduce shading of the soil, but allows for the penetration of light to growth points, and secondly, it increases the rate of nutrient cycling through defecation and trampling of dead material (but not always in the same place). However, trampling and overgrazing may result in soil exposure leading to erosion (Grab and Nüsser, 2001; Du Preez and Brown, 2011). Similarly, urine and faeses of livestock my cause vegetation to die, through high urea contents or coverage (Du Preez and Brown, 2011). Other effects include the alteration of plant lifespan, plant size, response to climatic variables and changes in abiotic factors such as soil nutrients (Mucina *et al.*, 2006). Increased light, reduced litter and increased nitrogen availability have been measured after grazing, and may increase species richness (Fynn *et al*., 2004; Veen *et al*., 2008). However, these may not influence resource variability on a small scale, as observed in Kansas, U.S.A. (Veen *et al.*, 2008). Defoliation may also decrease photosynthesis and transpiration as a result of a reduced ability to capture and use solar radiation (Caldwell *et al.*, 1983). Grazing also reduces the fuel load, and prevents fires of intensities suitable for restricting woody vegetation growth (Brown, 2000; Veen *et al.*, 2008; Davies *et al*., 2010). However, in less palatable tussock grasses, grazers selectively remove green shoots leaving unpalatable leaves to accumulate, thereby increasing flammability of the area (Kirkpatrick *et al*., 2011). The Drakensberg grasslands require frequent grazing to remove old plant litter and promote tiller growth (Everson *et al.*, 1985a; Manson *et al.*, 2007). Yet, grazing of burnt plots resulted in lower species richness than in grazed unburnt plots in the grasslands of KwaZulu-Natal (O'Connor *et al.*, 2011). In northern Norway, the highest number of alpine grass species are found in areas supporting the highest grazing pressure, as no single species was able to dominate (Löffler and Pape, 2008). This is supported by the finding that grazing reduces the dominance of C_4 grasses, but promotes greater plant diversity (Veen *et al.*, 2008). Similarly, the absence of grazing may both negatively

or positively affect plant diversity (Austrheim *et al.*, 2008). However, at small spatial scales, some species abundance may be reduced when grazers are removed, as observed in Kansas, U.S.A. (Veen *et al.*, 2008). The effects of grazing are also dependent on the species of grazer; for instance sheep are selective grazers between both plant parts and season (Hercus, 1964).

Control of palatable woody species and exotic species in moist spaces between tussock grasses is another advantage of grazing (Hunter and Boswell, 2003). This is achieved through grazing reducing population growth and increasing the mortality of palatable species (Breebaart *et al.*, 2006). In many areas of Lesotho, *Karroochloa purpurea* and *Catalepis gracilis*, species which are indicative of disturbance, have invaded from the Karoo (Killick, 1990). Individual species react differently to grazing according to their structure, characteristics, evolutionary traits and chemical composition (Bock *et al.*, 1995), as well as the frequency of grazing (Gillen *et al.*, 1990). These differences extend to influence the vegetation structure at population, community and landscape scales (Bock *et al.*, 1995).

The preference of grazers for resprouted burnt grasses has been established on the southern Great Plains, USA (Allred *et al*., 2011) and in the Hluhluwe-Umfolozi Park, South Africa (Archibald *et al.*, 2005). An increasing time since burnt correlated to a decrease in forage quality, while the forage biomass increased (Allred *et al.*, 2011). Grazers are frequently introduced into a post-burn site soon after green shoots appear and remove the newly produced biomass. Much of leaf growth after defoliation is the expansion of cells within the intercalary meristem, and in most cases the leaf tip will not reform (Hannaway *et al*., 2000). In some cases, new tissues are removed faster than can be produced, resulting in a decrease in production and a longer recovery period, such as in Bloemfontein (Snyman, 2004b). Grazing in one season may have a carryover effect into the next growing season, which may have the same or a greater effect than disturbances in the following growing season (Snyman, 2006). The carryover effect decreases with increasing recovery time between burning and grazing, as observed in Bloemfontein (Snyman, 2006). This, combined with a decrease in biomass due to grazing, leads to a general decrease in grass cover and biomass, as has been the case in Tasmania (Bridle *et al.*, 2001; Kirkpatrick *et al*., 2011). Reduced ground cover results in greater exposure of seedlings to heat and frost, increasing their mortality rate and leading to the degradation of natural vegetation and the landscape (Breebaart *et al.*, 2006). Conversely, it can also increase opportunities for non-palatable seedling establishment (Breebaart *et al.*, 2006). Yet, sustained seasonal grazing allows the limited forage to become further depleted, as in New Zealand (O'Connor *et al.*, 1997).

The resprouting ability of tussock grasses from roots, crowns and below ground storage organs enables survival from grazing and fire (Whelan, 1995). The near surface location of the shoot apexes permits tussock survival under frequent grazing conditions (Everson *et al.*, 1985b). Many plant deaths that occur are explained by the depletion of food reserves when new shoots are intensively grazed (Cremer, 2004). A single defoliation event in the form of clipping is illustrated in Figure 2.6. Substantial energy in the form of stored carbohydrates is required for the production of the first three leaves, until photosynthesis is able to take over (Trlica, 1999; Wolfson and Tainton, 1999). Grazing is known to decrease individual tussock nutrients by 35% and stored carbohydrates by 46% in the Murchison Mountains, New Zealand, which cannot be replaced using storage reserves (Lee *et al.*, 1988). Exactly how high grazing intensity needs to be to alter the vegetation has never been established in alpine ecosystems (Austrheim *et al.*, 2008).

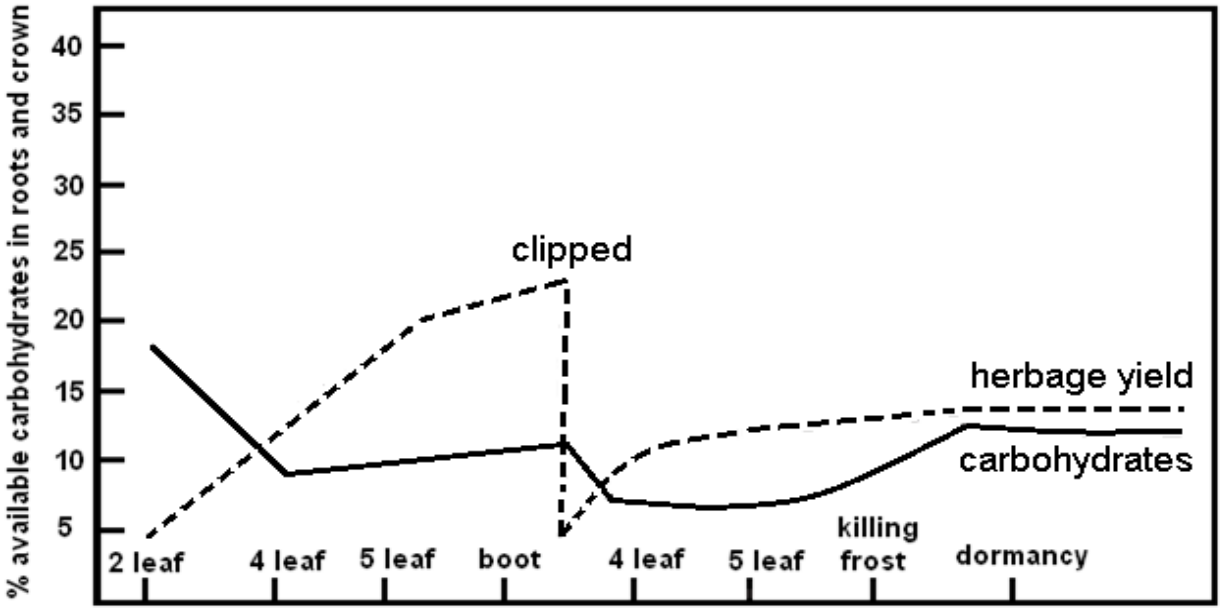


Figure 2.6: Carbohydrate levels and herbage yield of a grass plant after a single defoliation event (after Trlica, 1999).

The seasonality of grazing also plays a role in regrowth, species abundance and species richness, with different seasons producing different effects. Grazing during summer months may increase tiller turnover but not necessarily influence tussock density, as in Oxfordshire, England (Bullock *et al.*, 1994). However, winter grazing can negatively influence tussock density through the slower replacement of new tillers (Bullock *et al.*, 1994). In alpine regions this is thought to be more detrimental due to slower growth. During all four seasons in Lesotho, the least total biomass production has been measured under heavy stocking rates (14 ewes per ha per yr) (Mpiti-Shakhane *et al.*, 2001). The production of livestock is dependent on the quality and

quantity of available favourable grazing substrate. However, unfavourable grasses can be manipulated into being favourable through burning (Bond, 1997). In southern Africa, burning has been used to manipulate grassland composition to support larger numbers of grazers for many years (Bond, 1997), and especially so in Lesotho. Frequent and extreme defoliation can diminish grass reserves with detrimental consequences to individual plants and entire communities and ecosystems (Mucina *et al.*, 2006).

An assessment of influential factors on the effects of grazing by large herbivores was performed to determine whether large scale observations correspond to theory and management strategies, by Milchunas and Laurenroth (1993). They found that changes in species composition were linkd to aboveground net primary productivity, the history of grazers at each location and the intensity of grazing (Milchunas and Laurenroth, 1993). However, the location of grazing may be more influentaual than the manner in which grazing occurs (Milchunas and Laurenroth, 1993). As a result, the authors caution that many exceptions to broad scale models may occur (Milchunas and Laurenroth, 1993). Three other accepted models have attempted to model the relationship between grazing and plant diversity and community structure: the state and transition model (S-T model) proposed by Westoby *et al.* (1989), the MSL model propsed by Michunas, Sala and Lauenroth (1988), and the synethesis of the two models by Cingolani *et al.* (2005). The evolutionary history of grazers is considered important to the MSL and synethesis models and spanned the period from roughly one million years ago to 100 years ago (Milchunas and Laurenroth, 1993). The Holocene and associated warmer climatic contions are thought to have increased the herbivore species in the Lesotho highlands (Plug, 1997), suggesting a relatively short evolutionary history of grazers. Thus, the study region falls in to the synthesis model category of a short evolutionary history and low productivity. The model suggests that the resilience of the grasslands is not sufficient for long term grazing at moderate to high intensities (Cingolani *et al.*, 2005). The result may be species extinctions, altered community compositions, erosion and invasion by species with longer grazing histories (Cingolani *et al*., 2005).

2.13 Defoliation due to handcraft creation

In many areas of rural Lesotho, plants are used to meet the needs of people where other infrastructure over-exploitation and associated landscape destruction (Letšela *et al.*, 2003). The creation of conservation areas may act to restrict the area in which plants are gathered, thereby increasing the intensity of harvesting in particular areas (Letšela *et al.*, 2003). The harvested plants take the form of medication, food, firewood, building materials and handcrafts

(Guillarmod, 1971; Mofokeng and Ge, 2009). The leaves of mature plants of the *Merxmuellera* tussock grasses are hard, tough and unpalatable to livestock. However, these features make it ideal for use in handcrafts, mainly the construction of woven baskets (Letšela *et al.*, 2003) and traditional Lesotho hats (Figure 2.7). Sixty seven percent of people favour the *Merxmuellera drakensbergensis* (Schwweik.) and 43% favour *M. macowanii* (Stapf.) for handcrafts such as smaller baskets (Letšela *et al.*, 2003). These species are mostly used for crafts aimed at the tourist industry. In 1986 proper use factors (PUFs), indicators of the amount of available forage produced by a particular species, were assigned to species defoliated most often (Morris *et al.*, 1989). PUFs are dependent on the ability of a species to recover after defoliation and the palatability of neighbouring species (Morris *et al.*, 1989). *Themeda triandra* was assigned a factor of 0.5 and *Merxmuellera drakensbergensis* a factor of 0.15 (Morris *et al.*, 1989), but given that the crown of these grasses is seldom damaged, they are able to regenerate (Pomela *et al.*, 2000). Although it is suggested that sustainable use can address the needs of local people and cooperate with conservation efforts (Letšela *et al.*, 2003), this does not take the impacts of fire and associated grazing into account. Notably, areas recently harvested are frequently burnt to remove refuse and provide palatable shoots for livestock (Guillarmod, 1971).



Figure 2.7: Lesotho hats made within the study region using *Merxmuellera* sp. and sold to tourists.

2.14 Fire and grazing

The impact of environmental threats (e.g. fire, drought, grazing, flooding) acting together is often much greater than when occurring singly (Pomela *et al.,* 2000). Fire and grazing are often hard to separate for research purposes, due to their defoliation abilities and linkages at a landscape scale. However, the impacts of fire and grazing may differ with scale of observation (Veen *et al.*, 2008). On a small scale, burning and grazing have opposite impacts (Veen *et al.*, 2008): fire may increase plant production while grazing removes plant material. The advantages of fire to forage production have been mentioned in many literary documents (Wright, 1974). Burning generally increases palatability, digestability and nutrient content of grasses and thus increases their appeal to grazers (Van de Vijver *et al*., 1999; Cremer, 2004; Mbatha and Ward, 2010). The preference of cattle for grazing on burned sites was established by Laterra *et al.* (2003) and Vermeire *et al*. (2005), and for sheep by Spowart and Hobbs (1985) and Ross *et al.* (1997). In grasslands, this is largely due to the unpalatability of adult tussock leaves (Körner, 2004) and the increased palatability of seedlings and those recovering from fire (O'Connor *et al.*, 1997). Possible increases in the nutrient value and increased attractiveness of grasses to grazers after burning are indicated by weight gains in individuals within a grazing herd (Daubenmire, 1968; Vogl, 1974; Van de Vijver *et al.*, 1999), which may be likely due to an increase in accessibility of new shoots (Wright, 1974; Wright and Bailey, 1982). Consequently, animals tend to focus their grazing activities on burnt areas, which may lead to overgrazing and increased grass mortality, even though the grass growth rates may be greater (Wright, 1974; Kirkpatrick *et al*., 2011). Conversely, fewer grazers may be observed in unburnt areas, with numbers decreasing with increased burnt area size, as observed in KwaZulu-Natal grasslands (Archibald and Bond, 2004). Thus, to prevent overgrazing, the burn size should be greater than that required to support the livestock (Trollope, 2011). Many ecosystems contain few palatable grass species since few can complete their lifecycles (Mucina *et al.*, 2006), but burning can make long-lived species susceptible to grazing, often resulting in their death (Whelan, 1995). The reduction in preferential forage species availability may result in degradation of the current vegetation and landscape (Peláez *et al.*, 2009). In Lesotho, high intensity grazing results in new shoots being grazed almost immediately (Cwielong *et al*., 2000). The size of the burnt area becomes important as grazing is focused on these areas (Whelan, 1995; Archibald and Bond, 2004). The size, density and number of tillers of tussocks may decrease after burning in the first two growing seasons, such as observed on Mt Benger, New Zealand (Yeates and Lee, 1997), consequently reducing nutrient availability to livestock (Daubenmire, 1968).

Increased grazing access is an important outcome of burning, especially in tall-grasslands such as found in New Zealand (Yeates and Lee, 1997). The removal of leaf litter means new vegetative growth is continuous along the ground, making grazing less energy consuming and resulting in more grass being consumed (Daubenmire, 1968). However, the opening up of grassland communities and increased availability of light has the potential to provide space for the colonization of invasive species (e.g. Hercus, 1964; Laterra *et al*., 2003). An added advantage of burning for livestock is the removal of pests such as ticks and fleas which add to the attractiveness of burnt patches (Vogl, 1974; Trollope, 2011), whilst the resprouting of grasses after fire can increase the palatability of normally coarse grasses (Wright and Bailey, 1982). Owing to preferential grazing by sheep in Otago, New Zealand, shorter leaves (30-40cm) were observed in burnt sites than unburnt sites (50-80cm) (Ross *et al.*, 1997). In addition, regular burning combined with grazing frequently results in reduced vigour, stability and tussock biomass, and consequently decreases water quality (Mark, 1994). Grazing may also have a long-term impact on soil properties through a reduction in nutrient input to the soil. The impact of grazers to nutrient cycling through excretion may be less than the impact of grazing on recently burnt tussock productivity (Ross *et al.*, 1997). Yet, In addition, burning and grazing in bogs may greatly reduce the peat supporting wetland vegetation, thereby producing rocky shelves (Wahren *et al.*, 2001a).

'Alpine grazing reduces blazing' is a popular outlook in Australian alpine areas (Williams *et al.*, 2006), which relies on the principle that grazing reduces the fuel load, the probability of natural fires, and associated extreme high temperatures that may be produced. However, the effects are likely to be patchy and not at a landscape scale (Williams *et al.*, 2006). Supporting this, herbivory was demonstrated to have no effect on fuel levels in a KwaZulu-Natal grassland (Mulqueeny *et al*., 2011). Additionally, the palatability of the forage may also play a role, with grazers selecting only living shoots in unpalatable tussock communities, leaving flammable fuel to accumulate, as in Tasmania (Leonard *et al*., 2010). The exclusion of grazers in burnt patches may thus reduce flammability of grasslands, as also observed in Tasmania (Kirkpatrick *et al*., 2011). Yet, grazed plots contained less fuel than ungrazed plots resulting in smaller, less intensive burning events, in the sagebrush steppes, northeastern Oregon (Davies *et al*., 2010). The relationship between burning and grazing was expanded by Leigh and Holgate (1979), who found that the percentage of dead adult plants that were burned and grazed were significantly greater than those which were burned and ungrazed and those that were unburned and grazed in Australia. Conversely, an increased mortality in tussock grasses was observed in an ungrazed site

in the steppes, Oregon, and attributed to hotter fire temperatures due to an increase in dead plant matter (Davies *et al*., 2009). A method to determine mortality due to heat stress, rather than due to grazing, was introduced by Wright (1970), and involves using the ratio between duration of burning and the time required to kill the plant tissue. In addition to potentially killing individual tufts, burning also removes photosynthetic organs causing new growth to be fueled by stored compounds (Wolfson and Tainton, 1999; Miller, 2000). This is not a problem unless stored carbohydrates are low, as growth takes precedence over storage (Hannaway *et al.*, 2000).

During the rapid growth of new buds, stored starch is used, and if grazing occurs at this point, there is little remaining starch to support a second phase of sprouting (Whelan, 1995). Similarly, new bud growth is able to manufacture the carbohydrates necessary for continued plant growth and respiration, and if the new buds are removed, the growth and respiration of the plant is limited (Miller, 2000). During the recovery from fire, tussock grasses are required to draw on their own reserves and compete for nutrients from the surrounding soil with inter-tussock species (O'Connor *et al.*, 1997; Ross *et al.*, 1997; Yeates and Lee, 1997). Under grazing and burning regimes, inter-tussock species become dominated by herbs and other grasses, as observed in New Zealand (Yeates and Lee, 1997). The introduction of grazers soon after resprouting has occurred may be detrimental to tussock survival due to the lack of stored resources needed to partially fuel regrowth, such as observed in northwestern Patagonia, Argentina (Gittins *et al*., 2011). Grazing of regrowth within the first two years after burning was indicated to harm the nutrient status of tussock grasses, especially those growing on inhospitable soils, which represses and slows recovery on South Island, New Zealand (Mark, 1994). However, with regards to climatic climax grasslands, such as those within the study region, a shorter resting period is recommended (only until late summer following burning) by Trollope (1999). Grazing may affect the fire regime by elongating the time required for recovery between burning events, as well as increases the chances of plant mortality (Zimmermann *et al*., 2010).

Tussock population growth, and therefore community structure, is dependant on growth, fecundity and mortality of the individual plants (Bond and Van Wilgen, 1996). These three factors are affected by the combination of fire and grazing in Lesotho, and in different ways result in a general decrease in tussock population growth. It has been shown that repeated burning, followed by grazing, reduces tussock density and size in snow tussock grasslands in New Zealand (Duncan and Thomas, 2004). In Tasmania, post-fire intensive grazing has been observed to result in a decrease in tussock grasses and an increase in more palatable shorter

grasses (Kirkpatrick *et al*., 2011). However, in South Africa the opposite has been observed, with shorter more palatable grasses more common in unburnt grasslands (Archibald, 2008). An increase in homogeneity was observed under grazing and increasing burn frequency, in Kansas, U.S.A. (Collins and Smith, 2006). Burning may also alter the hydrological characteristics of an area. For instance, decreased summer flows in Otago, New Zealand, were observed in catchments after early spring burning (Duncan and Thomas, 2004). This was attributed to the vigorous growth and transpiration of tussock grasses after burning. Prolonged grazing after burning has altered tall-tussock grasslands, forests and shrublands to short tussock grasslands now covering many regions of New Zealand (O'Connor *et al.*, 1997). Both overgrazing and burning have altered landscapes and caused the decline of many indigenous plant species (Bowie and Frank, 2001; Watkinson and Ormerod, 2001). Conversely, an increase in species diversity, associated with burning and grazing has been observed in North American prairies (Collins, 1987; Veen *et al.*, 2008). Thus, correct management of grazing after burning can have important impacts on the landscape as a whole (Snyman, 2004a). Yet, where fire alone would have little or no consequence, when combined with the impacts of grazing and other ecological disturbances (e.g. drought), can have a pronounced influence on landscape genesis (Whelan, 1995).

2.15 Fire frequency and remote sensing

Satellite imagery has been used for the last three decades to evaluate vegetation and burning, especially large scale fires (Kushla and Ripple, 1998; Amenteras *et al.*, 2005). At the SAFnet conference in 2004, the need for remote sensing development for use in fire monitoring and its environmental effects was highlighted (Dube, 2004). Knowledge of the area covered by fires on global, regional and local scales is important due to its effects on global change (Amenteras *et al.*, 2005). Here, satellite images are useful as they represent current conditions where many maps are out-dated, and are more temporally and spatially accurate than maps (Justice *et al*., 1997). Satellite imagery allows larger areas to be observed within one image, making it a cheaper means of monitoring than the use of many aerial photographs (Kushla and Ripple, 1998; Kontoes *et al.*, 2009). Remote sensing is particularly useful for inaccessible areas, such as much of the Lesotho highlands, where large burns cover many kilometers (Key and Benson, 2006). Additionally, remote sensing of burn scars can be used in rehabilitation, quantification of potential fuel levels, burn severity and extent of biomass burning (Brumby *et al.*, 2002). Additionally, satellite images may be used to investigate the factors driving large scale burning events (e.g. De Angelis *et al.*, 2012) and the barriers to fire spreading (Archibald and Roy, 2009). Remote sensing and analysis can also indicate soil exposure, soil colour, soil moisture, plant

cover and plant moisture resulting from burning in suitable images (Patterson and Yool, 1998). Burn severity mapping is one of the most commonly used analyses of burn scars as it gives an indication of the recovery ability of vegetation (Bruhjell and Tegart, 2001). In 2007, burn scars were analysed in terms of severity for the Anaktuvuk River Fire, Alaska, and indicated that 80% of the burnt area was classified as being between moderately and severely burnt (Jones *et al.*, 2009). Similarly, burn scars were identified according to severity in order monitor vegetation recovery in more detail in the Black Hills National Forest, South Dakota (Chen *et al.*, 2011). The rapid and cost-efficient nature of remote sensing in monitoring landscape change explains their wide and increasing use (Lentile *et al*., 2006). The cost-effective nature of using remote sensing is highlighted in a study of burn scars to evaluate the post-fire treatment of the eastern Jemez Mountains, New Mexico (Clark and Kuyumjian, 2006).

A variety of satellites and sensors have been used to evaluate fires; for example the Advanced Very High Resolution Radiometer (AVHRR), NOAA Polar-orbiting Operational Environmental Satellite (POES), and Landsat Thematic Mapper (TM) , Enhanced Thematic Mapper (ETM+) (Brumby *et al.*, 2002) and the Advanced Wide Field Sensor (AWIFS) (Sedano *et al.*, 2013). The Moderate Resolution Imaging spectroradiometer (MODIS) sensor has been used in many burn scar mapping studies due to its availability (e.g. Archibald and Roy, 2009; Archibald *et al.*, 2010; Mohler and Goodwin, 2010; Govender *et al*., 2012). However, such a resolution would not be adequate for this study as it would not indicate burn scars smaller than $100m^2$. Landsat has historically been used due to its high resolution (15-30m) and availability (Brumby *et al.*, 2002; Petropoulos *et al.*, 2011). The sensor has six bands ranging from the visible to mid-infrared wavelengths and a seventh band depicting surface brightness temperature (Lentile *et al.*, 2006) (Table 1, Appendices). Landsat TM has proven a reliable source for mapping burn scars in semi arid regions of southern Africa (e.g. Ventura *et al*., 2003; Govender *et al*., 2012), Australia (e.g. Russell-Smith *et al.*, 1997; Sever *et al*., 2012; Srivastava *et al.*, 2012), the Mediterranean (e.g. Petropoulos *et al.*, 2011; Lozano *et al*., 2012) and North America (e.g. Bastarrika, 2011; Chen *et al*., 2011). Although, large archives exist to supply pre- and post burn images (Bobbe *et al.*, 2003), it is rare to find an adequate sequence of images to accurately map a fire regime for an area (Laris, 2005). As with all satellites, noise created by perturbing factors, such as reflectance variations, atmospheric particles, changes in surface variation and sensed observed dimensions, exist and need to be accounted for (Roy *et al*., 2006).

Today, monitoring programs such as the Global Burn Area 2000 (Grégoire *et al.*, 2003) and the GLOBSCAR (Simon *et al.*, 2004) report on burnt areas globally. More specifically, in the U.S.A., the LANDFIRE project aims to report on historical burn scars and assess the further implications of the detected burn scars, such as vegetation change and disease (Vogelmann et al., 2011). In Southern Africa, the Near Real Time Fire Information System for Southern Africa (AFIS) program detects and records active South African fires. However, these monitoring programs operate at coarse (500m) or medium (250m) resolutions (Bastarrika *et al*., 2011; Sedano *et al.*, 2013). The course resolution may result in an over estimation of the burn scars as smaller burn scars or the patchiness of the burn scars may not be recorded (Govender *et al*., 2012). Many of the current monitoring programs have become automated using frequent but in many cases, course data sets. For this study, Landsat and Spot 5 images were used due to the detail which they were able to provide to the burn scar identification and frequency maps.

The majority of burn scar studies have been conducted in forested areas (e.g. Brumby *et al.*, 2002; Fraser *et al.*, 2002; Justice *et al*., 1997; Kushla and Ripple, 1998; Patterson and Yool, 1998; Roy *et al*., 2006; Merino-de-Miguel *et al.*, 2010; Ruiz *et al*., 2012), while fewer studies have been carried out in grasslands or savannas (e.g. Trigg and Flasse, 2000; Ventura *et al*., 2003; Amenteras *et al.*, 2005; Laris, 2005; Smith *et al.*, 2005; Roy *et al*., 2006; Archibald and Roy, 2009; Mohler and Goodin, 2010; Takahata *et al.*, 2010). Regrowth of grasslands after burning is rapid, reducing the ability to identify burn scars over time. Similarly, burnt material such as ash and blackened plant stems may be removed by water or wind within a short space of time (Trigg and Flasse, 2000; Pereira, 2003). Hence, near-infrared radiation reflection of burn scars may not be able to differentiate between burnt and unburnt areas after one month (Mohler and Goodin, 2010). In Africa, coarse resolution studies of patchy, small burn scars tend to underestimate the extent of the burn scars, while studies of burn scars covering a large extent tend to overestimate burnt areas (Laris, 2005). It is important that the satellite used is capable of a resolution with pixel sizes smaller than the typical extent of fires in the study region (Fraser *et al.*, 2000). However, burn scars can be identified with both coarse and medium resolution if the frequency of satellite images is higher than re-growth of vegetation (Justice *et al*., 1997). Within the study region, the use of satellites, particularly Landsat, to monitor vegetation has been rarely used. Landsat images were used in the 1984 range inventory of Lesotho to determine where to collect basic plant data whilst aerial photographs were used for the identification of vegetation types (Martin, 1984). Similarly, and more recently, Landsat images have been used to monitor vegetation differences in the Lesotho highlands (Nüsser, 2002b).

Burn scars appear differently on different bands due to different brightness values (Figure 2.8) (Richards, 1993). Generally, the altered conditions after burning appear to the greatest extent in the reflective section of the magnetic spectrum: 0.45µm to 3.0µm (Petropoulos *et al.,* 2011). Ash and charcoal deposits, lack of vegetation, and alteration in vegetation structure, are characteristic of burnt areas and alter the reflectance of the landscape (Roy *et al*., 1999). Charcoal, which replaces vegetation, lowers reflectance from the visible spectrum (0.5-0.7µm) to near-infrared (0.8-1.2µm) (Lentile *et al.,* 2006; Smith *et al*., 2007). However, the reflectance may increase at small locations of intense burning, not often visible through Landsat as resolution is too coarse, due to the production of white ash (Smith *et al*., 2007). The near-infrared reflectance band correlates with the level of combustion completeness and extent of land burnt (Roy *et al*., 1999). Thus, the reflective and thermal characteristics of burn scars separate them from unburnt areas, allowing various indices to be used (Smith *et al*., 2007). Thresholds derived from one burnt area may be used for identification of burn scars in other areas. However, this is difficult to do as the thresholds are sensitive to differences in data and between burn scars (Roy *et al*., 1999). Burn scars differ spatially and temporally due to differences in vegetation type, intensity and duration of the fire, post-fire vegetative growth, and the rate of dissipation of ash and charcoal (Roy *et al*., 1999). Using fixed thresholds increases the ease and speed with which images may be processed, however, the thresholds need to be set for each new geographical region (Kontoes *et al*., 2009). The alternative to thresholds is the use of a decision/classification tree in which albedo, shadows, vegetation and burnt areas are all taken into consideration (Kontoes *et al*., 2009; e.g. Sedano *et al.,* 2013). Pixels appearing dark may be falsely identified as burn scars. These pixels may be caused by clouds, water or by the topography generating shadows (Sedano *et al.,* 2013).

Two different methods of monitoring fire through remote sensing exist: determination and study of 'hot spots' or active fires and of burn scars (Fraser *et al.,* 2000; Brumby *et al.,* 2002). One example, AFIS, provides information through remote sensing of active fires and records this information (Dube, 2004). Mapping of burn scars is more frequently used as the full extent of the burnt area can be established, as mapping 'hot spots' often fails to include some areas that are burnt (Fraser *et al.,* 2000). The resulting burn scar of a fire also remains visible for longer than an actively burning area allowing for a longer period for detection (Fraser *et al.,* 2000). Biased estimates of burn scar size are the result of the type of algorithm used, spectral heterogeneity (i.e. the contrast between the burnt areas and the unburnt areas) and the spatial heterogeneity (i.e. the mixture of vegetation or burnt and unburnt area within each pixel) (Laris, 2005). Many indices or

combinations of sensor bands have been developed to better observe landscape changes relating to fire (e.g. CBI-Composite Burn Index, BAI-Burn Area Index, CSI-Char Soil Index, GEMI-

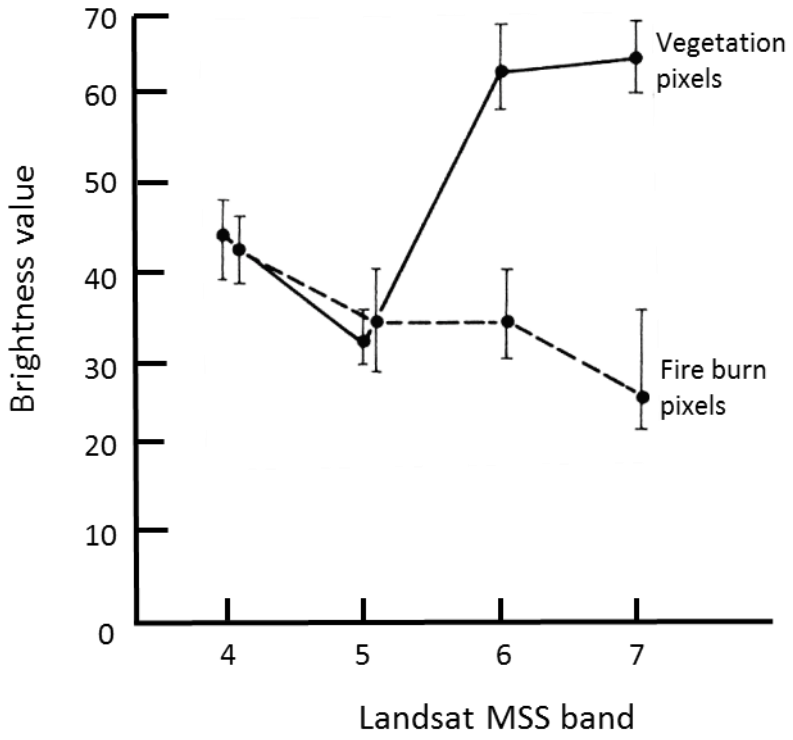


Figure 2.8: The representative spectral reflectance of healthy vegetation pixels and fire burnt vegetation pixels (after Richards, 1993, p. 283).

Global Environment Index) (Smith *et al*., 2007). Two of these, the NBR (and dNBR) and the NDVI (and dNDVI) will be discussed further. Additionally, many different classification algorithms have been used to identify burn scars, such as unsupervised classification, linear classification and non-linear classification.

2.15.1 Normalized Differenced Vegetation Index (NDVI)

The Normalized Differenced Vegetation Index (NDVI) has been widely used to successfully map burn scars (e.g. Diaz-Delgado *et al.*, 2002; Clark and Kuyumjian, 2006; Goetz *et al.*, 2006; Lentile *et al.*, 2006; Mohler and Goodwin, 2010; Chen *et al*., 2011). This is mainly due to the well-documented inaccuracies and limitations (Diaz-Delgado *et al.*, 2002). The NDVI uses the relationship between chlorophyll and reflectance values at different wavelengths to show vegetative ground cover. The differences between vegetation types are greater in the near infrared band than in the visible (Campbell, 2002) (Figure 2.9). It is believed that the 'cavities' within the leaf have a greater impact on the infrared band than the visible spectrum (Campbell, 2002). This index uses the property whereby near-infrared reflectance decreases after a burn event as chlorophyll and leaf tissue are reduced (van Wagendonk *et al.*, 2004; Escuin *et al.*,

2008). The NDVI is the 'ratio of infrared to red reflectance from the viewed surface' (Justice *et al*., 1997, p.4). The NDVI is a measure of photosynthetic activity (Fraser *et al.*, 2000), which is an appropriate method for use where alteration in green vegetation is the desired focus (Lentile *et al.*, 2006). Given that, the reflectance of exposed soil can significantly influence the results of NDVI, the Soil Adjusted Vegetation Index (SAVI) was formulated. Other indices using the red and near-infrared bands are the Burnt Area Index (BAI) and Global Environment Index (GEMI) (Escuin *et al.*, 2008).

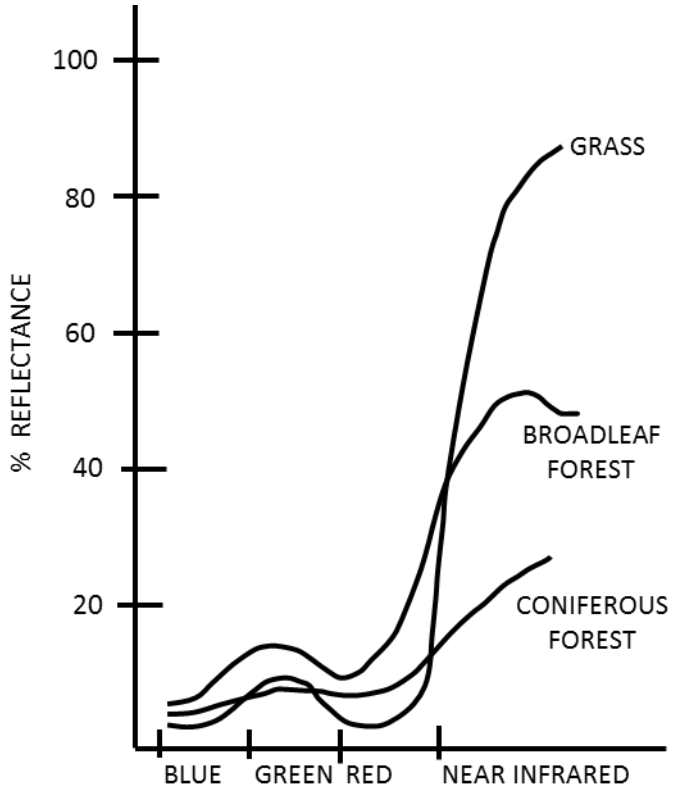


Figure 2.9: Different distinctions between vegetation types (after Campbell, 2002, p. 462).

The NDVI image shows values ranging between -1 and +1, with vegetated regions showing positive values and abiotic areas showing negative values (Escuin *et al.*, 2008). Burn scars show values within the negative region below 0, increasing in severity with increasing negativity (Diaz-Delgado *et al.*, 2003). Sparsely vegetated areas and unburnt senesced vegetation have been misclassified when using NDVI and SAVI indices (Smith *et al*., 2007), which is due to the low NDVI values of these areas (Ventura *et al*., 2003). Similarly, clouds are a problem in many satellite images as their shadows tend to reduce the extent of burn scars obtained using the NDVI (Ventura *et al*., 2003). Given the different spectral signatures of different regions, a single general signature for burnt surfaces is not possible, thus decreasing the comparability of images (Gulinck and Serneels, 1996).

Past uses of the NDVI in grasslands include classification, monitoring of landuse change, productivity, carrying capacity, management of pests, wildlife protection, and soil and water conservation (Nan, 2001). Traditionally, the NDVI of Landsat TM and ETM+ images have been used to identify burn scars (van Wagtendonk *et al.*, 2004). The NDVI of AVHRR is the mostcommon combination used in land classification research (Nan, 2001), however, the NDVI was outperformed by GEMI, GEMI3, VI3 (Vegetation Index 3) and channel 2 of AVHRR (Pereira, 1999). This demonstrates that although the NDVI has been used successfully in many studies, there are better indices which may be used, especially for grasslands.

2.15.2 Normalized Burn Ratio (NBR)

The use of Landsat bands 7 and 4 is more effective in depicting burn scars in canopy cover than the NDVI (Kushla and Ripple, 1998). Similarly, studies in Portugal using AVHRR have demonstrated that an improved burnt/unburnt distinction is observed when the mid-infrared band (3.55 – 3.93µm) replaces the red band in the NDVI (Pereira, 1999). The Normalized Burn Ratio (NBR) was developed by Key and Benson (Key and Benson, 2006) and uses the red and near-infrared bands (Key and Benson, 1999; Epting *et al.*, 2005; Escuin *et al.*, 2008; Vogelmann *et al*., 2011). These bands are the most influenced by burning but respond in opposite ways making them ideal for mapping burnt areas (Key and Benson, 2006). Tissue damage or a decrease in leaf chlorophyll reduces the reflectance in the near-infrared band (Epting *et al.*, 2005). Similarly, a decrease in vegetation shadow and moisture, as well as an increase in exposed soil, increases the reflectance of the mid-infrared band following burning (Epting *et al.*, 2005; Robichaud *et al*., 2007). The moisture sensitive nature of these bands has been used in the past to study and map wetlands, which makes it a very useful tool in areas where burn scars and wetlands are found (Dottavio and Dottavio, 1984). Decreased evapotranspiration and increased radiative absorption by the blackened scar also increases within the mid-infrared band (Lentile *et al.*, 2006). The mid- and near-infrared bands perform better than the combination of red and near-infrared bands (Epting *et al.*, 2005).

The NBR correlates better with field measurements than other indices, particularly the NDVI (Key and Benson, 1999; Key and Benson, 2004; Epting *et al.*, 2005). The NBR is also more sensitive to changes in reflectance than the NDVI (Escuin *et al.*, 2008). The bright reflectivity observed in the mid-infrared band (Landsat band 7) contrasts highly with the low reflectivity in the near infrared band (Landsat band 4), leading to a better display of burnt areas (Pereira, 1999). Similarly, this algorithm is good at separating burnt areas from wetlands and water bodies

(Pereira, 1999). Water and vegetation are displayed by positive values while burnt areas are chiefly displayed as negative areas (Pereira, 1999). The resulting image has a theoretical value range of -1.0 to +1.0 (Key and Benson, 2006). The value is positive when band 4 is greater than band 7, and near zero when the bands are nearly equal (Key and Benso, 2006). Negative values depict areas of 'severe water stress' and non-vegetative surfaces typical of burnt surfaces (Key and Benson, 2006). The short wave infrared band (Landsat band 7) is also less affected by smoke and haze than the red band used in NDVI (Hudak *et al*., 2007). However, the NBR relies on the mid-infrared sensor (band 7) on Landsat, which is unavailable on many other satellites (Escuin *et al.*, 2008). As with the NDVI, although to a lesser extent, drier areas are often classified as burnt areas (Pereira, 1999). Comparison of the NBR derived images is possible due to the removal of between image solar illumination effects and topographic effects within images (Key and Benson, 2006).

Today, the NBR (and dNBR) is the method of choice for US Federal land management agencies and the National Park Service, especially for mapping burn severity (Miller *et al*., 2009; Chen *et al*., 2011). It also forms the basis for the LANDFIRE project in the U.S.A. (Vogelmann *et al*., 2011). Similarly, Burned Area Emergency Response (BAER) teams use Burned Area Reflectance Classification (BARC) maps to allow for rapid responses to wildfires in the USA (van Wagtendonk *et al.*, 2004; Lentile *et al.*, 2006; Hudak *et al.*, 2007). These maps are created using NBR and dNBR indices, as they provide the best results and are quickly derived. The maps allow for burn area identification and areas at risk of soil erosion, which then permit appropriate erosion prevention initiatives to be undertaken (Hudak *et al.*, 2007; Robichaud *et al*., 2007). The principle was expanded to form the relativized dNBR (rdNBR) through normalizing dNBR images, which removes any bias towards pre-fire vegetation conditions (Miller *et al*., 2009). Using general remote sensing principles, the NBR has provided useful results in terms of quantifying burn scars and severity (van Wagtendonk *et al*., 2004). Similarly, the NBR produced the highest correlations to post-fire conditions over 50 sites when compared to the dNBR and other indices (Hudak *et al*., 2006). The NBR and dNBR are most useful for these maps, as the focus is on vegetative characteristics rather than soil attributes, thus allowing for the identification of ecological, economic and social areas at risk following burning (Lentile *et al.*, 2006). They also provide a reliable and repeatable, unbiased method for burn scar identification (Bobbe *et al.*, 2003). The dNBR was successfully used to map burn severity after the 2007 Anuktuvuk River fire in Alaska (Jones *et al.*, 2009). Similarly, the dNBR was used to construct a fire-history record for the Fraser Island, Australia (Srivastava *et al*., 2012). However, the NBR

has proven relatively inaccurate to a pre/post-burn study in a South African savanna context when compared to MODIS data for forest fires (Roy *et al.*, 2006). Yet, the NBR was successful in another South African study (Smith *et al.*, 2005). The NBR is considered more appropriate for measuring burn scars than the NDVI, given its ability to differentiate between burnt and unburnt pixels (Escuin *et al.*, 2008).

2.15.3 Other burn scar mapping indices

The MIRBI is apparently the most effective method to use for southern African savanna biomes with MODIS data, while the charcoal friction map is useful with Landsat ETM+ data (Smith *et al*., 2007). However, the use of this method at variable spacial scales produced unreliable results (Smith *et al*., 2007), and it may also be questioned whether this method is useful for grassland and sparser vegetation types. A multitemporal principal components analysis (PCA) has been used to map burn scars in Madikwe Game Reserve, where the land was overgrazed and fires suppressed (Hudak *et al.*, 1998). In this study, Landsat bands were converted into reflectance values before an NDVI analysis was used to successfully identify burn scars. The structural index (SI) (band 4 / band 5) was proven to be better than the NDVI when determining burn scars in canopy cover (Kushla and Ripple, 1998). Similarly, ratios of blue, green and red bands with SWIR were considered better than the NBR (Smith *et al*., 2005). The enhanced vegetation index uses the blue, red and near-infrared bands and is useful in forest vegetation (Chen *et al*., 2011). It would thus seem that the NBR may not be the best method of mapping in grasslands and shrublands (Lentile *et al.*, 2006). Recently, Support Vector Machines classifier was used successfully with Landsat images to produce burn scars in Greece (Petropoulos *et al.*, 2011). Additionally, MODIS Burned Area Index (BAIM) is one of the most commonly used (e.g. Merino-de-Miguel *et al.*, 2010).

2.15.4 Burn severity and remote sensing

The use of remote sensing in fire monitoring and assessment can be divided into perimeter and burned area methods, and those assessing post-fire surface characteristics (Lentile *et al.*, 2006). Burn severity analyses are useful in monitoring vegetation recovery, succession and rehabilitation (Roy *et* al., 2006). Burn severity is being increasingly determined from satellite imagery, especially in conjunction with the NBR (Hudak *et al*., 2006; Roy *et* al., 2006). Although burn severity has been highly correlated to the NBR for forested regions in many studies (Escuin *et al.*, 2008), the NBR is not neccessarily optimal for determining burn severity

(Roy *et al*., 2006). The cause of this was believed to have been spectral displacements occurring in many directions relative to the NBR isolines (Roy *et* al., 2006). The NBR was not created as a measure of severity, but rather as a burn scar detection index (Roy *et al*., 2006).The NBR has nevertheless been proven better than the NDVI for differentiating between moderate and severe burn severity (Escuin *et al.,* 2008). However, in lower density vegetation types, such as shrublands, the correlation was weak, which suggests that accurately performing a severity analysis of the Sani Pass region may be difficult. This study will not consider burn severity due to time constraints and difficulty in generating accurate severity figures, but rather focus on the location, extent and frequency of burns.

2.15.5 Post-fire vegetation monitoring

Monitoring of post-fire vegetation growth has been observed through remote sensing in the past and found to be a valuable management tool (e.g. Diaz-Delgado *et al.*, 2002; Pérez-Cabello and de la Riva, 2003; van Leeuwen *et al.*, 2010; Veraverbeke *et al.*, 2012a). For example, the recovery of plant populations was followed after a fire in a forested region of northern Spain in 1997; the rate of re-growth and the speed of the vegetation recovery was studied using Landsat TM images (Viedma *et al*., 1997). Monitoring post-fire vegetation recovery via satellite imagery may improve and alter change detection parameters through providing images before, during and after a fire (Chen *et al*., 2011). The recovery rate of vegetation relies on the burn severity (Diaz-Delgado *et al.*, 2003), climatic conditions (van Leeuwen *et al*., 2010) and vegetation type. A long-term study of fire scars and vegetation succession was conducted between 1989 and 2001 in northern Namibia using Landsat Quicklooks images (Verlinden and Laamanen, 2006). In order to follow vegetation growth and compositional changes, the NDVI is commonly used due to its high accuracy and ability to indicate photosynthetic activity (Goetz *et al.*, 2006; Vila and Barbosa, 2010; Veraverbeke *et al.*, 2012b). The NDVI is able to indicate the positive relationship between amount of burnt area and biomass of regenerated vegetative growth (Pérez-Cabello and de la Riva, 2003). The NDVI is the optimal tool for use in areas containing natural variation in vegetation in soil and vegetation type, for both long-term and short-term studies (Veraverbeke *et al.*, 2012b). This method was successfully used to study the relationship between fire severity and regeneration in Barcelona, Spain (Diaz-Delgado *et al*., 2003). The study demonstrated that a decline in NDVI values correlated with observed burn severity in the field. Similarly, the NDVI was demonstrated to accurately depict heterogeneous vegetation cover in the Peloponnese peninsula, Greece (Veraverbeke *et al.*, 2012b). Yet, the NDVI was indicated to have a lower correlation to field-based composite burn index data after two years, while the NBR was

indicated to correlate well for a longer period (Chen *et al.*, 2011). Similarly, the Tassel Cap Wetness index proved better suited to mediterranean vegetation than both the NDVI and the NBR (Lozano *et al.*, 2012), suggesting that perhaps vegetation recovery may vary with regards to the optimal index used with vegetation type. Remote sensing is useful in monitoring vegetation degradation over time, especially in high resolution using landsat TM, ETM+ and SPOT imagery (Nan, 2001; e.g. Diaz-Delgado *et al*., 2003). Long-term use of Landsat images to monitor vegetation can indicate changes in landuse patterns and alteration of climatic zones (Jönsson and Eklundh, 2002; 2004). However, this is difficult to do using NDVI in arid or sparse vegetated areas, such as in the Sani Pass region. The use of the NDVI to monitor vegetation may be more suited to mesic regions than to xeric regions (van Leeuwen *et al*., 2012). Despite this, the use of satellite imagery as a post-fire recovery management tool may be more cost effective and less time consuming than field-based monitoring (van Leeuwen *et al*., 2012). However, the special scale of the images needs to be taken into consideration before management practices are carried out (van Leeuwen *et al*., 2012).

2.15.6 Post-burn vs. pre/post-burn mapping

Three different types of analysis for the extraction of burn scars have been used; firstly a single date post-burn image is classified using derived indices or spectral tests; secondly, temporal analyses use pre and post-burn images; and thirdly, unsupervised or supervised classifications are used to differentiate between burnt and unburnt areas (Pu *et al*., 2004). There has been much debate as to whether the use of single date post-burn or differenced pre/post-burn images yield better results, as both methods have advantages and disadvantages. Single date post-burn indices are cheaper and less time consuming, and in addition, unlikely to contain as many errors as those using pre and post-burn images, such as those produced by differences in vegetation phenology, sun-sensor geometry, sensor calibration and atmospheric effects (Epting *et al.*, 2005). For example, a single post-fire Landsat TM image was used to map vegetation mortality with good accuracy by Patterson and Yool (1998). Many studies on historically burnt areas are undertaken by researchers with no knowledge of pre-burn conditions or burn severity, leaving single date post-burn studies as the only option (Escuin *et al.*, 2008). Single date images and the classifications used, have been included in local studies, but their use at larger scales or by automatic burn scar detection programs has proven problematic (Sedano *et al.*, 2013). In the current study, the region observed is at a small scale and the images may be classified manually at a low cost.

The use of pre-burn images allows better accuracy in mapping of older burns, senescent vegetation and water bodies with similar signatures (Epting *et al.*, 2005). This is conducted by subtracting the pre-burn composite image from the post-burn composite image (Fraser *et al.*, 2000). This process is commonly termed the Differenced Normalized Burn Ratio (DNBR/ dNBR) or the Differenced Normalized Differenced Vegetation Index (dNDVI) (Howard and Lacasse, 2004; van Wagtendonk *et al.*, 2004). Burn scars are identified through the change in reflectance measured for each pixel, falling either above or below an assigned threshold (Laris, 2005). Using images taken at different times creates many problems. Vegetation, biomes and often environmental changes, apart from burning, may occur between when the images were taken and consequently impact the results (Hudak *et al.*, 2007). Despite this, the pre-/post-burn method is the most commonly used method for identifying burn scars (e.g. Hudak *et al.*, 1998; Korontzi *et al.*, 2003; Howard and Lacasse, 2004; Key and Benson, 2006; Srivastava *et al.*, 2012).

Chapter 3:

Environmental Setting



CHAPTER 3:

Environmental Setting

3.1 Introduction to Lesotho

Lesotho is situated within the borders of South Africa between 28^{o} and 31^{o}S and 27^{o} and 30^{o}E (Pomela *et al.*, 2000). The country was under British rule from 1868 until independence in 1966 (Pomela *et al.*, 2000) and now has a population of over 1.8 million people inhabiting the relatively small country (30 555km^{2}) (Bureau of Statistics, 2008). Lesotho varies in altitude from 1520m in the west to 3482m in the east (Dempster and Richard, 1973). The majority of the country is mountainous and dissected by deep valleys forming valuable catchments (Guillarmod, 1972; Bureau of Statistics, 2012) (Figure 3.1). The Maluti and Drakensberg mountain ranges, which constitute the Lesotho highlands, form a major drainage divide, separating river catchments into those that flow east towards the Indian Ocean and those flowing west into the Atlantic Ocean (Sene *et al.*, 1998; Nüsser and Grab, 2002). The Lesotho highlands are an important watershed for southern Africa (Uys and Lechmere-Oertel, 2006), supplying rivers across southern Africa with fresh water and supporting hydro-projects such as the Lesotho Highlands Water Project.

Lesotho falls within the grassland biome of southern Africa (Martin, 1984; Mucina *et al.*, 2006) and the grasslands have been described as being irregular with tall and short grasses, tussock grasses and sod grasses, open grassland and areas of exposed soil and rock (Killick, 1990). Of the total area covered by Lesotho, 75% is non-arable due to cold conditions, erratic or low precipitation, and rocky thin soils on the slopes, and is thus agriculturally best used as rangeland (Mpiti-Shakhane *et al.*, 2001). Rangelands are a communal resource to which everyone has access, thus almost every rural household has livestock. Sheep and goats are generally found in greater numbers than cattle, although more households own cattle (70%) than sheep (45%) and goats (43%) (Tshabalala, 1995). Sheep and goats are mainly kept for wool and mohair production, with 50% of owners receiving income from livestock-related sales (Tshabalala, 1995). However, as observed in Austria, before livestock were introduced, the chief grazers of alpine environments (mainly deer and buck in Lesotho) are likely to have mainly focused on herb species (Austrheim *et al.*, 2008), whilst now the focus is more on the grasses. Livestock became increasingly more numerous in the nineteenth centuary (Quinlan and Morris, 1994). As a consequence, in many areas overgrazing has led to soil erosion. The decline of rangeland

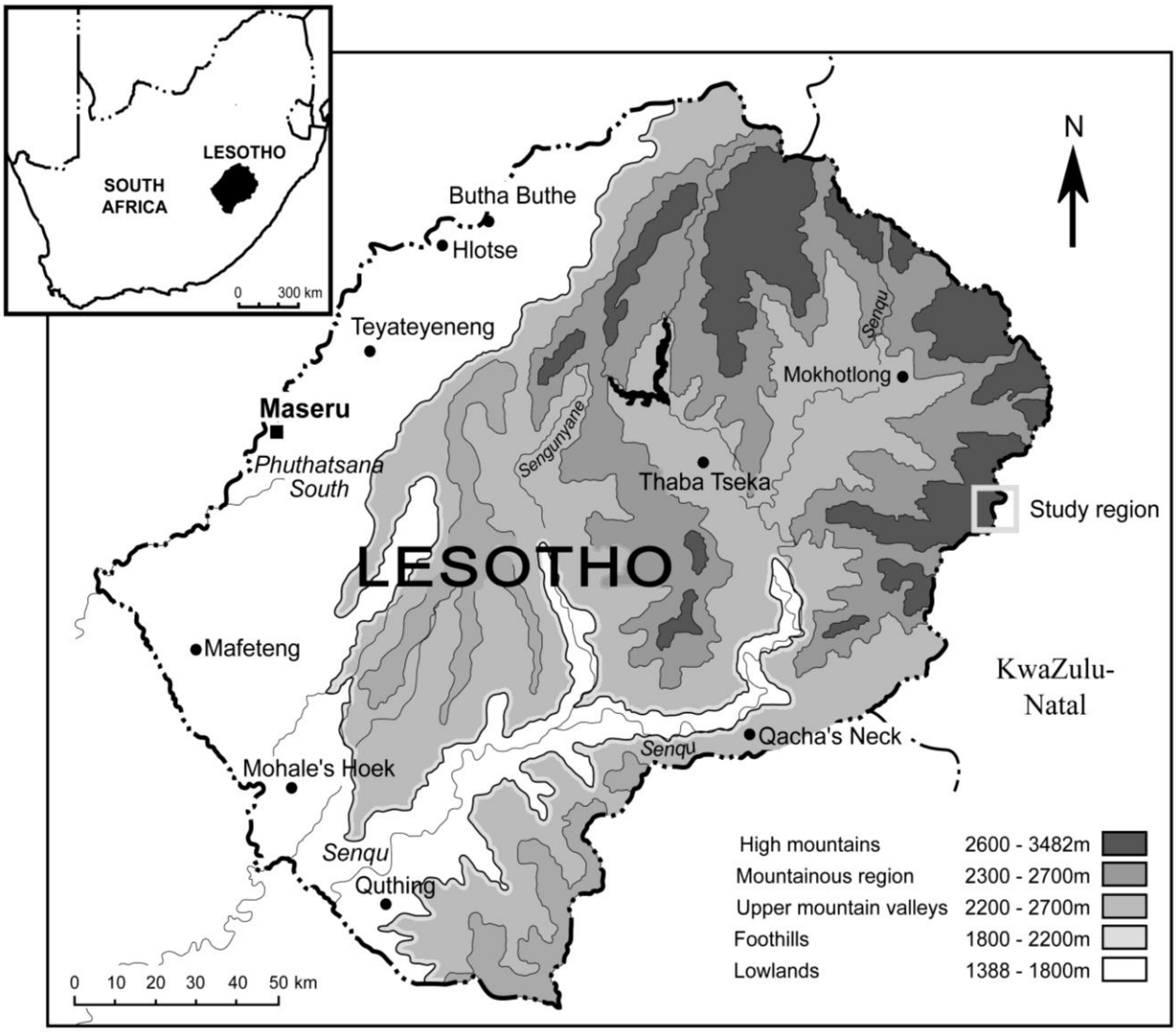


Figure 3.1: The physiogeographic regions of Lesotho (Wendy Phillips, Wits Cartography Unit).

conditions in Lesotho has resulted in a reduction in water production, which influences both the users of the rangelands and non-rangeland users (Motsamai, 1995).

3.2 The study region

The study region, containing eight smaller study sites, is located in the Sani Valley (29°35'S; 29°17'E) above Sani Pass in the Thaba-Tseka province of Lesotho, southern Drakensberg (Figure 3.1). The study region forms part of the centre of southern Africa's Afromontane region (McDonald, 1999; Bowie and Frank, 2001), the Drakensberg Alpine Center (DAC) (areas above 1800m a.s.l.) (Carbutt and Edwards, 2006), and also falls within the 'sourveld' vegetation region (Mucina *et al.*, 2006). It is estimated that 29.3% of plant species occurring in the region are endemic (Pomela *et al.*, 2000). Access to the study region is possible via Sani Pass where

accommodation is also available. Sani pass provides vehicular access between South Africa and Lesotho. The anthropogenic impact on the area is high, thus justifying any call for its conservation.

Sani top has a general altitude of between 2800-2900m and is classified as being within the alpine belt (between 2740m to 3480m) (Killick, 1978). The valley floor is characterized by topographically flat grasslands that at certain times of the year are marshy, interspersed with bogs and patches of heath (Hilliard and Burtt, 1987; Killick, 1978). The valleys are generally asymmetrical with steep slopes on which grazing posts have been established (Morris *et al.*, 1993). A small permanent grazing post is situated within the village to the northeast of the study site. Another grazing post is situated within the study valley and is situated on a north-facing slope on the side of the valley above 3000m. Further inland along the Mokhotlong road there is another grazing post in the village of Likhoetaneng, with livestock in the study region also using this grazing post. The grazing posts within the study area can be classified as either permanent or seasonal (Grab and Nüsser, 2001). The grass cover surrounding the grazing posts is depleted and conforms to the idea of decreased vegetation cover at high altitude grazing posts (Nüsser, 2002a).

The Sani region was selected for the study due to ease of access and the comparatively large literature base (e.g. Killick, 1978; Marker, 1994; Grab and Nüsser, 2001; Nel and Sumner, 2008). The region is relatively accessible through the Sani Pass, and additionally accommodation is available at Sani Top Chalets (Figure 3.2). The Kotisephola study region was added in December 2009 to add more depth to the study by including a higher altitude site (averaging 3230m a.s.l.). More specifically, the study sites, used within the baseline (chapter 7) and monitoring studies (chapter 8), were selected according to criteria discussed in more detail within Chapter 6 are indicated in Figure 3.3 and 3.4, and site name abbreviations in Table 3.1.

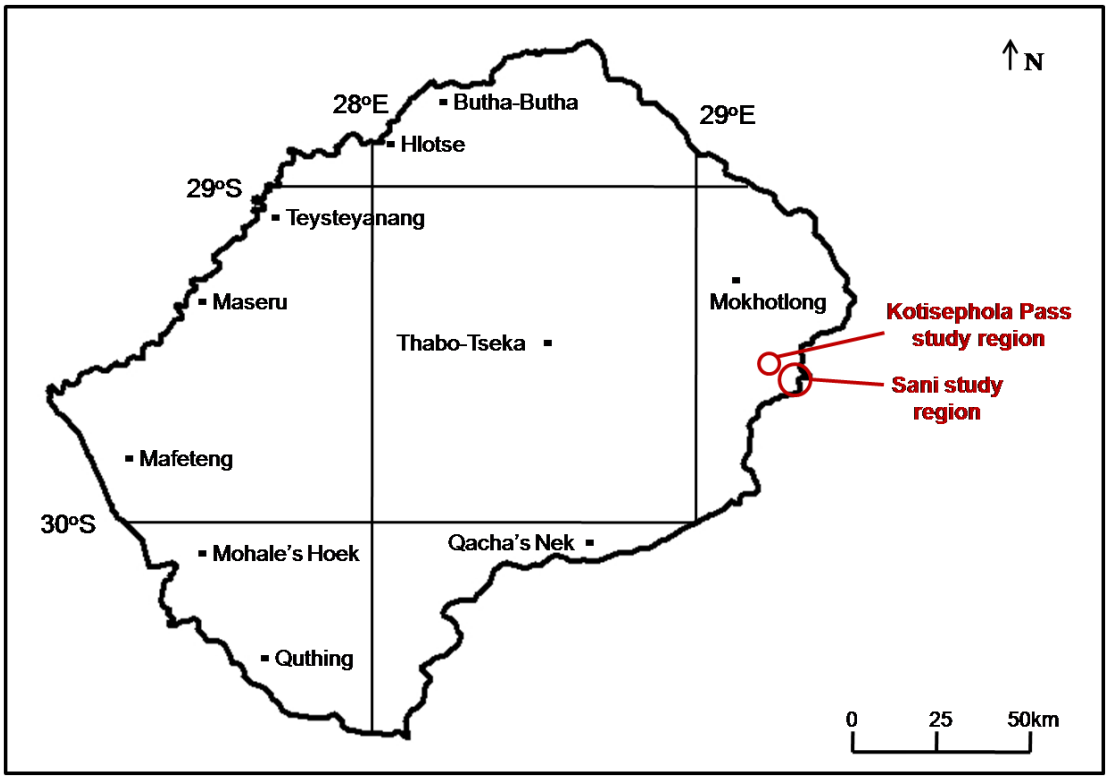


Figure 3.2: Location of the study regions selected in relation to a greater Lesotho context.

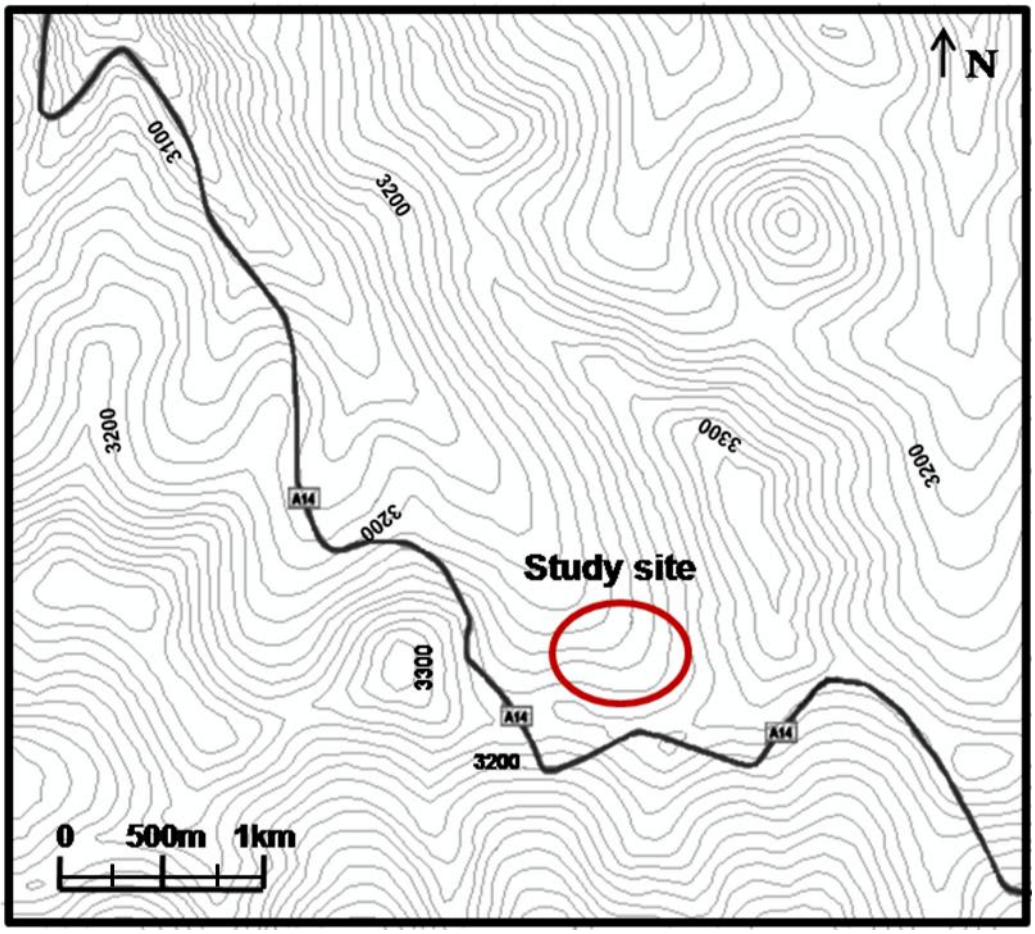


Figure 3.3: Map indicating the position of the Kotisephola Pass study site (contour interval: 20m).

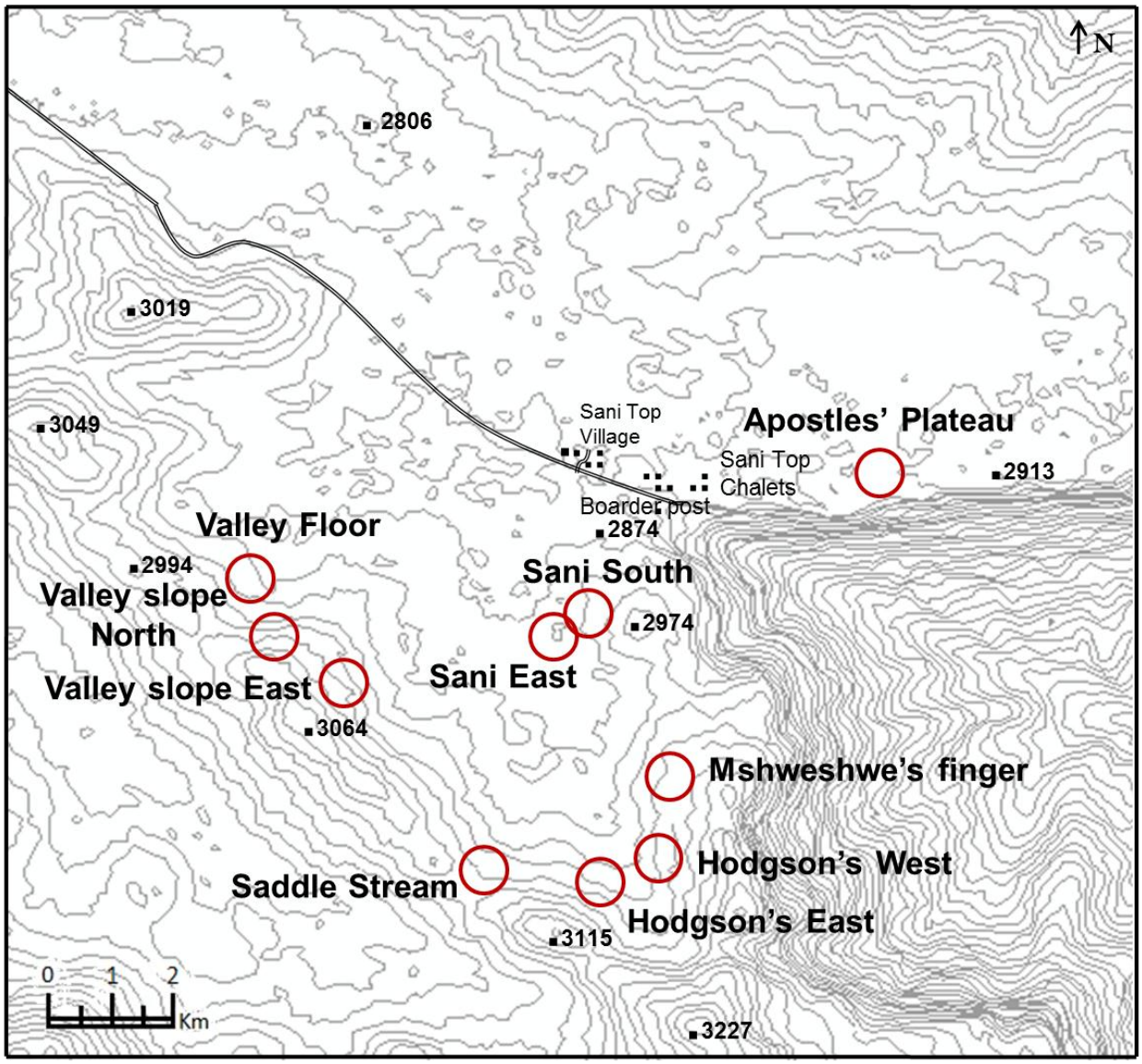


Figure 3.4: Diagram indicating the study sites (contour interval: 20m).

Table 3.1: Site name abbreviations used

Site Name	Abbreviation used
Sani east	S.E.
Sani south	S.S.
Valley slope north	V.S.N.
Valley slope east	V.S.E.
Saddle stream	S.Str.
Hodgsons east	H.E.
Hodgsons west	H.W.
Valley floor	V.F.
Mshweshwe's finger	M.F.
Kotisephola pass	K.P.
Apostles plateau	A.P.

3.3 Climate

3.3.1 General atmospheric circulation

Lesotho lies within the high pressure (anticyclonic) belt at approximately 30^{o}S and this has a substantial influence on the climate of the study region (Tyson *et al.*, 1976; van Zyl, 2003). The high pressure belt shifts north during winter and south during summer as it moves within the global tri-cellular circulation patterns (Preston-Whyte and Tyson, 1988; van Zyl, 2003) (Figure 3.5). The dominant high pressure cells are present off the South African coast to the east (South Indian Anticyclone) and the west (South Atlantic Anticyclone) and affect the circulation of moisture across southern Africa (Tyson and Preston-Whyte, 2000). The seasonal shift of the tri-cellular model allows for the occurrence of other perturbations such as an easterly low pressure

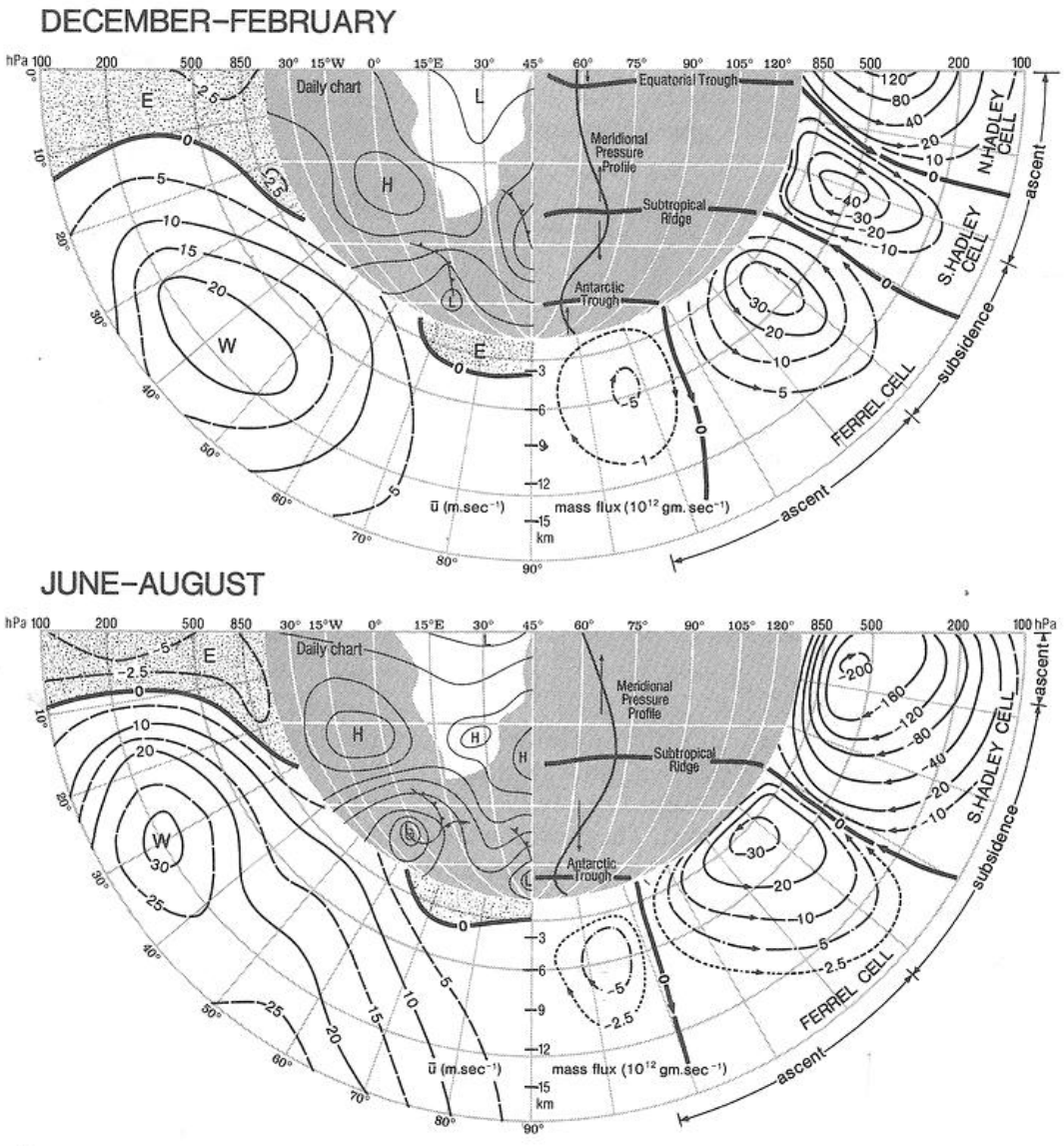


Figure 3.5: General circulation patterns in the southern hemisphere for December-February and June-August (after Preston-Whyte and Tyson, 1988).

which brings in moisture from equatorial regions (Tyson and Preston-Whyte, 2000) (Figure 3.6). Cold fronts, as part of mid-latitude cyclones, affect the southern regions of South Africa, primarily during the winter months, and often extend inland to influence Lesotho (Tyson and Preston-Whyte, 2000). In addition, cut-off lows infrequently occur, resulting in considerable rainfall and at times heavy snowfalls. Temperatures are affected by latitude, topography, altitude and ocean currents (van Zyl, 2003). The region typically experiences warm, wet and humid summers with partly cloudy skies (Guillarmod, 1971; McDonald, 1999; Lesotho Meteorological Services, 2008). Winters are cold and dry with moderate snowfalls adding to the precipitation (Killick, 1978; McDonald, 1999).

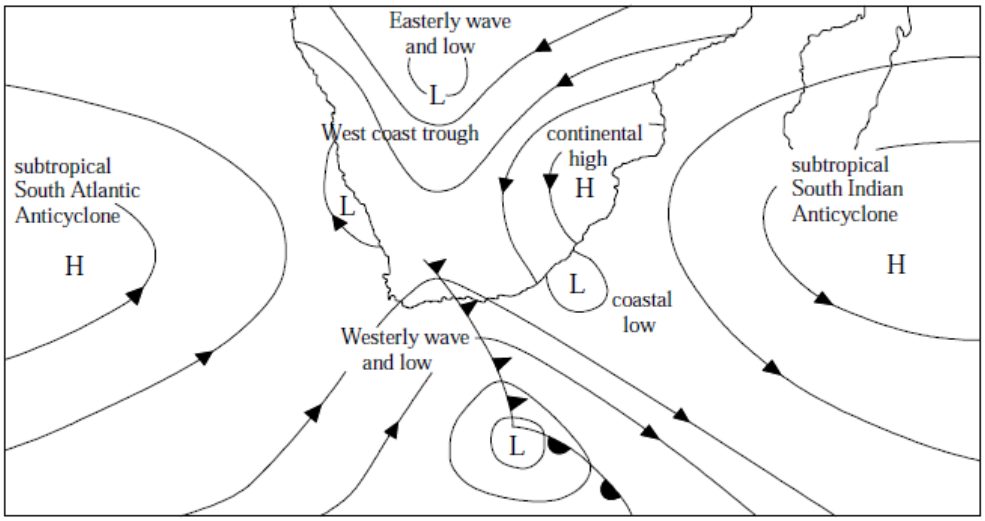


Figure 3.6: Common features of surface atmospheric circulation over southern Africa (after Preston-Whtye and Tyson, 1988).

3.3.2 Solar radiation and temperature

The region receives on average 70 to 80% of possible solar radiation as days are relatively clear (Tyson *et al.*, 1976; Lesotho Meteorological Services, 2008). However, the usually sudden onset of cloudy spring conditions drops the possible solar radiation to between 50 and 60%. Solar radiation is estimated between 5,700J/m^2 and 7,700J/m^2, and is therefore not a constraint to photosynthesis. In the highlands, the temperature remains cool throughout the year due to the effects of altitude (Schmitz and Rooyani, 1984). The mean annual air temperatures (MAAT) range from 7^{o}C in the highlands to 15.2^{o}C in the lowlands (Lesotho Meteorological Services, 2008). According to Nel and Sumner (2008), Sani Pass in the study region had a MAAT of 5.8^{o}C between 2001 and 2005 (Table 3.2).

Freezing temperatures are common in winter (between May and August), with 65 freezing cycles experienced during winter 2003 (1 May to the 30 July) and 76 during 2004 (1 May to the 31 August) (Letšela *et al.*, 2003; Nel and Sumner, 2008). Ground freezing occurs on approximately 200 days per year (Grab, 2002) but is absent during the summer months (Nel and Sumner, 2008). Soil freezing is mostly confined to the upper 15cm where shrub and grass roots are located. Smaller soil temperature ranges occur when covered with vegetation at Sani Top (Nel and Sumner, 2008). In the past, instances of cold damage to crops have been reported throughout the year (Showers, 2005), and in conjunction with associated landform development, have led to the area being classified as a marginal periglacial region (Nel and Sumner, 2008). Climatic graphs determining the main vegetation regions (Lesotho highland basalt grassland and Drakensberg afroalpine heathland) covering the study region can be seen in Figure 3.7 (Mucina *et al.*, 2006).

Table 3.2: Table showing the air and soil temperatures at a depth of 2.5cm at Sani Pass between 2001 and 2004 (adapted from Nel and Sumner, 2008).

	Summer 2001/2 (1 Nov to 17 Mar)	**Summer 2002/3 (1 Nov to 31 Mar)**	**Winter 2003 (1 May to 10 Jul)**	**Winter 2004 (1 May to 31 Aug)**
Mean air Temp.	6.7	6.6	-2.9	-2.1
Mean soil Temp.	11.9	10.6	1.5	0.7
Min. air Temp.	-1.6	-5.6	-11.0	-11.3
Max. air Temp.	20.8	21.0	13.2	12.2
Min. soil Temp.	4.0	1.4	-2.1	-3.0
Max. soil Temp.	24.3	23.8	13.5	12.5
No. of freezing cycles in air	4	6	65	76
No. of freezing cycles in soil	0	0	40	48
No. of frost days in air	0	1	6	17
No. of frost days in soil	0	0	5	0

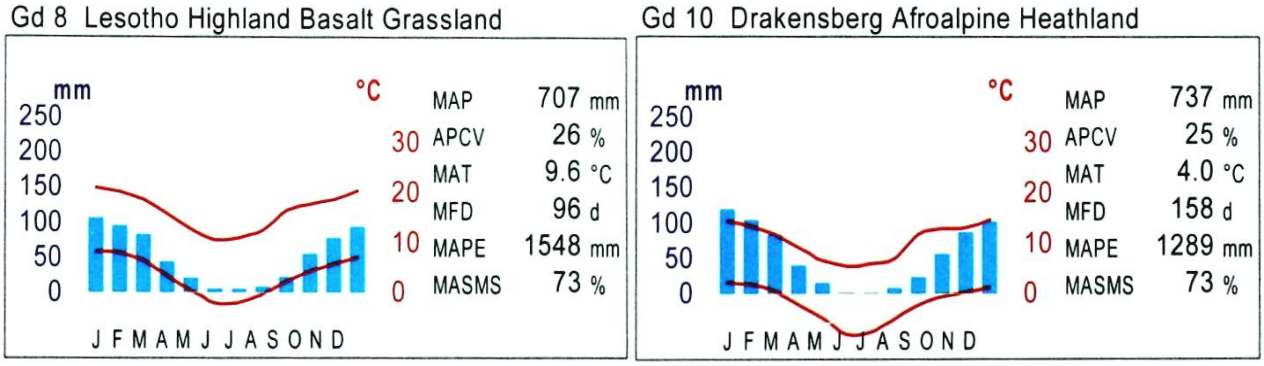


Figure 3.7: Climate of the Drakensberg grassland regions of Lesotho Highland Basalt Grassland (Gd 8) and Drakensberg Afoalpine Heathland (Gd 10). Red lines show the mean daily maximum and the mean daily minimum, and the blue bars show the mean monthly precipitation (Mucina *et al*., 2006).

3.3.3 Wind

Winds experienced within the study region are normally local unless pressure gradients of the large scale pressure systems are strong. Diurnally, the regions along the escarpment edge may be affected by the warm plain-mountain winds by day and the cool mountain-plain winds by night (Tyson *et al.*, 1979; Preston-Whyte and Tyson, 1988). The pronounced escarpment significantly influences and controls the flow of air from coastal regions into the interior plateau. The wind is dominantly from between the north and west during winter (predominantly northwesterly) and north and east during summer (Sene *et al.*, 1998; Grab, 2010). The dominance of large scale pressure systems usually occurs during the passage of mid-latitude depressions. Ahead of cold fronts, the wind is generally warm and northwesterly, whilst behind cold fronts, they tend to be strong southwesterlies (Tyson *et al.*, 1979). Wind speeds are low in summer and highest in winter (Grab, 2002). The strongest wind speeds (mean of $6ms^{-1}$ to $8ms^{-1}$) have been observed between August and October (Grab, 2010). Additionally, wind gusts may reach $36.8ms^{-1}$, as recorded in April 2001 (Grab, 2010). Winds generally prevent the accumulation of atmospheric moisture, resulting in low humidity (Guillarmod, 1969).

3.3.4 Precipitation

Rainfall predominantly occurs between October and April (Killick, 1978b; Lesotho Meteorological Services, 2008), with 83% falling during these months (Leipzig, 1996). The wettest period is from January to March and the driest period occurs in June and July (Nel and Sumner, 2008). However, precipitation is highly variable both temporally and spatially. The variability may be partially explained by the influence of the El Niño/Southern Oscillation as well as the

Southern Oscillation Index influencing preceding rainfall (Nel, 2009). Precipitation may occur at any time of the year, resulting in leached nutrient poor soils (Mucina *et al.*, 2006). Summer rainfall is mainly the product of line storms or orographic storms, while winter precipitation is mainly a consequence of mid-latitude depressions passing over KwaZulu-Natal (Guillarmod, 1971; Nel and Sumner, 2006; 2007; 2008). Both thunderstorms and orographic uplift produce rainfall which primarily occurs in the late afternoon (Tyson *et al.*, 1979; Sene *et al.*, 1998) (Figures 3.8 and 3.9). Afternoons along the escarpment are typically characterized by cloud spreading as mist towards the west (Hughes and Hughes, 1992; Pomela *et al.*, 2000). Previously, rainfall for the study region was estimated to be between 1000mm and 2000mm per annum (e.g. Tyson *et al.*, 1979), yet more recent records suggest that although rainfall is high (mean annual rainfall of between 600mm and 1000mm; Mucina *et al.*, 2006; Bureau of Statistics, 2012), it rarely reaches the original figures (Nel and Sumner, 2008). The topography also influences the amount of rainfall recorded across the region, with the highlands receiving more rainfall than the lowlands (Nel and Sumner, 2008). It is estimated that for every 100m rise in altitude, precipitation increases by ca. 42mm (Nel and Sumner, 2006). Despite this, erosive storms account for only 8% of the summer rainfall, yet this figure increases with decreasing altitude, suggesting more intense precipitation at lower altitudes (Nel and Sumner, 2007). The east-facing slopes also receive more precipitation than west-facing slopes due to a rainshadow effect, especially over the central regions of Lesotho (Sene *et al*, 1998). However, in the past decade the Sani pass region has experienced the drying up of usually perennial streams, possibly a consequence of drought, water extraction, or even through the water draining actions of ice rat burrows (Grab and Nüsser, 2001).

Although hail is a frequent occurrence during the summer months as a result of updrafts (Dempster and Richard, 1973), the intense hail events are usually of short duration (Tyson *et al.*, 1979). Freezing temperatures are common during winter months and lead to ca. 111 frost risk days in the lowlands and ca. 276 days in the highlands (Lesotho Meteorological Services, 2008). On the Lesotho plateau, the first frost events usually occur in mid to late April (Tyson *et al.*, 1979). Similarly, the last frosts are generally experienced in October and extend towards the end of the year with increasing altitude (Tyson *et al.*, 1979). Both needle ice and frost are nearly daily occurrences between May and August (Grab, 2002). Snowfalls generally occur between May and September, but melts after a few days (Guillarmod, 1971). Yet in previous decades snow was known to lie for several months on the south-facing slopes in the highlands (Guillarmod, 1969; Dempster and Richard, 1973). Although on average it may snow eight times

per annum (Sene *et al.*, 1998), the contribution of snow to the water table is unknown (Nel and Sumner, 2008). Despite the relatively high precipitation, the relative humidity generally remains low (<30%) as a result of rocks covering soils, shadows and low temperatures limiting evaporation (Killick, 1978b; Crawford, 2008). The low relative humity is also a result of the dominant high pressure belt occurring over the central regions of South Africa, and is particularly dominant during the winter months (Tyson and Preston-Whyte, 2000).

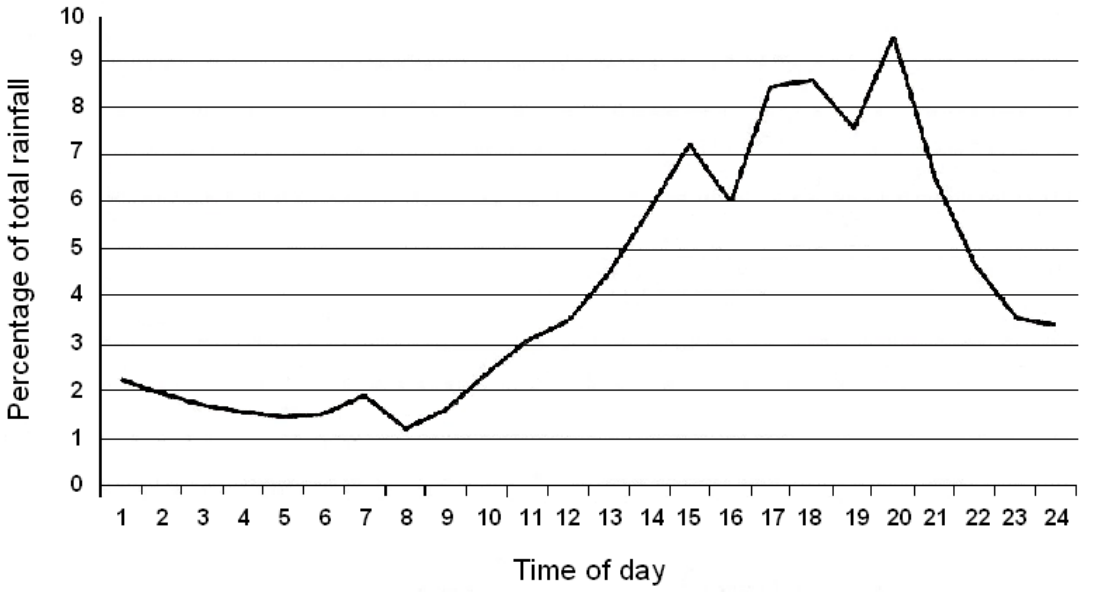


Figure 3.8: Timing of the daily rainfall for Sani Pass (adapted from Nel and Sumner, 2008).

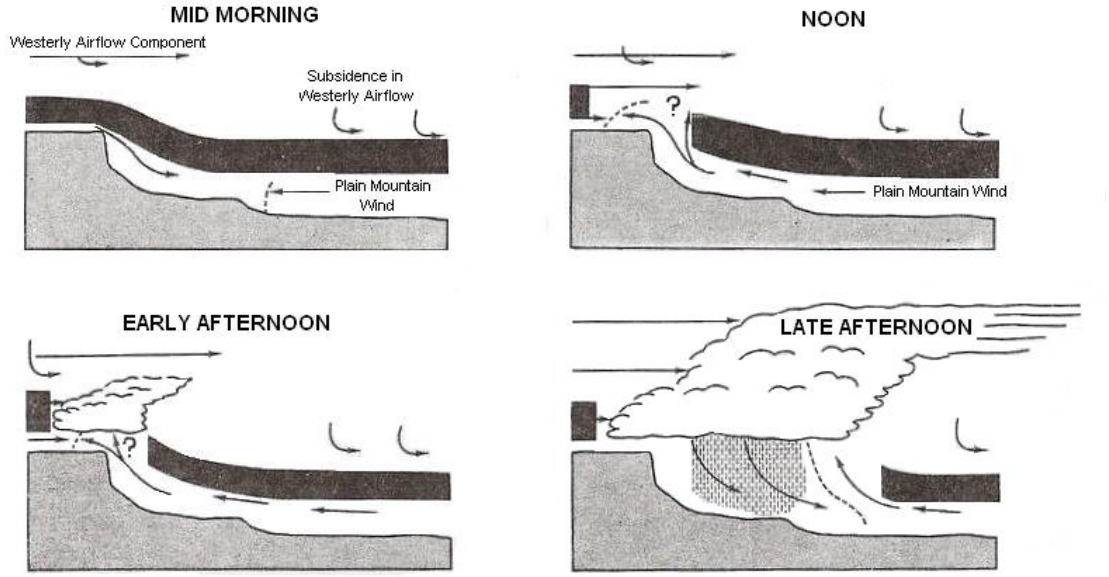


Figure 3.9: The stages of development of oragraphically-induced thunderstorms over the escarpment. The dark layer indicates stable discontinuities in the atmosphere (after Tyson *et al.*, 1976).

3.4 Geology and Soils

3.4.1 Geology

During the late Paleozoic and Mesozoic eras, the area known today as Lesotho was part of the large landmass, Gondwanaland (Partridge, 1998). Centrally located, the area was composed of Karoo strata including the Ecca and Beaufort groups (Partridge, 1998) (Figure 3.10). During this period, the landmass was at a high elevation and many areas were exposed to erosion, yet the Karoo Basin was relatively protected (Partridge, 1998). Today, Lesotho lies between the central and eastern regions of the Karoo Basin and on top of the Stormberg Group (Schmitz and Rooyani, 1987). The Stormberg group was formed during the last stages of the Cape Fold Belt formation between the late Permian and early Jurassic, and reflects a climatic change to a more arid environment (Schmitz and Rooyani, 1989; Partridge, 1998; McCarthy and Rubidge, 2005).

The Stormberg Group forms the basis of Lesotho and is composed of the Molteno, Elliot and Clarens formations, with the Molteno strata being the oldest (Schmitz and Rooyani, 1989; Marake *et al.*, 1998). When the compression force, which lasted throughout the deposition of the Karoo Supergroup, relaxed, sedimentation ceased approximately 182 million years ago (McCarthy and Rubidge, 2005). The crust ruptured releasing lava and marked the beginning of the fragmentation of Gondwana (McCarthy and Rubidge, 2005). During the early Jurassic or late Triassic, magma flowed up through fissures and into sedimentary strata, spreading over the Karoo Basin and forming the Lesotho Formation and Drakensberg Group (Schmitz and Rooyani, 1989). Conical volcanoes were rare and for the most part lava flowed from long fissures in the crust (McCarthy and Rubidge, 2005). The lava flows covered the Clarens Formation in layers with varying thickness from 0.5m to greater than 50m (Pomela *et al.*, 2000). Horizontal lava flows, used by the magma to reach the surface in the form of dolerite sills and dykes, are common, and thus many hillsides are stepped (Chakela, 1981; Schmitz and Rooyani, 1987; Grab *et al*., 2005).Three major and several minor lava flows contributed to the structure of the Lesotho Formation, which attained a maximum thickness of approximately 1600m (Guillarmod, 1969; Schmitz and Rooyani, 1987; Marake *et al.*, 1998; McCarthy and Rubidge, 2005).

The Drakensberg forms the edge of the escarpment, a feature whose origin has been much debated in the past and continues to be debated. An early theory as proposed by Suess (1904), claimed that the escarpment was the result of uplift along a major fault, but was later disproved.

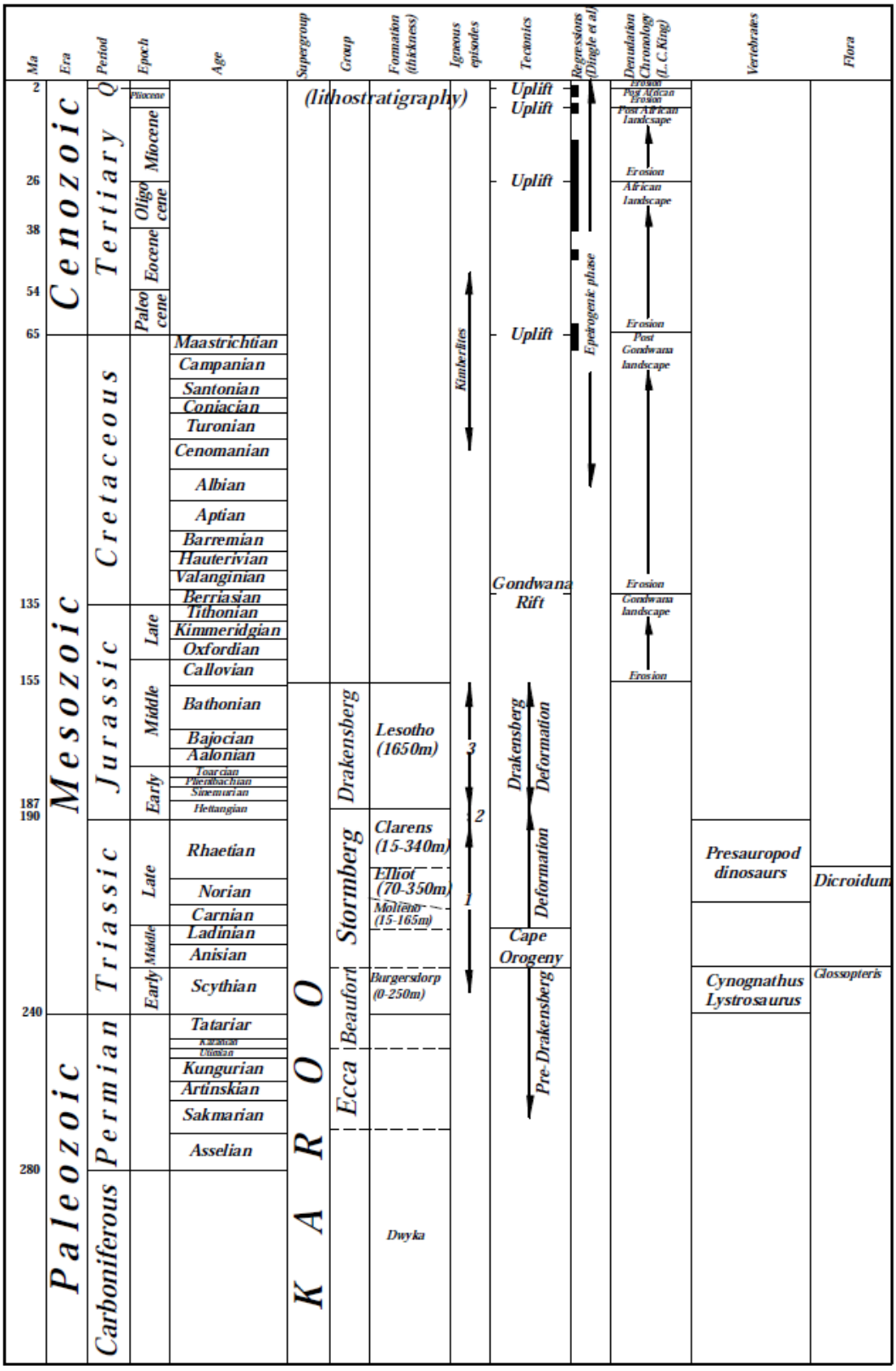


Figure 3.10: Geological time scale (after Schmitz and Rooyani, 1987).

Following this, King (1941; 1944; 1947) hypothesized that erosion along the rifted edge of the newly formed southern African coast (Natal coastal monocline) was the cause, and evidence supporting this was provided by Partridge and Maud (1987). Erosion was believed to have occurred after at least five uplifting events over the past 135 million years, the last of which occurred ca. 2 million years ago (Pomela *et al.*, 2000; McCarthy and Rubidge, 2005). The periods of uplift suggest the Drakensberg mountains would be present today even if volcanic activity had not occurred (King, 1953), and since then the erosion rate has been further debated (e.g. Flemming *et al*., 1999; van der Beek *et al.*, 2002; Kounov *et al.*, 2007). Cosmogenic isotopes were used to show that if a uniform rate of erosion was responsible it would be too slow for the escarpment to move from the coastal region to its current position (Flemming *et al*., 1999). This finding was supported by a conceptual numerical model demonstrating minimal escarpment retreat (van der Beek *et al.*, 2002). Variations in the erosion rate have been suggested in the form of erosion cycles and uplift (e.g. King and King, 1959; Partridge and Maud, 1987; Moore and Blenkinsop, 2006). A decrease in the erosion rate, during the Tertiary era, was suggested to be linked to a climate shift towards more arid conditions (Partridge, 1998). Inland-facing scarps imply that erosion as a result of continental rifting is not the only process occurring (Moore and Blenkinsop, 2006). A drainage divide formed as the result of rifting and which occupies the position of the current escarpment, has been suggested (Flemming *et al.*, 1999; Brown *et al.*, 2002; van der Beek *et al.*, 2002; Moore and Blenkinskop, 2006). However, lithological controls are thought to have limited erosion, which would have been compounded by aridity during the Cretaceous, uplift along the C-S axis and the vegetative influence, which would have influenced the current location of the escarpment (Moore and Blenkinskop, 2006). The Lesotho Formation previously covered all of Lesotho but has since been dominated by uplift and erosion which is exposing the underlying strata. Both erosion and weathering have created a landscape which is under constant change (Marneweck and Grundling, 2003). Today the remaining divided and eroded basalt plateaus of the Drakensberg Group have formed the Maluti and Drakensberg ranges (McCarthy and Rubidge, 2005) (Figure 3.11).

3.4.2 Soils

The soils of an area are representative of the underlying geology and thus vary with rock type, aspect, altitude and vegetation. The highland soils generally reflect the basalt parent material (Grab, 2002). The soils are formed from the parent material through weathering colluviation and eluviation (Pomela *et al.*, 2000; Schmitz and Rooyani, 1989). Lesotho has 26 soil types of which 80% are located above 2000m a.s.l. (Schmitz and Rooyani, 1978) (Table 3.3). The soils of the

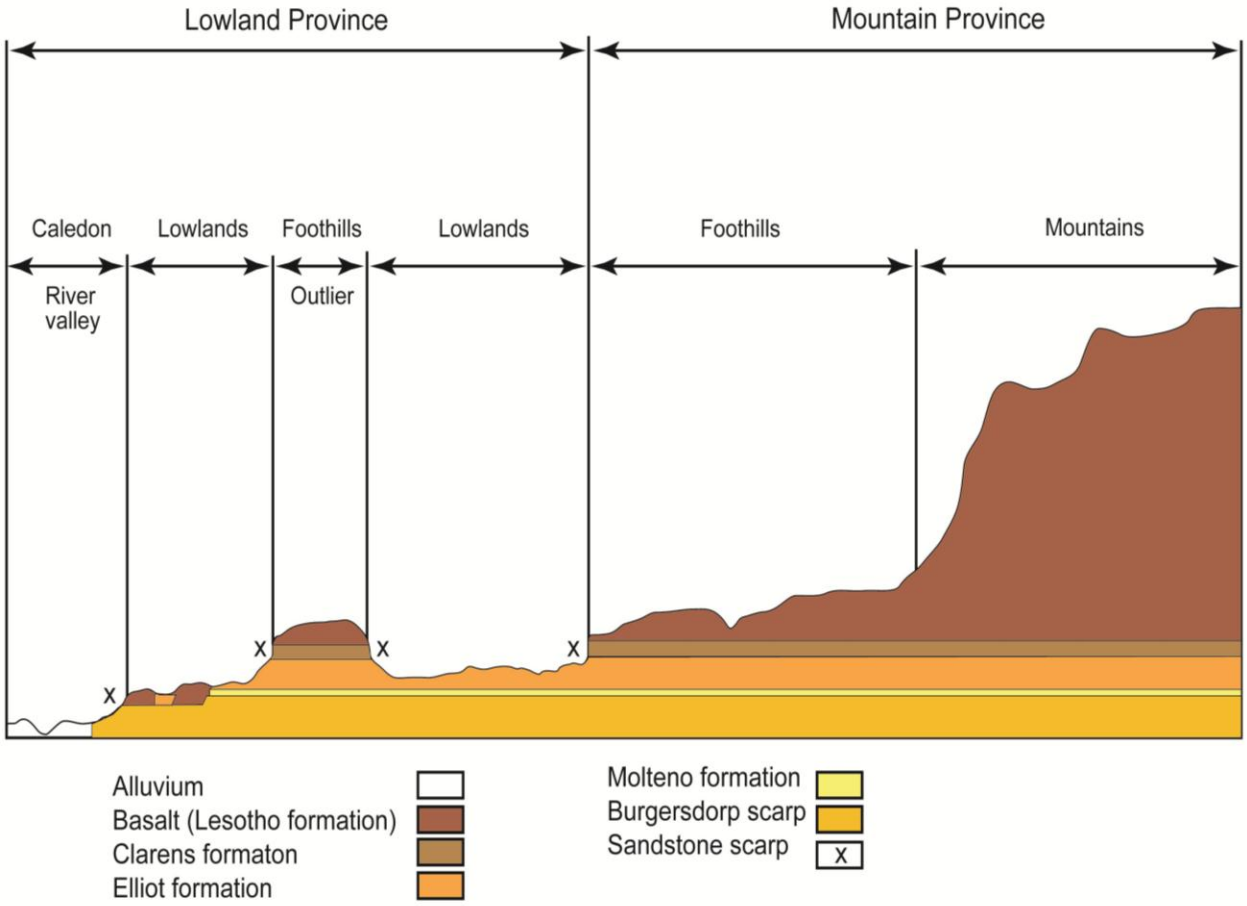


Figure 3.11: Simplified diagram depicting the geology of Lesotho (after Binnie and Partners, 1972; Schmitz and Rooyani, 1989).

(Sani region are classified as Mollisols, and are "soils with a deep, dark friable (easily broken up) surface horizon that has formed under grassland vegetation (Hausenbuiller, 1978). These generally dark coloured soils produce a relatively thick topsoil (Schmitz and Rooyani, 1989).

These soils are considered to be rich with much undecomposed organic material (Bond *et al.*, 2004). Sediment accumulation at the base of some north-facing hollows is composed of peat (<80%), dark rich clay, gravel and varying quantity of stones (Marker, 1994). The Lesotho Formation has a low permeability, thus the soils are seasonally moist with a high stream and spring density (Schmitz and Rooyani, 1989). The accumulation of soils in valley bottoms results in an increasing soil depth towards the valley floor (Schmitz, 1984; Morris *et al.*, 1989; Leipzig, 1996). Slopes are observed to have thin soils (<50cm) while the valley bottoms have deeper soils (>50cm) (Grab, 2002).

Table 3.3: The geomorphological units and associated soil series (after Schmitz and Rooyani, 1987).

Geology	Geomorphological Unit	Associated Soil Series
Lesotho Formation	Steep and middle slopes	Popa and Matana
	Accumulation glacis	Fusi and Thabana
	Planation surface	Machache, Nkau, Sefikeng, Tumo, Mataba, Seforong, Ralebese and Matela
	Alluvial deposits	Phechela, Khabo, Sofonia and Maseru dark
Clarens Formation	Structural plateau	Matela, Berea, Ntsi, Qalaheng, Thoteng and Theko
	High structural plateau	Lekhalong, Tenola and Sani
Bergersdorp, Molteno and Elliot Formations	Accumulation glacis	Maliehe, Bosiu, Majara, Moshoeshoe, Tsiki, Sephula, Takholo and Maseru
	Planation surface	Leribe, Matela, Qalo, Hololo and Rama
	Alluvial deposits; high terrace	Mataba, Seforong, Ralabese, Kubu and Khabo
	Alluvial deposits; middle terrace	Khabo, Kubu, Bela, Phechela, Maseru and Maseru dark
	Alluvial deposits; low terrace and flood plain	Caledon, Soforina, Kolonyama and Phachela
	Dolerite dykes and sills	Ralebese

3.5 Vegetation

Alpine climates have resulted in plant cover which is highly adapted to stress tolerance through low palatability, long lifespans, slow growth and effective resource use (Crawford, 2008). Added to the isolation typical of alpine areas, endemism is high. Lesotho vegetation has a high level of diversity as alpine areas are known for their endemism as well as the role they play in speciation (Leipzig, 1996).The Drakensberg Alpine Center (DAC) extends across Lesotho and includes the Eastern Cape Drakensberg (areas above 1800m a.s.l.) (Carbutt and Edwards, 2006; Figure 3.12). The DAC contains over 2800 specific and intraspecific species of which currently 16% are endemic (Carbutt and Edwards, 2004). However, much of the diversity is being lost through forest removal, collection of medicinal plants and for firewood, replacement of crops with high yield crops (Leipzig, 1996), erosion, over grazing and over burning. Overgrazing has been observed for many decades and the area affected is increasing due to population pressure and a

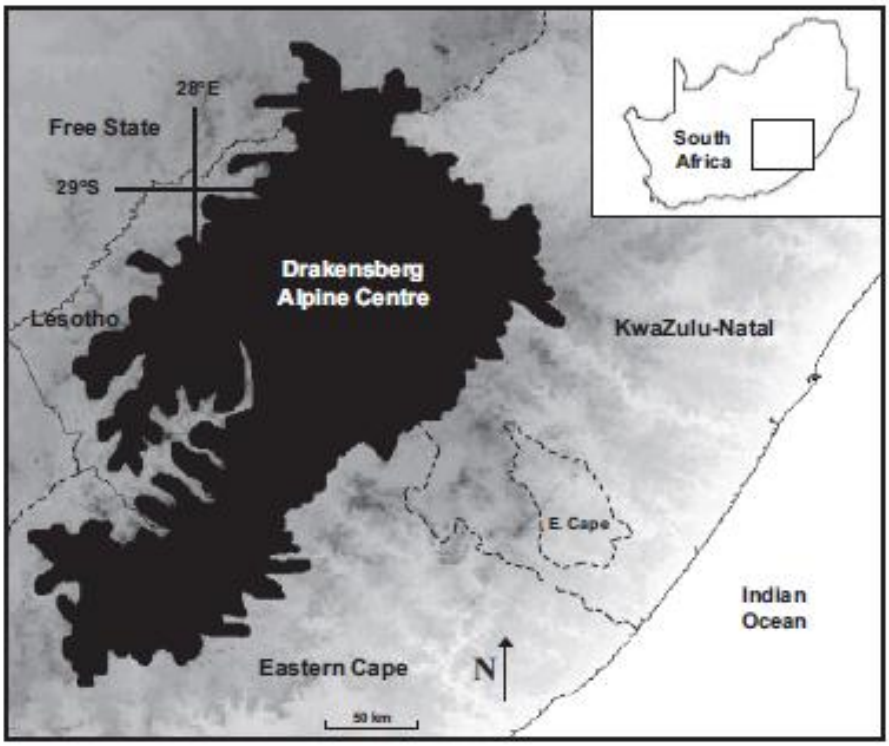


Figure 3.12: Diagram showing the Drakensberg Alpine Center (after Carbutt and Edwards, 2006).

shortage of suitable forage (van Zinderen Bakker, 1981). Despite this, only 0.21% of Lesotho is legally protected, and the opinion is that all the biodiversity should be considered under threat (Pomela *et al.*, 2000; Carbutt *et al.*, 2011). More recent statistics confirm the degradation of the environment by labeling 54% of Lesotho as critically endangered and a further 25% listed as endangered (Bureau of Statistics, 2012). Nevertheless, the alpine regions of Lesotho are still considered 'relatively intact,' while the lowlands are threatened and degraded (McGinley, 2007). Yet, erosion and the encroachment of Karoid species are becoming more evident in the alpine belt. In 1984, the range inventory described the mountains as having areas in good condition and areas in which overgrazing is evident (Martin, 1984). By 2009, the Drakensberg montane grasslands were described as being critically endangered (Gibson, 2009). The deterioration in vegetation cover has been recognized as the consequence of a lack of knowledge and information used in the formation of management policies (Motsamai, 1995). The lack of enforcement and monitoring ability at both institutional and community level has also been blamed.

3.5.1 Vegetation classification

The vegetation of Lesotho is generally classified as grassland with scattered pockets of forest (Ranthamane, 2008). The plant communities occupy specific topographic locations which best suit their needs (Morris *et al.*, 1993). The tree line is observed to be at 2300m a.s.l. on the eastern

side of Lesotho (van Zinderen Bakker, 1981), and the study area is thus above the treeline (Killick, 1978). The area was originally classified as Themeda-Festuca alpine veld by Acocks (1975), but more recently has been classified as alti mountain grassland (Granger and Bredenkamp, 1996). The alti mountain grassland vegetation type covers 7 118km^2 in Lesotho and 119 188km^2 in southern Africa, of which only 1.08% is conserved in Lesotho. Alti mountain grassland occurs above the tree-line between 2500m and 3480m a.s.l., and consists mainly of tussock grasses (Granger and Bredenkamp, 1996). The vegetation in the Sani region has more recently been classified into Lesotho highland basalt grassland and Drakensberg afroalpine heathland (Mucina *et al.*, 2006) (Figure 3.13). Lesotho highland basalt grassland occurs over the majority of the basalt plateau and is generally found above 2900m a.s.l. The vegetation is not significantly influenced by soil type, but rather is strongly influenced by slope orientation and altitude (Carbutt and Edwards, 2004).

3.5.2 Wetlands

The basalt parent rock of the Leotho highlands prevents much infiltration, thus allowing water to remain within the soil for long periods (Bond *et al.*, 2004). Where water collects, wetlands develop in depressions, on mid-slopes, in valley heads, at valley bottoms, near channels and on footslopes (Bainbridge and Marneweck, 1996). Wetlands occur in the form of bogs and fens, and many contain hummocks. Pools are also a common site, in the study region in the summer months, many containing floating or submersed vegetation (Killick, 1978). Although no link has been observed between wetland type and aspect (Marneweck and Grundling, 2000), the number of C_3 grasses in wetlands increases whilst C_4 grasses decrease with increasing altitude (Sieben *et al.*, 2009). This was attributed to a decrease in the photosynthetic ability of C_4 grasses at altitudes above 2 400m a.s.l. Wetlands occupy a considerable area of the Lesotho Highlands and have their own unique vegetation. Most contain algae, bryophytes and other small mat-forming plants (Marake *et al.*, 1998; Marneweck and Grundling, 2000; Marneweck and Grundling, 2003). Short sedges and mixed sedge and grass species are common within shallow areas and around the edges of deeper mires. Wetlands are a valuable resource to Lesotho, as they have an important role in the hydrological system (purification and regulation of flow) and provide a high proportion of the grazing resources (Nüsser, 2002a). Locally, soggy areas usually found in valley bottoms are termed '*mokhoabong*' and are commonly designated for grazing. The wetlands have a high biomass and a carrying capacity 2.5 times higher than the surrounding grasslands (Nüsser, 2002a). However, many of the wetlands are now degraded due to overgrazing, gully erosion and

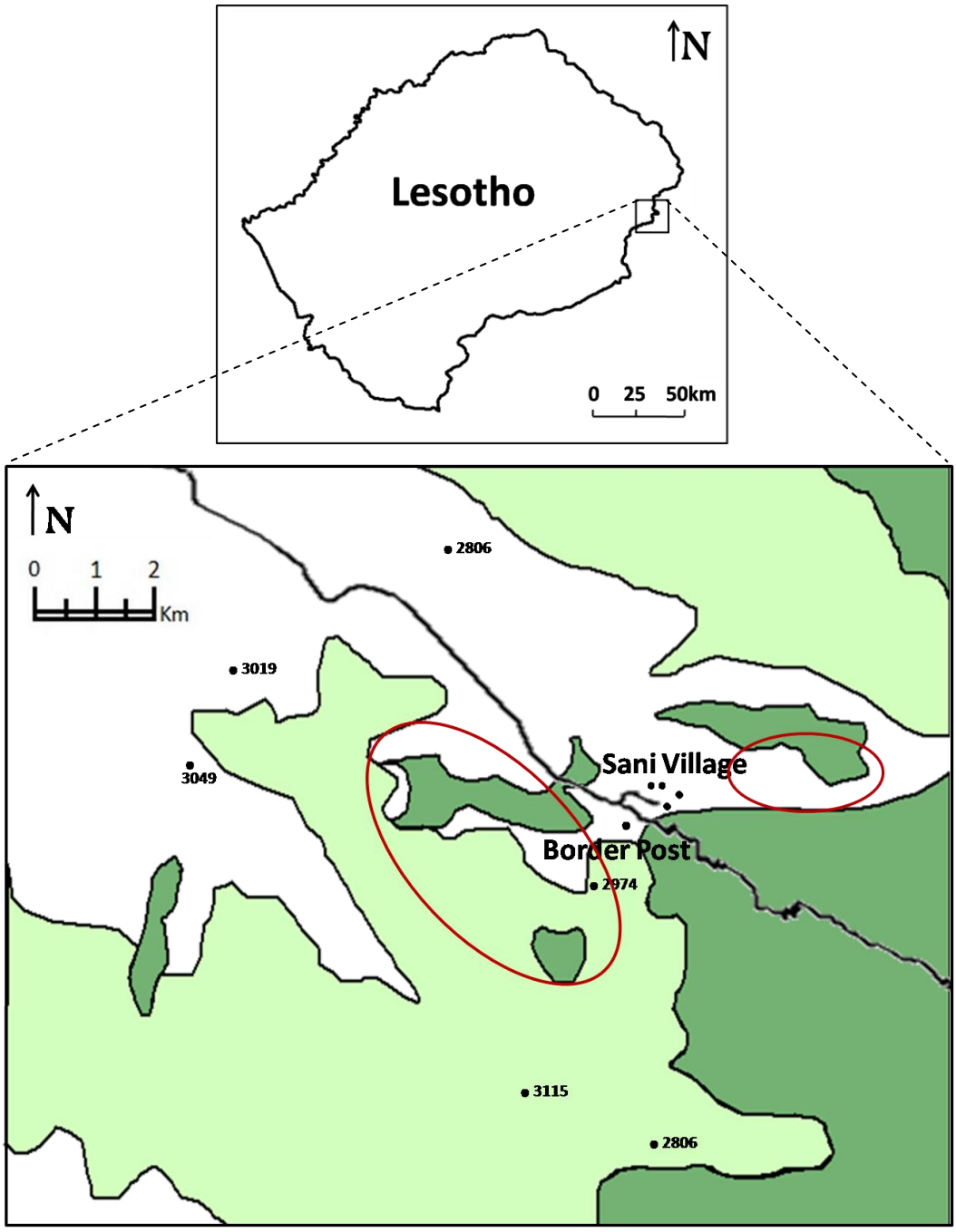


Figure 3.13: Vegetation of the study region (areas circled in red): white representing Lesotho Highland Basalt Grassland, light green representing Drakensberg Afroalpine Heathland and dark green representing uKhahlamba Basalt Grassland (after Mucina *et al*., 2006).

invasive plants (Grab and Deschamps, 2004; Du Preez and Brown, 2011). According to Schwabe (1995), 49% of observed mires contain species which are indicative of disturbance. The destruction of wetlands reduces the flow maintenance, erosion control, filtering and storage capacity of the Lesotho landscape (Marneweck and Grundling, 2000). Additionally, biomass may decline and the composition of species may become altered (Du Preez and Brown, 2011).

3.5.3 Vegetation in the Sani Pass region

Themeda triandra dominates the lower to middle altitudes whilst *Festuca caprina* dominates the higher altitudes, with *Merxmuellera macowanii* frequenting drainage patterns (Mucina *et al.*, 2006). Drakensberg afroalpine heathland is also found above 2900m a.s.l. in the northeastern areas of Lesotho, although good examples occur in the study region (Mucina *et al.*, 2006) (Figure 3.13). The vegetation found within this classification is dominated by *Merxmuellera disticha*, with *Merxmuellera drakensbergensis* appearing along water courses (Mucina *et al.*, 2006). However, throughout the study region, overgrazing, burning and low rainfall has led to the influx of Karoo vegetation (Dempster and Richard, 1973; Granger and Bredenkamp, 1996; McGinley, 2007). However, rainfall trends along the escarpment edge and Drakensberg region are highly seasonal, with decreases observed in autumn rainfall during the latter half of the twentieth centuary (Nel, 2009). Thus, low rainfall may not be responsible for the influx of Karoid vegetation, rather earlier onset of the dry season may be partially responsible.

3.5.4 *Merxmuellera drakensbergensis*

Tussock grasses have been selected as the focus of this study as these are targeted for burning, due to the ease with which they burn during the winter months and a relatively long smoldering duration, additionally individuals are easily identifiable. Many of the tussock grasses found in the Drakensberg and Maluti mountains are xerophytic, as can be seen from the tightly convoluted leaves of the *Merxmuellera* species (Bews, 1918) (Figure 3.14). Tussocks have the ability to resprout, despite both burning and grazing, resulting in a long lifespan. The same species, *Merxmuellera drakensbergensis*, will be used in all of the samples due to different patterns and rates of growth amongst different species, as well as different abilities to cope with fire. The species belong to the Poaceae family and have a C_3 photosynthetic pathway (Hoare, 2008). This indicates that grass growth commences as soon as soil temperatures are above freezing (Trlica, 1999). Species from the *Merxmuellera* genus are perennial tussock grasses found in warm regions, however, many are restricted to alpine environments (Schmitz, 1984). They are representative of the Cape Floral Center and have a secondary floral center within the DAC (Carbutt and Edwards, 2001).

M. drakensbergensis is commonly referred to as Molala-hlolo amongst the local Sotho people, “broom-grass” in English (Letšela *et al.*, 2003), or Drakensberg Danthonia (Granger and Bredenkamp, 1996). However, other local names used include ‘moseha’, ‘moseea’, ‘mosea’ and

'mohlaba-pere' (Schmitz, 1984). This species forms rigid vertical tussocks with bristly and involute or canaliculate leaves tapering to a sharp apex (Ellis, 1981). The large wiry tufts measure up to 1m high during flowering in December and January (Schmitz, 1984; Killick, 1990). The leaves are distinctive as they split and curve directly above the ligule (Killick, 1978; Ellis, 1981; Killick, 1990) and are present unless recently burnt (Ellis, 1981). Despite its unpalatable nature, it is frequently burned to produce more palatable shoots (MDTP, 2007). At the study sites, this species was distinguished from similar species (*M. disticha*, *M. stereophylla* and *M. macowanii*) through the arrangement of hair tufts along the lemma (Figure 3.15). These tussock grasses have a large carryover of organic matter between consecutive years. It is harvested by local Basotho in the Sani region, primarily for handcrafts such as baskets, brooms, hats, ropes and mats (Letšela *et al.*, 2003). This species is also widely used in the highlands as thatching grass for huts and grazing posts (Nüsser, 2002b). In many areas of Lesotho, *M. drakensbergensis* is sold to a manufacturer and consumer as an export commodity, providing valuable income (Maliehe, 1997). Under the legal notice 36 of 1969, *Merxmuellera* species are not listed as being protected (Leipzig, 1996; Pomela *et al.*, 2000). A study undertaken by Schwabe (1995) demonstrated, through the use of aerial photography, that marginal wetland vegetation was meager and in many places entire communities of *Merxmuellera drakensbergensis* had vanished. However, no further research has examined the state of *M. drakensbergensis* in the Drakensberg/Maluti Alpine Center.

Although several authors (Killick, 1978; Ellis, 1981; Schmitz, 1984; Hughes and Hughes, 1992) have observed that *Merxmuellera* frequently grows along stream banks, more often it grows on basalt outcrops rather than along stream banks (Killick, 1990; Morris *et al.*, 1989). Others have suggested that tussock grasses prefer deeper soils than the thin regolith covered basalt outcrops (Bainbridge and Marneweck, 1996; Maliehe, 1997). In the study area, *Merxmuellera* is predominantly found in seasonally waterlogged areas or dry soils, rather than permanently wet soils (Guillarmod, 1971; Bainbridge and Marneweck, 1996), and is observed in the study region where the tussocks occur in bogs rather than in the lower lying, flatter fens (Hughes and Hughes, 1992; Schwabe, 1995). Alternatively, *Merxmuellera* may grow in areas surrounding the edges of waterlogged soils (Nüsser, 2002b). Yet, others have observed that *M. drakensbergensis* tufts often grow along the edges of bogs and may invade the center of bogs below the seepage line, growing over a layer of black peat (Hughes and Hughes, 1992; Marneweck and Grundling, 2003). Through observations, it is thought that this species has a wider ecological tolerance in the Sani rregion than elsewhere (Killick, 1978; Ellis, 1981).



Figure 3.14: The structure of *Merxmuellera drakensbergensis* (Schweick) Conert. (after Schmitz, 1984).



Figure 3.15: The lemma of *Merxmuellera drakensbergensis* (Schweick) Conert. (after Schmitz, 1984).

Merxmuellera disticha, a very similar species, has been described as being an unpalatable species, but which may dominate and exclude other grass species such as *Themeda triandra* (Acocks, 1971). This could suggest that *M. drakensbergensis* may act in a similar manner. *M. drakensbergensis* was assigned a low proper use factor (PUF) of 0.15 in 1986 by the Land Conservation and Range Development Project (Morris *et al.*, 1989). Thus, despite having a high availability (Table 3.4), *M. drakensbergensis* is a relatively unpalatable grass with a low consumption potential (Morris *et al.*, 1989). *M. drakensbergensis* is termed a 'weak increaser' as it increases in abundance under increasing grazing intensity but decreases under heavy grazing intensity (Morris *et al.*, 1989). *M. drakensbergensis* is believed to alter its abundance with minor changes in moisture or fertility (Morris *et al.*, 1989). Despite extended searches, existing growth rate data could not be found for *M. drakensbergensis*.

3.5.5 Growing season

Photosynthesis is the conversion of carbon dioxide and water by sunlight to glucose and oxygen. Thus, growth is merely the conversion of carbon dioxide, water and soil nutrients into plant matter (Everson and Everson, 1987). High UV radiation, strong winds, large temperature ranges, cool conditions, shallow soils and low atmospheric water vapour are characteristics of alpine ecosystems and act to reduce both the rate of growth and the growing season (Letšela *et al.*, 2003; Crawford, 2008). Added to this, the amount of available sunlight, water and temperature vary throughout the year, restricting growth to only the most favourable conditions. In alpine regions, growing seasons are typically unpredictable, cold and short (Billings, 1974), whilst growth in Lesotho is generally restricted by temperature and available moisture (Showers, 1989). Although it has been suggested that once sufficient rain has fallen to soften the soil, growth can become accelerated (Letšela *et al.*, 2003), soil moisture does not correlate to growth in the lower montane grasslands at 1800m a.s.l. (Everson and Everson, 1987). In Lesotho, growth may also be limited by the low minimum temperatures at night, or through snow (Morris *et al.*, 1989). Low temperatures may limit microbal action within the soil, thus decreasing the availability of nutrients for growth (Blank *et al.*, 2007). Thus, moisture and temperature limit growth and plants must thus adapt to such stresses (Showers, 1989). A correlation was found between growth, soil temperatures and air temperatures in what was termed the Natal Highland Sourveld (Everson and Everson, 1987), suggesting that growth may therefore be limited by temperature. Slope orientation may also have an impact on vegetation, with north-facing slopes experiencing a slightly longer growing season (Morris *et al.*, 1989; Carbutt and Edwards, 2004). The growing

Table 3.4: Usable forage and stocking rates for vegetation types occurring within the alpine zone (after Morris *et al.*, 1989).

Vegetation Type Name	Average Total Usable Forage kg/ha/annum	Range kg/ha/annum	Mean Stocking Rate for the Growing Season (6 Months) AU/ha	Ha/AU
Hyparrhenia + *Cymbopogon* + associated species	313.2	163 - 464	0.2	5.7
Eragrostis + *Aristida* Species	772.7	159 - 2093	0.4	2.3
Themeda triandra and *Elionurus muticus*	1335.9	810 - 1933	0.7	1.3
Festuca caprina + other Grasslands	715.6	365 - 1291	0.4	2.5
Chrysocoma and Artemisia Species	337.4	131 - 650	0.2	5.3
Merxmuellera species	1205.6	741 - 2185	0.7	1.5
Limited vegetation, shallow soil + rockland	95.8	60 - 160	0.1	18.8

AU – Animal unit. The animal units in Lesotho are: cattle = 1.0AU, horses = 1.0AU, donkeys = 0.8AU, sheep = 0.2AU and goats = 0.2AU.

season experienced by an individual plant may thus be related to the microclimate and microenvironment in which it is situated. The growing season within the study region extends from October to March (Nüsser, 2002a), between the first rains and the first frost, yet has an unpredictable length (Showers, 1989, Letšela *et al.*, 2003; Carbutt and Edwards, 2004). In winter most of the vegetation undergoes dormancy induced by low temperatures (Morris *et al.*, 1989).

3.5.6 Grazing systems

The people of Lesotho have used the mountainous regions for grazing during the past 100 years, moving herds with seasonal climates and plant growth cycles (Quinlan, 1995b). Developed in the 1920s, the grazing post resource management scheme or transhumance system has been followed by almost all herdsmen (Swallow and Bromley, 1995). Previously, young boys and men were

trusted to move the herds to mountainous grazing posts in December, and returned to lower altitudes in April (Quinlan, 1995b; Nüsser, 2002b). Herding is a culturally important activity as a means of socialization by young men (Quinlan, 1995a). Today, livestock are typically moved from villages (usually below 2 400m a.s.l.) into the mountain grasslands (usually between 2750m and 3100m a.s.l.) during late October and remain until late May (Deschamps, 2006). However, some livestock are overwintered in high mountain cattle posts (Morris *et al*., 1989), a trend which appears to be on the increase (pers. obs.).

Many chiefs practice *leboella*, a rotational system permitting grazing or the use of a resource in one area, and allowing the remaining areas to recover (Quinlan, 1995a; Marake *et al.*, 1998). Failure to obey *leboella* results in a monetary fine, the amount of which depends on the size and quantity of livestock which were allowed to graze (Motsamai, 1995). However, the *leboella* system no longer functions well due to a lack of enforcement, low membership to governing bodies, difficult registration of *leboella* practices, lack of support from chiefs and a 'free for all' approach by non-members (Mdee, 2004). Consequently, grazing records for the study region are not available. The lack of coordinated and planned grazing rotations has led to overstocking and overgrazing with consequent severe soil erosion (Mbata, 2001). The preservation of resources is practiced in the lowlands since the late eighteenth century, and introduced by Paramount Chief Moshoeshoe I in the form of the *maboella* system (Mbata, 2001). *Maboella*, or reserved grazing areas, are areas set aside for the growth of thatching grass, preservation for winter grazing and growth of wood resources, usually in the summer months (Quinlan, 1995a; Swallow and Bromley, 1995; Marake *et al.*, 1998). A *molisa oa lethobo*, or warden, was appointed by the chief to enforce the *maboella* rules (Swallow and Bromley, 1995). However, despite many communities currently actively using this system, it has collapsed in many instances due to increasing population pressure, increasing livestock numbers and lack of traditional authority (Swallow and Bromley, 1995; Marake *et al.*, 1998). Colonial and post-colonial laws and courts reduced the legal support for chiefs and their wardens, alongside which many agencies and committees have attempted to 'modernize' the traditional systems (Swallow and Bromley, 1995). Financial rewards for chiefs and wardens faded out, and without incentive, many stopped enforcing *maboella*. In the 1970s, permits were introduced to control grazing post locations, livestock numbers, grazing times and assist with bans on burning (Swallow and Bromley, 1995). The *maboella* system was replaced in 1986 with the amended Range Management and Grazing Control Regulations (of 1980), which is believed to have weakened the anti-erosion legislation put forward in the Land Husbandary act (1969) (Pomela *et al.*, 2000).The Range Management

and Grazing Control Regulations do not specify the conservation efforts covered and is left open to interpretation (Wizsch and Ambrose (1992). If the traditional Chief and the Village Development Council differ in view with regards to grazing practices, degradation is a possible outcome (Pomela *et al.*, 2000).

Many grazing posts (*motebos*) were used by chiefs as areas of settlement for their growing follower populations, and with time the grazing posts were forced further up into the mountains (Quinlan, 1995a). Since the early 1900s, this led to an increasing decline in grassland conditions in the lowlands, and a movement of more livestock to mountainous regions. In the 1930s, the first ecological survey noted a deterioration of grasses, resulting in the introduction of grazing principles such as controlled stocking rates and rotational grazing (Quinlan, 1995b). However, these were unsuccessful and little attention was given to conservation of the grasslands as degradation was not as visible as within villages (Quinlan, 1995b). The great drought of 1933 highlighted the extent of vegetation change and the need for conservation. Despite observations of increasing livestock numbers and decreasing stock health, little was done to conserve the grasslands (Showers, 2005). Small conservation projects and the national anti-erosion campaign of 1936 were introduced, with only local impacts (Showers, 1989). After independence in 1966, the government continued to be more concerned with livestock production than grassland conservation, whilst the recent government has also focused on livestock production in mountainous regions (Quinlan, 1995b; MDTP, 2007).

Over 80% of the natural vegetation in Lesotho comprises rangelands, making them the focus of countless conservation and management practices (Uys and Lechmere-Oertel, 2006), one of which is the moderately successful Sehlabathebe Grazing Association (Swallow and Bromley, 1995). Established in 1982 with the support of USAID (United States Agency for International Development) and their Land Conservation and Management Project, this system allowed for more communal involvement (Swallow and Bromley, 1995). Range Management Areas (RMAs) were mountain grassland areas above 2750m a.s.l., designated to the grazing of livestock by a group of owners who have formed into a Grazing Association (Quinlan and Morris, 1994; Pomela *et al.*, 2000). RMAs were introduced in the 1980s and were endorsed again in 2002 by the Land Review Commision (Mdee, 2004). Their goal was to provide grazing for livestock, while still taking into account the sustainability and improvement of vegetation cover (Motsamai, 1995; Pomela *et al.*, 2000). The RMAs allowed communities to control their own grasslands, which was accomplished through rotational grazing and reduced livestock numbers

per area (Marake *et al.*, 1998). Rangelands in the highlands of Lesotho were divided into 22 areas controlled by principal chiefs who control access to the grazing areas (Marake *et al.*, 1998). However, the effectiveness, accomplishment and aptitude of chiefs have been questioned by authors such as Swallow and Bromley (1995) and Tshabalala (1995). In many areas the chieftain system is becoming less recognized as leadership roles (Swallow and Bromley, 1995). In 1986, the system was altered to allow the project manager to administer finances, direct range riders and organize the grazing rotations (Swallow and Bromley, 1995). Prevention of transhumance of livestock from the lowlands to the mountainous regions was proposed in the 1990 National Livestock Policy Implementation Plan (Tshabalala, 1995), yet was never put into practice. This policy did not take into account the impact it would have on livestock owners in both the mountainous areas and lowlands and thus was never implemented. Within the last 10 years, RMAs were renamed Environmental Resource Management Areas to include all forms of resource use and not only grazing (Department of Environment, 2009). Until recently, neither the chiefs nor government enforced strict controls over alpine grasslands (Quinlan, 1995a). The chiefs have also been limited in their control due to economic and political contexts of livestock production, and thus RMAs have for the most part been discarded (Nüsser, 2002b).

Over the last 50 years the number of grazing-posts has substantially increased, especially in highland regions (Morris *et al.*, 1989; Quinlan, 1995a). This is believed to be the result of settlement expansion and the increase in livestock production (Quinlan, 1995a). In order to better protect the natural environment and achieve improved results for livestock owners the Livestock and Range Management Policy was introduced and states that:

"*the overall goal is to achieve greater self reliance and increased incomes for livestock owners while protecting and regenerating the underlying natural environment and resource base* (MDTP, 2007, p. 1)*."*

However, burning is not included in the strategies to achieve this in any form, despite greater awareness of its influence on vegetation and soil erosion.

3.5.7 Invasive species and encroachment of shrubs

Cold montane climates restrict the growth of woody species preventing succession from grasslands to woody bush (Bond *et al.*, 2004). Similarly, colonization by invasive species has been demonstrated to decrease with increasing altitude (Mark *et al.*, 2011). Tussock grasses are also able to outcompete or reduce the ability of invasive species to dominate a landscape through accumulation of litter, dense structure, tall height and high biomass (Pokarzhevskaya, 1998).

However, herbs tend to invade where grasses can no longer survive (Scherrer and Pickering, 2005) or where individuals die. Burning tends to open up areas for invasive species through the loss of plant matter and an increase in nutrient availability (Blank *et al.*, 2007). Similarly, roads and mountain passes (such as the Sani Pass and its associated roads) assist the spread of invasive species (Carbutt and Edwards, 2004). Shrubs are almost always associated with cattle posts and are related to overgrazing (Martin, 1984). North-facing slopes are also considered to be more prone to invasive species as this is where livestock concentrate their grazing on the sweeter species (van Zinderen Bakker, 1981). Alternatively, invasive species may be better adapted to the warmer, drier conditions occurring on north-facing slopes in the southern hemisphere. Bare patches of soil produced through trampling or the daily routes of livestock, encourage the invasion of dryland species (Quinlan, 1995b), with *Chrysocoma ciliata* being well established around grazing posts (Morris *et al.*, 1989). However, Carbutt and Edwards (2001) argue that the Drakensberg alpine region contains many aspects of the Fynbos floral kingdom, and that the montane regions are an extension of its natural boundaries. Consequently, many species considered to be invasive, may not necessarily be so.

Today, it is estimated that more than 359 680 hectares (16%) of Lesotho is covered by invasive species such as *C. ciliata* (Ranthamane, 2008). The presence of *C. ciliata* is taken as an indicator of rangeland deterioration (Pomela *et al.*, 2000). However, *C. ciliata* and *Pentzia cooperi* are both indicators of the subtropical vegetation belt (Morris *et al.*, 1993). This may possibly indicate a temperature increase, permitting these species to live at higher altitudes. Karoid shrubs such as *C. ciliata*, *Pentzia cooperi*, *Eumorphia sericea* and *Selago flanaganii* are invasive species found in many areas of Lesotho. These invaded communities often have little defense against rainfall (Herbst and Roberts, 1974; Morris *et al.*, 1989), but are considered strong increasers (i.e. they increase in abundance with increasingly heavy grazing pressure) (Morris *et al.*, 1989). *Chrysocoma ciliata* L. is also known as Karoo Bitter Bush and is used interchangeably in the literature. The presence of this shrub has been recorded for many decades and is considered a valuable fuel in Lesotho (van Zinderen Bakker, 1981). In many areas of Lesotho, *Karroochloa purpurea* and *Catalepis gracilis*, species indicative of disturbance, have invaded from the Karoo (Killick, 1990). *Karroochloa purpurea* is also an invasive species encroaching the grasslands of the Sani region (Deschamps, 2006), and apart from being an indicator of disturbance, these shrubs are also responsible for further disturbance. The invasive karroid shrubs are generally unpalatable to livestock (Dempster and Richards, 1973), which results in increasing pressure being placed on surrounding grasses.

Invasive species, past disturbances and climate change can substantially alter the way in which an ecosystem responds to disturbances such as fire (Kirk *et al.*, 2009). Fire may promote multiplication of invasive species from seeds (Lunt, 1990). To this end, the spread of Karoid species is controlled, at least in part, by the combination of burning and temperature extremes (McGinley, 2007). The effect of fire on shrubs is responsible for the altitudinal expansion of the alpine zone through the lowering of the tree line (Nagy and Grabherr, 2009). Yet, fire has been proposed as a tool to control *Chrysocoma ciliata* and *Pentzia cooperi* and other fynbos shrubs, as they are fire sensitive (Guillarmod, 1971; Morris *et al.*, 1989). *C. ciliata* has decreased in density by 90% after burning in the river basins to the east of Matebeng Pass, in the Drakensberg (Morris *et al.*, 1989). However, the shrub reduction effects were contradicted by the widespread sprouting of the shrub *Athanasia thodei*, due to burning (Morris *et al.*, 1989). Despite the beneficial effects of burning on shrubs, incorrect management can negate the effects (Morris *et al.*, 1989). The result is that burning merely removes the symptom of mismanagement rather than the cause. Management should not be solely dependent on fire but should be incorporated with weeding, poisoning and grazed by suitable herbivores (Lunt, 1990). However, these bring problems of their own; labour costs, poisoning of ecosystems and over population of grazers.

3.5.8 Burning in Lesotho

Little is known about the contemporary burn frequency for the alpine regions of the Maluti mountains, and even less is known about past burning regimes (Morris *et al.*, 1989). Before the permanent settlement of people in the highlands, the main cause of burning is believed to be lightening, which still frequently occurs during summer months in particular (Vogl, 1974; Morris *et al*., 1989; Pomela *et al*., 2000). Today, burning in the Drakensberg has been attributed to, and depends on, the type of land use, and is thus mostly anthropogenic in origin (Everson *et al.*, 2004). Lesotho fires tend to be patchy (generally less than 10ha), with some areas being burnt more than once a year, whilst others have been excluded from fire for many years (Cwielong *et al.*, 2000; Pomela *et al*., 2000). The small nature of these fires was attributed to overgrazing, leading to a lack of fuel accumulation and thus reduced spreading (Cwielong *et al.*, 2000; Pomela *et al*., 2000). Previously, several authors confirmed the occurrence of annual or biannual widespread fires (Guillarmod 1969, 1971; Morris *et al.*, 1989). Guillarmod (1971) describes these fires as being carried by the westerly winds across mountains and lasting for many days, eventually dying out on river banks or in marshes. It is believed that the vegetation of the northern Drakensberg has over time been more affected by fire than the southern regions (Bond *et al.*, 2004). This may be due to the effects of altitude on the specialization and coping

mechanisms of alpine vegetation on tolerance to fire. Many alpine grasses have high tannin contents and low palatability, and decompose slowly, thus resulting in the accumulation of easily combustible fuel (Bond *et al.*, 2004; Mucina *et al.*, 2006). Areas on the eastern escarpment, although regularly grazed, are less populated than the lower and western regions, and thus have a higher fuel load, promoting the spread of fire (Morris *et al.*, 1989). The burning events cannot be described as 'prescribed' nor as wildfires, as they are not planned burns, but are usually ignited by herdsmen.

Most communities tend to burn annually (MDTP, 2007), and yet, during the summer of 1987/1988 no fires were observed except for small scale burning of *Merxmuellera drakensbergensis* to scare away jackals from herds (Morris *et al.*, 1989; Nüsser, 2002b). Similarly, elders and herdsmen have reported that burning of an area did not occur annually and that burning for green growth for grazing purposes was not used (Morris *et al.*, 1989). However, people working on the South African side of the border disagreed with this, adding that burning has always been an annual event in spring (Morris *et al.*, 1989). Often, fires started on the South African side of the border, for management purposes, are observed to cross the border during strong winds and burn inland along the escarpment. Similarly, regular burning of *Merxmuellera* species for palatable green shoots was noted by Guillarmod (1969) and more recently by Nüsser (2002b).

According to Guillarmod (1971), burning is a positive factor for the high altitude grasslands, and it is argued that most grass species found in the Lesotho highlands are adapted to fire and require regular burning to maintain their optimum health (Smith, 1985; Morris *et al.*, 1989; Paysen *et al.*, 2000; Uys *et al.*, 2004). However, there is little control leading to the negative effects of frequent burning. For example, regular uncontrolled burning of slopes without the use of appropriate conservation systems may result in slope erosion (Ranthamane, 2008). There is almost no literature or available information concerning the burning that has taken place during the recent past, or the policies relating to rangeland burning (Cwielong *et al*., 2000). However, under the Range Management and Grazing Control regulations of 1980, legal notice number 30 states that unauthorized grass burning in both villages and cattle post areas is banned and offenders face sizable fines (Morris *et al.*, 1989). This notice was reinforced in 1986, stating that Principal Chiefs operating under the Ministry of Agriculture had the right to enforce bans on burning (Swallow and Bromley, 1995). Haphazard burning was also prohibited in rangelands unless prescribed by rangeland specialists (Motsamai, 1995). The general grass burning management in

Lesotho is believed to have been based on the idea that controlled burning at long intervals was beneficial to the control of shrubs and unpalatable grasses (Morris *et al.*, 1989). The current study region was apparently annually burnt in the late 1980s, while the lower pastures were burnt less frequently (Morris *et al*., 1989). The establishment of the Maloti-Drakensberg Transfrontier Conservation and Development Program in 2002 aimed to better control unplanned burning, yet currently no literature is available concerning their policies.

The 2007 Mokhotlong-Sanqebethu Managed Resource Area Management Plan aimed to limit uncontrolled and unregulated burning through the introduction of controlled burning, implemented through community building and the introduction of a by-law. The plan outlines the by-law as follows:

"Any person who willfully burns, destroy or otherwise damages pastures, natural habitats, fauna and flora or anything whose management and protection is addressed by these by-laws, commits an offence and is liable on conviction to a fine not exceeding M1,000 or to imprisonment for a period not exceeding six months (MDTP, 2007, p. 39)*."*

3.6 Fauna of the study region

Lesotho has several animal species, some of which have become valuable tourist attractions (e.g. Sloggett's Ice Rat, Figure 3.16). The species diversity has changed dramatically over the years due to hunting, habitat loss and competition with livestock for grazing (Marake *et al*., 1998). Pomela *et al.* (2000) estimate that there are 1279 invertebrate species, of which 134 are endemic. Approximately 285 bird species are found in Lesotho, but most are considered scarce and rare (NES, 2000). However, Pomela *et al.* (2000) argue that 318 species occur in Lesotho, whilst historical records account for a further 22. Today, Lesotho has 41 species of reptile (Earth Trends, 2003) (e.g. Figure 3.17), 19 amphibian and 14 fish species (Pomela *et al.*, 2000). Of these, the Maloti Minnow and Drakensberg River frog (*Amieta dracomontana*) are endemic (NES, 2000). *Bufo gariepensis* (Karoo or Drakensberg toad), *B. gutturalis* (guttural toad), *Cacosternum boettgeri* (common caco), *Rana angolensis* (common river frog) and *R. vertebralis* (aquatic river frog) are some of the more common species found in the study region (Hughes and Hughes, 1992; Pomela *et al.*, 2000). The wetlands of the study region provide nurseries for many juvenile frogs and attract birds to the area, such as white storks (*Ciconia ciconia*) (Pomela *et al.*, 2000).

Larger mammals have mostly been hunted to local extinction, yet small populations of smaller antelope still exist (NES, 2000). Lesotho previously had 63 mammal species, 19 of which are locally extinct in the country and a further two which are globally extinct (Pomela *et al.*, 2000). The Grey Rhebok (*Pelea capreolus*), Black Backed Jackal (*Canis mesomelas*) and Sloggetts' Ice Rat (*Otomys slogetti*) are some of the more common mammal species still found in the study region (Pomela *et al.*, 2000).



Figure 3.16: Sloggets' Ice Rat (*Otomys slogetti*).



Figure 3.17: Lang's crag lizard (*Cordylus pseudocordylus* variant *langi*).

Chapter 4:

Climatology and Soils



CHAPTER 4:

Climatology and Soils

Plants differ from each other in terms of their tolerance to ecological factors both biotic (e.g. competition) and abiotic (e.g. soil type) (Crawford, 2008). The most influential factor is the climate, through its effect on seasons, species distributions and growth rates. Climatic variables and soil moisture have been demonstrated to be more influential in the growth of plants in burnt grasslands than in unburnt grasslands (Briggs and Knapp, 1995). Hence, climatic and soil knowledge may assist in the explanation of results observed in various sub-components of this PhD project. Temperature data were collected for the Sani and Kotisephola regions, whilst precipitation, wind and solar radiation were measured at the Sani Top station. The soils of each site were compared using five characteristics: soil moisture, particle size distribution, biological material content, leachate pH and nutrient concentration.

4.1 Methodology

4.1.1 Climatology

Climatic data were collected from a horizontal surface in the Sani top region for comparison with measured growth rates. A Davis Instruments weather station was constructed, reinforced and placed at Sani Top. For security reasons, it was constructed next to an existing meteorological service station (Figure 4.1). The Davis weather station recorded rainfall, temperature (outside, inside, dew point and the windchill factor), wind (speed and direction), atmospheric pressure, UV index and solar radiation at hourly intervals. The instruments were initially calibrated by the manufacturer and installed as new instruments. Additionally, a second air temperature recorder, TGP-4020, was attached to the Davis weather station as a precaution.

Data are also displayed on the consol monitor placed in the Sani Top Chalet lounge where visitors were able to view the current weather conditions. The weather station was assembled in March 2010 and remains in operation. Initially, technical errors were a problem, which limited records between March 2010 and May 2010. Weatherlink 5.8.1 (and upgraded Weatherlink 5.9.2) software was used to download the data, and software faults were encountered which delayed the downloading of data. Once downloaded, the data were exported out of the weather station software program into Microsoft Excel for Windows 7 and graphed to show the measured

variables. However, once downloaded, the temperature data were noted to be patchy, with some dates not recorded and some data notably incorrect. Thus, the temperature data used are from the additional temperature data logger. However, some temperature dates are missing from the additional data logger due to loggers being damaged by cattle in one instance, and lack of data recording space in another. The missing data occurred between 21 October 2009 (from 11:00am onwards) and 1 April 2010 (0:00am), and between 19 January 2012 (from 14:00pm onwards) and 3 March 2012 (09:00am). The missing climatic data accounted for 15.4% of the total planned logging time.

During the course of the study, a TGP-4020 temperature gauge was added to the Kotisephola region as part of another study. The data recorded between May and October each year may aid the explanation of the results observed in that region, and hence were included. The climatic data were correlated with the measured growth rates from the monitoring study, so as to ascertain the possible climatic influence on growth.



Figure 4.1: The weather station undergoing regular maintenance.

The solar radiation data was observed to be greater than the solar constant (1.37kWm^{-2}) during the summer months, hence the solar sensor was recalibrated and the maximum solar radiation records were found to be incorrect (Tyson and Reston-Whyte, 2000). A recalibration value was obtained and applied to the data already obtained and produced viable results.

4.1.2 Soils

The soil in which the study species grows has a direct influence on the biomass and their growth through the nutrients it provides. Thus, soil samples were sampled for nutrient content, moisture content and particle size distribution, which may provide greater insight into the growth of *M. drakensbergensis* tussocks. Samples were taken twice within the study period; in early May 2009 and early May 2011. The soil samples were collected in late autumn, as during the summer months (October to March) the soil was observed to become saturated at each of the sites. Differences in soil moisture content between sites were also anticipated towards the drier months, as during the summer months most sites were saturated with little difference between sites. One sample was collected from the center of each site (10 sites in 2009 and 12 sites in 2011 due to the inclusion of M.F. Up and the K.P. site), between the tussocks. In the case of the Sani East (S.E.) site, the soil sample was collected directly next to the tussocks due to a small number of individuals and small inter-tussock spaces. Soil samples were taken for an area approximately 10cm wide and to a depth of 15cm (approximately 1kg), placed in soil bags and labeled. All seams and openings of the bags were sealed to prevent moisture loss. Each soil bag was additionally placed within a plastic container to prevent any additional moisture loss. Glass containers were not used due to their fragility. Additionally, once in the lab, the soil samples were placed in a freezer to prevent any further moisture loss.

4.1.2.1 Moisture content

Soil moisture is determined by precipitation, infiltration and runoff and has a direct effect on growth and the establishment of plants (Ravi *et al.*, 2007). Plants can obtain water from both the soil and the atmosphere; however, soil moisture is the primary source of water for plant use (Araya, 2007). The total amount of soil water available for uptake by vegetation is a function of the water content of the soil and the volume taken up by plant roots (Chapin III *et al*., 2002). Soil moisture also influences nutrient diffusion into plant roots, hence influencing growth (Wilkinson *et al*., 2000). Hence, soil moisture may vary between sites and play a role in the results obtained. Moisture content of the soil at each site was collected using an instant moisture reader in the

field during May 2009, 2010 and 2011, as coincided with the trips to the study region for the monitoring study. The moisture readings may provide insight into *M. drakensbergensis* growth patterns as it is a potential limiting factor to growth during the months leading up to and during the dry season (Briggs and Knapp, 1995). However, due to freezing temperatures experienced in June and July, measurements were made in April and May when the temperatures at midday were observed to be slightly higher, allowing for any potentially frozen soil moisture to thaw for plant up take. Readings were taken randomly throughout the sites to include the community periphery and central regions. The moisture probe was placed 8cm into the ground (a line was drawn on the probe to ensure the same depth was measured at each site), readings were taken and recorded. Fifty readings were taken at the first site, but it was established that a lower number of readings would be quicker, whilst still achieving the same average per site; thus only 25 readings were taken at the following sites. In 2011, the Kotisephola pass site was initially sampled at approximately 15:30 in the afternoon, but was frozen as a result of both the wind chill and the cooler temperatures at this high altitude. This site was resampled at 13:40, which was deemed more suitable given the thaw at midday.

The soil samples collected for laboratory analysis (one from the center of each site; section 4.1.2) were also measured for soil moisture to confirm the results obtained in the field. The frozen samples were weighed and then placed in a pre-heated oven at 80^{o}C for 48 hours until dry (Figure 4.2). The samples were reweighed directly after removal from the oven and the dry weights recorded. The wet weight and the dry weight were then subtracted and the percentage moisture contained in the original sample calculated (equation 1).

$$\text{Percentage soil moisture} = \frac{\text{Wet weight} - \text{dry weight}}{\text{Wet weight}} \times 100 \qquad \text{(Equation 1)}$$

An Analysis of Variance (ANOVA) test was selected to determine if any soil moisture differences existed between sites using the soil moisture results from the soil samples and from the soil probe readings. Using SPSS version 21, the data were investigated to check that all the assumptions of an ANOVA were met. The assumptions tested were to indicate the non-presence of outliers, normal distribution (conducted using the Shapiro-Wilk test due to the small number of sites tested) and the homogeneity of variance (conducted using the Levene test) (Puri, 2002). However, the results from the soil samples could not be tested for normal distributions due to the

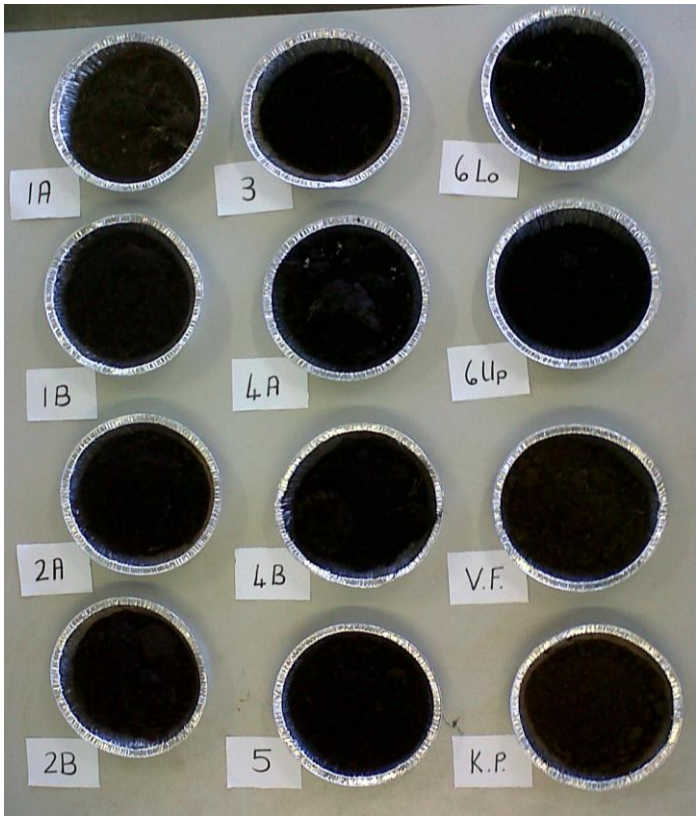


Figure 4.2: Soil samples from each site ready for oven drying.

small number of values per site. Similarly, the moisture probe data did not meet the assumption of normal distribution in the case of several sites (S.E., S.S.,V.F., M.F. Up, M.F. Lo, A.P. and K.P.). Additionally, the moisture probe data was significant ($p<0.001$) for Levene's test suggesting variations were not homogeneous. Appropriate transformations of the moisture probe data continued to fail to meet the assumptions, hence a non-parametric Kruskal-Wallis test was performed for both data sets (Townend, 2002; Dytham, 2011). The null hypothesis (H_0) used was that there is no difference between the medians of the soil moisture probe readings from each of the sites. The standard deviation was determined to establish the variation in soil moisture within each site, using the moisture probe results. The results were then simply compared to aspect (in the form of values from 1 to 4.5 with 1 representing north-facing sites, 4.5 representing northwest-facing and 0 representing flat land) so as to view whether any of the soil moisture differences between sites were the result of aspect, using a scatter plot.

4.1.2.2 Particle size distribution

A particle is an entity with an easily recognizable surface, consisting of either one uniform substance or smaller joined entities (Loveland and Whalley, 2000). The particle distribution of a representation of soil from each site was calculated through dry sieving. The soils were dried in

an oven at 80^{o}C for 48 hours and then weighed. The sieve column was shaken in a mechanical shaker at 15rpm for 10 minutes. The 10 sieves used ranged in mesh size from 4.750mm to 0.038mm to indicate the particle size range (Table 4.1). The results, in the form of percentage soil retained in each sieve tray (equation 2), were then graphed to provide a graphical distribution of the particle gradation at each of the sites. An Analysis of Variance (ANOVA) test was used to determine if any differences existed between sites, using SPSS (version 21). The data met the required assumptions of normal distribution, equal variances and non-presence of outliers (Puri, 2002; Townend, 2002). The null hypothesis (H_0) used was that there was no difference between the particle size distributions of each site.

Table 4.1: The sieve sizes used to calculate the particle distribution of each soil sample.

Sieve Number:	Mesh size in mm:	Mesh size in µm:
1	4.750	4700
2	2.000	2000
3	1.000	1000
4	0.500	500
5	0.250	250
6	0.125	125
7	0.063	63
8	0.045	45
9	0.038	38
10	<0.038 collection tray	<38 collection tray

$$\text{Percentage soil retained in each sieve} = \frac{\text{Sieve weight}}{\text{Total sample weight}} \times 100 \qquad \text{(Equation 2)}$$

Despite its common use, perceived accuracy and consistency, there are limitations to this method. Clay particles are light and may become airborne within the sieve column during shaking and weighing, resulting in under estimations of clay particles. However, with the initial total weight known, one can calculate the percentage shortfall of clay particles from the retained sieve weights. Particle shape can also have an impact as this method assumes particles are spherical. Long, thin particles may either be trapped in large meshes or move sideways through the large meshes to the smaller meshes, thus perhaps marginally influencing results.

4.1.2.3 Organic content

Fifty gram samples of each site sample (mixed to prevent any bias from surface materials) were measured out and placed into small ceramic crucibles. These were placed in a preheated kiln at 500^{o}C for 24 hours. Under high temperatures, organic material burns and samples were re-weighed once removed from the kiln to ascertain how much organic material had been lost. The results were tabulated for further comparison.

4.1.2.4 Leachate pH

The pH of the leachate for each soil sample was calculated in the laboratory to determine whether differences between sites could explain differences in the growth, shape, size and distribution of the studied tussocks. Labeled test tubes were filled with 15.0g of collected soil, and 10ml of pH neutral water (7.00) was subsequently added and mixed (Figure 4.3). The samples were then left at room temperature for 2 hours, mixed and the pH was measured using a calibrated Consort C532 pH meter.

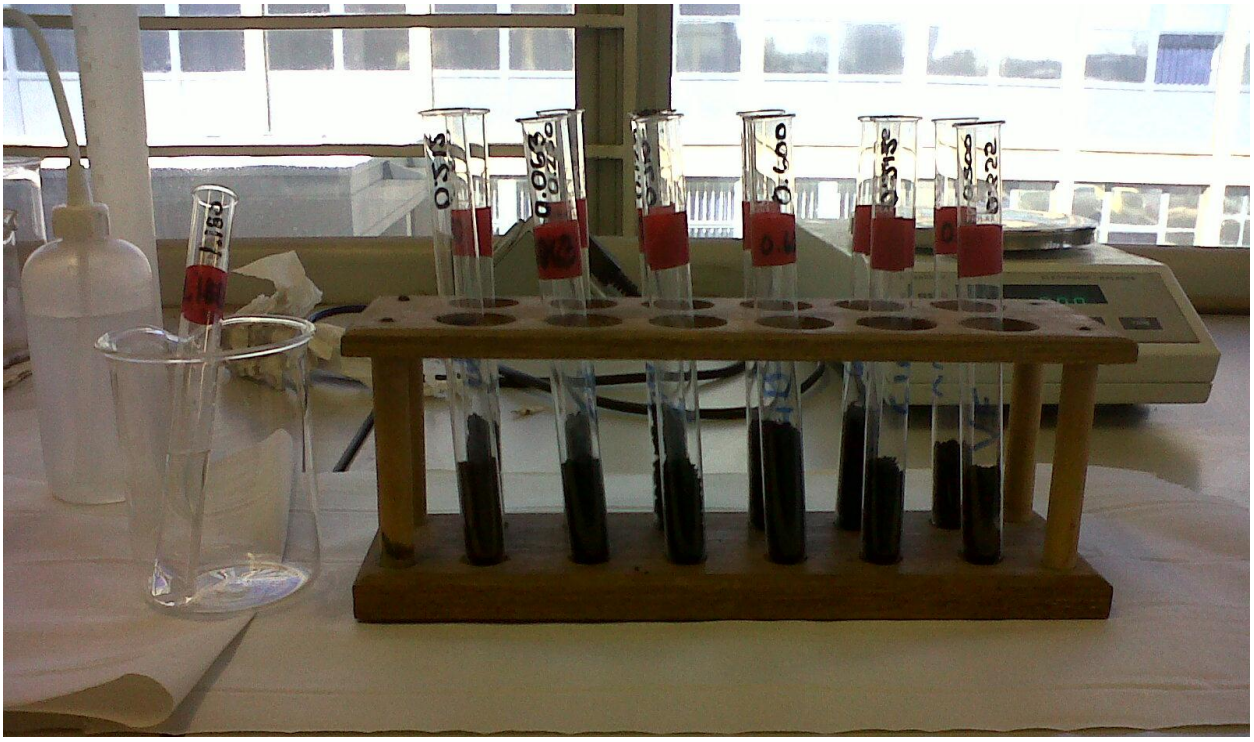


Figure 4.3: Test tubes containing 15.0g samples from each site, with a control sample in the beaker before the 10ml distilled water was added.

4.1.2.5 Soil chemistry

The nutrient cycle may be influenced by burning, either positively or negatively, as has been suggested by Cremer (2004) and Manson *et al.* (2007). To this end, it is worth comparing the influence of burn frequency on tussock growth, biomass and soil nutrient levels. It may also provide a greater insight into growth rate differences between sites. Five hundred gram samples from each site were sealed in lined boxes and sent to the Soil Fertility and Analytical Services department of the KwaZulu-Natal Department of Agriculture, Environmental Affairs and Rural Development, Pietermaritzburg. Total soil concentrations of phosphorus, potassium, calcium, magnesium, zinc, manganese, copper and nitrogen were measured for each soil sample. The results were tabulated to assist with the comparison between sites and an ANOVA was undertaken to see whether significant differences exist between the sites.

4.1.2.6 Principal component analysis

Principal component analysis (PCA) is a multivariate technique which is often used to indicate which measured factors vary the most between samples (Townend, 2002). A PCA can be used to reduce a data set, allowing the researcher to focus on the factors which provide the most information (Townend, 2002). In the current study, a PCA was performed to indicate which soil characteristics varied the most between the study sites, and to potentially identify groups of related characteristics. The PCA correlated the soil variables obtained in this chapter; namely soil moisture, organic content, leachate pH, soil chemistry and particle size distribution. SPSS version 21 was used to perform the analysis using a correlation matrix due to the presence of different units and Oblimin rotation to increase the interpretability of the resulting factors.

4.2 Results

4.2.1 Climatology

4.2.1.1 Air temperature in the Sani region

Air temperature records were measured from 8 March 2009 to 16 May 2012 at Sani Top (Figure 4.4 and 4.5). The absolute temperature range throughout the data collection period is 38.4^{o}C, as the highest temperature recorded was 23.4^{o}C (9 Novmber 2011) and the lowest recorded temperature was -15.0^{o}C (16 June 2010; Table 4.2). The average daily temperature range was 22.3 ^{o}C, but ranged between 14.7 ^{o}C (January 2011) and 30.8^{o}C (October 2010). The mean annual temperature for the complete data year, 2011, is 6.6 ^{o}C. The hours recorded below 0 ^{o}C

Table 4.2: Summary of temperature data with percentage hours per month below 0°C and 2°C for the Sani region.

Year	Month	Max. (°C)	Min. (°C)	Mean (°C)	Mean daily range (°C)	% Hours per month below 0°C	% Hours per month below 2°C
2009	**March**[1]	16.4	0.2	8.7	16.3	0.0	1.3
	April	15.0	-4.1	6.3	19.1	6.8	20.0
	May	11.5	-4.9	3.2	16.4	28.0	47.2
	June	10.5	-10.3	1.3	20.9	43.1	61.3
	July	12.2	-16.4	-0.9	28.6	60.4	70.4
	August	13.8	-8.1	2.5	21.9	36.8	54.0
	September	18.1	-6.4	7.6	24.4	9.0	18.8
	October[2]	18.4	-1.6	7.7	20.0	1.2	6.3
	November						
	December						
2010	**January**		No data available				
	February						
	March						
	April[3]	20.3	-5.4	7.2	25.7	6.9	13.5
	May	15.4	-8.6	4.8	24.0	19.2	28.8
	June	16.5	-15.0	1.5	31.5	42.6	57.4
	July	11.1	-12.0	0.9	23.1	50.4	61.7
	August	16.5	-14.2	3.2	30.8	31.3	41.5
	September	20.1	-5.7	9.0	25.9	4.2	10.7
	October	21.4	-9.5	8.9	30.8	5.8	11.2
	November	21.3	-2.6	10.0	23.8	0.8	3.1
	December	18.4	0.8	10.0	17.6	0.0	0.3
2011	**January**	17.6	2.9	10.4	14.7	0.0	0.0
	February	17.9	1.7	9.9	16.2	0.0	0.2
	March	18.3	2.6	10.6	15.7	0.0	0.0
	April	16.5	-6.0	5.7	22.5	4.7	13.5
	May	12.9	-9.4	3.4	22.3	20.3	38.7
	June	10.8	-12.3	1.4	23.0	38.5	57.4
	July	10.1	-11.4	-1.6	21.5	62.9	78.4
	August	16.9	-13.8	1.9	30.7	39.5	48.7
	September	17.9	-6.7	6.9	24.6	8.6	18.1
	October	22.2	-5.0	8.9	27.2	11.0	21.0
	November	23.4	-1.3	10.2	24.6	8.3	20.3
	December	19.8	3.1	11.1	16.7	0.0	2.6
2012	**January**[4]	20.6	3.9	12.7	16.7	0.0	0.0
	February		No data Available				
	March[5]	17.4	-0.0	8.6	17.4	0.13	1.61
	April	15.7	-3.5	5.1	19.2	23.89	49.86
	May[6]	16.3	-5.2	6.4	21.5	23.61	52.52

[1] First recording: 8 March 2009 00:00am; [2] Last recording: 21 October 2009 10:00am; [3] First recording: 1 April 2010 00:00am; [4] Last recording: 19 January 2012 13:00pm; [5] First recording: 3 March 2012 09:00am; [6] Last recording: 16 May 2012 17:00pm.

indicate a strong seasonal trend with no hours below freezing recorded in midsummer each year (December to February), while less than 10% of days recorded temperatures below 0 ^{o}C in April (except 2012) and between September and November (except October 2011). The most number of hours below freezing were recorded in July 2009 (60.4% of the month), July 2010 (50.4% of the month) and July 2011 (62.9% of the month). Fewest hours per month below 2 ^{o}C were observed between September and March/April each year (less than 21%). The months experiencing the highest percentage of hours below 2 ^{o}C were June and July each year (more than 55%). Temperatures below -10 ^{o}C were recorded on 6 days in 2009, 7 days in 2010 and 11 days in 2011 (all in June, July and August).

Air temperature records were obtained for the Kotisephola region from 23 May 2011 to the 15 October 2012, excluding the period between 8 October 2011 and 17 May 2012 (Figure 4.6). Despite the lack of summer data, the data still adds value to the study as the most likely periods to decrease below the growth thresholds occur during the winter months. The temperature range throughout the period was 30.3 ^{o}C, with the highest temperature recorded at 17.7 ^{o}C (7 October 2012), while the lowest recorded temperature was -12.5 ^{o}C (24 June 2011, Table 4.3). The average daily temperature range is between 15 ^{o}C and 24 ^{o}C for the study period (May to October). The mean air temperature for the 2011 winter period is -0.5 ^{o}C, whilst that for 2012 is 1.9 ^{o}C. The air temperature decreased to below 0 ^{o}C for more than 70% of each month between May and July 2011, with 92.9% of July below 0 ^{o}C. During the following year, only June experienced temperatures below 0 ^{o}C for 70% of the month. Less than 20% of days were below 0 ^{o}C for September and October each year and this trend is expected to continue into the summer months. Similarly, May to August 2011 experienced temperatures below 2 ^{o}C for more than 70% of each month, with June and July recording more than 90% of the days below 2 ^{o}C. In 2012, only June and July are observed to have temperatures below 2 ^{o}C for more than 80% of each month. Temperatures below -10^{o}C were recorded during 8 days in 2011 and 6 days in 2012 between May and August.

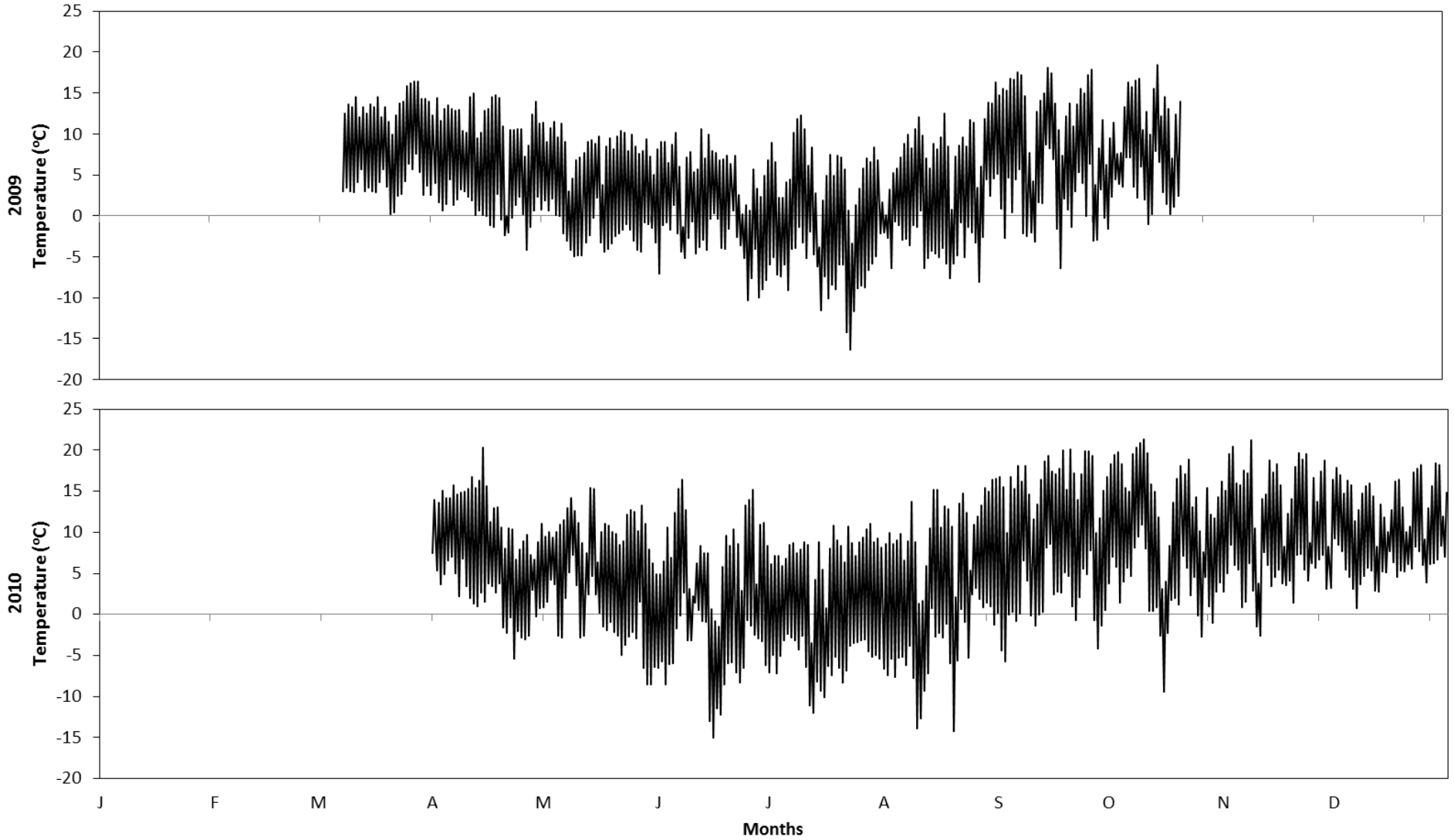


Figure 4.4: Average daily temperature, maximum and minimum temperatures for 2009 and 2010 from the Sani region.

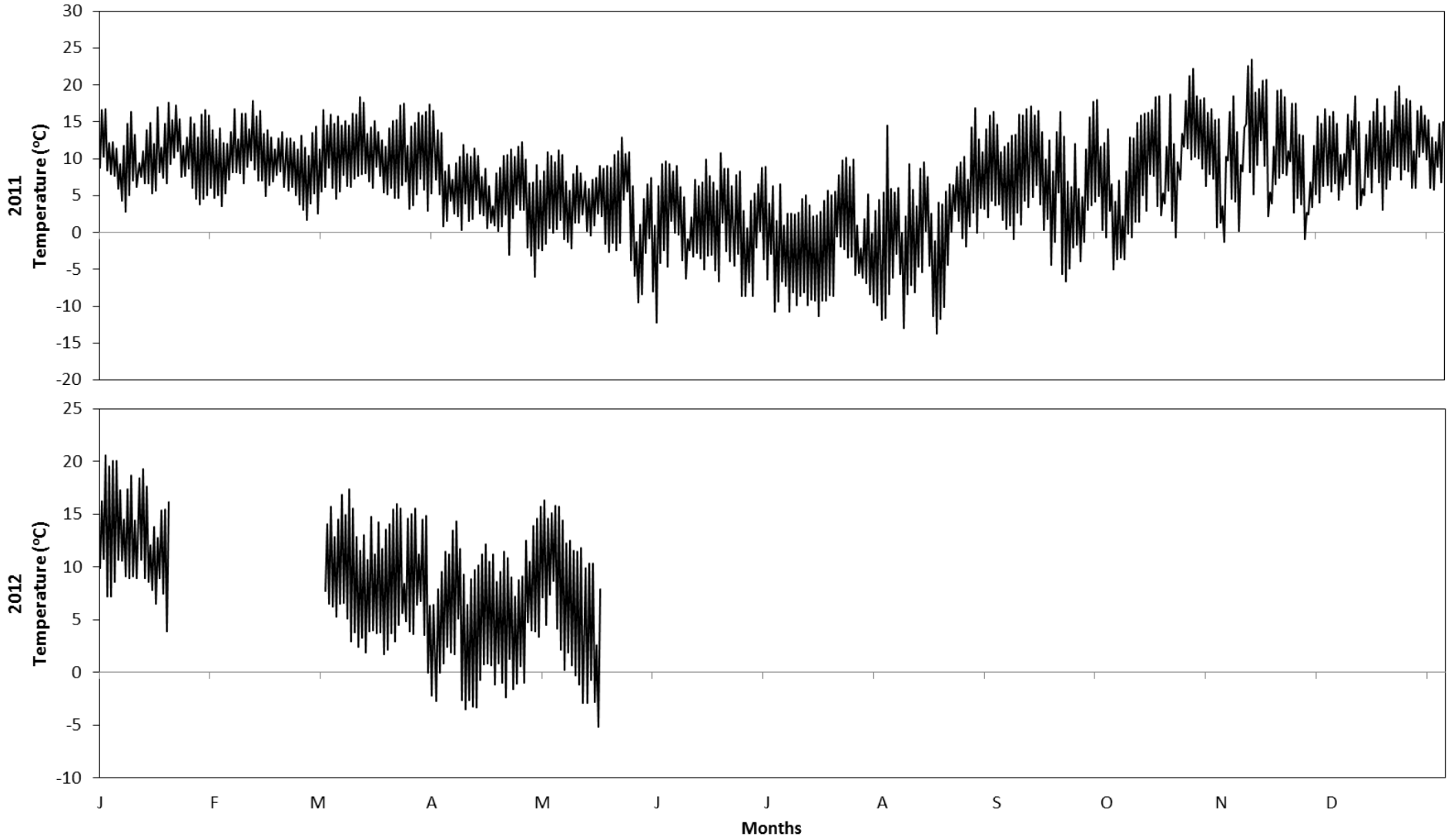


Figure 4.5: Average daily temperature, maximum and minimum temperatures for 2011 and 2012 from the Sani region.

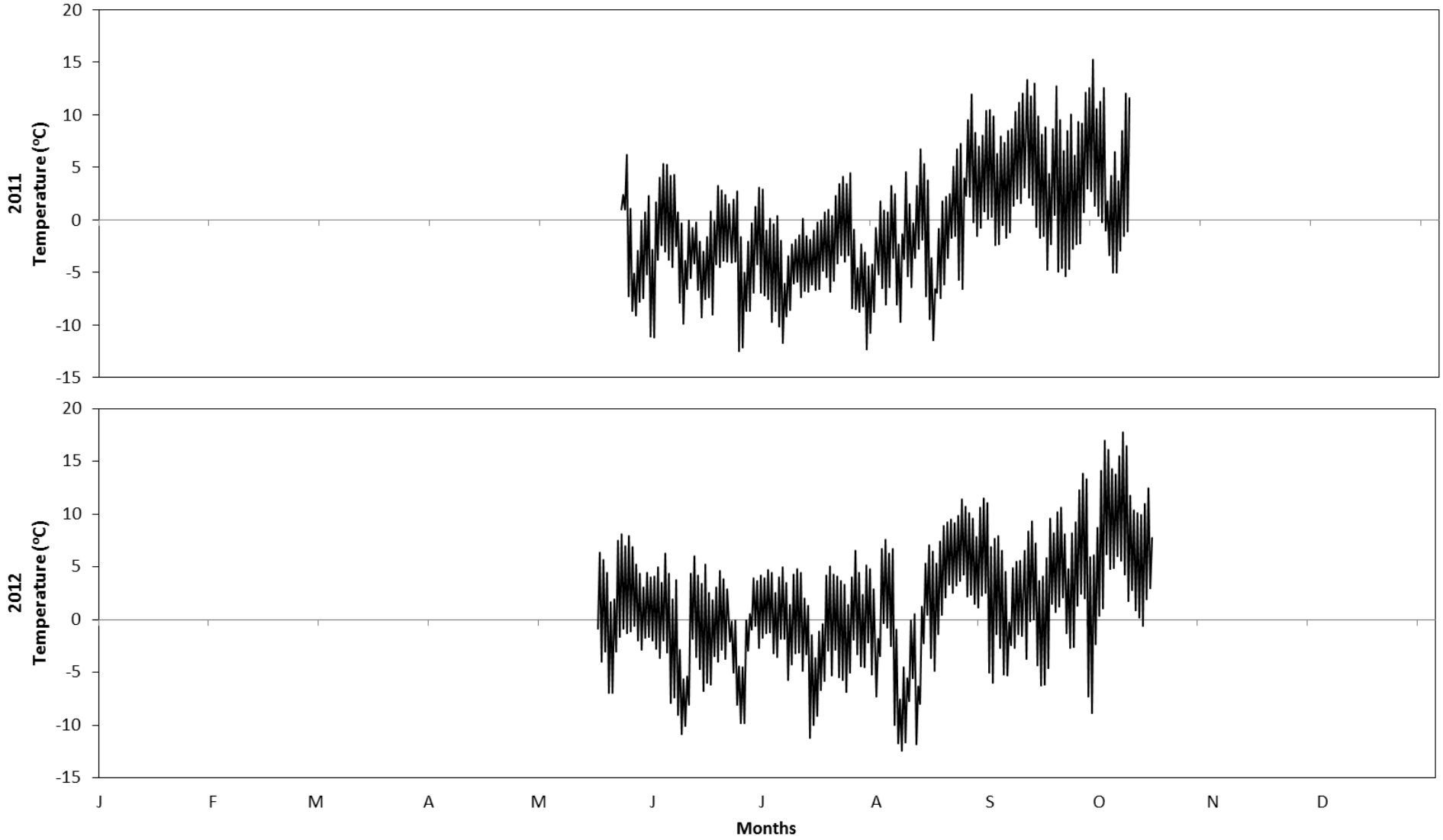


Figure 4.6: Average daily temperature, maximum and minimum temperatures for 2011 and 2012 from the Kotisephola region.

Table 4.3: Summary of temperature data with the percentage hours per month below 0°C and 2°C for the Kotisephola region.

Year	Month	Max. (°C)	Min. (°C)	Mean (°C)	Mean daily range (°C)	% Hours per month below 0 °C	% Hours per month below 2 °C
2011	**May**[1]	6.2	-11.2	-3.0	17.4	71.3	84.7
	June	5.3	-12.6	-2.7	17.8	83.5	93.1
	July	4.5	-12.3	-4.1	16.8	92.9	96.9
	August	11.9	-11.5	-0.4	23.4	54.4	73.0
	September	15.2	-5.4	4.5	20.6	18.9	34.0
	October[2]	12.6	-5.1	2.5	17.6	37.0	63.0
	November						
	December						
2012	**January**			No data available			
	February						
	March						
	April						
	May[3]	8.1	-7.0	1.3	15.1	33.3	59.8
	June	6.3	-10.9	-1.5	17.2	70.1	85.8
	July	6.5	-11.2	-0.8	17.7	61.8	82.1
	August	11.5	-12.4	1.5	23.9	36.8	48.0
	September	13.9	-8.9	2.6	22.7	33.1	48.6
	October[4]	17.7	-0.6	8.0	18.3	0.3	6.7

[1] First recording: 23 May 2011 17:00pm; [2] Last recording: 8 October 2011 15:00pm; [3] First recording: 17 May 2012 00:00am; [4] Last recording: 15 October 2012 08:00am.

4.2.1.2 Precipitation in the Sani region

The highest rainfall recorded between 1 May 2010 and the 31December 2010 was 179mm during the month of December (Figure 4.7). May 2010 received almost no rainfall. It was noted that much of the precipitation which fell between June and August in both 2010 and 2011 fell as snow. In 2011, the highest rainfall recorded was 311mm in January, which exceeds that recorded by Nel and Sumner (2008) in previous years. Similarly, the total annual rainfall received was 997mm, which was also higher than the annual means for the period between 2001 and 2005 (Nel and Sumner, 2008). However, the total rainfall for 2011 falls within the range indicated by Pomela *et al.* (2000), Mucina *et al.* (2006) and the Bureau of Statistics (2012). The total rainfall from 1 May 2010 to 31 December 2010 was 376mm. Mist was frequently observed within the study region and is potentially important to tussock grasses as their leaves capture such precipitation. However, the rain gauges are likely not sensitive enough to provide an accurate indication of net precipitation owing to mist.

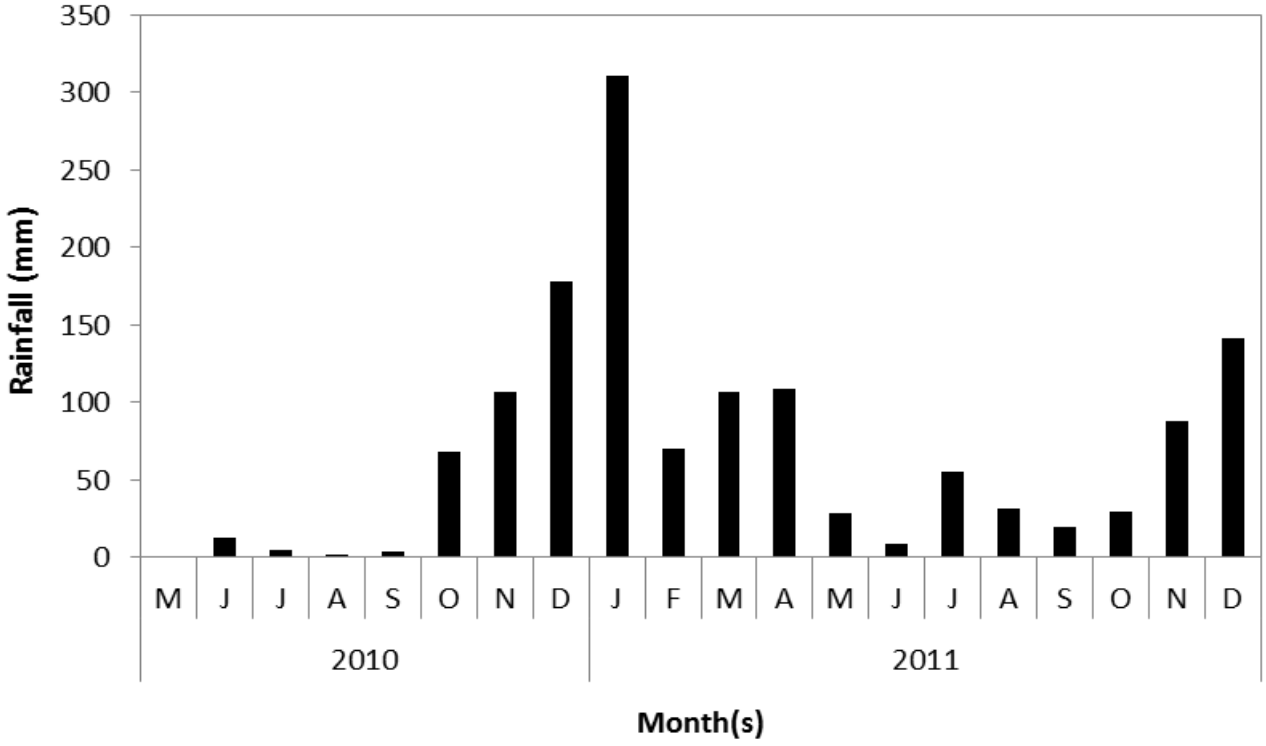


Figure 4.7: Total rainfall (mm) measured in each month during 2010 and 2011.

4.2.1.3 Wind in the Sani region

The mean monthly wind speeds indicate the greatest average wind speeds occur during winter (May to August) and reach a maximum in September 2010 (7.45 ms^{-1}) and August 2011 (4.7 ms^{-1}) (Table 4.4). Conversely, the lowest wind speeds are observed during the summer months (October to April) with the lowest mean recorded in December 2011 (1.42 ms^{-1}). The wind speeds were observed to increase in short gusts reaching a maximum of 19.42 ms^{-1} in May 2010 and 19.29 ms^{-1} in June 2010. The dominant wind directions experienced throughout 2010 and 2011 are northwesterly (14.1% and 12.3%), north-northwesterly (29.2% and 25.1%), northerly (17.7% and 14.3%) and north-northeasterly (3.2% and 3.1%) (Figure 4.8 and 4.9). Greater than 40% of the wind direction frequency was experienced in a north-northwesterly direction in October 2010 (41.7%), May (44.5%), September (46.5%) and October 2011 (45.5%). The wind direction frequency in 2010 was below 10% in southerly directions except in May, July, August and in December. In 2011, more than 10% of the wind recorded had a southerly direction during the summer months (January, February, November and December). The fastest average wind speeds (lasting an hour) throughout 2010 were recorded in a north-northwesterly direction (19.4ms^{-1} in May; 19.3ms^{-1} in June; 13.5ms^{-1} in July; 19.1ms^{-1} in August; 18.3ms^{-1} in September; 18.2ms^{-1} in October; 10.9ms^{-1} in November). In 2011, the fastest average wind speeds were recorded in a northerly to northwesterly direction (14.0ms^{-1} in July and 14.9ms^{-1} in September).

Table 4.4: Average monthly wind speed and maximum gusting wind speed.

Year	Month	Average Wind Speed (ms^{-1})	Maximum Speed (ms^{-1})
2010	**May**	4.7	19.4
	June	6.3	19.3
	July	5.1	13.5
	August	6.6	19.2
	September	7.5	18.3
	October	5.1	18.2
	November	3.0	10.9
	December	2.1	4.0
2011	**January**	2.0	3.3
	February	1.8	3.7
	March	2.4	5.1
	April	2.1	7.5
	May	2.4	7.5
	June	3.5	11.3
	July	4.8	14.0
	August	4.7	11.8
	September	4.4	14.9
	October	1.6	9.1
	November	2.5	16.3
	December	1.4	5.1

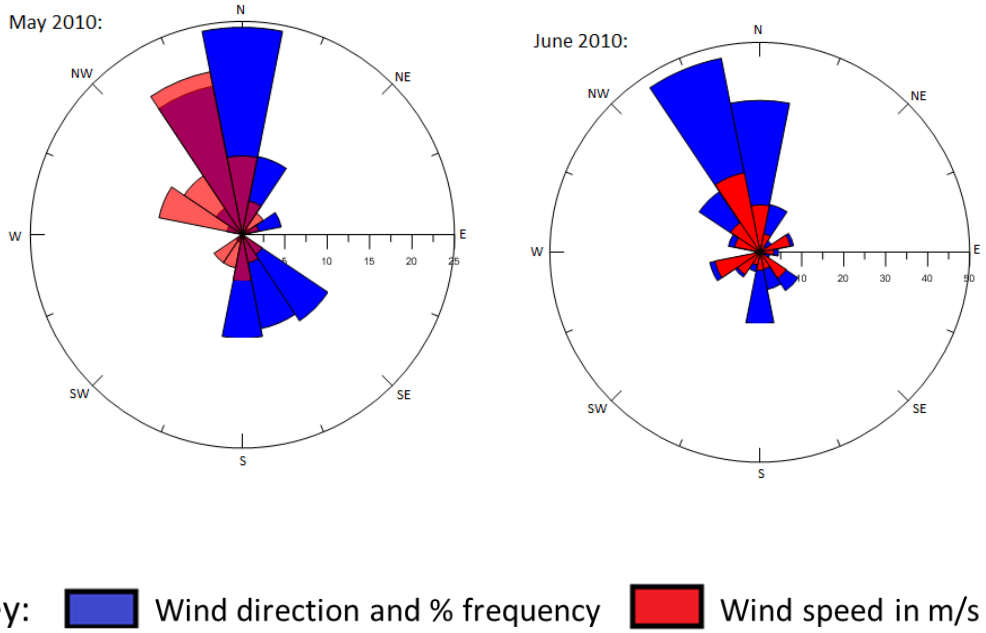


Figure 4.8: Wind direction, percentage frequency and average wind speed for 2010.

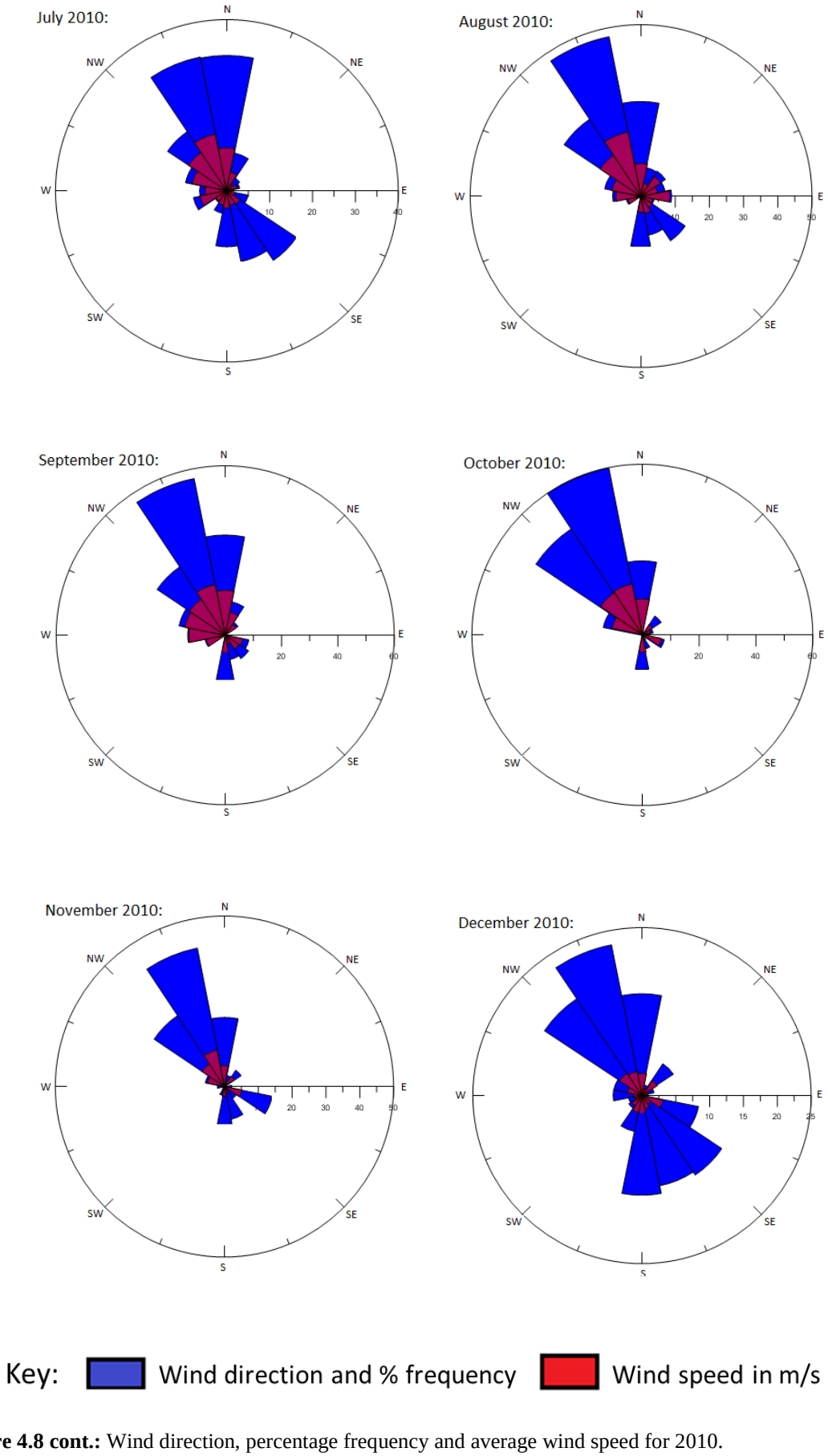


Figure 4.8 cont.: Wind direction, percentage frequency and average wind speed for 2010.

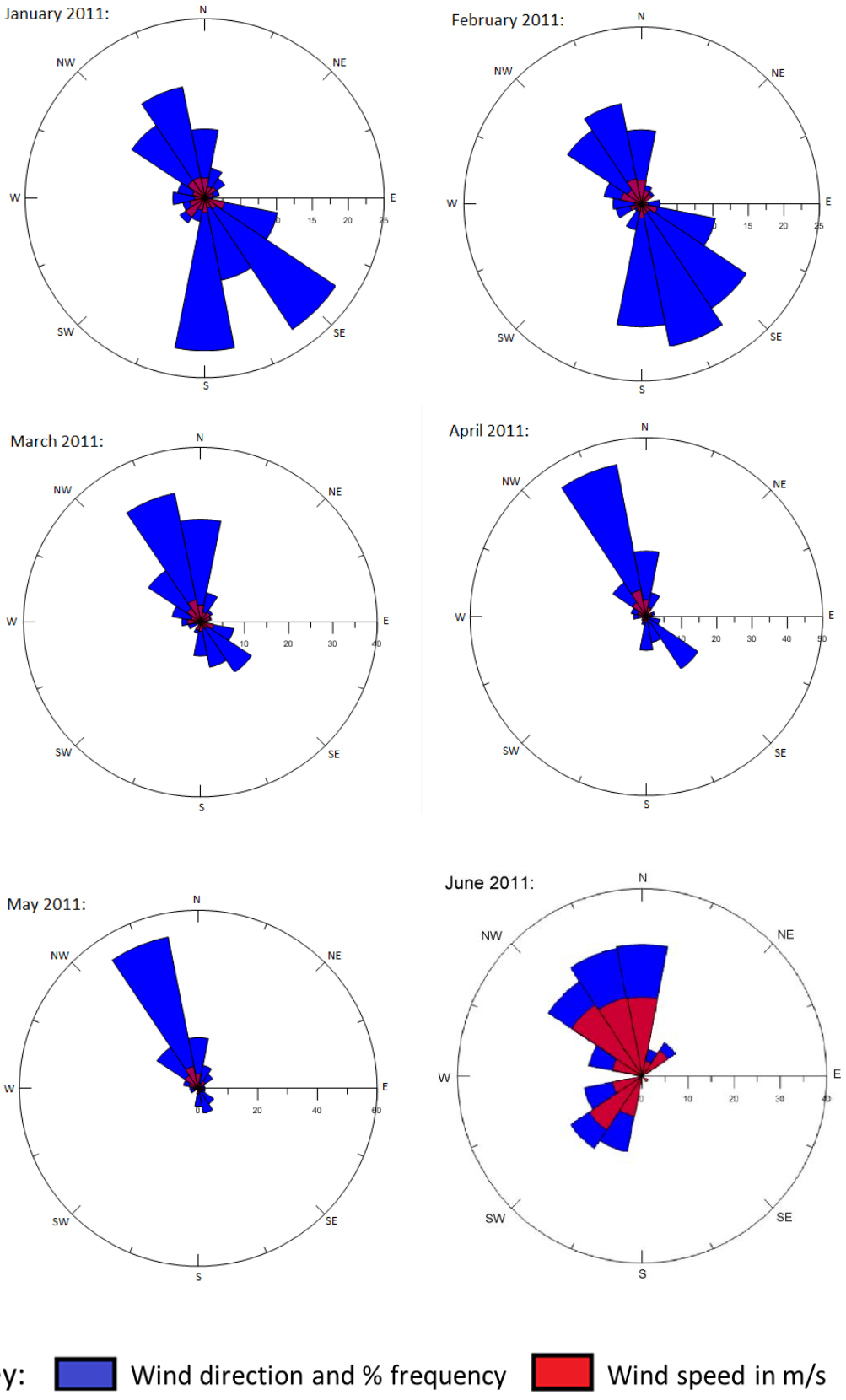


Figure 4.9: Wind direction, percentage frequency and average wind speed for 2011.

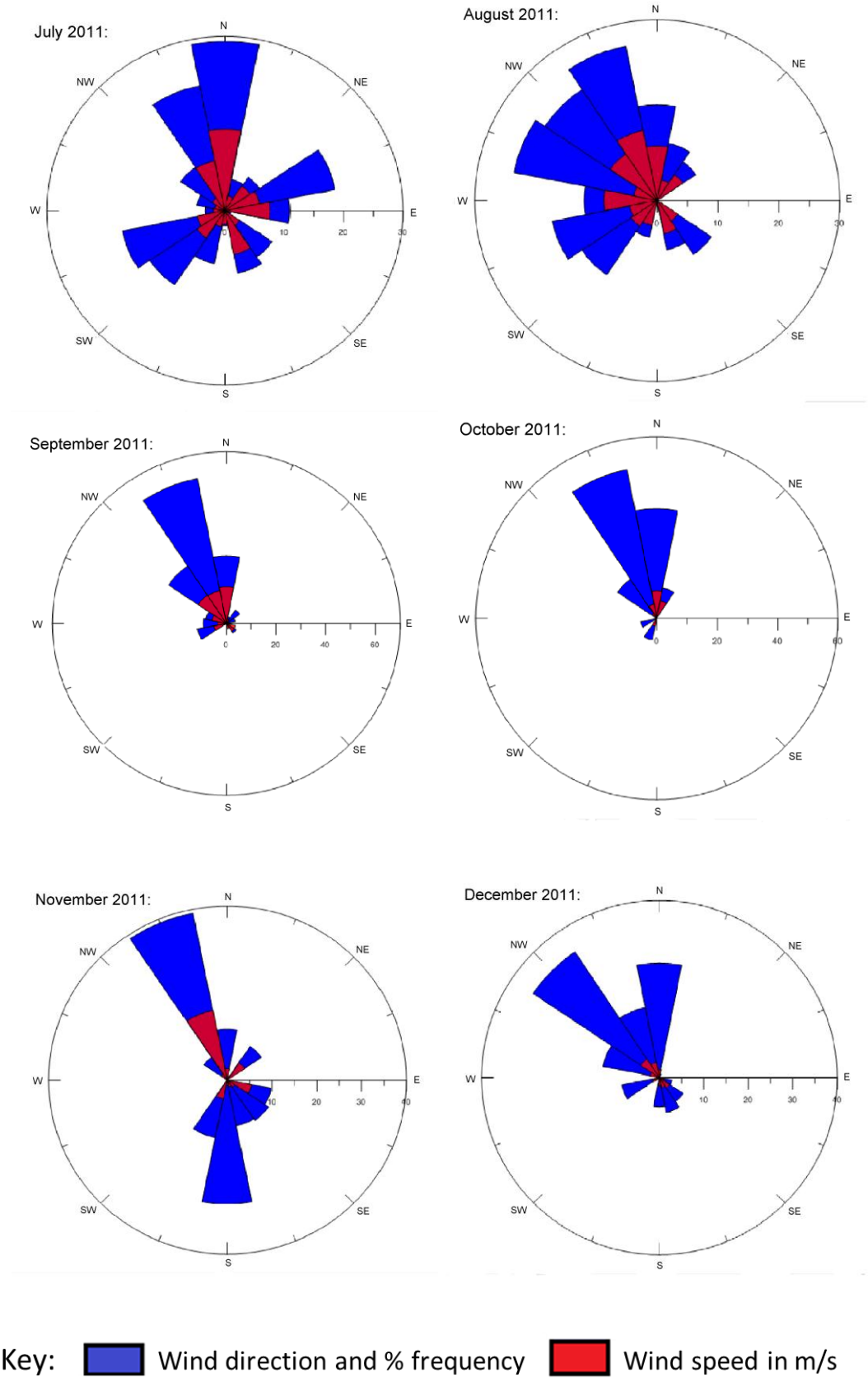


Figure 4.9 cont.: Wind direction, percentage frequency and average wind speed for 2011.

4.2.1.4 Solar radiation at Sani Top

The average monthly solar radiation from a horizontal surface did not decrease below 160W/m^2 from the start of recording on 1 May 2010 to the end of the study period 31

December 2010 (Figure 4.10). The average monthly solar radiation dropped below 200W/m^2 from April to July each year and was above 200W/m^2 from August to March each year. Afternoon cloud cover was observed to decrease the hourly solar radiation received during November and December each year. The decreases were observed generally after 3:00pm and occasionally increased to low values (average of less than 220 W/m^2 per hour) before decreasing further to indicate sunset. The lowest average monthly solar radiation was measured during May 2011 (161W/m^2). Similar low average monthly values were recorded in June 2010 (165.2W/m^2) and April 2011 (167.4W/m^2), indicating that autumn and early winter months may receive the lowest incoming solar radiation levels. Similarly, the monthly solar radiation maximums are lowest during May 2010 (722 W/m^2) and June 2011 (742 W/m^2). The greatest amount of radiation was observed on the 31 January 2011 (1 240W/m^2) between 12:00pm and 02:00pm. The maximum daily solar radiation is usually experienced between 10:00am and 12:00pm during both the winter and summer months.

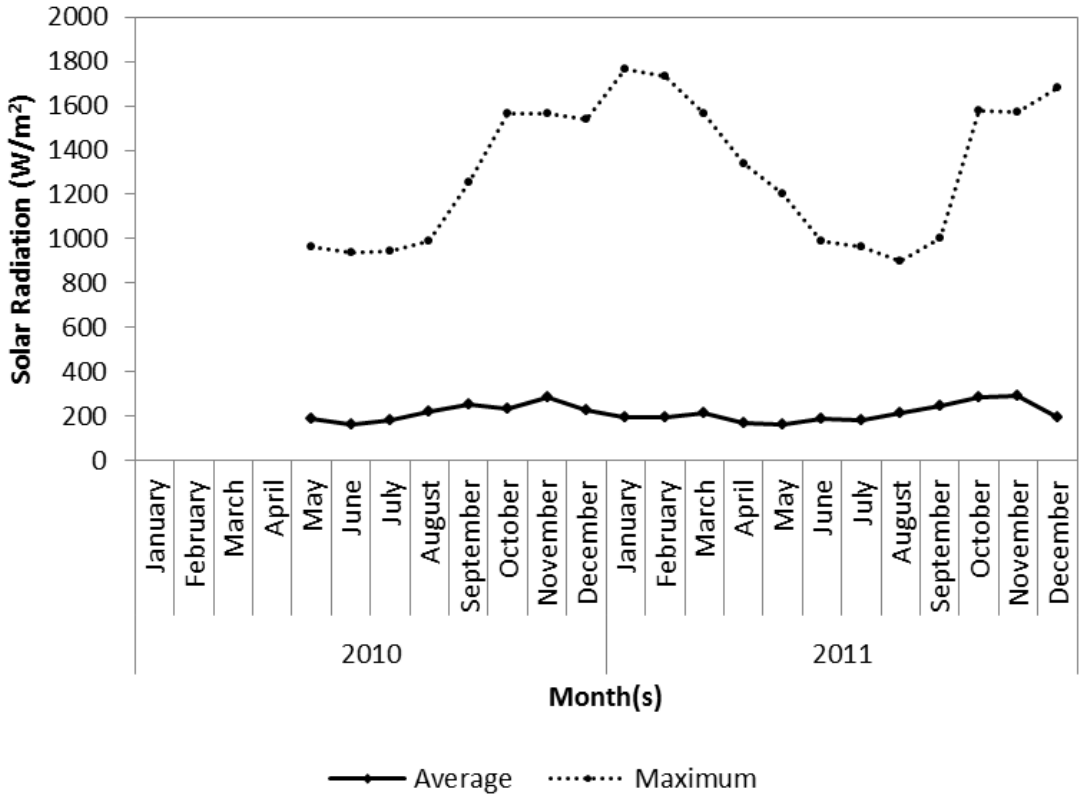


Figure 4.10: Average monthly solar radiation and maximum monthly solar radiation values for each month.

4.2.2 Soils

4.2.2.1 Moisture content

The moisture content was measured for each site in May 2009 and May 2011 using soil samples (Figure 4.11). In 2011, a soil probe was used to determine the soil moisture within each site as well as the variation of soil moisture within the sites. However, during measuring

with the soil probe in the field the soil was observed to be frozen, especially when measured in the early morning or late afternoon. The result is that the moisture probe measurements were lower than expected at some sites. Soil samples were also taken and these results are more likely to be accurate. The greatest soil moisture values were observed for site H.E. in 2011 (soil sample = 48% and the average moisture probe readings = 44%). The A.P. site experienced the lowest soil moisture in 2011 when the sample was observed to contain only 13%. The result of the Kruskal-Wallis test was a p-value of 0.725, thus the null hypothesis was not rejected (Table 4.5). Thus, there was no significant difference between the sites with regards to the soil moisture of the samples analyzed in the laboratory.

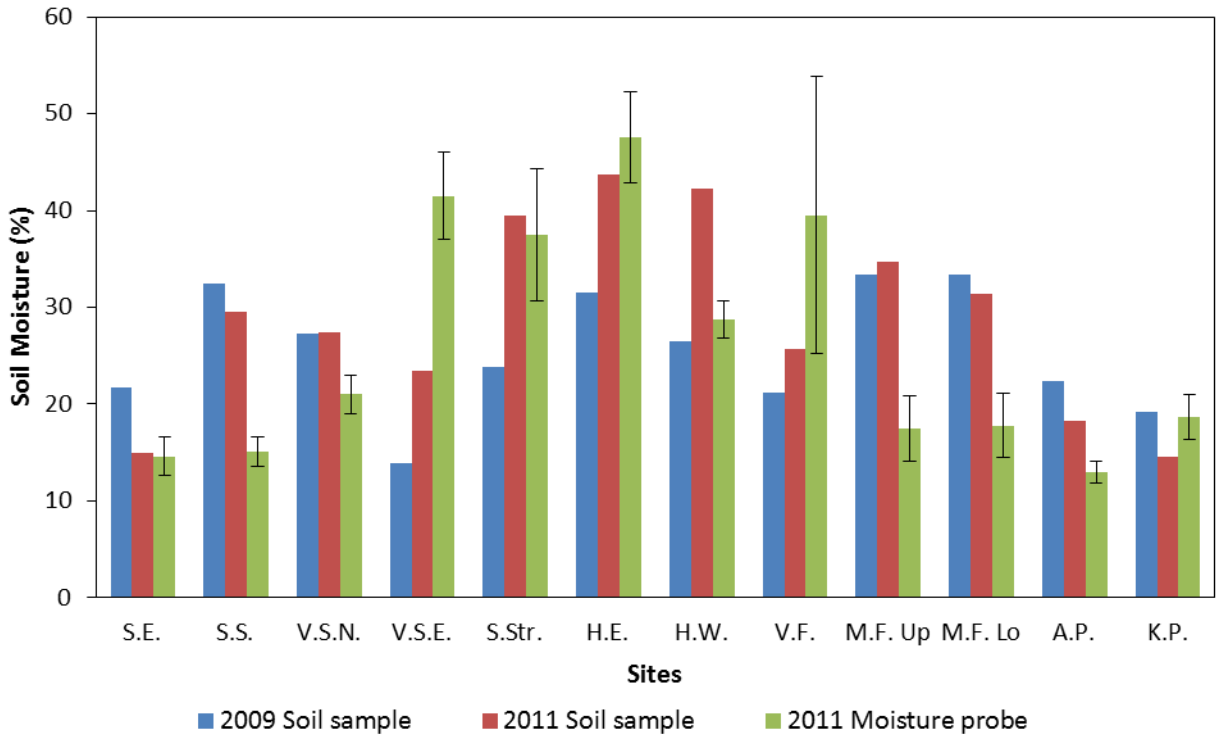


Figure 4.11: Percentage moisture content for 2009 and 2011 from soil samples and the average moisture content using the moisture probe with standard deviation.

The standard deviation and the coefficient of variation of the soil moisture probe readings (2011) indicated that the deviation within a site was greatest at site V.F. (SD = 28.56%) followed by site S.Str. (SD = 13.70%), and lowest on the A.P. site (SD = 2.31%) (Figure 4.11). The result of the Kruskal-Wallis test was a p-value less than 0.001, thus the alternative null hypothesis was not rejected (Table 4.6). Thus, there was a significant difference between the sites with regards to the 2011 soil moisture probe readings.

A simple scatter plot of the soil moisture sample results (obtained in 2009 and 2011) and the average moisture probe results for each site (2011) against aspect may be viewed in the appendices. The scatter plot suggested that the southerly-facing sites generaly experienced a higher soil moisture than northerly-facing sites in 2009 and to a lesser degree in 2011 using

the soil samples. However, this finding was reversed in 2011 when the average soil moisture probe readings were used.

Table 4.5: Results of the Kruskal-Wallis test for significant differences between the soil samples observed at each of the sites in 2009 and 2011 (n = 12).

	n	df	χ^2	p-value
Soil samples	300	11	7.868	0.725

Df: degrees of freedom

*P<0.05; **P<0.01; ***P<0.005; †P<0.001

Table 4.6: Results of the Kruskal-Wallis test for significant differences between the soil moisture probe readings observed at each of the sites in 2011 (n = 25 per site).

	n	df	χ^2	p-value
Soil moisture probe	12	11	206.988	<0.001†

Df: degrees of freedom

*P<0.05; **P<0.01; ***P<0.005; †P<0.001

4.2.2.2 Particle size distribution

The soil structure and texture influence the pore size where both water and air are stored (Araya, 2007). The efficiency of water absorption and gaseous exchange occurring at the root surface is related to the pore size (Araya, 2007). The water and air found between soil particles is also essential for the uptake of nutrients by plants (Wilkinson *et al.*, 2000). Hence, the rate and amount of plant growth may be related to the soil particle size distribution. Differences in the soil particle size distribution between sites may result in variation in the growth rates at each site. The particle distribution of the soil samples brought back from each of the study sites are presented in Figures 4.12 and 4.13.

In May 2009, when the samples were collected, additional sites had not yet been added, thus the V.F., M.F., A.P. and Kotisephola Pass sites are not included. The results from the 2009 soil particle size distribution are consistently greater than those observed in 2011. However, this is due to fewer sieves being used and if accounted for the results become very similar. K.P. had the highest gravel (>2.000mm) content (60.3%), while site H.E. had the lowest(16.8%). Site H.E. also had the highest sand (<1.000mm and >0.063mm) content (76.3%), with S.Str. (49.2%), K.P. (37.2%) and the A.P. site sharing the lowest (43.9%).

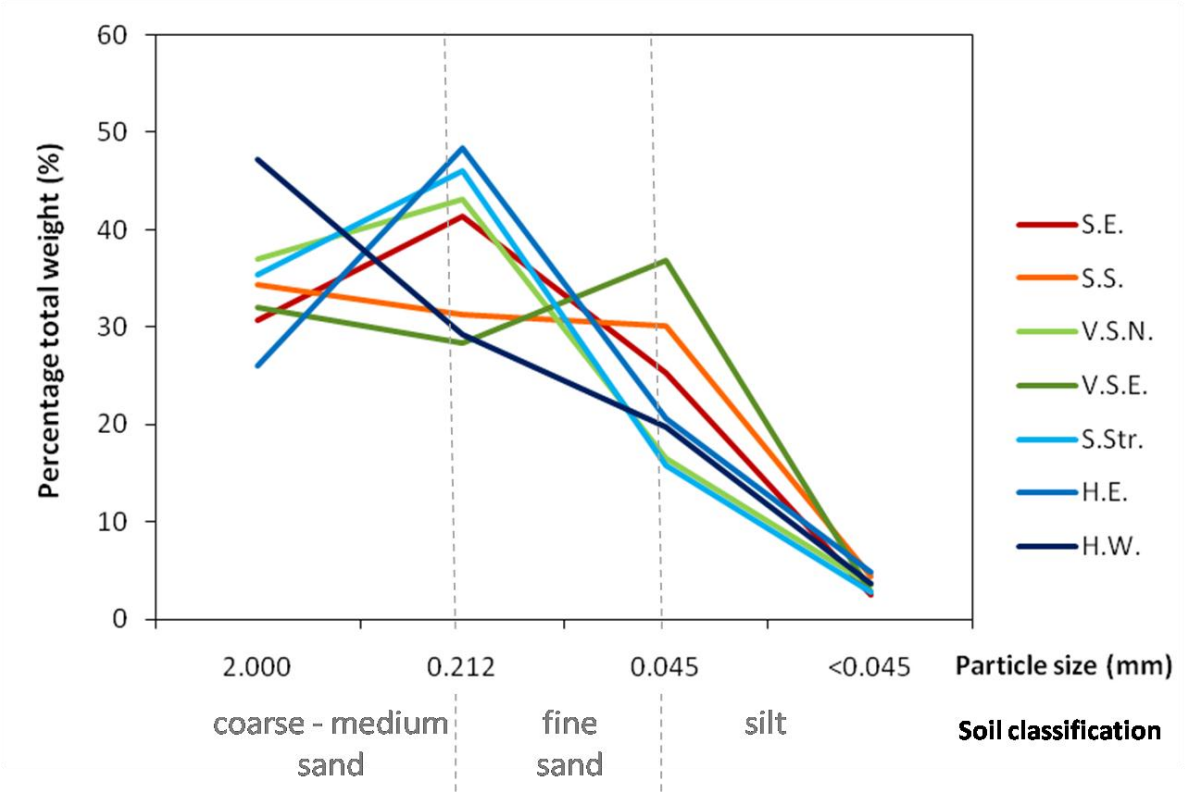


Figure 4.12: Particle size distribution of the samples taken in 2009.

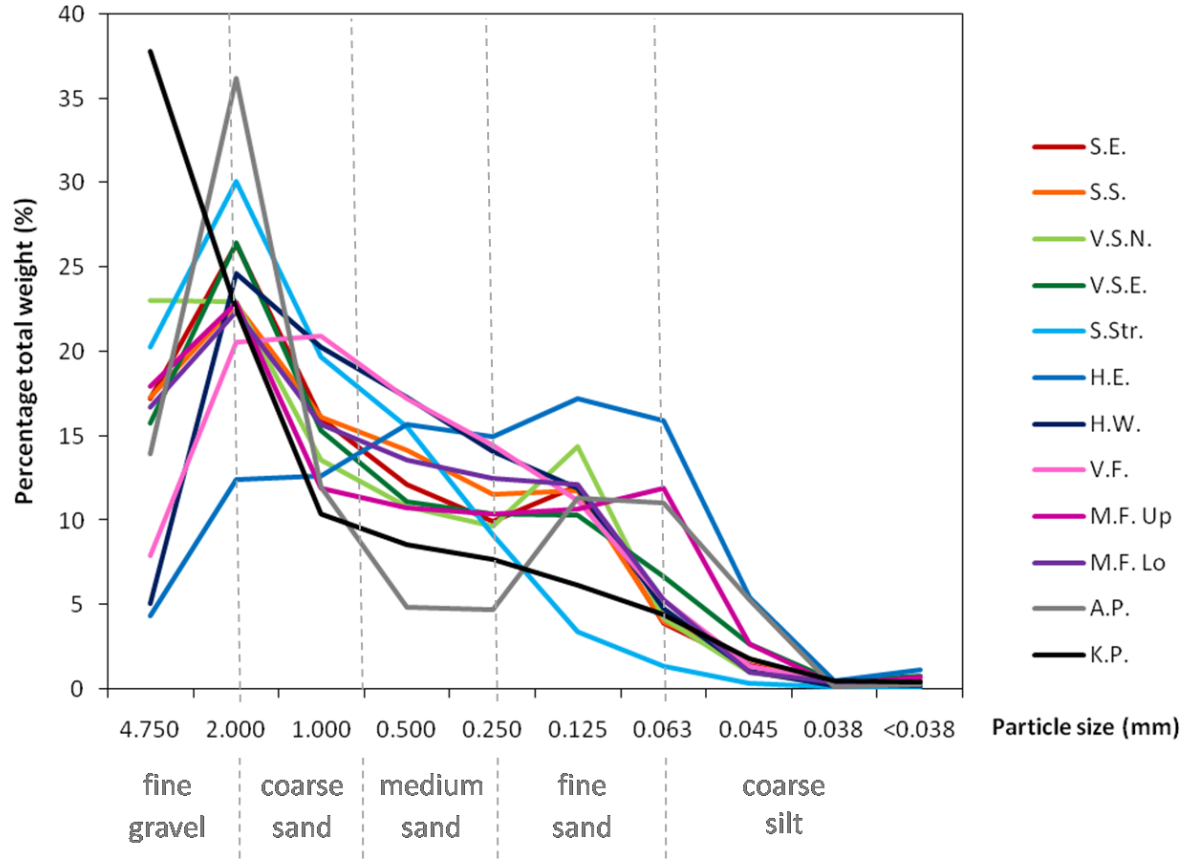


Figure 4.13: Particle size distribution of the samples taken in 2011.

Overall, small amounts of clay (<0.045mm) were observed but were highest at site H.E. (7.1%) and lowest at site S.Str. (0.7%). A one-way ANOVA was performed to determine whether a significant difference existed between the soil particle distributions at each site. The p-value obtained was 1.000, hence the null hypothesis was not rejected indicating the soil particle size distributions of each site were similar (Table 4.7).

Table 4.7: The results of an ANOVA comparing the soil particle size distributions observed at each of the sites.

Source of Variation	SS	*df*	MS	F	P-value
Between Groups	7.521	11	0.684	0.009	1.000
Within Groups	7784.16	108	72.076		

SS: sum of squares; Df: degrees of freedom; MS: mean square; F: the ratio of MS_{group} to MS_{error}; P: significance value.
*P<0.05; **P<0.01; ***P<0.005; †P<0.001

4.2.2.3 Organic content

The organic content was obtained using mid-infrared spectroscopy. The highest percentage organic content was observed at site S.S. (23%), and closely followed by that at site S.E. (21.6%; Table 4.8). Site V.S.N. had a very low organic content of 0.7% while nearby site V.S.E. had an organic content of 14.3%. The sites along the banks of streams had low organic matter contents: site S.Str. had 7.3% and site V.F. had 2.7%. Site H.E. had a lower content (2.6%) than that at site H.W. (14.1%), whilst the upper M.F. site was observed to have a greater content (17.4%) than that of the lower M.F. site (12.2%). The A.P. site had a high (20.2%) organic matter content whilst the K.P. site had an even higher value (21.4%).

Table 4.8: Percentage organic matter from each sample site.

Sites	S.E.	S.S.	V.S.N.	V.S.E.	S.Str.	H.E.	H.W.	V.F.	M.F. Up	M.F. Lo	A.P.	K.P.
% Organic Matter	21.6	23.0	0.7	14.3	7.3	2.6	14.1	2.7	17.4	12.2	20.2	21.4

4.2.2.4 Leachate pH

The pH of the soil in pH neutral distilled water is presented in Table 4.9. The mean value was 4.80, while the standard deviation between the samples was 0.307. The standard error of the mean was 0.089. The most acidic site was the A.P. site (pH = 4.45), closely followed by the lower M.F. site (pH = 4.48), whilst the most neutral site was the upper M.F. site (pH = 5.54), followed by site V.F. (pH =5.02). None of the sites had alkaline soil.

Table 4.9: pH values for each site.

Sites	S.E.	S.S.	V.S.N.	V.S.E.	S.Str.	H.E.	H.W.	V.F.	M.F. Up	M.F. Lo	A.P.	K.P.
pH	4.53	4.55	4.84	4.92	4.96	4.9	4.66	5.02	5.54	4.48	4.45	4.69

4.2.2.5 Soil chemistry

The chemical composition results from the Soil Fertility and Analytical Services laboratories indicate differences did exist between some sites for total soil concentrations of different elements (Table 4.10). The elements tested are all required for plant growth and survival, and thus will influence the study species at each site, especially where it is limiting. Macronutrients are used by plants in comparatively large amounts (e.g. nitrogen, phosphorus, potassium, calcium and magnesium) while micronutrients are used in smaller amounts (e.g. zinc, manganese and copper) and most are available from the soil (Rost *et al*., 2006; Eash *et al.*, 2008). Some macro and micronutrients were not tested either because they were unlikely to be limiting or were too costly for this study (e.g. sulphur, iron, boron, chlorine and nickel).

4.2.2.5.1 Macronutrients

All but four sites were indicted to have nitrogen contents above 0.6% (Table 4.10). Site S.E. had a 0.6% concentration, site V.S.E. had 0.4% and site V.F. had 0.6%, whilst the K.P. site had only 0.2% nitrogen concentration. With regards to phosphorus concentrations, the results indicated low (P<10ppm) to medium (P is between 10 and 20ppm) values within the soil samples for all sites tested in Lesotho. The highest value was observed for site V.S.N. (15.2ppm), while the lowest value was observed for the Kotisphola Pass site (1.9ppm). Sites situated close together or at similar altitudes did not share similar phosphorus values. The study sites range from high to low potassium levels (K<150ppm, Table 4.10), with the highest values measured for sites V.S.N. (364.6ppm) and V.S.E. (283.5ppm). The lowest value was indicated for site V.F. (49.0ppm), followed by the upper M.F. site (57.7ppm) and K.P. (57.7ppm). Sites situated on slopes were observed to have slightly higher potassium values than those near upper slopes or in flat areas. The calcium concentrations were high (4 800ppm-9 600ppm) to medium (800ppm-4 800ppm). The highest concentration was observed at site S.Str. (5 300ppm), and the lowest at the A.P. site (482.2ppm). Sites situated close together have similar values, yet sites S.Str. and V.F. have slightly higher values than at other sites. Magnesium was observed at medium (60ppm-300ppm) and high (>300ppm) levels across all sites. The highest level was observed for site V.F. (777.1ppm) and the lowest level at the A.P. site (83.3ppm, low = <60ppm). There were no observable trends in magnesium content in relation to site altitude, position in relation to water and aspect.

Table 4.10: Chemical composition of the soil for each study site.

Site	Sample density g/mL	P Ppm	K Ppm	Ca Ppm	Mg Ppm	Zn Ppm	Mn Ppm	Cu Ppm	N %	Org. C %
S.E.	0.79	12.7^{m}	154.4^{m}	1025.3^{m}	101.3^{m}	3.4^{m}	54.4^{m}	6.3^{m}	0.58	>6
S.S.	0.74	6.8^{l}	66.2^{l}	1020.3^{m}	117.6^{m}	0.9^{vl}	20.3^{m}	5.8^{m}	>0.6	>6
V.S.N.	0.79	15.2^{m}	364.6^{m}	2798.7^{m}	293.7^{m}	1.1^{vl}	15.2^{m}	4.3^{m}	>0.6	>6
V.S.E.	0.97	8.2^{l}	283.5^{m}	3672.2^{m}	590.7^{h}	1.6^{vl}	28.9^{m}	4.6^{m}	0.42	5.5
S.Str.	0.74	8.1^{l}	120.3^{l}	5300.0^{h}	585.1^{h}	2.4^{l}	48.6^{m}	5.1^{m}	>0.6	>6
H.E.	0.83	10.8^{m}	88.0^{l}	3726.5^{m}	418.1^{h}	2.8^{l}	91.6^{m}	6.0^{m}	>0.6	>6
H.W.	0.64	14.1^{m}	182.8^{m}	2210.9^{m}	242.2^{m}	3.3^{m}	87.5^{m}	6.3^{m}	>0.6	>6
V.F.	0.96	8.3^{l}	49.0^{l}	4567.7^{m}	777.1^{h}	2.9^{l}	16.7^{m}	3.1^{m}	0.57	>6
M.F. Lo	0.71	12.7^{m}	187.3^{m}	1480.3^{m}	140.8^{m}	1.0^{vl}	15.5^{m}	4.6^{m}	>0.6	>6
M.F. Up	0.78	7.7^{l}	57.7^{l}	2289.7^{m}	391.0^{h}	0.5^{vl}	14.1^{m}	4.1^{m}	>0.6	>6
A.P.	0.90	5.6^{l}	71.1^{l}	482.2^{vl}	83.3^{m}	1.2^{vl}	18.9^{m}	4.6^{m}	>0.6	>6
K.P.	1.04	1.9^{l}	57.7^{l}	3093.3^{m}	463.5^{m}	0.8^{vl}	14.4^{m}	6.1^{m}	0.22	3.7

^{vl} Very low levels; ^{l} Low levels; ^{m} Medium levels; ^{h} High levels; ^{vh} Very high levels

Phosphorus, Potassium and Magnesium levels from Horneck *et al.* (2011) and Marx *et al.* (1999)

Calcium, Zinc, Manganese and Copper levels from Jones (2001)

4.2.2.5.2 Micronutrients

The lowest concentration of zinc was observed at the lower M.F. site (0.5ppm), followed by that of the K.P. site (0.8ppm) and site S.S. (0.9ppm). Sites V.S.N. (1.1ppm), lower M.F. site (1.0ppm) and the Apostles Plateau site (1.2ppm) were all below 1.5ppm. The highest concentration was indicated for site S.E. (3.4ppm), closely followed by that of site H.W. (3.3ppm). Manganese values were medium (6ppm-150ppm) for all sites within the study region (Table 4.10). The lowest value was measured at the upper M.F. site (14.1ppm), followed closely by that at K.P. (14.4ppm), whilst the highest values were recorded for sites H.E. (91.6ppm) and H.W. (87.5ppm). The copper concentration was considered to be on the low side of optimum for all sites as the values were all greater than 3ppm (Jones, 2001). Sites situated close together appear to have similar values. The highest value was indicated for sites S.E. and H.W. which both had a value of 6.3ppm, whilst site V.F. had the lowest value (3.1ppm).

4.2.2.6 Principal component analysis

A PCA was performed to indicate which soil characteristics varied the most between the study sites. The PCA indicated that together the first four factors explain 75% of the variance (Table 4.11). The eigenvalues indicate that factor one explain 30% of the variation of the

original soil characteristics and factor two explain 19%. Seven characteristics loaded highly onto factor one: manganese and zinc concentration, soil moisture probe readings, very coarse and coarse sand, pebbles and copper (Table A.1 in the appendices). Factor two contained magnesium, calcium, soil moisture probe readings, very fine silt, fine sand nitrogen concentration and organic content. The third factor contained coarse silt, very fine sand to very coarse sand and pebbles. Only phosphorus and potassium loaded onto factor four. Pebbles, fine sand, course sand, nitrogen and organic content had negative factor scores, indicating that these characteristics decreased as the other characteristics increased.

Table 4.11: Eigen values and variance resulting from the PCA.

Factor	Eigenvalue	Variance (%)	Cumulative Variance (%)
1	6.857	29.8	29.8
2	4.427	19.2	49.1
3	3.927	17.1	66.1
4	2.123	9.2	75.4
5	1.528	6.6	82.0
6	1.514	6.6	88.6
7	1.164	5.1	93.7

4.3 Discussion

4.3.1 Climatology

4.3.1.1 Temperature and plant growth

Temperature is important to plant growth through its facilitation of metabolic reactions (Körner, 2006). Carbon fixation is one such reaction which may be limited by low temperatures (Chapin III *et al.*, 2002). However, rather than the reduction of metabolic processes, plants may link their growth to hormones which respond to temperature (Penfold, 2008; Hua, 2009). Low temperatures reduce the production of gibberellin and its accumulation, thereby reducing growth (Hua, 2009). Growth is the result of cell division, enlargement and differentiation, resulting in new tissues (Körner, 2006). Yet, growth also requires nutrients and nutrient uptake may also be influenced by temperature and root growth, especially in nutrient poor soils (Wilkinson *et al.*, 2000). The time taken for a cell to replicate at 0^{o}C is almost infinite despite the occurrence of photosynthesis (Körner, 2006). The formation and melting of ice crystals reduces the water available for cellular functioning and alters the turgor pressure (Hayashi, 2007). Similarly, the freezing of fluids within plant cells may prevent or merely reduce carbon dioxide assimilation in cold adapted plants (Körner,

2006). Hence, below a temperature of 0°C, no growth is likely to occur (Woodward *et al.*, 1986; Körner, 2003; Hayashi, 2007; Penfold, 2008). The lower limit to growth thus lies between 0°C and 2°C (Körner, 2006). Yet, despite temperatures above 2 °C, growth only becomes visible to the naked eye above 6°C (Körner, 2006). At lower altitudes this threshold may increase to approximately 5 °C (Woodward *et al.*, 1986).

The air temperatures of the Sani and Kotisephola regions frequently fall below 0°C, yet this has not been observed to kill the tussocks occurring in the study region, hence freezing tolerance strategies must be present. Grasses from high altitudes (2 500m to 4 200m a.s.l.) have exhibited a supercooling capacity to between -3 °C and -6 °C and injury between -7 °C and -18 °C (Márquez *et al.*, 2006). More specifically, *Danthonia secundiflora*, closely related to the study species, indicates a supercooling capacity to -4.5 °C (±0.4 °C) and injury at -10.9 °C (±1.2 °C) (Márquez *et al.*, 2006). Survival of cold adapted plants below -10 °C or -20 °C is rare (Hayashi, 2007). Temperature may be most influential to leaf elongation at night when temperatures reach their lowest (Busso and Richards, 1992). The majority of the hours recording temperatures below 0°C in this study were recorded during the night and particularly early hours of the morning.

The upper temperature limit to growth generally occurs between 46 °C and 56 °C (Körner, 2006); temperatures which have not been recorded within the study region both in the current study and in past climatic studies. The upper temperature limit for many alpine and arctic plants lies between 35 °C and 40 °C (Hayashi, 2007). Tussock grasses are known to alter the adjacent air (e.g. Hedberg and Hedberg, 1979; Monteiro *et al.*, 2011) and soil temperatures (e.g. Chapin III *et al.*, 1979) by as much as 8 °C in the case of soil warming in the central tussock zones. Hence, despite air temperatures of below 0°C, it is possible for slow growth to occur within the central tussock tissues. Temperature may also influence, particularly in terms of the rate of development, other plant processes such as tiller formation and flowering (Körner, 2006; Turnbull *et al.*, 2012). Similarly, sensitivity to temperature extremes may vary according to the life cycle stage, with mature plants being more resistant (Osmond *et al.*, 1987; Woodward and Williams, 1987).

4.3.1.2 Precipitation and plant growth

Water is essential to many plant processes such as the synthesis of carbohydrates as well as physical functions such as cooling (Araya, 2007). Water plays a role in the photosynthetic process, through splitting water into hydrogen and oxygen which releases electrons required for growth (Chapin III *et al.*, 2002). Leaf elongation has been directly linked to water

availability, with increases in available water increasing growth (Woodward and Williams, 1987). Seedling survival is also dependent on water availability while mature plants are more likely to survive periods of stress (Osmond *et al.*, 1987).

Water may limit growth as a lack of water reduces the ability of plants to match carbon dioxide levels with radiation levels (Chapin III *et al*., 2002). Plants in arid areas have adapted to prevent water loss through behavioural (e.g. reduced stomata activity) and physical (e.g. reduced leaf area) means (Chapin III *et al*., 2002). The ability of plants to deal with water stress is partially dependent on the life strategy of each species (Chapin III *et al*., 2002). Essentially, water limitations may reduce leaf elongation by reducing cell division and expansion rates, and hence, reduce the growth rate (Wolfson and Tainton, 1999; Skirycz and Inzé, 2010). Reducing the growth rate of tissues may increase the survival rate of the plant (Skirycz and Inzé, 2010). Alternatively, water may be in excess resulting in waterlogged soils, where the soil pore spaces are filled with water (Araya, 2007). Waterlogging results in limited air within the soil, and consequently limits gaseous exchange (particularly concerning oxygen) occurring at the roots (Osmond *et al*., 1987; Araya, 2007). Limited growth may result from either water deficits or water excess associated with limited or abundant precipitation. The availability of water for growth is also an important factor to be considered. Frozen water is unavailable for plant use until it melts, hence snow cover does not promote growth. However, snow cover can be beneficial to plant survival through insulation against extremely low temperatures and desiccation (Körner, 2006). The depth and the duration of the snow cover essentially determines the survival and effects on growth of tussocks (Körner, 2006).

Rainfall following a burning event can have a negative impact on infiltration through an increase in runoff and evaporation and thus reduce soil moisture relative to that of an unburnt area (Mark, 1994; Whelan, 1995; O'Dea and Guertin, 2003; Vermeire *et al.*, 2005). Increased runoff may also lead to increased erosion. Additionally, burning can significantly reduce water-use efficiency resulting in lower production (Snyman, 2003). The results of grazing when tussocks are experiencing water stress are species specific (e.g. Flemmer *et al*., 2002), with the resposes of *M. drakensbergensis* under water stress remaining largely unknown.

4.3.1.3 Wind, burning and tussock grasses

The presence of wind and the wind speed can determine the type of fire that occurs through its influence on fire intensity and behaviour. In the presence of wind, the fire may be classified as a head fire (burning with the wind direction) or a back fire (burning against the

wind direction), and each will have particular effects on the vegetation (Everson *et al*., 1985b; Whelan, 1995; Snyman, 2003). Head fires tend to reach greater flame heights than back fires and spread rapidly, while back fires spread slowly, burning at a higher intensity (Whelan, 1995; Trollope *et al*., 2002; Snyman, 2003). Hence, back fires may be more harmful towards vegetation by increasing the amount of damage to tissues. Post-burning conditions may also alter the wind characteristics of an area. Wind speed and duration may increase at ground level after burning due to the removal of aerial plant organs (Whelan, 1995).

Wind may play both a direct and indirect role towards growth rates. The mixing of air layers by wind may cause a cooling effect or alter a favourable growing temperature (Wilson, 1959). This may be particularly influential in late autumn, winter and early spring when temperatures are suitable for growth for shorter periods of time. Wind may also be destructive towards tussock leaves, especially during the winter months when water stress may cause the leaves to become brittle and easy to break (Wilson, 1959). This may apply more strongly to the aerial parts of tussocks (Wilson, 1959). Additionally, in grasslands wind forms a mechanism by which seed may be distributed (Mark, 1964; Hay *et al.*, 2008).

4.3.1.4 Solar radiation and plant growth

Solar radiation provides the energy necessary for photosynthesis and thus plant growth (Granger and Schulze, 1977; Robinson, 2007). The level of solar radiation reaching plants varies with season, diurnal changes, water vapour content, pollution, topography, elevation and latitude (Granger and Schulze, 1977; Tyson and Preston-Whyte, 2000; Robinson, 2007). The low values experienced in winter each year may also be the result of increasing aerosols in the air as a result of the increasing wind speeds observed and reduced precipitation. The solar radiation received reaches a maximum in December and January each year as months receive the longest hours of solar radiation. The peaks observed in June 2010 and July 2011 may be the result of increased albedo due to snow falls which occurred (Tyson and Preston-Whyte, 2000). The peaks are associated with increases in precipitation (Figure 4.7) which may also reduce the aerosols, increasing the solar radiation. The average monthly solar radiation values for December fall within the rage indicated by Taele *et al.* (2007; 220.3W/m^2 to 329.2W/m^2) as observed in Maseru, while the June values recorded (165.2W/m^2 in 2010 and 189.2W/m^2 in 2011) are slightly greater than those observed by Taele *et al.* (2007; 119.2W/m^2 to 156W/m^2) and Gopinathan (1991; 138.9W/m^2 to 173.6W/m^2) across Lesotho.

Plants have adapted to the variations in solar radiation through altering their light-harvesting, energy creation and energy storage processes (Osmond *et al.*, 1987). North-facing slopes receive the most sunlight, especially in mid winter when the solar angle is low (Hilliard and Burt, 1987). This will have an effect on vegetation distribution. Both excess light and reduced light may form limitations on growth. Excess solar radiation may result in a drying of the soil and potentially drought, thus resulting in water stress for the plants. UV-B damage has also been suggested as a cause for low growth rates (Hofstede *et al*., 1995). Conversely, photosynthesis has been observed to decrease with decreasing radiation (Körner, 2006). Similarly, low light intensity can result in the accumulation and inefficient use of nitrogen within plant tissues (Wilkinson *et al*., 2000). Hence, the growth rate will decrease with decreasing intensity. Light intensity has also been linked to inflorescence production (Mark, 1964). Tussocks receive the most solar radiation at the periphery while the central regions may be shaded (Ryel *et al*., 1994; Monteiro *et al.*, 2011). Yet, the cost of receiving the most light also means increased exposure to climatic features and grazing (Monteiro *et al.*, 2011). Additionally, the distance between individual tussocks may also play a role, with shading occurring between individuals in clumps (Ryel *et al.*, 1994).

4.3.1.5 Alpine Environments and Climate change

Alpine environments were recognized early on to be in danger of considerable species loss due to climate change (Pauli *et al*., 1996; Guisan and Theurillat, 2000; Grabherr *et al*., 2010). Despite this, little is known about the effects of a warmer climate on alpine regions (Klanderud and Totland, 2005). However, climate is the main dictator of species occurrence and distribution in alpine environments. Thus, alpine vegetation is more likely to be affected by changing climates than vegetation at lower altitudes.

Species adjust to climate change through shifts in distribution (or range) or adaptation of their phenology and/or physiology (Davis and Shaw, 2001; Bellard *et al*., 2012). Range shifts often take the form of migrations in animals to areas more suited to their climatic range of tolerance. However, the capacity of vegetation and other stochastic organisms to move is limited and species specific (Kardol *et al*., 2010). What follows is a slow movement involving seed dispersal, seedling establishment, growth and eventual seed production and dispersal (Davis and Shaw, 2001). However, in alpine environments where climatic gradients are steep and ranges are decreasing, seed dispersal may play a much smaller role in predicting future ranges (Engler *et al*., 2009). There is a fine line between species migrating with optimum conditions due to climate change and invasion of invasive species into areas more suited to their growth. The rapid colonization of invasive species often is detrimental to the

abundance and even existence of slower moving native species. Similarly, increasing growing season length may be more advantageous to lower altitudinal species which are then able to outcompete higher altitudinal species (Grabherr *et al*., 2010). The result may be a decrease in species diversity, altered species compositions and reduced grassland extent (Hoiss *et al*., 2013). The fossil record shows that there are limits to adaptation when climate change is rapid (Davis and Shaw, 2001). The potential for adaptation may be lost in many plant species with rapid climate change (Jump and Peñuelas, 2005). This is especially true for fragmented landscapes where species are long lived and have slow life cycles (Figure 4.9).

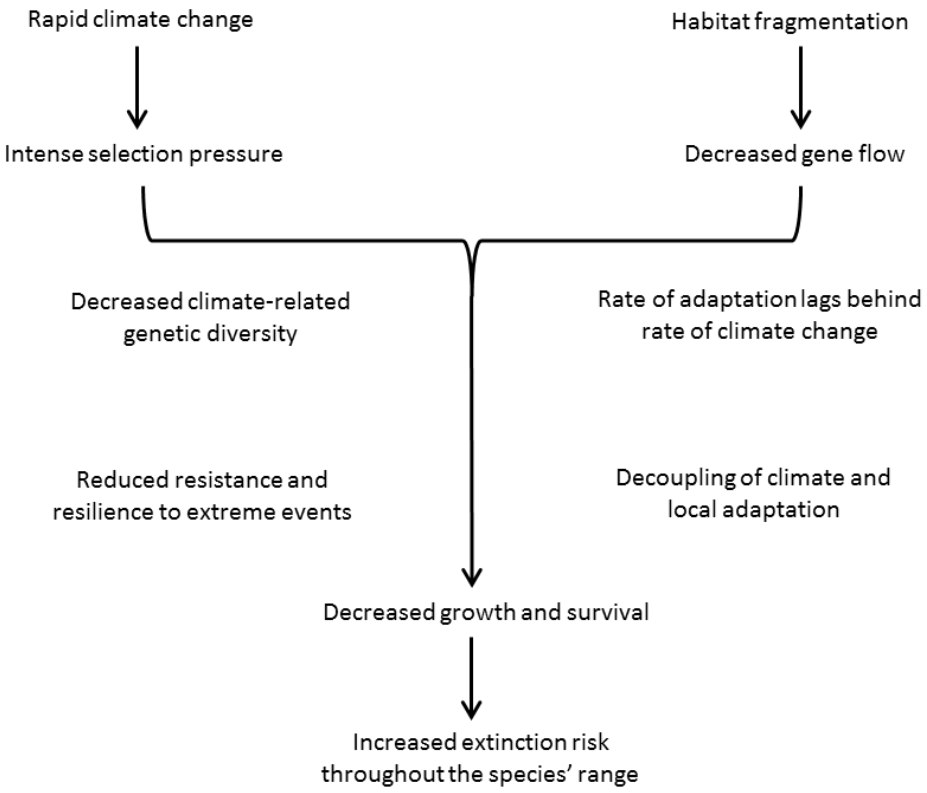


Figure 4.14: Diagram showing the relationship between rapid climate change and range fragmentation (Jump and Peñuelas, 2005).

Rising temperatures cause shifts in range towards higher elevations (Pauli *et al*., 1996; Guisan and Theurillat, 2000; Walther *et al*., 2002; Engler *et al*., 2011; Grabherr *et al*., 2011; Hoiss *et al*., 2013). The consequence is the formation of 'alpine refugia' where alpine plants are restricted to mountain tops and separated by areas of lower altitude and higher temperature. Range fragmentation may aid survival of species through facilitation of genetic adaptation (Saavedra, 2002), hence, the high endemism common to many alpine regions (Crawford, 2008). However, the formation of 'refugia' acts to prevent genetic flow between populations and communities. Reproduction and migration do not occur readily as seed would need to be transported through either wind or by animal traffic (Crawford, 2008). Thus, the isolation of plant species may be both biological and physical (Crawford, 2008).

As temperatures rise, species are able to move up slopes to higher altitudes whilst at the highest altitudes plants have nowhere to go, highlighting the importance of local topography (Engler *et al*., 2011). Slopes get steeper, permafrost becomes more prevalent, snow cover increases and soils become shallower with increasing altitude in some mountain environments (Guisan and Theurillat, 2000). In such situations, the further up species are forced to move the less surface area there is for colonization (Guisan and Theurillat, 2000; Hoiss *et al*., 2013). Data from the European Alps in 1992 and 1993 were compared to data for 1996 and show increases in species richness for the majority (70%) of summits (Pauli *et al*., 1996). It is suspected that this study looked at the movement of sub-alpine species into alpine zones and did not look at 'true' alpine species. Later analyses using 'true' alpine species and sub-alpine species showed that the 'true' alpine species were at greatest risk to extinction as they began with smaller ranges and had nowhere to go (Guisan and Theurillat, 2000; Dullinger *et al*., 2012).

The migrational rate of many alpine species is modeled to be slower than the upward movement of isotherms generating a lag between them (Pauli *et al.*, 1996). As migration of many species may be too slow, the role played by adaptation is essential (Jump and Peñuelas, 2005). Adaptation may be either phenotypic or genetic. Genetic adaptations are slow and dependent on survival of new individuals as well as genetic flow and species abundance (Jump and Peñuelas, 2005). Range fragmentation acts to reduce both species numbers and genetic flow. The results are often inbreeding and increased genetic drift resulting in both extinctions and speciation. Genetic adaptations may vary between individuals and populations (Jump and Peñuelas, 2005).

The timing of life strategy activities or phenology has been well documented in Europe since the 1960s (Walther *et al*., 2002). Species are able to alter the timing of life cycle events on daily and seasonal scales (Bellard *et al*., 2012). Temperature is the main trigger of flowering for many alpine species, as this signals the start of the growing season (Hülber *et al*., 2010). Studies have demonstrated a general trend towards earlier onset of spring, and hence activities such as earlier flowering and leaf production (e.g. Grab and Craparo, 2011). The lengthening of the growing season into the winter months is indicated but results are contradictory between species and location (Walther *et al.*, 2002). Changes in phenotypic functioning could mean that flower and pollinator occurrence occur at different times or with a reduced overlap (Saavedra, 2002). This would reduce the reproduction of the plants and may negatively influence the pollinators and have a knock on effect for seed eaters.

Additionally, the altered flowering times may also mean altered interactions between plants and increased competition for resources (Hülber et *al.*, 2010).

Higher temperatures associated with climate change should cause an earlier snow melt and thus an earlier start to the growing season (Saavedra, 2002). Earlier snow melt may result in drier soils being experienced early in the growing season and warmer soil temperatures than in the past.The response of plants will likely be in temporal shift in phenophases. A study on the alpine species *Delphinium nattallianum* in the Colorado Rocky Mountains observed a small but significant increase in seed numbers and weight with earlier snow melt and flowering (Saavedra, 2002). The New Zealand species *Chionochloa* recorded an increase in the number of inflorescences, and thus seeds produced, but at the cost of reduced seed viability and size (Rees *et al.*, 2002). Observations from further research suggest that in the longer-term, density of plants may be more negatively impacted than the number of seeds produced (Saavedra, 2002). However, phenotypic changes tend to ensure a species survival with short-term changes rather than long-term changes in climate (Jump and Peñuelas, 2005). However, earlier snowmelt and adaptation to a longer growing season may leave plants more vulnerable to spring cold days, as observed in the Swiss Alps (Wipf *et al.*, 2009). Coupled with earlier snow melt and milder winters is the increased persistence of herbivores within habitats which were previously unsuitable for their existence in the winter months. However, alpine grasslands were observed to increase in growth and thus productivity under warming conditions, on the northeast Tibetan plateau (Guo *et al*., 2012). Yet, this may lead to an increase in grazing of grasslands which has effects on grassland composition, erosion and the fire regime. Changes in phenology and range shifts can cause disruptions to communities and between species interactions.

It is difficult to distinguish between changes due to land use and those due to climate change (Crawford, 2008). Land use changes limit the flow of genes (and thus adaptation) and movement of species to more suitable habitats and may increase the number of extinctions experienced (Davis and Shaw, 2001; Jump and Peñuelas, 2005). Currently, alpine ecosystems may be threatened more by land use change than by climate change (Grabherr *et al.*, 2011). Similarly, 'island' communities such as those at mountain summits may be left outside their tolerance ranges with no place to go, and may thus become extinct (Jump and Peñuelas, 2005). Additionally, failure to adapt through changes in range, phenology and physiology may result in extinction at local, regional or global scales (Bellard *et al*., 2012). A hybrid model using species range and seed dispersal knowledge was used to predict alpine vegetation in the European Alps, and found a reduction in range of 44 to 50% by the end of

the century (Dullinger *et al*., 2012). Endemic species were observed to face the greatest risk of extinction through the greatest range reductions (Dullinger *et al.*, 2012). Supporting these findings, 19-46% of montane species, 31-51% of subalpine species and 36-55% of alpine species were observed to lose more than 80% of their niches by the end of the century under four different modeling conditions (Engler *et al*., 2011). With between 15% and 30% of sampled species becoming 'committed to extinction' according to a study by Thomas *et al*. (2004), climate change is likely to have a significant impact on the survival of the species within the study area. Thus, the impacts of climate change need to be considered in this study, despite the area being within a protected area. Protected areas can only offer a limited amount of protection against rapid climate change (Dudley, 2003). Hence, the management of alpine grassland areas may play an important role in the future composition and ranges under a changing climate (Hoiss *et al*., 2013).

4.3.1.6 Climate change and *Merxmuellera drakensbergensis*

Merxmuellera drakensbergensis (and similarly *Merxmuellera macowanii*) are found in alpine areas with a high soil moisture content. A reduction in soil moisture may negatively influence *M. drakensbergensis* distributions. Consequently, *Merxmuellera* populations are likely to become smaller and more fragmented, which is likely to impede genetic flow and reduce resistance to disease (pests and disturbance) and assist genetic drift. However, the Drakensberg Mountains may experience increased precipitation and potentially rainfall intensity (Hewitson and Crane, 2006; Haensler *et al*., 2011). Thus, *Merxmuellera* distributions may expand. Additionally, a recent study indicates nitrogen availability may also possibly increase due to greater activation of the nitrogen mineralization process by microbial action under increased temperatures (30^{o}C; Carbutt *et al*., 2013). This may result in increased plant growth of sub-alpine species under warmer conditions.

Alterations to the *Merxmuellera* communities are likely to affect other populations through interspecific relationships, such as mutualism (Bellard *et al*., 2012). During the course of the current study, many smaller reptiles (e.g. the striped skink: *Mabuya striata*) and insects were observed to shelter within the *Merxmuellera* communities. Hence, alterations to the physiology, range and phenology of the study species may impact the survival of other species within the ecosystem. The production of seed by *Merxmuellera* individuals and communities is significant. However, the observed low success rate of seedlings reduces both the genetic flow and the migrational capacity of the species. New seedlings represent lush green grass and may thus be grazed along with other favourable grazing species. This would greatly reduce the success rate of seedlings becoming adult individuals. In order for

successful seedling establishment to increase, it is suggested that the number of herbivores be reduced and time for recovery towards the end of the growing season be increased. The low survival rate of *M. drakensbergensis* seedlings suggests that vegetative reproduction through fragmentation of senescent adults would be the main means of migration. However, this is a very slow progressive movement as it may take years for adults to fragment. This suggests that migration may not be fast enough and populations may become isolated. A study involving the sedge species *Carex bigelowii* at Toolik Lake, Alaska, showed that an increase in mean air temperature of 4-5^{o}C resulted in a significant decrease in the number of robust seeds produced (Hobbie *et al*., 1999). This may suggest that increased temperatures associated with anthropogenic climate change may further reduce the ability of tussock grasses to migrate and adapt to the warmer conditions. However, at the same time, more seed was produced through an increase in the number of inflorescences produced (Hobbie *et al*., 1999). What requires further study is to determine how many of these seeds, with reduced weight and assumed reduction in viability, are able to germinate successfully.

The alpine areas may eventually become invaded by sub-alpine heath, according to Pauli *et al.* (1996). Chapin *et al.* (1995) was able to show changes had been made in terms of species dominance under conditions of slightly warmer climates near Toolik Lake, Alaska. While Klanderud and Totland (2005) used simulated warmer temperatures and observed changes in terms of species dominance in Hardangervidda, southwestern Norway. It is possible that sub-alpine species are able to migrate at a faster rate than alpine species due to faster life cycles. Similarly, species with longer life cycles or slower dispersal and reproductive strategies are more vulnerable to being isolated as they are less able to adapt and migrate (Jump and Peñuelas, 2005). Thus, faster moving sub-alpine species are likely to invade the current ranges of alpine species. The current alpine species may become isolated on 'islands' of cool conditions while the coolest areas disappear under an increase of 2^{o}C (Scherrer and Körner, 2011). Competition for cool areas will increase (Scherrer and Körner, 2011) with faster growing species able to outcompete slower growing alpine species unable to take advantage of warmer conditions. However, productivity may increase with increasingly early starts to the growing season. Temperature increases of up to three Kelvin may not increase plant metabolic rates resulting in few observed effects (Körner, 2003). The progression of grasses to shrubs as shown by Chapin *et al.* (1995) may be the result of other factors such as an altered fire regime. Fire is required to prevent the encroachment of shrubs and thus a lack of burning may permit the encroachment of shrubs. Yet, despite the loss of the coolest environments, many species may find other suitable micro-climates within a few meters of their current locations due to the high topographical variability in alpine environments

(Scherrer and Körner, 2011). Additionally, extinctions may be delayed until 2080-2100 due to the ability of alpine species to shift their niches to more suitable climates (Engler *et al.*, 2009).

4.3.2 Soils

4.3.2.1 Limitations

Single samples were taken from within the central regions of each *Merxmuellera* community across the study valley, but this does not provide a good indication of the variability within each site. The organic carbon content, leachate pH and the various chemical concentrations may vary the most between the central regions of each community and those at the periphery. In future research, altering the soil sampling strategy to include a greater number of samples spread across the sampling sites would increase the accuracy of the results, and potentially provide greater information which may assist in the explanation of results later in the study. For further research, a grid sampling technique is suggested within the inter-tussock spaces at intervals dependant on the community size. This will allow for samples to be collected from a variety of locations within each community, without bias towards community size, whilst still maintaining a number of samples that is temporally and financially cost effective.

4.3.2.2 Soil moisture

Statistical tests indicated there were no significant differences between the soil moisture observed within the samples collected in 2009 and 2011 ($p = 0.725$). However, the soil moisture probe readings (undertaken in 2011) indicated significant differences between the different sites ($p<0.001$). In the field, the soil moisture within many sites was observed to be frozen in the early morning and late afternoon. Thus, the soil probe readings measured the available soil moisture to the plants rather than the total soil moisture. The availability of soil moisture to plants will become less influential on the growth of the *M. drakensbergensis* individuals as air and soil temperatures, as well as precipitation, increase towards the summer months, and soil moisture is more readily available throughout the day. Between 20-80% of days in January experience soil moisture stress across Lesotho, and increases to between 80 and 100% in July (Marake *et al.*, 1998). Additionally, during early spring and autumn the difference in site moisture may have an influential role in *M. drakensbergensis* growth. The driest sites will be the most affected, as is shown through samples at site V.S.E. in 2009 and site S.S. in 2011. The drier the site, the greater the need for moisture to stimulate growth, thus the more important freezing becomes to tussock growth. The wettest sites will thus be the least effected and were recorded as being site S.S. in 2009 and sites S.Str., H.E. and H.W. in 2011. Site M.F. (both upper and lower) was consistently recorded as having a high moisture

content. The soil moisture content may play a role in the recovery of the *M.drakensbergensis* communities between burning events and may explain differences observed between sites.

4.3.2.3 Particle size distribution

The soil particle size distribution of each site are similar (p = 1.000). Thus, the soil particle sizes are unlikely to play a role explaining the differences observed in *M. drakensbergensis* growth between sites.

4.3.2.4 Organic content and leachate pH

According to Schmitz and Rooyani (1989) the organic content of the Lesotho highlands varies between 10 and 20%, depending on the microclimate and location. The organic contents observed in this study ranged between 1% and 23%, confirming the findings of Schmitz and Rooyani (1989). The presence of grassland has helped to develop a high organic matter content within the soil, which is higher along the valley floor than on the slopes (Herbst and Roberts, 1974; Schmitz and Rooyani, 1989). The organic matter assists to increase the soil pH, which is generally neutral (Ranthamane, 2008). Highland soils generally have a high pH due to high in aluminum, silica and calcium concentrations (Guillarmod, 1969). Soil pH influences the availability of nutrients and the solubility of phosphate molecules and metallic micronutrients (Chapin III *et al*., 2002). The soils of the study region had a mean leachate pH of 4.8. Acidic soils may be a product of the type of parent rock and high rainfall, and are thus often highly leeched of soluble molecules. The acidity of the soils makes it difficult for bacterial growth reducing the rate of decomposition, allowing the organic content to increase.

4.3.2.5 Soil chemistry

Nitrogen is considered an essential element in the creation and functioning of amino acids, nucleic acids, chlorophyll and proteins (Rost *et al.,* 2006; Eash *et al.,* 2008). Hence, nitrogen is required in larger quantities than other nutrients (Wilkinson *et al.,* 2000). Increasing nitrogen can enhance growth and demand for other nutrients (Wilkinson *et al.,* 2000). Soil nitrogen is only available to plants in the form NH_4^+ and NO_3^-, and levels of available nitrogen to plants depend largely on the amount of bacteria in the soil (Black, 1957; Rost *et al.,* 2006). In the cooler climate of the Lesotho highlands, bacterial and fungi functioning would be limited or slowed, reducing the amount of nitrogen available for plant use, particularly during the winter months (Carbutt *et al*., 2013). This may explain the low concentration (0.2%) observed in the soil at the colder Kotisephola Pass site. Additionally,

the nitrogen content of leaves was observed to increase with increasing altitude in various subarctic, temperate, subtropical and tropical mountain environments (Körner, 1989). This would also decrease the amount of nitrogen in the soil. Nitrogen deficiency is generally indicated by a yellowing of the lower leaves, reduced number of leaves and stunted growth (Black, 1957; Rost *et al.*, 2006). Yellowing of the lower leaves was not observed at the K.P. site despite the low nitrogen concentration.

Phosphorus is used in energy transfer mechanisms within the plant (Eash *et al.*, 2008), as well as in nucleic acids (Rost *et al.*, 2006). Hence, low phosphorus levels may result in slow growth rates (Eissenstat and Caldwell, 1987). The lowest value was observed for the Kotisphola Pass site (1.9ppm) and is potentially limiting to growth. A phosphorus deficiency is evident through reddish-purple leaf colouring in young plants, a factor not observed in the study species at any of the sites (Eash *et al.*, 2008). The amount of available phosphorus to plants is related to the level of biological activity within the soil. Sites situated close together or at similar altitudes do not share similar values.

Potassium is an essential element, and thus is required in large quantities for water uptake and transport within plant tissues (Wilkinson *et al.*, 2000; Rost *et al.*, 2006; Eash *et al.*, 2008). Potassium may be limiting towards growth at site V.F. (49.0ppm), followed by the upper M.F. site (57.7ppm) and K.P. (57.7ppm). The lower values may be explained by the role of potassium in cold temperature survival, with plants at the cooler, higher altitudes requiring greater amounts of potassium from the soil (Wilkinson *et al.*, 2000). Potassium deficiency is often indicated by brown edges of lower leaves, a factor difficult to distinguish from natural dieback in the study species (Eash *et al.*, 2008). Sites situated on slopes are observed to have slightly higher values than those near upper slopes or in flat areas.

Acidic soils frequently experience low calcium levels (Horneck *et al.*, 2011; Marx *et al.*, 1999), yet that is not the case in the study region as all values are high (1 500ppm-2 500ppm) to medium (1 000ppm-1 500ppm) (Table 4.10). Sites situated close together have similar values, and sites S.Str. and V.F. which are near streams have slightly higher values than those at other sites. The flatter the land, the lower the indicated calcium value. Calcium is important in the formation of calcium pectate, a cell wall component, and a reduced calcium availability is reflected in few apical root tips and terminal buds (Eash *et al.*, 2008).

Magnesium is observed at medium (60ppm-300ppm) and high (>300ppm) levels across all sites and is thus not a limiting factor to growth. It is an important nutrient in chlorophyll

production and the uptake of phosphorus, yet very high levels can reduce the amount of water able to enter the soil (Rost *et al.*, 2006; Eash *et al.*, 2008). Zinc plays a role in the metabolism of the plant hormone auxin, which is involved in cell elongation (Eash *et al.*, 2008). Zinc concentrations above 1.5ppm are considered adequate for crop growth (Horneck *et al.*, 2011), however, levels are only above this value at some of the sites (Table 4.10). The lowest concentration is observed at the lower M.F. site (0.5ppm), followed by that of the K.P. site (0.8ppm) and site S.S. (0.9ppm). Additionally, sites V.S.N. (1.1ppm), lower M.F. site (1.0ppm) and the Apostles Plateau site (1.2ppm) are all below the 1.5ppm threshold. A zinc deficiency often manifests as white or yellow areas between veins on the upper leaf surface, a condition not observed at any of the sites (Eash *et al*., 2008).

Manganese values are medium (15ppm-29ppm) to very high (>50ppm) for all sites within the study region (Table 4.10). Manganese has been observed to differ within plant tissues between burnt and unburnt areas (O'Connor *et al.*, 2004). Yet, there is no observable trend between site positions and manganese values. Manganese is used in plants in enzyme functioning and electron transport (Eash *et al.*, 2008). Values of copper concentration above 0.6ppm are considered adequate for plant growth (Horneck *et al*., 2011), thus copper availability is unlikely to be a limiting factor at each of the sites as the values are all greater than 1.5ppm (Jones, 2001). Sites situated close together appear to have similar values. Copper deficiency may be observed as yellowing of stunted young leaves as a result of decreased enzyme functioning (Eash *et al*., 2008); a factor difficult to separate from dieback.

4.3.2.6 Principal component analysis

The PCA indicated that the majority of the variation between the *M. drakensbergensis* communities at each site could be explained by factors one, two and three (66%). Factor one included micronutrients and soil moisture probe readings, while factor two included macronutrients and soil moisture probe readings. Factor three included fine sands and coarse silt. These factors may prove useful in explaining the observed growth rates in Chapter 9. The ordination diagram may be seen in the appendices (Figure A.2)

4.4 Summary

Climatic and soil data were quantified due to their influence on growth. The results may help to explain some of the findings presented later. The following are the main findings:

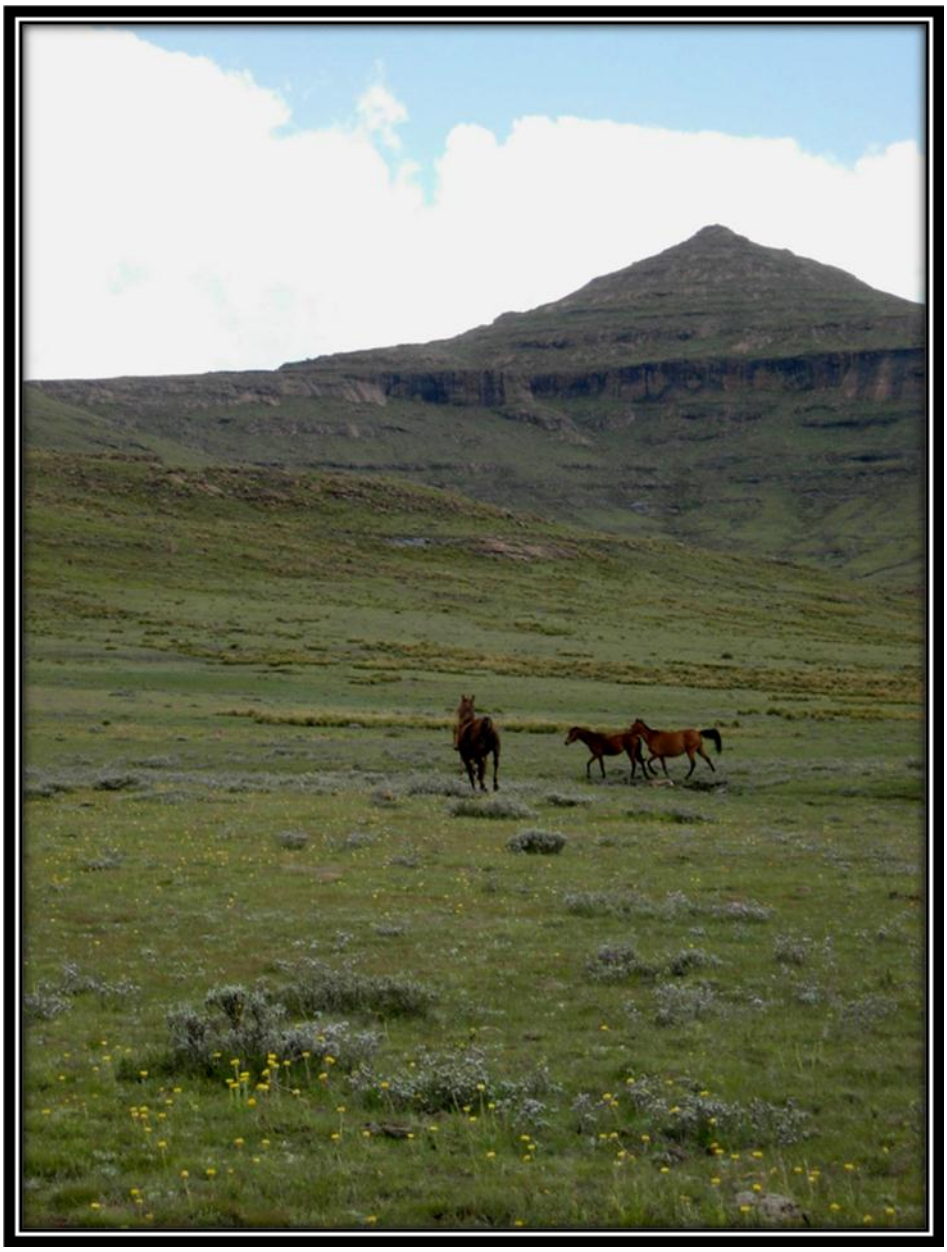
1. In the Sani region, the months experiencing the greatest percentage of hours below 2 ^{o}C were June and July each year and below 0 ^{o}C in July (more than 55%).

Temperatures below -10 ^{o}C, when damage to plants is most likely, were recorded on 6 days in 2009, 7 days in 2010 and 11 days in 2011, all in June, July and August.

2. In the Kotisephola region, between May and July in 2011, more than 70% of the days were below 0 ^{o}C, the limit to growth, with 92.9% of July below 0 ^{o}C. Ninety percent of June and July were below 2 ^{o}C in 2011 and 80% in 2012. Potentially damaging temperatures below -10^{o}C were recorded for 8 days in 2011 and 6 days in 2012 from May to August.
3. Precipitation was recorded during all months, with snow mainly falling during the winter months. Winter rainfall and snowmelt was less than 15mm between May and September 2010 and less than 32mm between May and October 2011, potentially limiting growth. More than 65mm were experienced between November and April each year, hence water is not limiting during the growing season.
4. The dominant wind directions experienced were northwesterly, north-northwesterly, northerly and north-northeasterly. Mean monthly wind speeds were greatest during winter (May to August) and reach a maximum in September in 2010 and August in 2011. The wind speeds were observed to gust at greater speeds in 2010 than in 2011, potentially resulting in more damage to plant tissues.
5. The average solar radiation never decreased below 160W/m^2 per month from the start of recording until the end of the study period, and is not limiting to plant growth.
6. Soil moisture available to plants, measured in May 2011 using a soil moisture probe, were observed to be significantly different between sites ($p = <0.001$).
7. No significant differences in soil particle size distribution between each of the sites ($p = 1.000$) were observed.
8. Potassium concentrations are low at several sites (S.S., S.Str., H.E., V.F., M.F. upper, A.P. and K.P.), yet are only potentially limiting towards growth at the upper M.F. site and the K.P. site. Similarly, phosphorus concentrations are low at several sites (S.S., V.S.E., S.Str., V.F., M.F. upper, A.P. and K.P.), but only potentially limiting to growth at the A.P. site and the K.P. site.
9. Calcium concentrations are low but not limiting at the A.P. site. Manganese concentrations are low but not limiting at the upper M.F. site and the K.P. site. Zinc concentrations are potentially limiting to growth at sites S.S., the upper M.F. site and K.P., yet zinc plant deficiency indicators were not present in the study species.
10. *M. drakensbergensis* may migrate to higher altitudes (via vegetative propagation) under conditions of warming, and may thus become increasingly fragmented. Invasive karoid species pose a potential threat to the high Drakensberg region.

Chapter 5:

Grazing Study



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Grazing Study

Grazing plays a role in the recovery and defoliation of the *Merxmuellera* tussocks. In this study, the grazing mammals and the grazing invertebrates are the most relevant. However, mammals other than livestock only have a minor impact on *M. drakensbergensis* in the study area (e.g. *Otomys sloggetti*). However, the grazing of livestock is considered important to this study. Although the effects of grazing and fire are both important to consider in the study, it is difficult to determine which has the greatest impact on the studied tussocks, making an analysis of only the impacts of burning rather problematic. Clearly, grazing and its impacts cannot be ignored.

Extensive livestock agriculture involves more people in Africa than anywhere else in the world (Swallow and Bromley, 1995). Africa has 21% of the world's sheep and 14% of the world's cattle (Swallow and Bromley, 1995). More specifically, the Lesotho highlands contain 54% of the countries sheep (majority), 34% of the goats, 39% of the horses and 28% of the cattle (Pomela *et al.*, 2000). Yet, in the Lesotho highlands, there is little governmental control over livestock numbers as grazing land belongs to a ward chief and those he delegates grazing rights to (Letšela *et al.*, 2002; Du Preez and Brown, 2011). Thus, the results are potentially high livestock numbers and "severe" damage to wetland vegetation and its functioning (Du Preez and Brown, 2011). However, in the highlands of Lesotho, not all of the available land is suitable for livestock grazing. Much of the land is steep, rocky and may not support the growth of preferred grazing species. The result is that wetlands become a vital resource for herdsmen (Du Preez and Brown, 2011). The more livestock introduced into these areas, as conditions continue to deteriorate and numbers increase in the lowlands, the more pressure and overgrazing that is likely in the highlands. Compositional and vegetation cover has been documented to have changed on warmer slopes in the mountainous regions of Lesotho due to overgrazing (Martin, 1984; Morris *et al.*, 1989).

Given increasing grazing pressure, the *Merxmuellera* individuals and other forage species are being grazed at a greater intensity. Grazing of the inter-tussock vegetation removes protective vegetation from around the sides of the tussocks, increasing their exposure to potential soil erosion, which in turn affects their rate of growth and shape. A small permanent grazing post is situated within the village at Sani Top, and a further grazing post is situated within the lower Sani Valley north-facing slope at 2991m a.s.l. The grass cover surrounding the post is depleted and conforms to the idea of decreased vegetation cover at high altitude grazing posts

(Nüsser, 2002). By using a grazing study component, it is hoped that the hypotheses detailed in the introductory section (Chapter 1) may be better confirmed. This chapter investigates recent grazing patterns in the Sani valley study region. The potential value of invasive species as grazing indicators will also be observed.

5.1 Methodology

In order to investigate the recent grazing patterns within the Sani valley study region, a behavioural grazing study was conducted in 2011, allowing grazing routes and time spent along grazing routes and at particular sites to be determined. An observer was placed in a position from where the entire valley is visible and manually recorded all grazing activity within the valley from sunrise (approximately 04:30am) to sunset (approximately 6:30). Records were made of the number of livestock, type of livestock, their position within the valley, time spent within that region and whether or not they were grazing during that period. The valley was divided into a grid using a GIS, where each block measures 10m x 10m, overlaid on a three dimensional map. Each block was allocated a co-ordinate so the observer merely needed to note the block co-ordinates to obtain an accurate indication of where the livestock were observed within the valley. Observations of livestock grazing within the study valley was undertaken for three consecutive days in January 2011 and repeated (when possible) throughout the year. The livestock (sheep, goats and cattle) herds are distinguishable by the colouring of horns. The males of each herd have their horns painted with different colours depicting different owners. Females may be unmarked or have painted markings on their sides, tails, ears or heads. This makes individual herds distinguishable, thus enabling easier monitoring of grazing. The numbers and frequencies of observation of each livestock type were tabulated and assist with identifying which livestock species may be the most destructive towards *M. drakensbergensis* growth and survival. The obtained data were also used to give an indication of the maximum stocking rates occurring (LU/Ha). Live stock units (LU) were used as opposed to animal units (AU) as the stocking rates as only animals kept for agricultural purposes were used. Maximum stocking rates were not generated for sites where livestock were passing through as grazing was often not continuous, with some individuals not grazing at all.

Interviews were undertaken with the herdsmen in the village in early January 2011. Five herdsmen were interviewed to verify the accuracy of the results, as well as provide a comprehensive view on their grazing strategies. As the majority of herdsmen do not speak fluent English, a translator (available at the border post or through the San Top Chalets) was used to improve the understanding of questions asked and increase the accuracy and extent of

their answers. The questions posed were open ended and allowed for answers which were different to those expected. The same questions were asked to each herdsman, and any new information which was relevant to the project was pursued further. The herdsmen were asked whether they are responsible for the livestock on a daily basis, whether they follow grazing paths and whether these are followed daily, weekly or monthly. In many cases the herdsmen were able to indicate the locations where their livestock were left to graze and where they were moved to and from. They were also asked what time their herds start grazing in the morning, as well as where their livestock spend the night. Herdsmen were asked whether they ever burn to provide grazing for their livestock or if they allowed livestock to graze already burnt areas, and if so, how long livestock were allowed to graze these sites. GPS collars would have been useful to the grazing study but could not be obtained due to budget constraints.

Along with the information provided by the herdsmen and the observations of livestock movements, handheld Garmin Etrex GPS units were used to mark the positions of pathways and routes used in the movement of livestock. The GPS recorded waypoints at the start of each path, every 10m along the path and at the end of the path. In some instances pathways were observed to have endpoints where the vegetation altered or bare ground was observed. In these cases the pathways disappeared and an endpoint was named at the last place the path was visible. The information gathered, together with the behavioural study, enabled grazing maps (routes, frequency and duration of grazing at each site) to be produced using a GIS, as well as provide better insight into the grazing pressure within *M. drakensbergensis* communities in the study region.

The GPS data from handheld GPS units and information obtained from herdsmen on livestock routes were plotted onto a map using ArcMap 3.0 software. To assist with the positioning of the grazing routes and paths, an aerial image used in the fire frequency map was exported from Idrisi Taiga using the export tool as a .Tif file into ArcMap. The GEOREFERENCE toolbox was used to apply the same referencing as was used in the fire frequency map creation, to ensure continuity and accuracy. The georeferencing was checked according to feature points and associated co-ordinates with that of the aerial image in Idrisi Taiga. A shape file was then generated through the creation of a personal database using ArcCatalog 10. A line feature class was added and named ‘roads’. In ArcMap, the feature class layer was overlaid on the aerial image and the Transport symbol selected (symbol B02, black). The ‘roads’ layer was activated and the EDITOR tool was activated. Lines were traced on existing road networks to produce a roads shape file. A second feature class was

generated as a line file and saved as ‘pathways’. In the same way as with the ‘roads’ file, the editor tool was used to plot the pathways observed in the field using the GPS and indicated using black dots. A third feature class, ‘routes’, was used to portray the suspected routes used between pathways using the information from observations and interviews with the herdsmen. The notes gathered from the valley observations gave more detail to the durations the livestock spent at each location and was added to the GIS maps where possible.

The *M. drakensbergensis* individuals of each community (indicated as study sites) which had been grazed, were recorded. This was undertaken every second month from January 2009 to December 2011 and additional sites were added as the study progressed. The number of grazed individuals was transferred into a percentage value of the number of grazed individuals per community per monitoring record. The percentage grazed at each monitoring recording may crudely indicate the potential frequency to which *M. drakensbergensis* tussocks are being grazed, as well as quantify the extent to which each community is being grazed. The records may also indicate whether there is a temporal aspect to tussock grazing. Photographic evidence was also used to demonstrate grazing occurring at each site, as well as its effects on the tussocks.

In order to indicate the presence of invasive species as a potential consequence of overgrazing, the presence and biomass of two invasive species were examined. The dry-weight-rank method was used to indicate the percentage biomass of two invasive species, *Pentzia cooperii* and *Chrysocoma ciliata*. This method was chosen as it is a non-damaging and cost effective method (‘tMannetjie and Haydock, 1963; Ratliff and Frost, 1990; Dowhower *et al.*, 2001). The dry-weight-rank method allowed for yearly changes in the same communities to be recorded and detected in 2009, 2010 and 2011. The method has been successfully performed in the Mpumalanga highveld by Barnes *et al.* (1982) and in tussock grasslands in northern Australia by Ash and McIvor (1998). However, it has not been tested in Lesotho prior to this study. Due to the size of the vegetation within the study region, a 1m by 1m quadrat size was selected as this would allow for three or more species to potentially be present within each sample. The quadrat was randomly placed 30 times within each of the selected *M. drakensbergensis* communities (12 study sites), as this covered almost all or more than the area occupied by each community. The dominant species present within each quadrat were ranked into first, second and third place according to estimates of their contributing dry-weight (‘tMannetjie and Haydock, 1963; Ratliff and Frost, 1990). The incidences of the ranked presence of the two invasive species were added to provide the estimated proportion of dry-weight within the community. The first ranked values were then multiplied by 70.19,

21.08 for second ranked values and 8.73 for values ranked third to produce the percentage dry-weight of each species within each site (‘tMannetjie and Haydock, 1963). The results for species other than *P. cooperii* and *C. ciliate* were tabulated for each year where present at each site.

5.2 Results of the grazing study

Communication with the herdsmen (as per the methodology of this study) was not easily accomplished. The major challenge was predictably that of the language barrier, and sometimes even with a translator it was difficult to reach a point of understanding. At the same time, it was difficult to centre the conversation on grazing, grazing routes, livestock and grass, as they preferred to ask questions of the interviewer. Information was also difficult to gather as a young white female was the one who wished to know - they did not wish to explain their “business”. On other occasions the herdsmen were aggressive and threatening, using sharpened golf clubs to deter the researcher and translator. However, some young herdsmen and an older villager were happy to share their knowledge. Their knowledge is mentioned throughout this chapter and adds to the richness of the knowledge of the grazing study.

5.2.1 Livestock observed in 2011

Observing the herdsmen and their livestock revealed that there are four large flocks of sheep in the immediate Sani area. These flocks each contain between 50 and 120 individuals, excluding lambs in the 2011 lambing season. Alongside the sheep flocks, there are also three cow herds in the area, each group containing between 5 and 25 cattle and not all are kept for dairy purposes. There are also 6 donkeys that spend most of their time confined to the village or are used as transport for goods by the herdsmen travelling between villages. Angora goats are another frequent sight in the area and are usually found in two herds, despite more than one owner. In 2011, the angora goats were the last to arrive in the summer grazing valleys, as conditions remained very dry well into December. Consequently, more palatable grasses had not grown much and in many places remained only a dull green- brown as opposed to the rich green of the more usual summer pastures.

5.2.2 Grazing during the winter months

Cattle were observed to be the first livestock moved down to lower altitudes as they risk becoming sick in cold conditions (Personal communication with a herdsman). According to the herdsmen, sheep are tough and are able to tolerate snow which frequently covers the

study region in the winter months. Thus, the transhumance grazing system varies according to livestock type, and so sheep were observed in the study valley during the winter months (Figure 5.1). Neither the angora goats nor the horses are viewed as tough, but may still be observed in the months leading up to or out from the months of June, July and August, if temperatures permit, and only in locations sheltered from the cold wind (Figure 5.2). A herdsman interviewed claimed that the goats are less tolerant of dry conditions.



Figure 5.1: Sheep grazing in the Sani valley in June 2009.

5.2.3 Grazing routes

The positions of visible pathways were recorded using a handheld GPS and indicate the routes most often used by herdsmen (Figure 5.3). Hikers also make use of these pathways, especially the pathway from the Sani Top Lodge to the valley behind Hudson's peak; from there the hikers follow the pathway to Hodgson's peak. The dashed red lines indicate routes taken where the pathways disappeared due to either bare, rocky ground or dense vegetation, usually in the form of *Merxmuellera* communities. However, in some places the ground does not account for the lack of pathway and it is thought that livestock were permitted to spread out and graze before being moved on to the paths and new locations. The locations of these pathways were indicated by the herdsmen interviewed and recorded using a GPS.



Figure 5.2: Angora goats on the sheltered slopes on the north side of the study valley (June 2010).

Sheep and Angora goats are roused at sunrise, translating to a time of ca. 05:00 in mid summer. The herds are guided into valley bases and allowed to roam within the confines of the valley during the course of the day. In some cases the livestock are herded during the day to new grazing grounds (e.g. Figure 5.4). At approximately 16:30 or 17:00 in the afternoon the herdsmen begin to guide the herds back, usually along frequently used pathways, towards the village or grazing post where they have arranged to house their herd for the night. This allows the herdsmen to move their herds into 'kraals' before the sun sets for added protection, required especially when there are lambs in the herd. Herds are also temporarily guided into village or grazing post kraals if a severe storm approaches (severe was defined by the interviewed herdsmen as low dark clouds producing marble sized hail – as shown with fingers, and lightening). Herdsmen without arrangements with the kraal owners move their clustered herds to sheltered positions for the night. The interviewed herdsmen confirmed that fires were often lit within *Merxmuellera* tussocks for warmth, protection from jackals and for light. Yet the tussocks were usually not burnt throughout the entire night, and frequently not burnt if a full moon was present and temperatures were relatively warm.

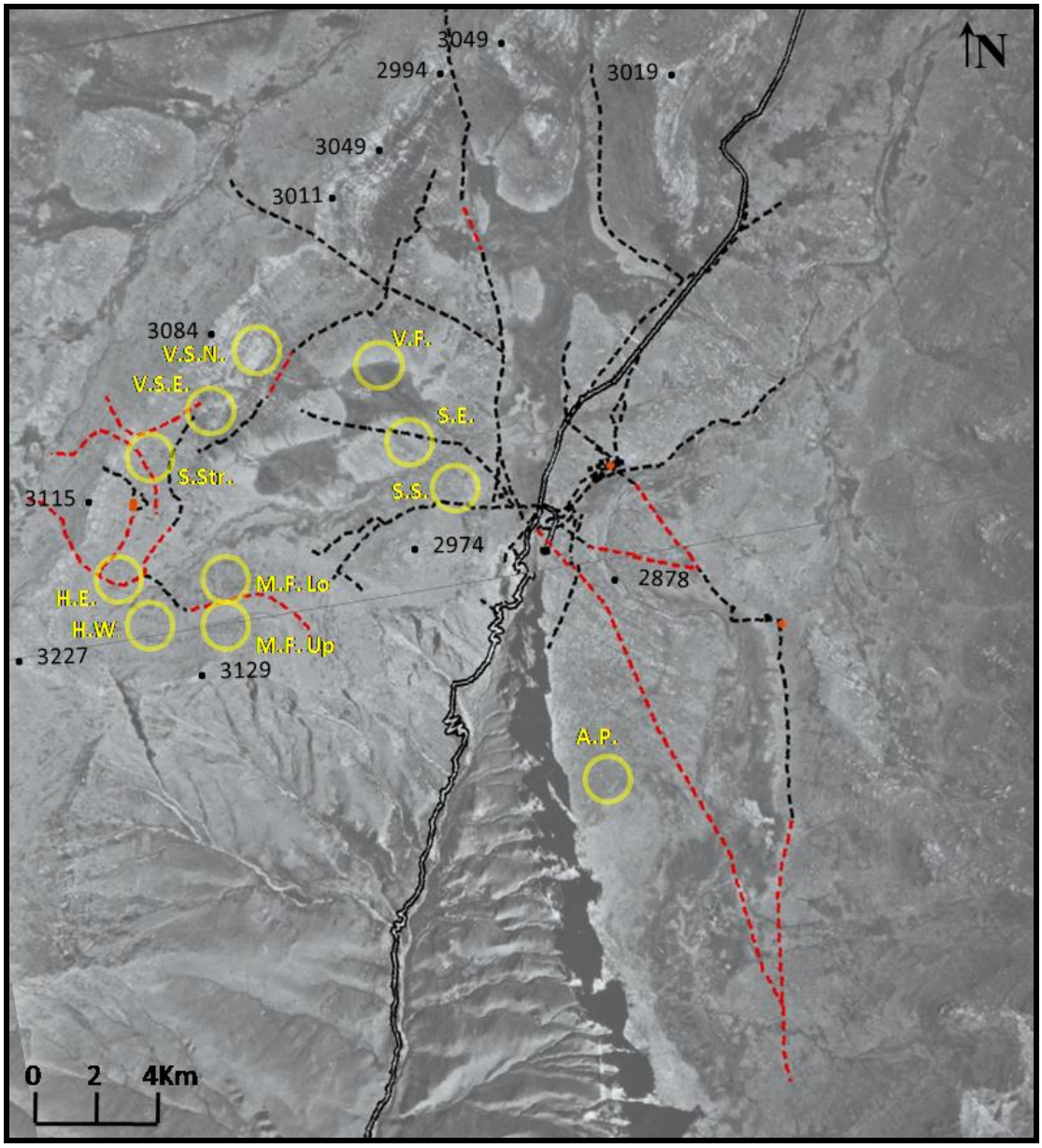


Figure 5.3: Grazing routes used by the herdsmen. Black dashed lines indicate the presence of pathways; red dashed lines indicate the pathways indicated by herdsmen despite not appearing as pathways, black squares indicate the presence of buildings, orange dots indicate grazing posts.

The frequency of route usage is indicated by the visibility of pathways and the observations made by the researcher (e.g. Figure 5.5). The most frequently grazed area year round was the central valley base of a study valley. Livestock were allowed to graze freely within the valley and very few herdsmen were observed to watch their livestock. Along with livestock, natural herbivores were observed in the valley (e.g. grey rhebok). When questioned, the herdsmen claimed that the valley was the safest place for the livestock, especially foals and lambs, hence the high number of foals and high frequency of observation during the early summer months. This may also explain the high numbers of sheep and goats present during the summer months. However, the herdsmen admitted that there was little forage available for the livestock particularly at the beginning and end of summer, and almost nothing during

winter months. Horses and cattle were observed on the valley slopes while this was a rare occurrence for sheep and goats. When queried, a herdsmen pointed out the "weak legs" of the sheep and goats, and added that they did not prefer to take their flocks through mountain passes as this could cause injury. The herdsman added that it was essential for flocks to be returned to the village grazing post at night during winter, thus the flocks were kept close to the village.

The locations of grazing livestock were recorded as well as the duration spent grazing, with special focus on the study sites. The central valley proved the most popular place for grazing livestock (Table 5.1). Afternoon thunderstorms are common during the summer months along the escarpment edge and in most cases the livestock remained in the valley bottom. During the winter months, livestock were herded out of the grazing stations into the central valley or the valley flats to the northeast of the village each morning. Flocks were then permitted to graze until they were rounded up and returned to the grazing posts before sunset and temperatures dropped. This shortened the time which was available to the livestock for grazing compared to that observed during the summer months.

Many of the study sites were observed to contain livestock only in passing as they were herded to alternative grazing in adjacent valley flats. This included the S.Str. site and the M.F. sites, both of which are observed to be near grazing routes (Figure 5.3). Sites H.E. and H.W. are observed to be near a pathway, yet it was not uncommon for cattle and goats to be observed grazing within and around the *Merxmuellera* communities on either side of the stream. However, this was usually of a short duration (<1 hour) with livestock remaining to graze for several hours only on one occasion. It was assumed, as no herdsmen were present that the cattle had wondered there, perhaps in search of better forage, on their own accord, having been left at the bottom of the slope to graze in the valley base.

The maximum stocking rates were calculated and confirm that the valley centre is the most grazed, with a maximum of 1.05LU/Ha in summer. This figure does decrease slightly during the winter months but remains high (0.93LU/Ha). The maximum summer stocking rates are similar to those obtained by Quinlan (1989) but are greater than that for 1999 in the Sanqebethu Valley (north of the study region) observed by Nüsser (2002). The maximum stocking rates provide an indication of the potential grazing press ure placed upon the study valley, however, it must be noted that the actual values experienced throughout the day may be lower. The numbers of individuals within a herd/flock were also recorded as this would provide an indication of the grazing pressure. The maximum number recorded in each

Table 5.1: Locations where livestock was observed, including the study sites, frequency of occurrence, the average duration and the type of livestock observed.

Location	Frequency	Average Duration*		Livestock	Maximum Stocking Rate (LU/Ha)
		Summer	Winter		
Valley centre	Almost daily	9 hours	7 hours	Sheep, goats, horses, cattle and donkeys	†1.05 (sheep: 0.64; goats: 0.29) ‡0.93 (sheep: 0.64; goats: 0.29)
Site S.S.	Often (mostly in passing)	30 minutes	1 hour	Sheep and horses	
Ridge behind S.S.	Often (mostly in passing)	4 hours	2 hours	Sheep and cattle	3.26 (sheep: 2.70; cattle: 0.56)
Site V.S.N.	Seldom	8 hours	6 hours 30 minutes	Horses	0.71
Site S.Str.	In passing	<20 minutes	<20 minutes	Sheep and cattle	
Site H.E.	Seldom	2 hours	2 hours	Horses	0.33
Site H.W.	Seldom	7 hours	5 hours	Cattle	1.49
Site V.F	Often	5 hours, 30 minutes	4 hours	Sheep and horses	1.43 (sheep: 1.26; horses: 0.18)
M.F. sites	In passing	<20 minutes	Not observed	Sheep, goats and cattle	
Flats NE of village	Often	8 hours, 30 minutes	7 hours, 30 minutes	Sheep, goats, cattle and donkeys	0.33 (sheep: 0.20; goats: 0.09)
NW end of valley	Almost daily	8 hours	8 hours	Sheep and goats	0.43 (sheep: 0.29; goats: 0.13)

* Average duration rounded off to the nearest half hour; †Summer stocking rate; ‡Winter stocking rate



Figure 5.4: Herdsman with his dog moving cattle from the northeast of the study valley behind Hodgson's peak to the valley in the west (September 2009).



Figure 5.5: Pathway to the south of Moshweshwe's finger indicating a route that is frequently used by livestock (June 2010).

herd/flock may indicate the maximum grazing pressure which the valley experiences (Table 5.2). A flock of 134 sheep was recorded for the first time in October 2009 in the center of the study valley. In June the following year the same herd was observed again (identified through

the purple identification markings) to the northeast of the village, yet the number had been reduced to 131 sheep. The large flock is believed to be guided into adjacent valleys during the summer months and return to graze the study valley and surrounding areas during the winter months. Horses are a form of transport in the area and only a few owners are wealthy enough to own a horse. Thus, the horses are kept close to the village in case they are required. Similarly, donkeys are used as pack animals and are chiefly used to carry fire wood from along the Sani pass or shrubs from the lower valleys to the west.

Table 5.2: Maximum number of individuals recorded per grazing herd/flock.

Livestock	Maximum Number Recorded per Herd/Flock	Date of observation
Sheep	134	23 May 2011
Goats	60	26 June 2010
Cattle	21	21 October 2009
Horses	7	22 October 2009
Donkeys	3	23 May 2011

5.2.4 Grazing and *M. drakensbergensis*

Despite the low palatability of *M. drakensbergensis* and its prickly tips, livestock were observed grazing tussocks on more than one occasion. Tussocks were usually only grazed when the more palatable grasses were scarce or once post-burning resprouting had begun (Pers. obs.). The percentages of grazed tussocks recorded at each monitoring event do not indicate a clear pattern across all study sites (Figure 5.6). However, almost no grazing of *M. drakensbergensis* was observed for October in 2010 and 2011, which suggests livestock preferred grazing other available forage. Yet, the data for 2009 do not support this trend.

On average, low percentages of grazing were recorded during the 2008/2009, 2010/2011 and for the 2011/2012 growing seasons. However, this was not the case for the 2009/2010 growing season when grazing of *M. drakensbergensis* communities was high (average percentage varies between 34% and 50%). The greatest intensity of grazing across all sites was recorded for May 2011. The lower M.F. site (Av. = 36%) was subjected to a greater number of individuals being grazed than the upper M.F. site (Av. = 29%), perhaps due to the location of the upper M.F. site on a steeper slope than the lower site, while the lower site is within close proximity to a grazing route path (Figure 5.3 and 5.5). Similarly, the frequency

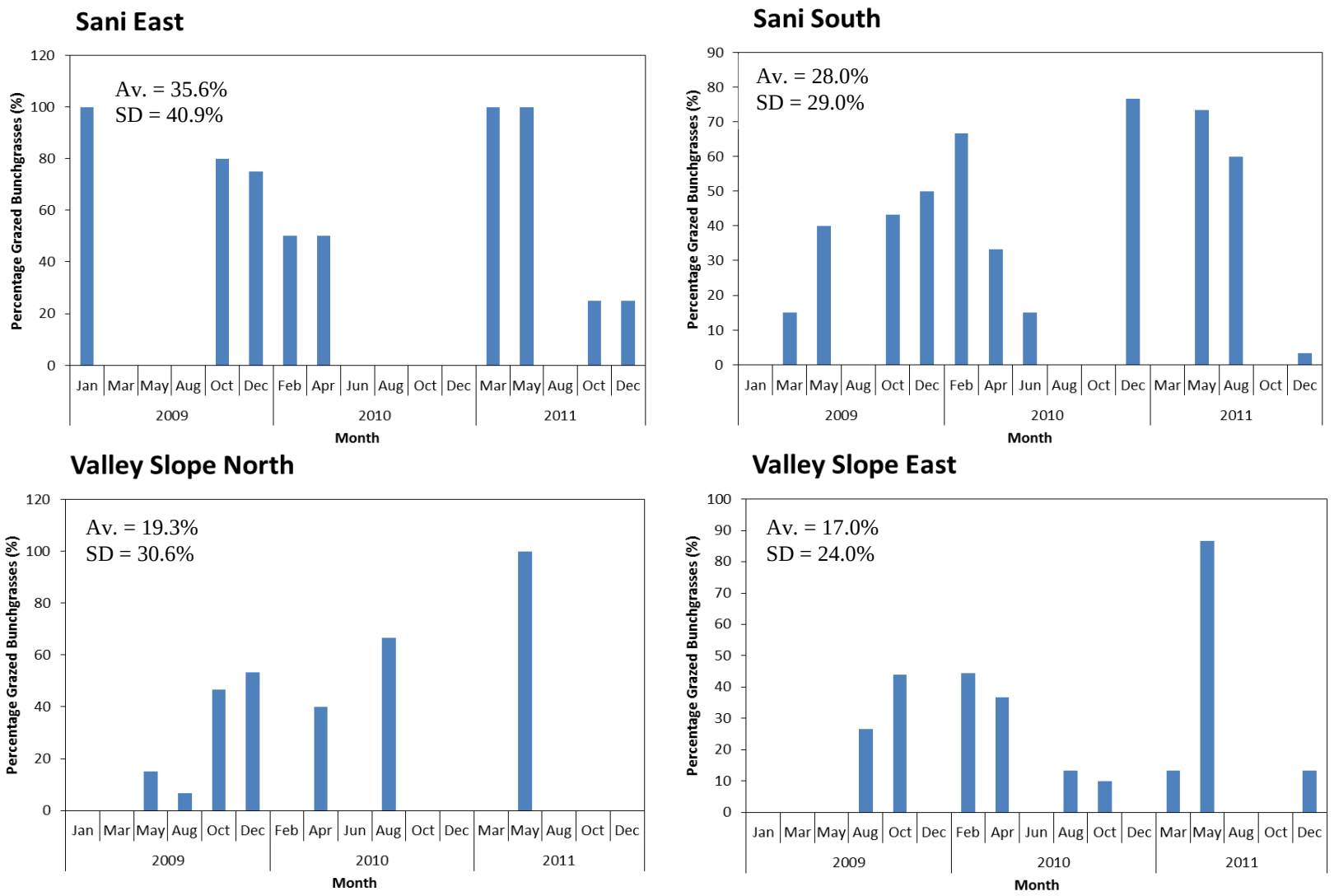


Figure 5.6: Percentage of monitored tussock grasses grazed at each site when monitoring took place (Av.: Site average; SD: Standard deviation).

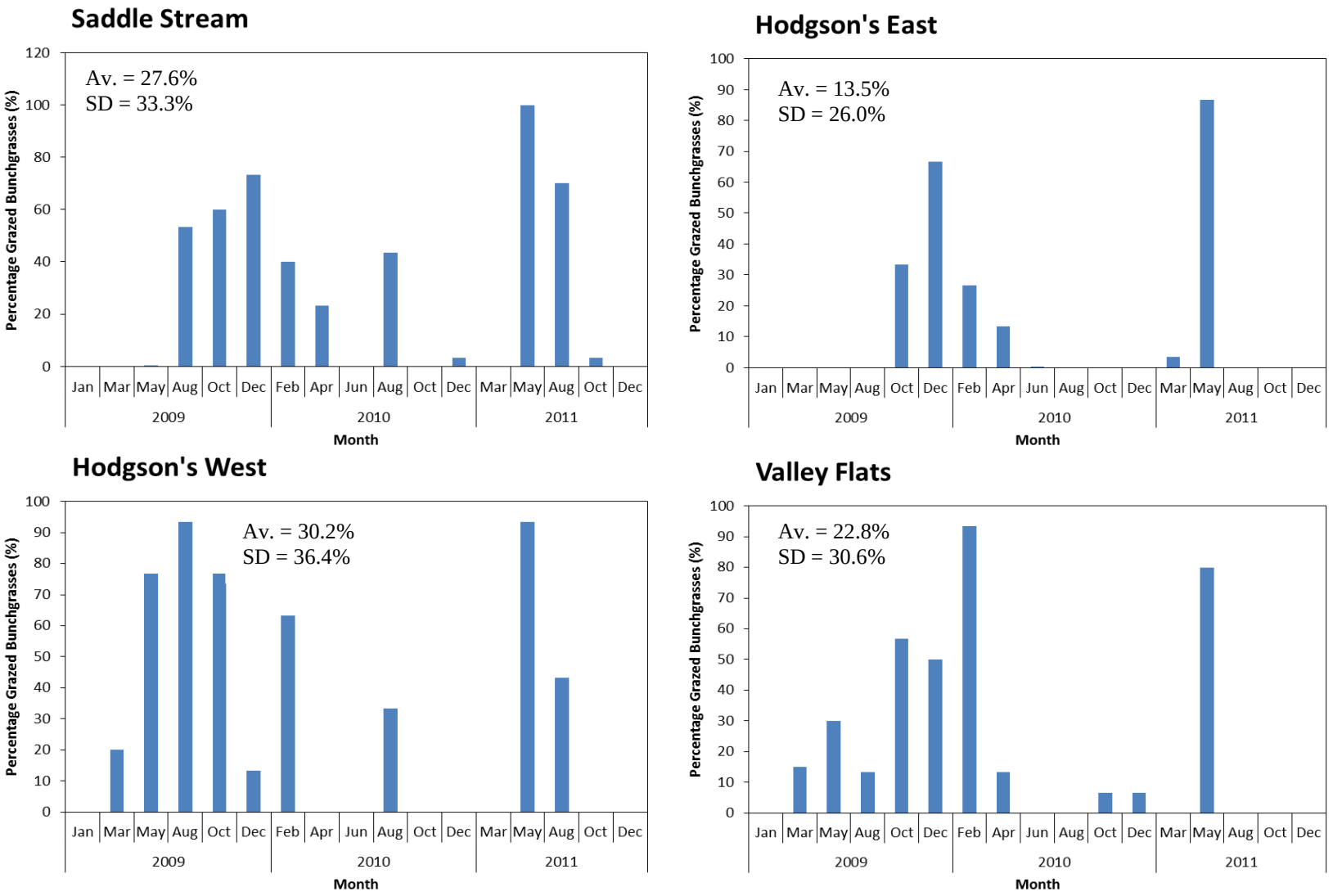


Figure 5.6 cont.: Percentage of monitored tussock grasses grazed at each site when monitoring took place (Av.: Site average; SD: Standard deviation).

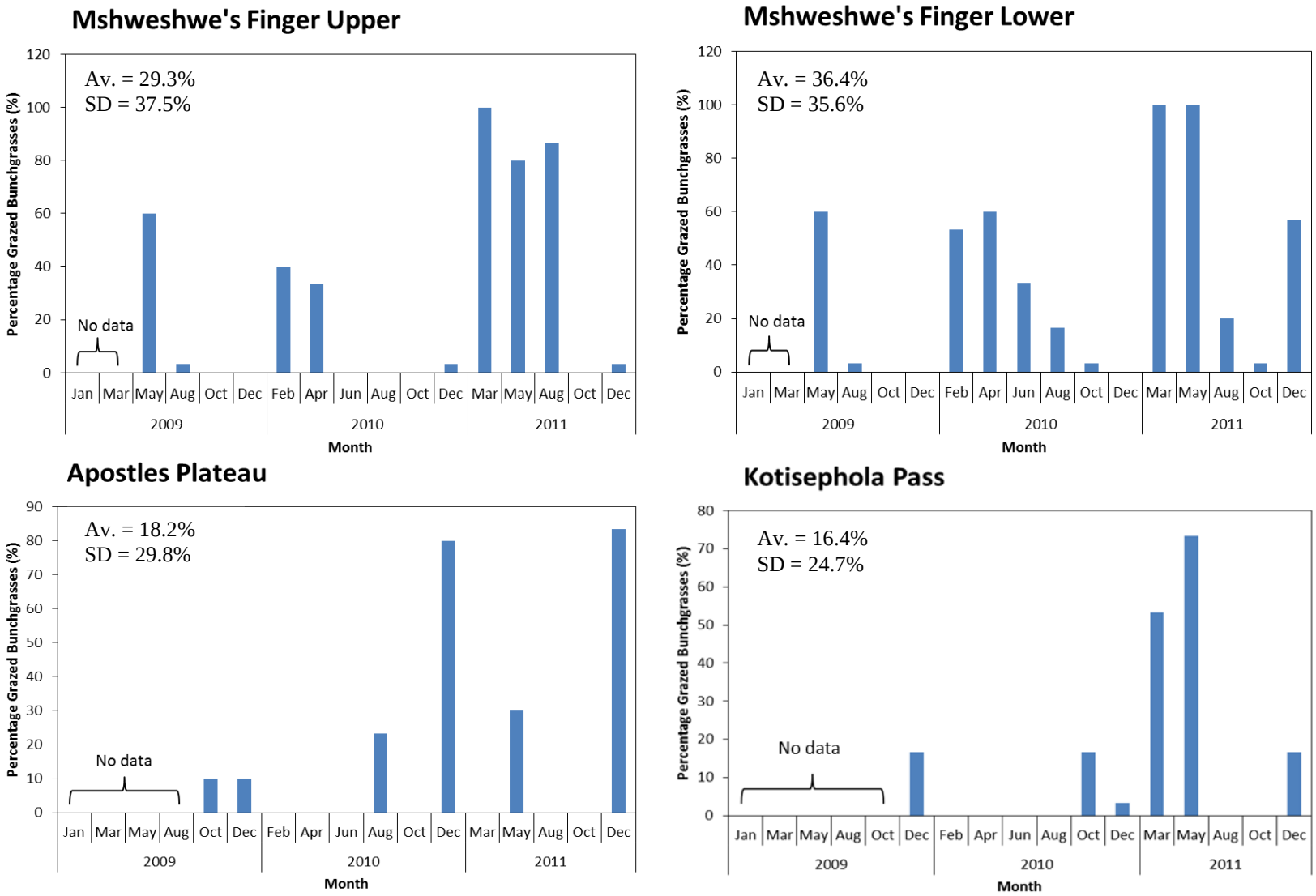


Figure 5.6 cont.: Percentage of monitored tussock grasses grazed at each site when monitoring took place (Av.: Site average; SD: Standard deviation).

of grazing at the lower M.F. site was greater, possibly for the same reason. Site H.E. received the lowest intensity of grazing (Av. = 14%), perhaps due to a slightly steeper slope than the H.W. site (Av. = 30%). Thus, out of preference, livestock may choose to graze on the gentle slope, which is supported by the higher percentages of grazed individuals at the H.W. site.

Resprouting *M. drakensbergensis* tussocks were grazed with little preference for outer or middle leaves, yet the livestock avoided unburnt tussocks when resprouting individuals were present. Unburnt tussocks were grazed on the outer edges than in the centers, perhaps due to their prickly nature. New shoots emerge on the outer perimeter whilst older shoots tend to gather in the centers. This observation may thus be a preference for newer leaves with an increased palatability. Sheep and goats are the livestock that seem to most regularly graze the tussock grasses, but mostly graze *Merxmuellera* tussocks along their edges (Figure 5.8). However, cattle were observed trampling and sleeping on the tussocks, which in some cases results in greater destruction than grazing. Less grazed *M.drakensbergensis* communities are grazed predominantly during the winter months when many of the other grasses are dormant and in many cases overgrazed. Livestock was observed grazing within *Merxmuellera* communities during the winter months, yet were not commonly observed in these locations during the summer months. Grazing is evident as blunt edges on leaf blades as opposed to the typical pointed tip of most *Merxmuellera* species (Figure 5.9).

Less evidence of grazing was generally observed at the Kotisephola Pass site, perhaps due to the consistently harsher weather conditions and fewer livestock here. The herdsmen confirmed the literature (e.g. Acocks, 1971; Morris *et al.*, 1989) stating that the livestock, particularly sheep, did not demonstrate a preference for *Merxmuellera* tussocks. However, they did also add that sometimes the grazing was "so bad there is nothing for them to eat" but had not been given permission to move the herds to further grazing pastures. This is when they claimed that cattle and Angora goats would consume the *Merxmuellera* tufts. However, they did confirm that the new shoots after burning were "liked very much" by all livestock. The literature predicts that resprouting shoots will have a higher palatability and thus may be grazed more readily. This was observed at every burnt site studied, but particularly the burnt V.S.E. site and the A.P. site (Figure 5.10). Yet, it was observed that the herdsmen did not deliberately graze their livestock within the burnt locations within the study valley during the study period. Rather, the burnt tussocks were grazed as herds were guided to other grazing locations, or by individuals who had made their own way into the resprouting burnt patches.



Figure 5.7: An angora goat grazing the edges of a *M. drakensbergensis* individual (May 2011).



Figure 5.8: Cattle grazing the green shoots off year old burnt *Merxmuellera* tussocks.



Figure 5.9: An example of a *M. drakensbergensis* tussock after being grazed by an Angora goat.

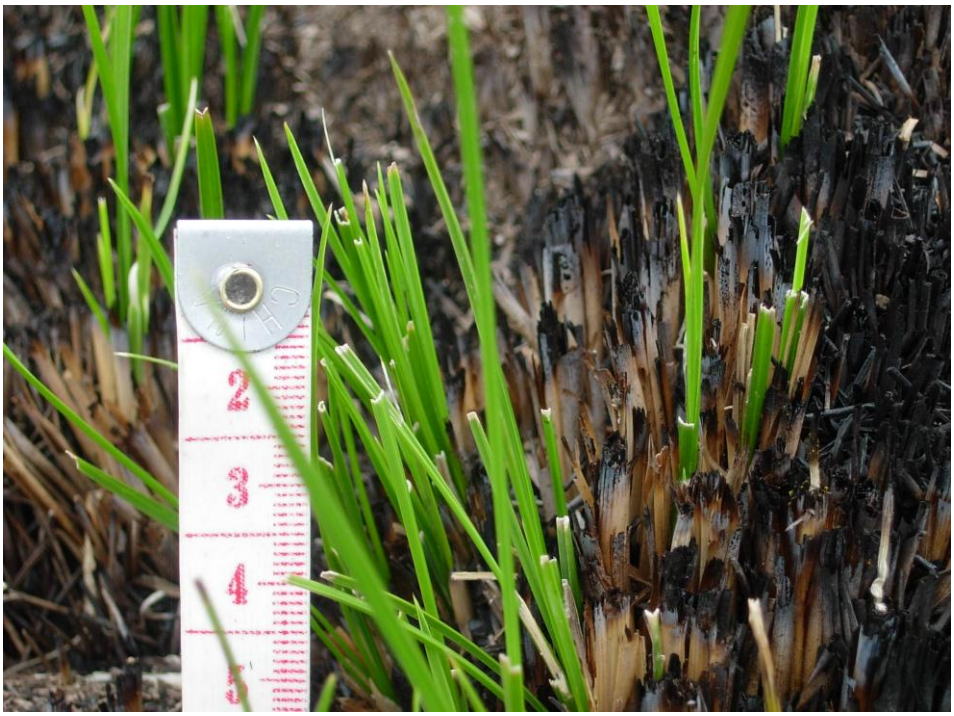


Figure 5.10: An example of a *M. drakensbergensis* tussock after being grazed following soon after resprouting following burning at the A.P. site.

5.2.5 An unlikely grazer

Slogget's ice rat (*otomys sloggetti robertsi*) was observed to contribute towards the loss of tussock tissue. Leaf blades and inflorescence stalks were commonly observed on the ground at the entrance to burrows. Despite being herbivorous, it is thought that rather than eating the blades of grass, as grasses are a less preferred food, the ice rats use them for nesting (Schwaibold, 2005; Hinze *et al.*, 2006; Figure 5.11). Additionally, the *Merxmullera* tillers may be removed to clear entrances and in other burrowing activities (Mokotjomela *et al.*, 2009). The ice rat's diet includes grass inflorescences (Schwaibold, 2005), and this may also partially explain the leaf blades observed lying on the ground as inflorescence stems are cut, enabling them to reach the seeds. Crawford (2008) believes that the burrowing and grazing of small herbivores can be beneficial as they increase the open spaces allowing for seedling growth and vegetative regeneration.



Figure 5.11: A Sloggets ice rat burrow demonstrating their destructive ability in creating easy access to their burrows below*Merxmuellera* tussocks.

5.2.6 Invasive species as overgrazing indicators

Invasive species tend to colonize heavily grazed areas and thus can be considered indicaters of overgrazing. In the Lesotho highlands, plants such as *Heilichrysum trilineatum*, *Erica dominans*, *Chrysocoma ciliata* and *Pentzia cooperi* are considered such indicators (Morris *et al.*, 1993; Nüsser, 2002b). The percentage dry-weight biomass was calculated at sites where *P. cooperi* and *C. ciliata* were present (Table 5.3).

Table 5.3: Percentage of *Pentzia cooperi* and *Chrysocoma ciliata* coverage at each site where present.

	Pentzia cooperi			***Chrysocoma ciliata***		
Year Month	**2009 December**	**2010 December**	**2011 December**	**2009 December**	**2010 December**	**2011 December**
S.E.					1.2	1.2
S.S.						
V.S.N.		32.1	20.0		32.1	33.7
V.S.E.					22.5	22.9
S.Str.	0.5	9.5	15.5		4.3	10.5
H.E.			17.3	12.0	12.3	13.4
H.W.				4.8	7.6	9.8
V.F.		11.6	9.0		10.4	16.0
M.F. Up						
M.F. Lo					10.4	10.5
A.P.						
K.P.						

P. cooperi was noted within the village, particularly near the grazing post or where sheep congregate. However, no examples were recorded within the central valley floor, perhaps due to its saturated nature during the summer months. The presence of both *P. cooperi* and *C. ciliata* are recorded at the study sites surrounding the seasonally saturated soils. Yet, invasive species were not present at site S.S., M.F. upper, A.P. and K.P., and the reasons for this are not clear. Each year the percentage cover increases for both species, with the greatest increases observed at site S.Str. where *P. cooperi* increased by 9.0% between December 2009 and December 2010. Losses in *P. cooperi* cover were measured at site V.S.N. (12.1% loss) and at the V.F. site (2.6% loss). The reasons for these decreases are unknown and require further study. Sampling error may have played a small role but does not account for the entire losses. The shrubs are potentially collected as fuel, yet many residents of the study region preffered to gather wood from the Sani Pass. The general increase in invasive species coverage as overgrazing indicators suggests that overgrazing is a problem on the slopes of the study valley. These shrubs are unpalatable and may further reduce the amount of palatable forage available (Quinlan, 1995b).

5.3 Discussion

Winter grazing at high altitudes has historically been avoided, with herdsmen returning to low altitudes to avoid the herd being exposed to harsh winter conditions (Quinlan, 1995b; Swallow and Bromley, 1995; Tshabalala, 1995; Pomela *et al.*, 2000). There is thus little legislation and almost no management strategies dealing with winter grazing posts (Quinlan, 1995b). This study observed sheep and goat herds remaining in the highlands over the winter months, an observation which has also been made by Morris *et al.* (1989), Quinlan and Morris (1994) and Bisaro (2007), who added that herds generally graze within or near the village between June and October. The changing transhumance system through winter grazing has been observed since 1980 (Nüsser, 2001), and thus is not a new concept. Yet there is very little knowledge of the impacts which this will have on forage availability, species richness and plant density. A lack of palatable grasses, both in the lowlands and in the highlands, is considered the primary restriction to maintaining herds in the highlands during winter months (Quinlan, 1995a; Nüsser, 2002b). Despite a general decrease observed between the usage of pastures in 1988 and in 1999 (Nüsser, 2002b), a small increase, especially in winter grazing, has been observed for the inland Merareng Valley, to the north of the study valley (Nüsser, 2002b). It is anticipated that the study valley has also experienced an increase in livestock numbers and annual grazing durations through the years of study.

The season in which grazing takes place is important in terms of the ability of the tussock grasses to recover (Ash and McIvor, 1998). Thus, winter grazing has been suggested to be detrimental to highland landscapes (Quinlan, 1995b). Negative effects of grazing in spring have been demonstrated in northern Australia, with carry-over effects recorded the following spring (Ash and McIvor, 1998). Conversely, it has been reported that winter grazing, when tussock grasses are dormant, may have little effect on productivity or species diversity during the following growing seasonin English and Australian grasslands (Bullock *et al.*, 1994; Ash and McIvor, 1998). This suggests that perhaps winter grazing within the study valley may not be as detrimental as first thought. However, defoliation during winter may increase the amount of stored resources the tussocks require for the following spring, potentially reducing inflorescence numbers. Winter grazing has been demonstrated to increase tiller death rates and result in earlier spring tiller emergence in Oxfordshire, England (Bullock *et al.*, 1994). Further research is required on this topic to determine whether the continued presence of livestock through the winter period, as well as during the summer period, is detrimental to this landscape.

Sustained seasonal grazing may cause the limited forage to become further depleted through reduced recovery time and a decreased ability of tussocks to replenish stores, as observed in New Zealand grasslands (O'Connor *et al.*, 1997). The predicted result of year-round grazing and the over utilization of palatable forage species has been observed within the study region by the researcher and by Schwabe (1995) and Nüsser (2002b). Similarly, despite a reduction in the number of grazing posts in the Sanqebeth Valley in eastern Lesotho, an increase in grazing pressure was recorded between 1988 and 1999 (Nüsser, 2002b). It was suggested that this was due to a shift to almost constant grazing out of 'permanent' grazing posts (Nüsser, 2002b). According to a map in Grab and Nüsser (2001), the number of grazing posts within the Sani valley, to the north of the road, has decreased slightly (31% fewer grazing posts were observed by 2011), yet many seasonal grazing posts have become used year round. The altitude of winter grazing posts seems to have increased. Winter grazing posts are said to be between approximately 2 700m a.s.l. and 2 550m a.s.l. (Quinland and Morris, 1994), yet winter grazing posts at 2 900m a.s.l. or even higher occur in the same study area. Perhaps this is an indication of climate change and/or degradation at lower altitudes forcing winter grazing to higher altitudes.

Livestock (particularly sheep) have a significant impact on plant growth and species composition within alpine areas (Hercus, 1964; Mysterud *et al*., 2011). Additionally, overgrazing may lead to vegetation changes and has been recognized as occurring in Lesotho (Guillarmod, 1969; van Zinderden Bakker, 1981; Morris *et al.*, 1993). Overgrazing may decrease the competitive ability of palatable species, thus allowing less palatable species to increase their extent (Breebaart *et al.*, 2006). Thus, evidence of overgrazing exists in the form of indicator invasive species such as *Pentzia cooperi* and *Chrysocoma cilliata*. Additionally, the single presence of *Erica dominans* (single community near the Sani Top Chalets) and occasional occurrence of *Helichrysum trilineatum* recorded by Killick (1978) are both now common to the study region. This finding is in agreement with the literature which has identified that 50% of the Lesotho highlands is said to contain invasive Karoo shrubs (van Zinderden Bakker, 1981). Their presence has increased not only during the course of the last few decades but since the study by Deschamps (2006). In 2006, in an adjacent valley to the study valley, *C. ciliate* was observed surrounding grazing posts, but not on valley slopes (Deschamps, 2006). However, this study has indicated the growth in the biomass of not only *C. Ciliata*, but other invasive species along the study valley sides and floor, particularly along the warmer slopes as found by Quinlan and Morris (1994). Additionally, the invasive species recorded have been noted to occur alongside drainage patterns (Grab and Deschamps, 2004). The observations made in this study support this, yet invasive species were

also recorded at locations more than 20m from stream banks (e.g. sites S.E., V.S.N., V.S.E. and the lower M.F. site), highlighting their spread through wind and animal dispersion. As many of these species are indicators of overgrazing, this suggests that the impacts of overgrazing are prevalent.

A recent study in the Lesotho highlands observed that patches excluded from grazing have a higher biomass production and average height than those experiencing grazing, yet species numbers remained constant (Du Preez and Brown, 2011). However, the exclusion period was only 12 months but it was suggested that grasses may outcompete smaller flowering plants if grazing is excluded for a longer period (Du Preez and Brown, 2011). Additionally, the grasses may be outcompeted by shrubs, reducing the number of grasses (Watkinson and Ormerod, 2001). Moderate levels of grazing have also been indicated to marginally increase growth rates of tussock grasses (Olson and Richards, 1988; Patton *et al.*, 2007). Additionally, grazing increases light penetration to the crowns, prevents collapse and increases tiller production (McNaughton, 1991). In the absence of grazing, tussock grasses may become harmful to themselves through self-shading and thus reduce the growth rates (Caldwell *et al.*, 1983). Thus, excluding livestock from the Lesotho highlands may also not be the solution. Low levels of grazing may form a necessary form of defoliation, especially in the absence of fire, to ensure grassland diversity remains high.

After burning, new growth may appear earlier than that of unburnt tussocks (Daubenmire, 1968), and appear more attractive to grazers than new growth on unburnt tussocks (Allred *et al.*, 2011). A dramatic increase in grazing of similar species *M. distcha* was recorded in karroid *Merxmuellera* mountain veld (Novellie, 1990), suggesting that the study species will be similarly grazed after burning, as was observed. A lag period between burning and grazing of new shoots has been suggested as a method to best maintain grassland health in Argentina (Peláez *et al.*, 2009). However, this was not observed to occur within the study region as half of burnt *M. drakensbergensis* tufts were grazed within two months of burning and the majority after six months. Grazing of regrowth soon after burning may reduce the survival and slow the recovery rate of alpine tussock grasses due to the slow rate of growth and the reallocation of stored resources to leaf tissues, as observed on South Island, New Zealand (Mark, 1994).

Sufficient reserves exist within alpine plants to allow them to withstand grazing, despite harsh climatic, edaphic and topographical conditions (Crawford, 2008). However, a study conducted

on *Chionochloa pallens* suggests that alpine tussock grasses may be better adapted to recover from burning than grazing (Lee *et al*., 1988). Recovery from grazing is linked to extent of tissue removal, timing, frequency, soil characteristics and climatic factors (Lee *et al*., 1988; Wolfson, 1999). Fifteen years or longer has been suggested as the required time to recover to the pre-defoliated state for *C. pallens* (Lee *et al*., 1988; Lee *et al.*, 2000). This species is similar to that of the study species, suggesting that many more than one growing season may be required for *M. drakensbergensis* to recover from severe grazing. Fortunately for the study species, severe grazing of unburnt individuals to crown level was not recorded during the study period for any of the study sites.

The personal observations and evidence of grazing of *Merxmuellera drakensbergensis* may indicate that the preffered, more palatable grasses are being overgrazed. However, this would require further study of the grazing patterns on *M. drakensbergensis* as well as other more palatable grasses. The observations disagree with the findings that despite a heavy grazing pressure (0.5ha LSU^{-1}) a very similar species, *M. disticha*, coverage remained the same (Mpiti-Shakhane *et al.*, 2002). However, this study was only a year-long study and the results may change as the more palatable species become further grazed and depleted. But not all the individuals will experience the same amount of grazing within a *M. drakensbergensis* community (Gillen *et al.*, 1990). Individuals which are repeatedly heavily grazed may die sooner than those grazed less often. This suggests that tussocks found in the centres of communities are less likely to be grazed than those along the edges, particularly near grazing pathways. There is little evidence in the literature on the impacts that grazing will have on *M. drakensbergensis*. Similarly, there is poor knowledge as to whether grazing has an effect on inflorescence numbers and seed viability of the study species. However, some studies have observed that grazing may result in a negative change in inflorescence numbers (e.g. Bullock *et al.*, 1994).

This study has indicated that grazing numbers and frequencies have increased slightly during the three years of practical study within this region (2009-2011). Increased livestock numbers suggest an increased demand for palatable grasses (Volesky, 1994). However, when these grasses have been overgrazed, the livestock may graze less palatable, less nutritious species such as *M. drakensbergensis*. The increase in livestock numbers is believed to be mainly in the form of sheep and goats, which would alter the cattle to sheep grazing ratio. A ratio with greater numbers of sheep to cattle has been observed to result in a decrease in plant diversity near Kokstad, KwaZulu-Natal (O'Connor *et al*., 2011). Hence, the increase in sheep and goat

numbers may not be beneficial to the region. The increase in livestock numbers correlates to a decline in wool and mohair production by the animals, and therefore a decrease in products (Mpiti-Shakhane *et al.*, 2002). Overgrazing and damage to wetlands and slopes may therefore be a growing concern for both the future of the Lesotho highlands and the livelihoods of the people who rely on the land. According to local inhabitants of the Sani village, the grazing routes were altered at the end of May 2011. However, personal observations as well as the interviews with the herdsmen did not provide evidence to support this. Yet, it was observed that the grazing markers had been repainted white and in some cases moved slightly. New grazing markers were also observed towards the end of the A.P. site plateau.

Overgrazing, large herd numbers and population growth are not new problems facing the Lesotho highlands (Showers, 1989; Schwabe, 1995; Mbata, 2001; Bisaro, 2007), yet these problems are increasing each year. Improved management is required within these areas to maintain both biodiversity and livelihoods. The delegation of control over the grazing rights suggests that reducing livestock numbers and grazing frequencies may be very difficult, a thought shared by Du Preez and Brown (2011). Yet, a decrease in livestock numbers would mean healthier grasslands and livestock (Bisaro, 2007). However, seasonality has been demonstrated to have a greater impact on productivity than the number of individual livestock grazing an area, at an altitude of 2 275m in Lesotho (Mpiti-Shakhane *et al.*, 2002). Similarly, under heavy stocking rates (14 ewes ha^{-1} $year^{-1}$), the grass species were not significantly altered, but biomass was negatively impacted (Mpiti-Shakhane *et al.*, 2002).

5.4 Summary

The grazing study quantified the grazing routes, type of livestock, grazing trends and the colonization of invasive species used as overgrazing indicators. The following are the main findings:

1. Sheep, followed by Angora goats, form the majority of the livestock grazing within the Sani region, Lesotho.
2. The transhumance grazing system varies according to livestock type: sheep were observed in the study valley during the winter months, while cattle and Angora goats were observed in early winter and late winter.
3. Livestock are guided to grazing areas and the frequency of use is often indicated by pathways. The valley bottoms are used more frequently than slopes for grazing. The

duration of grazing is dependent on weather conditions (i.e. temperature, storms e.t.c.) and forage availability.

4. None of the study sites were grazed purposefully; rather they were grazed in passing as livestock were guided to new locations.
5. *M. drakensbergensis* tussocks were usually only grazed when the more palatable grasses were scarce or once post-burning resprouting had begun. No visible trend was observed as to the time of year when the tussocks were grazed at each site.
6. A general increase in invasive species coverage (*P. cooperii* and *C. ciliate*), as overgrazing indicators, suggests that overgrazing is a problem on the slopes of the study valley. *P. cooperi* increased by 9.0% between December 2009 and December 2010 at site S.Str., while unexplained losses in *P. cooperi* cover were measured at site V.S.N. (12.1% loss) and at the V.F. site (2.6% loss).

Chapter 6:

Fire Frequency and GIS Map



CHAPTER 6:

Fire frequency analysis

The construction of a fire history, in this case through the production of a burn frequency map, can assist greatly with understanding current burning regimes (Whelan, 1995). It can also assist with the design of appropriate fire management systems. In the recent past, the re-creation of fire histories has been undertaken through the use of satellite imagery, as is used here. The fire frequency maps were used to identify suitable sites to monitor and study *M. drakensbergensis* tussock growth, shape and burn recovery. Burn frequency data are also subsequently used to assist in discussions concerning tussock growth rates and shape. This chapter aims to produce a burn frequency map (1993 to 2011) for the study region using remote sensing. The produced map would enable the area burnt by each frequency to be calculated. The role aspect plays in burning was also observed through the location of burn scars on slopes. The chapter also aims to determine the burn scar size and the number of burn scars present at the end of winter (end of August to early September) each year. The locations of Merxmuellera communities within the study region were indicated using aerial photographs. The location of the communites, as well as field identification of *M. drakensbergensis* communities allowed for the selection of study sites for further indepth study in the following chapters.

6.1 Methodology

A GIS map depicting fire frequency was produced for approximately 22 049km^2 of the Sani Top and Kotisephola Pass regions using Landsat images. Landsat TM and ETM+ images were used as they were easily available, inexpensive and have an adequate spatial resolution of 15 to 30m (Kerr and Ostrovsky, 2003). The spatial and spectral properties of Landsat images are effective tools for burn scar mapping (e.g. Kontoes *et al.*, 2009; Minchella *et al*., 2009; Petropoulos *et al.*, 2010; Chen *et al*., 2011). Landsat images have been available since 1982, allowing for access to 18 years worth of images (1993 to 2008), as opposed to the shorter term availability of MODIS images (Dubinin *et al.*, 2010). However, it is rare to find an adequate sequence of images to accurately map a fire regime for an area (Laris, 2005), which was the case when searching for images with appropriate dates for eastern Lesotho. The images were obtained from the Satellite Applications Center (now SANSA) at Hartebeespoort and are listed in Table 6.1. Two images were required for each date as the Sani study region was divided between two rows (80 and 81) within the same flight path (169) (Figure 6.1). Snow covered the study region during the winter

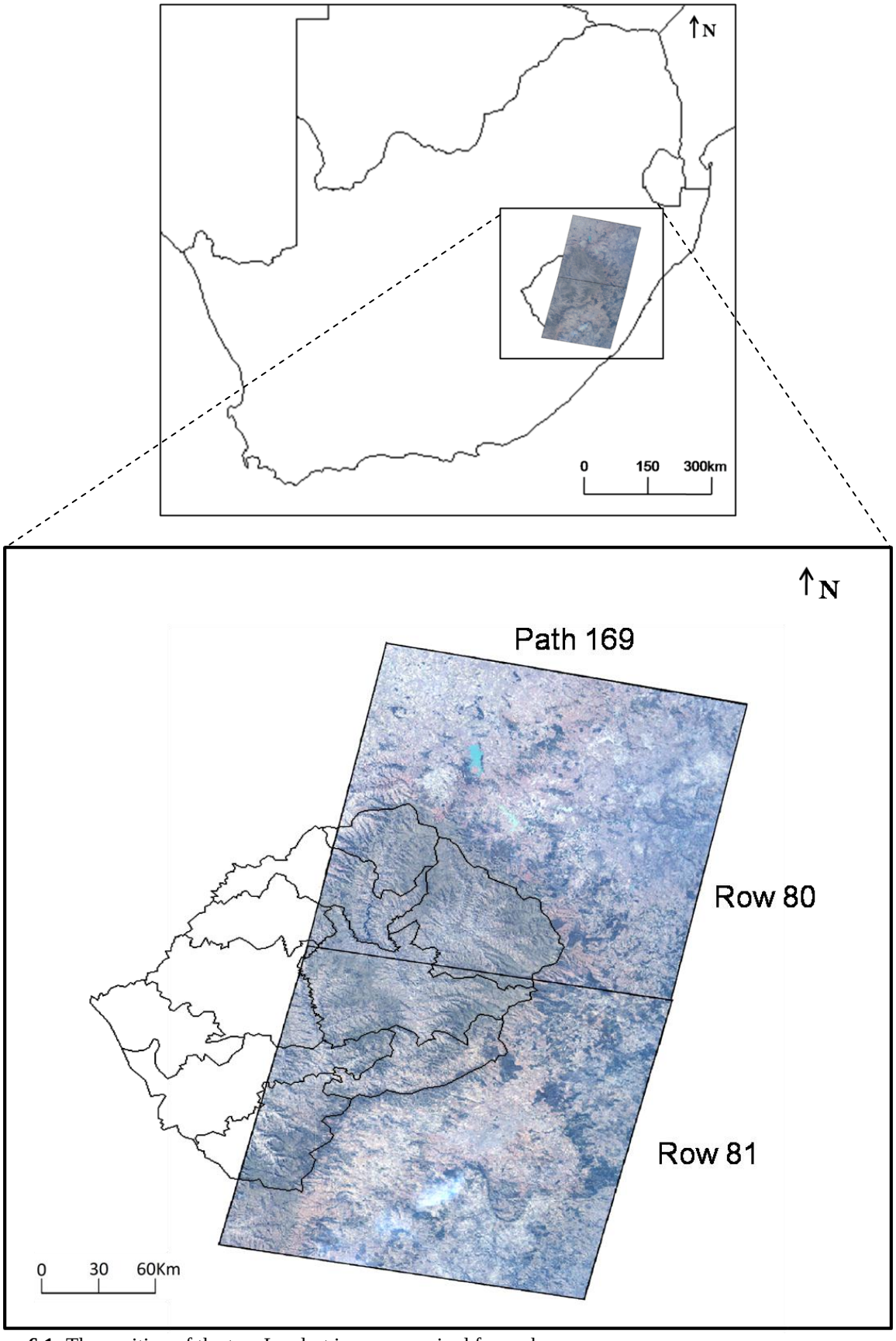


Figure 6.1: The position of the two Landsat images acquired for each year.

Table 6.1: Landsat images obtained for orbit 169, path 169 and rows 80 and 81.

Sensor	Date	Cloud Coverage (%)
TM	27/08/1993	0
TM	30/08/1994	0
TM	02/09/1995	0
TM	19/09/1996	0
TM	22/08/1997	0
TM	25/08/1998	<1
TM	28/08/1999	0
TM	30/08/2000	0
ETM	25/08/2001	<1
ETM	13/09/2002	<1
TM	08/09/2004	0
TM	29/09/2005	<1
TM	31/08/2006	0
TM	03/09/2007	0
TM	05/09/2008	0

images of 1997 and 1998, obscuring any burn scars. A thin cover of snow was observed in the 2006 images, but was restricted to high lying areas, with unburnt vegetation visible through the snow. Alternative image dates were selected where images contained snow cover. The Scan Line Corrector (SLC) mechanism on Landsat-7 ETM+ failed in May 2003, and has been termed 'permanent' as an attempt to correct the failure in September 2003 was unsuccessful (Markham *et al*., 2004). The SLC is fixed in one position allowing for gaps in the coverage at the image edges. However, the data remain of a similar quality to those before the failure (Markham *et al*., 2004). Landsat 5 did not pass over the study region at an appropriate date for this study; consequently no images were available for 2003. Landsat images were not available from 2009 due to reduced activity and hence reduced availability of images. Spot 5 images were used to supplement the Landsat images as the HRG-5 sensor recorded the visual, near infrared and shortwave infrared images required for the NBR methodology used in this study. Spot 5 images were used for 2009, 2010 and 2011 (Table 6.2). There were no Spot 5 images available for 2012 as the only suitable images were taken after a snowfall (10 August 2012). All the images obtained had already been corrected for any noise experienced and were north orientated. They were also projected at a geographic world, UTM projection with a WGS1984 datum.

Table 6.2: Spot 5-HRG images obtained for orbit path 137 and orbit row 409.

Sensor	Date	Cloud Coverage (%)
HRG-5	11/11/2009	0
HRG-5	02/09/2010	1
HRG-5	12/09/1011	11

Single post-burn images were used, as using both pre-burn images as well as post-burn images would have been too expensive for this study. Difficulty would have also been encountered as the vegetation is dormant in the post-burn images taken near the end of the winter period, but may still have been active in the late summer/autumn pre-burn images, thus leading to the incorrect identification of burn scars. The burn scars resulting from past fires, rather than active fires, were detected as this enables the entire burnt area to be recorded. Winter images are known to provide better results rather than spring images, as this allows for easier classification given that the difference between burn scars and vegetation are more prominent (Gulinck and Serneels, 1996). The images were chosen for dates at the end of winter, but not yet within the growing season, which begins in October, as recommended by Nüsser (2002; Table 6.1 and 6.2). Delaying the image dates improves the probability of including all winter burns. However, the success of burnt area classification is dependent on the time between burning and when the satellite image was captured, as burnt vegetation and ash will be altered by meteorological effects and regrowth (Hudak *et al.*, 2007). The only exception was in 2009, as the only available image was for the 11 November 2009. The burn scars are still clearly visible during the growing season which allows the images to be used in this study. In the rare event of burning occurring during the summer period, the burn scars were still visible in the image for the following year and were included. AFIS (Advanced Fire Information System) records were used to ensure all summer burn scars were included.

The satellite images were obtained in data format and had to be converted into a suitable format for use within a Geographical Information System (GIS). For this project, Idrisi Taiga was chosen as the GIS software, given its high quality imaging capabilities, availability and range of tools. The Landsat images were imported as individual bands into Idrisi Taiga as raster images using the GENERICRASTER tool (Figure 6.2). The Spot 5 images were imported as four bands using the GeoTiff conversion tool. The obtained images were georeferenced according to the

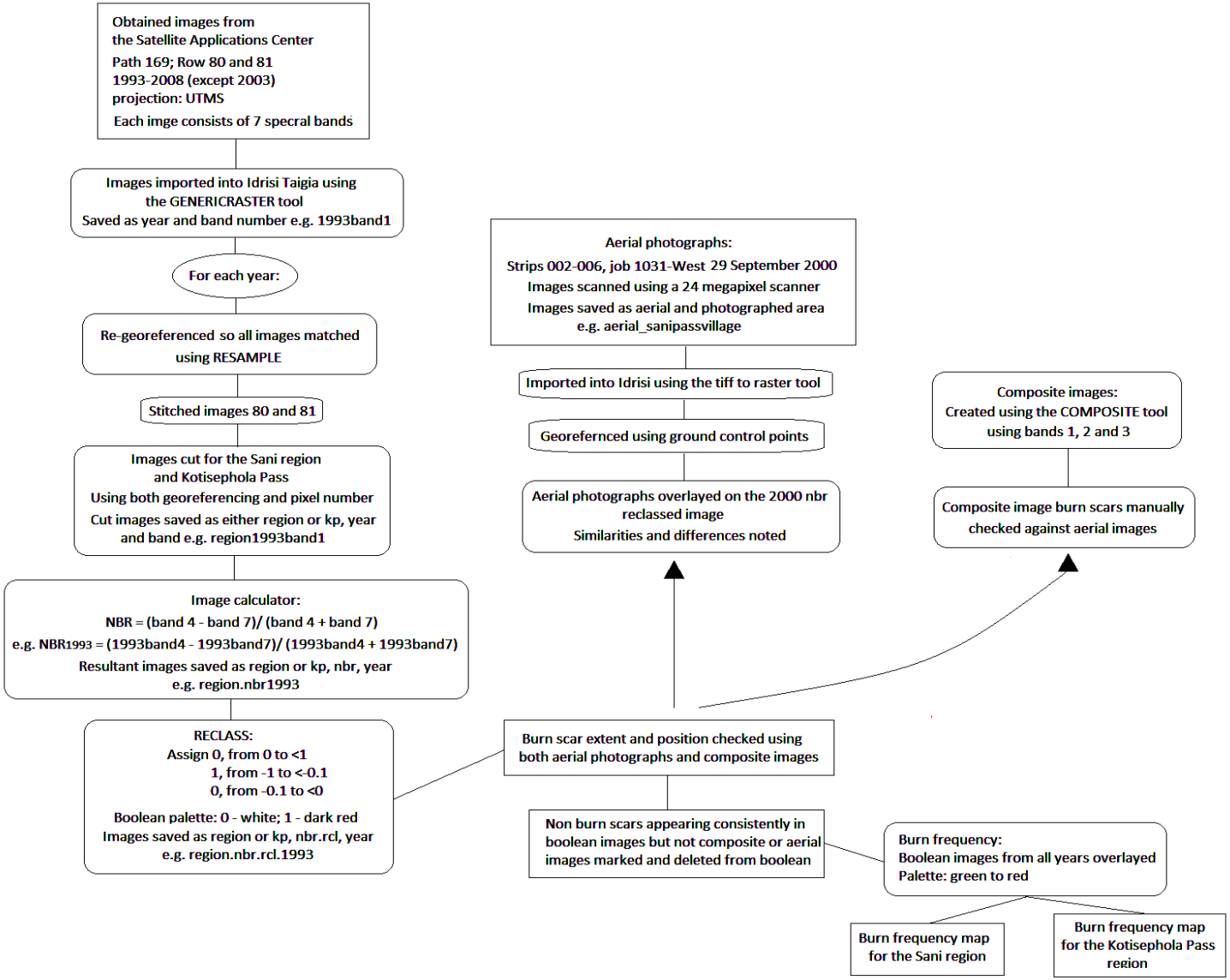


Figure 6.2: Flow diagram indicating the process used to create the burn frequency maps for both the Sani and Kotisephola Pass regions.

header files provided, as a UTMS (UTM-29S) projection (datum used was WGS84). However, co-ordinate discrepancies were observed between each of the images and thus all the images were geo-corrected to ensure consistency between the images. Discrepancies would mean that the images would not overlay correctly and thus the burn scars would not be accurately positioned, resulting in errors within the final burn frequency map. The images were geo-corrected using the RESAMPLE tool. The 1994 image georeferencing values were consistent with the values obtained by thorough ground truthing and those on the DEM file, thus was used as a reference to which all other images were re-georeferenced. Six ground control points were used from different locations (e.g. rock outcrops and stream confluences) within the images: Sani top sign position, a visible outcropping near Hodgson's north peak, two river confluences, a road junction and a peak to the north of the Sani top study valley. Nearest neighbour assignment was used to reassign values. An RMS (Root Mean Square) error value was obtained and the resultant differences from that of the 1994 image are displayed in Table 6.3.

Table 6.3: The root mean square value error values for each image and the difference in meters.

Year	RMS	Difference (m)
1993	0.0014	0.0044
1995	0.0008	0.0013
1996	0.0002	0.0003
1997	0.0004	0.0009
1998	0.0006	0.0006
1999	0.0007	0.0003
2000	0.0004	0.0007
2001	0.0276	0.6674
2002	0.0074	0.0398
2004	0.0013	0.0038
2005	0.0019	0.0075
2006	0.0014	0.0043
2007	0.0006	0.0009
2008	0.0013	0.0020
2009	0.0006	0.0007
2010	0.0011	0.0018
2011	0.0009	0.0016

The use of two satellite path rows (80 and 81) for the Landsat images meant that the obtained images of the Sani region required cutting and stitching. The images were stitched together with

image 80 positioned to the north of 81. The images were cut according to co-ordinates as well as rows and columns to ensure all the images had the same number of pixels to allow for easier overlaying. The images were cut into two separate images for each year; one for the Sani region and another for the Kotisephola Pass region. Using separate images allowed for an increase in detail and thus an increased analysis accuracy and more detailed selection of study sites.

Two different methods which have been used in previous studies were tested in order to obtain the best results. Both methods were undertaken in order to determine which produced the most accurate and visually clear results. The two methods were the Normalized Difference Vegetation Index (NDVI) and the Normalized Burn Ratio (NBR). The NDVI was initially undertaken as this reflects the green vegetation (Salvador *et al.*, 1999) and has previously been successfully used in burn scar identification (e.g. Diaz-Delgado *et al.*, 2002; Clark and Kuyumjian, 2006; Goetz *et al.*, 2006; Lentile *et al.*, 2006; Chen *et al*., 2011). The second method undertaken was the NBR, also due to its previous success in burn scar identification (e.g. Dottavio and Dottavio, 1984; Epting *et al.*, 2005; Lentile *et al.*, 2006; Chen *et al*., 2011). Firstly, the NDVI was calculated for each image using the VEGINDEX tool in Idrisi, using equation 3.

$$NDVI = (NIR - R) / (NIR + R) \qquad -1 \leq NDVI \geq 1 \qquad (3)$$

$$NDVI_{Landsat} = (band\ 4 - band\ 3) / (band\ 4 + band\ 3)$$

$$NDVI_{Spot\ 5} = (band\ 3 - band\ 2) / (band\ 3 + band\ 2)$$

A boolean image was then generated, highlighting the position of only the burn scars. To do this, the image was reclassed according to the burn scar values and thus a suitable threshold needed to be established, following the creation of the NDVI images, as per Fraser *et al.* (2000). Many different threshold values were tested (e.g. the burn scars were within the ranges of -0.5 to 0, -0.5 to -0.05) and processed with the RECLASS tool. The reflectance of a small snow patch on a peak in two images (1996 and 1999) appeared as bare soil and later as a burn scar. This demonstrated that the NDVI had a tendency to label landcover with similar reflectance values as a burn scar, despite the area not having been burnt; a problem also observed by Fraser *et al.* (2000). Similarly, areas of bare ground were also classified as burn scars, and this was expected according to the literature. The use of the soil adjusted vegetation index (SAVI) to overcome this problem proved unsuccessful in areas containing snow cover. Thus, the Normalized Burn Ratio (NBR) method was tested.

Although the use of the NBR has raised concerns when applied to grasslands or shrublands (e.g. Lentile *et al.*, 2006), it produced results with a high burn scar clarity, visibility of both recent and older burns for the current study region. It was also simple to perform, reducing the time cost of the study. Previously, the NBR had also provided better correlations to post-burn vegetation than soil characteristics (Hudak *et al.*, 2007), thus making the NBR appropriate for use in this study. It was preferable to use both pre and post burn images (commonly termed the dNBR) for identification of burn scars (Hudak *et al.*, 2007), but the single date NBR was used in this study as it reduced both financial and time constraints, but was as effective (as demonstrated by Epting *et al.*, 2005). The NBR was calculated using equation 4 and applied to each image through the 'image calculator' tool.

$$NBR = (NIR - MIR) / (NIR + MIR) \qquad -1 \leq NBR \geq 1 \qquad (4)$$

$$NBR_{Landsat} = (band\ 4 - band\ 7) / (band\ 4 + band\ 7)$$

$$NBR_{Spot\ 5} = (band\ 3 - band\ 4) / (band\ 3 + band\ 4)$$

To separate areas of sparse vegetation cover and burnt areas, an unsupervised classification was used, as per Justice *et al*. (1997). The RECLASS tool was used to classify the burn scars and not the surrounding landscape within each cropped image. The values which were initially chosen follow suggestions by Key and Benson (2004), where pixels with a negative value were identified as burn scars. Yet, it was observed that strongly negative values were not necessarily burn scars. Further adjustment through trial and error was made to the simple original parameter and its decimal values until the burn scar was accurately depicted when compared to the composite images using the OVERLAY tool and visual observations. The final RECLASS parameters for the Landsat images selected were as follows: a value of 0 was assigned to pixel values from 0.0 to values asymptotically approaching 1, a value of 1 was assigned to values from -1 to values asymptotically approaching -0.1, and a value of 0 was assigned to values from -0.1 to values asymptotically approaching 0. The Spot 5-HRG images required slightly different parameters: a value of 0 was assigned to pixel values from -1 to values asymptotically approaching 0.18 (0.15 in 2010), a value of 1 was assigned to values from 0.18 (0.15 in 2010) to values asymptotically approaching 0.55 (0.3 in 2010), and a value of 0 was assigned to values from 0.55 (0.3 in 2010) to values asymptotically approaching 1. A boolean palette (saved as Booleanred-white) was created to indicate the burnt areas, where burn scars with a value of 1 were displayed using dark red, while unburnt areas were displayed as white.

Once boolean images were produced for both the Sani Pass and Kotisephola Pass regions for each year, the burn scar extent and position was verified. For this, both aerial photographs and composite images were used. The extent covered by aerial photography is much smaller, allowing for more detail on burn scar extent and margins (Figure 6.3). Aerial photographs (strips 002-006, job 1031-West 29 September 2000) were obtained from a previous study and scanned using a 24 megapixel scanner. These images were then imported to Idrisi Taiga using the .tiff to raster tool. The photographs were georeferenced according to the same ground control points used in the georeferencing of the satellite images. Similar to the NBR images, the aerial photographs were reclassed to indicate only the burn scars. This was undertaken using the burn scar pixel values from within the aerial images and the RECLASS tool. This allowed the reclassed aerial photographs to be overlayed onto the cropped 2000 NBR image and the reclassed image, with the similarities and differences between the burn scars noted. There was a 94% similarity in burn scar identification between the reclassed aerial photographs from 2000 and the 2000 reclassed image, with the NBR image classifying a few rock outcrops and small areas of bare soil as burn scars.

Colour composites were also used to visually identify and note the shape and size of burn scars, before the NBR was applied, as recommended by Key and Benson (2006) and Dubinin *et al.* (2010). Similarly, they were used after the NBR was applied to verify the results through visual comparison. Thus, the burnt and unburnt vegetation classes were determined through a combination of manual and computational determination techniques, as used by Amenteras *et al.* (2005). The colour composites were generated using the COMPOSITE tool for bands 1, 2 and 3. The burn scars visually appeared consistent with those obtained on the NBR images and the reclassed images. Eight small ($<225m^2$) clusters were depicted as burn scars on the reclassed images, yet did not appear on the aerial images. These false burn scars were also noted when visually compared to the colour composites. They appeared consistently throughout the years of study and were removed from the final boolean images using Microsoft Paint.

The NBR images were visually compared to the online AFIS (Advanced Fire Information System) records of then current burning to determine whether any correlation existed between recorded fires and the map produced. This technique was also used in a European context using the Fire Information for Resource Management System (FIRMS) to test the accuracy of the satellite data (Sedano *et al.*, 2013). It was noted that the AFIS images were an under-estimation of the number of burns indicated by the burn scars each year, this is perhaps due to the method of

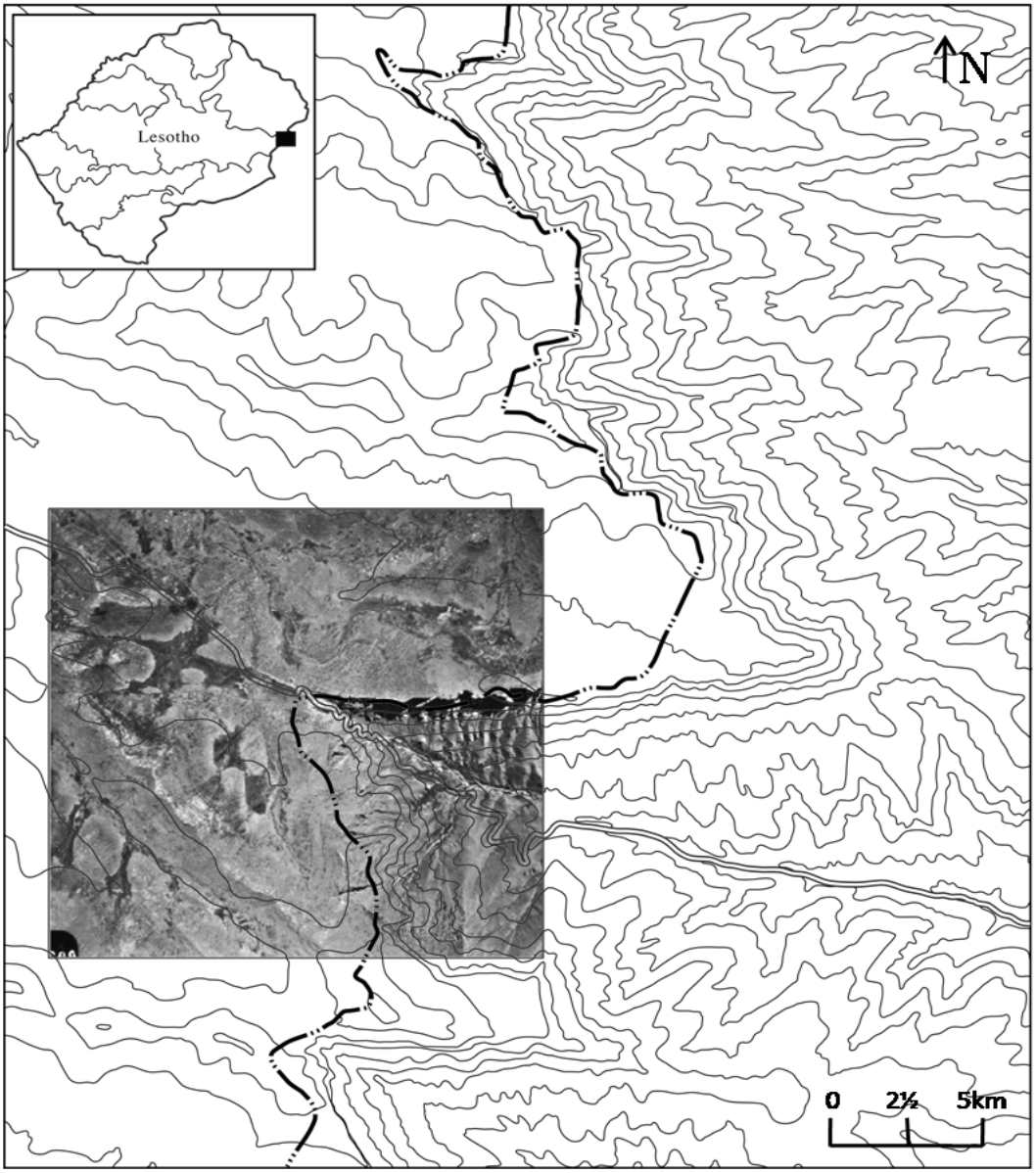


Figure 6.3: The extent of the study region covered by each aerial photograph, an example from Sani Top.

recording (placing single blocks over an area observed to be actively burning), which may reduce both the number of separated burns and the extent of a burn, as well as the resolution used (4.8km using the Modis Satellite system or 1km using the Polar Orbiting system; Steenkamp and Frost, 2008). It has also been observed that much of the burning occurs at night for warmth and these values may be underestimated. A 30m Digital Elevation Model (DEM) was obtained from the ASTER database (version 2, 0.4 arc second DEM, a product of METI and NASA) and was used to generate contour lines of the study regions through the CONTOUR tool, which were overlaid onto the Boolean images to indicate the position of the burnt areas within topographic settings.

Further analysis was conducted on the Boolean images for each year to provide more information on the size and distribution of the burn scars observed each year. In order to gather the information, the images were imported into ArcGIS Desktop version 10 (a product of ESRI, 2010) as it is an easy to use and appropriate software program with the capability of performing these analyses. Each image was imported as a raster layer through ArcCatalogue and saved. The spatial reference parameters were checked in ArcCatalogue to confirm that the images were a Geographic projection with a world datum of WGS1984. The raster images were then added to ArcGIS Desktop 10 and projected using the *Data management* tool. The output co-ordinate system selected was UTM for the southern hemisphere, zone 29 south, and again the datum used was WGS1984. Once the images had been projected, the Boolean images were converted from raster to polygon images using the *Conversion Tools* utility. Again the geodatabase selected was WGS_1984_UTM_Zone_29S, and the output image was saved. The projected polygon images were then able to be analyzed for their average size, number of burn scars and average distance to the nearest burn scar. The *Calculate area* utility found within *Spatial Statistics Tools* was performed with the output saved. The attribute table was then opened and exported to Microsoft Excel for summarizing and would provide an indication of the size of each individual burn scar. Similarly, the polygon images were used to calculate the average distance to the nearest burn. This was performed using the *Average nearest neighbour* utility within the *Analyzing patterns* tool in the *Spatial Statistics* toolbox. A Euclidean distance was used as opposed to a Manhattan distance as this was a more direct line between burn scars. Due to the skewed nature of the data, the medians obtained from the nearest neighbor analysis were compared to determine whether differences existed between the years. The nearest neighbour data did not meet the assumption of normal distribution, thus a non-parametric Kruskal-Wallis test was conducted.

The number of burns located on each aspect as well as in flat areas was also calculated in ArcGIS version 10 (Figure 6.4). This was performed through the creation of a personal geodatabase in ArcCatalogue and generated a polygon feature class layer for each aspect (e.g. NORTH) within each of the two geodatabases. In ArcMap 10, a clear NBR image of the region (1993) was used as a template for drawing polygons identifying areas of each aspect. The feature class layers were then used to discover areas where the aspect polygons overlapped with the burn scar polygons created earlier. This was conducted using the *Intersect* tool within the *Overlay* tools (*Analysis Tools*), and the results produced a map depicting which burn scars fell within the polygons depicting each particular aspect. The attribute values tables were exported as reports for summarizing. The flat areas are indicated by the areas not covered by any of the aspect

polygons, and the burn scars which fell into this area needed to be calculated. The polygon feature files for each aspect were combined using the *Union* tool (within *Overlay* tools, *Analysis Tools*), to form a single feature image. The *Symmetrical difference* analysis was performed using the individual polygon burn scar images for each year and the unified aspect feature layer. The resulting image depicted the burn scars outside the aspect polygons (i.e. on flat land) and the data were exported from the attribute values tables into Microsoft Excel to be summarized. This procedure was undertaken for each aspect within each year and the results tabulated. Significant differences between the aspects were determined using a Friedman test as the data did not meet the assumptions of normal distributions and homogeneity of variance. Additionally, the data may be considered to contain related samples as the same areas were observed each year. However, the area occupied by each aspect was different and thus the data was weighted according to the percentage each aspect occupied. The Friedman test was undertaken using SPSS version 21.

The Landsat images were cut to the same number of rows and columns resulting in images composed of the same number of pixels. The Spot 5 images were cut to represent the same area using landmarks and co-ordinates. However, due to the difference in resolution (Landsat: 15-30m; Spot 5: 10m) the number of pixels in the Spot 5 images were different to that of the Landsat images. Using ArcGIS, the Spot 5 images were resampled using the raster processing *Resampl*e tool and where needed further altered using the raster *Rescale* tool, until the Spot 5 images matched the number of rows and columns of the Landsat images. In order to produce a burn frequency map from the Boolean images, the individual images for each year were overlaid via addition (the images were added in chronological order) using the IMAGE CALCULATOR tool within Idrisi Taiga. The product was an image depicting the number of times each pixel (at 15-30m resolution) was burned during the 15 years prior to site selection (1993 to 2008; with the exception of 2003). An appropriate palette had to be created to appropriately depict the different burn frequencies. A multi-colour palette was tested but the result was a confusing mosaic of 15 different colours, thus a green to purple distribution palette was generated. An additional palatte was created with the addition of a third colour, red, to better indicate the different burn frequencies in the burn frequency maps from 1993 to 2011. Along with indicating burn frequency, the maps were also able to indicate the time since last burnt, which is important as the species diversity changes with time since the last burning event. Further, the vegetation after a burn event is largely related to the vegetation type which was present before burning (Whelan, 1995), thus the burn history of an area may serve as an indication of future vegetation changes that might be expected in the area.

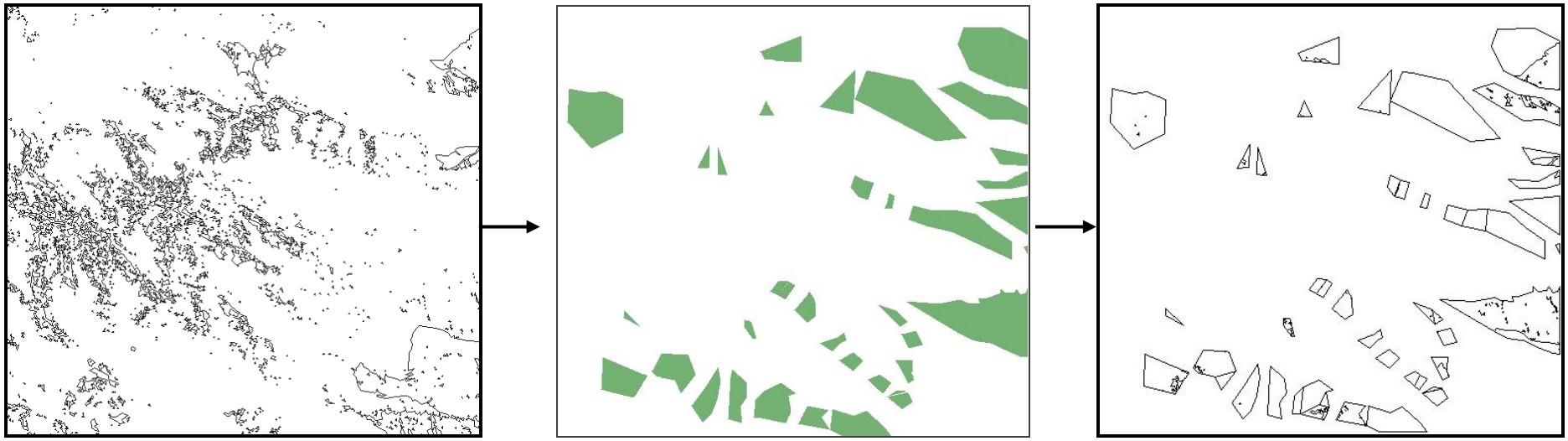


Figure 6.4: The method used to determine which burns appeared on each aspect.

A distribution map of *Merxmuellera* sp. was generated for *Merxmuellera* species in general. A 24 megapixel scanned aerial photograph of the Sani region was imported into the GIS software (Idrisi Taiga) through the .tiff to raster function and georeferenced according to the same ground control points used to georeference the satellite images. The image was cut slightly to remove any distortion near the edges and a grayscale palette applied. The image was then reclassed to depict areas where *Merxmuellera* tussocks were present. This was done through initially identifying the tussock grasses on the satellite image, assisted by GPS co-ordinates obtained in the field, and noting their spectral value for several points within the image. The tussocks were also easier to pinpoint as they were of a darker green than the surrounding short sod grasses. The RECLASS tool was then used to filter out the pixels with other values than that of the tussock. A value of 0 was assigned to pixel values from 0 to just less than 20 003, a value of 1 to values from 20 003 to just less than 27 005 and a value of 0 to values from 27 005 to just less than 57 708. The resultant boolean image (using the red-white palette) produced depicted the location of *Merxmuellera* tussocks within the study region. This was overlayed by a DEM (obtained from the ASTER database, version 2) produced contour map to give the location of the red pixels a spatial perspective. However, the location of the study species, *Merxmuellera drakensbergensis*, and the communities it forms had to be established manually due to the inability of Landsat to differentiate between different species as observed by Sever *et al.* (2012).

6.1.1 Image Analysis

The burn frequency maps for both the Sani and Kotisephola pass regions were used along with the map of *Merxmuellera* distribution to select the most suitable individual study sites. The first step towards study site selection was the identification of *Merxmuellera drakensbergensis* communities within the broader *Merxmuellera* community, through field observations. Tussock monitoring sites were chosen within the *M. drakensbergensis* communities so as to represent different times since the last burn in order to observe the impact of infrequent and frequent burns. The sites were also selected on an altitudinal basis to enable comparison between growth rates and shape between different heights above sea level. Slope aspect was also taken into consideration to enable comparison between these. Where possible, similar topographical settings were selected for two sites, preferably near each other, but differ between either burn frequencies, the time since the site was last burnt, aspect or altitude (Figure 6.5). For example, two sites were selected below Hodgson's north peak at the same altitude, and with the same burn frequency and time since last burnt, yet differ in terms of aspect.

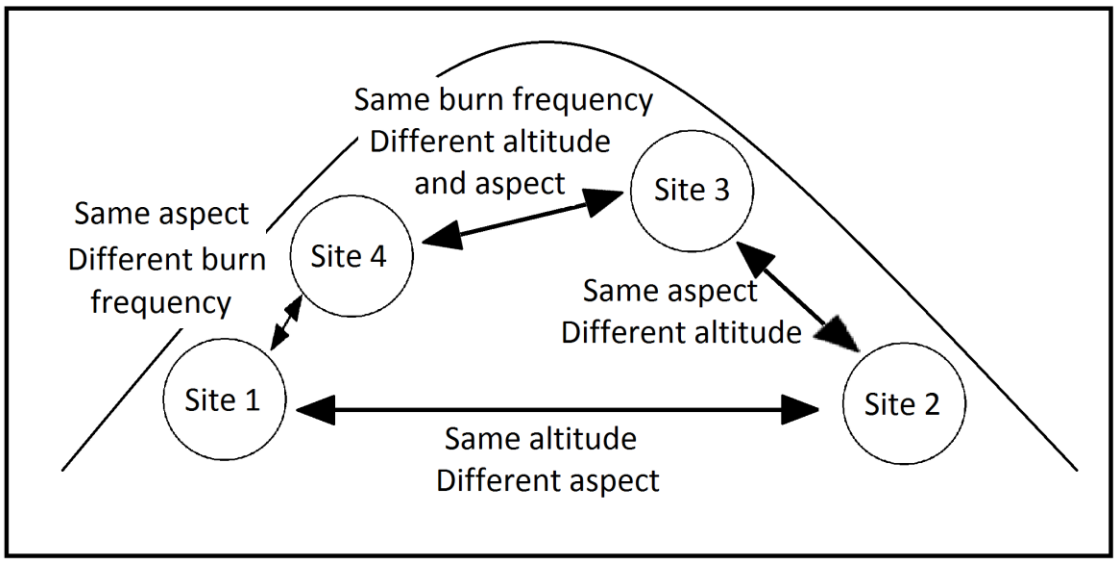


Figure 6.5: Example of how sites were selected.

6.2 Results of the GIS study

Burn scars in regions with steep slopes, heterogeneous vegetation and intricate topography are difficult to measure using traditional field based methods (Escuin *et al.*, 2008). Thus, satellite images were used to construct a fire frequency map. Two methods were tested to determine which method was the most accurate for the identification of burn scars within the study region. These two methods were the NDVI (equation 3, chapter 6.1) and the NBR (equation 4, chapter 6.1). The output images were cut to highlight the study region in increased detail. The resulting images were compared to each other and to both composite images and aerial photographs in the case of the year 2000. The initial observation was that the burn scars appear more visible in the NBR image than in the NDVI image, where burn scars appear as darker areas (>0.50) as opposed to the lighter (around 0.00), less visible, areas in the NDVI images (Figure 6.6). Attempts at reclassifying the burn scars through the RECLASS tool proved more complex with the NDVI than the NBR. A lower ccuracy is measured between the classified NDVI images, colour composites and the aerial photographs (87% accuracy), than with the NBR (94% accuracy). In the literature, both the NBR and dNBR were also observed to have a lower accuracy for low severity burnt areas than unburnt or higher burn severity areas (Bobbe *et al.*, 2003). The GIS map for the current study supported this result, as the reclassing had to be altered to include the burnt areas with lower pixel values visible in the colour composites and the aerial photographs. Once it was established that the NBR was the most appropriate for this study, it was applied to the images obtained for each year. Two smaller images were produced for enhanced detail;

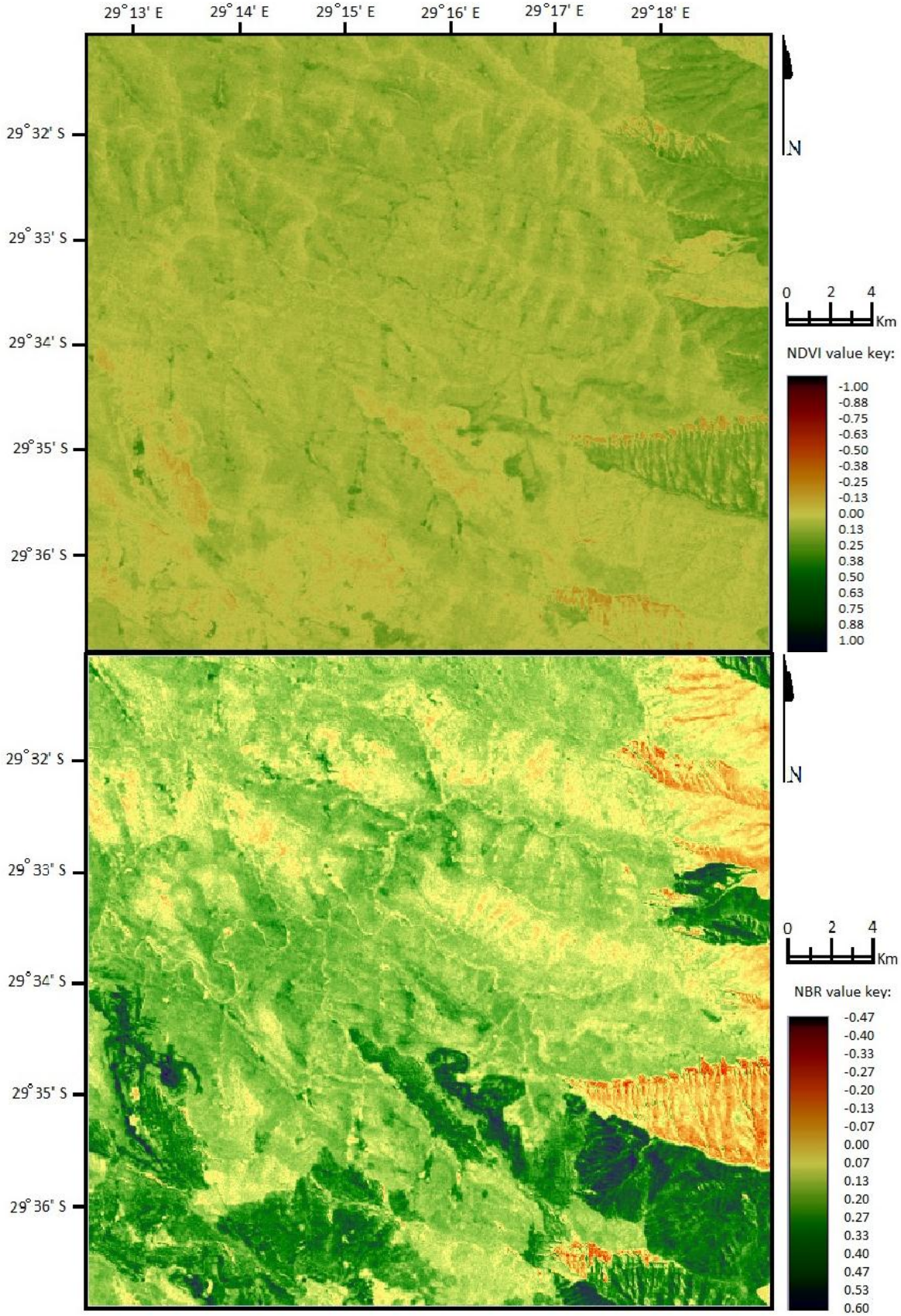


Figure 6.6: NDVI image (top) and the NBR image (bottom) of the Sani region in 1998.

namely fire frequency maps of the Sani and Kotisephola Pass regions. These were analyzed separately so that as much detail could be included as was possible.

6.2.1 Sani region burn scars

The percentage area burnt and the percentage area remaining unburnt in the Sani pass region each year are indicated in Figure 6.7. Burnt areas represent less than 30% of the area in the Sani region each year (omitting 2003). More than 10% of the study region was burnt in 2007 (23%) and 2010 (13%), with the largest burnt area recorded for 1998 when 25% was burnt. Substantial snow fall events as observed by Mulder and Grab (2009) were not significantly correlated ($p = 0.2621$) to the annual percentage burnt area between 1993 and 2004. Yet, five substantial snowfall events during 1996 may explain why only 0.3% of the Sani region was burnt. Similarly, eight substantial snow falls may explain the 2.2% burnt area observed in 2002.

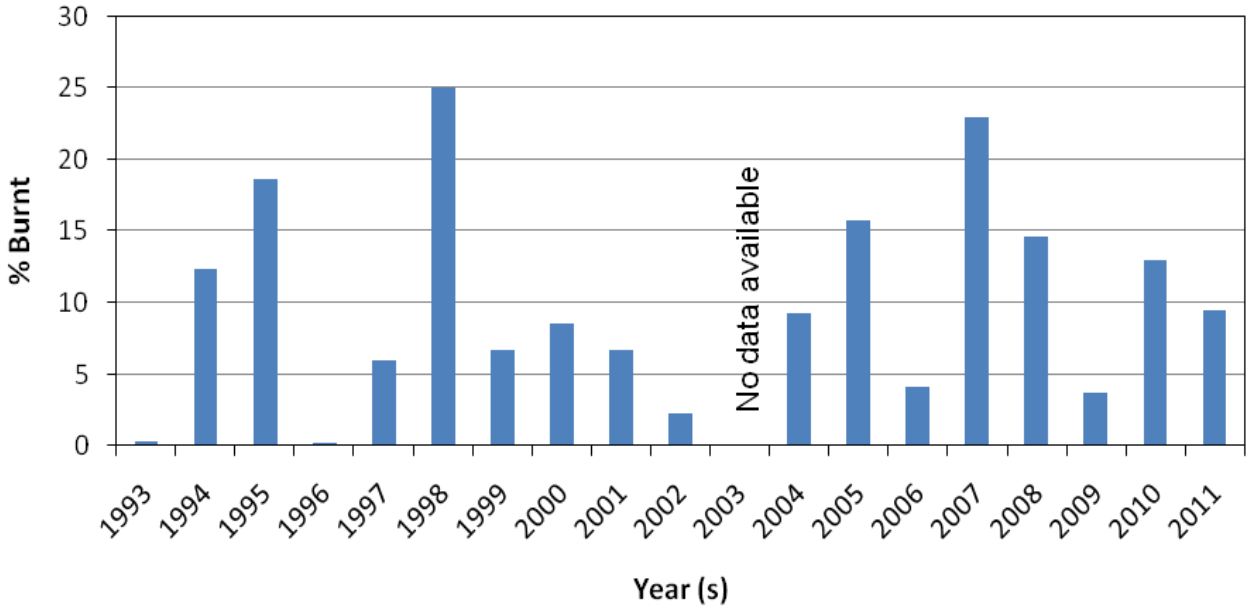


Figure 6.7: The percentage burnt area in the Sani Pass region betwen 1993 and 2011.

The reclassed boolean NBR images confirm the findings presented in Figure 6.7, with almost no burns observed in 1993 and 1996 (Figure A.3). Large burn scars (>20 000m^2) are observed on the Sani Pass valley slopes in 1994, 1997, 1998, 1999, 2005, 2007, 2008, 2009 and 2011 (Figure A.3, A.4 and A.5). However, these burns only breached the edge of the escarpment and extended inland in 1994, 1998, 1999, 2007 and 2009. In 2005, the burn scars extend inland for up to two kilometres and cover the Hodgson's Peak area. Patchy burns ranging in extent between 1 000m^2 to 3 000m^2 were observed in 1995, 2005, 2007, 2008 and 2010. Large burn scars (10 000m^2 to 20 000m^2) are observed inland during 1999, 2001, 2004 and 2010. Many smaller patchy burn

scars (<100m^2) cover the greater Sani region in almost all years, except for 1993, 1996 and to an extent 2009. The burn scars appear divided by streams and in some instances by roads. Similarly, rocky peaks remained unburnt in the majority of the years.

6.2.2 Kotisephola region burn scars

Less than 20% of the Kotisephola Pass region was burnt each year between 1993 and 2011 (Figure 6.8). The least amount of burning took place in 1996 when only 0.5% of the area was burnt, and is closely followed by that of 1993 when only 0.8% of the area was burnt. The greatest area burnt in the Kotisephola Pass region took place in 2008 when be 17.2% was burnt. More than 10% was burnt in 1995 (12.8%), 2007 (13%), 2008 (17.2%) and 2010 (13.1%). The annual percentage burnt area does not significantly correlate ($p = 0.1976$) to the number of substantial snowfall events between 1993 and 2004, as observed by Mulder and Grab (2009). However, the low percentages of burnt area in 1996 (0.5%) and 2002 (1.6%) may be explained by the high number of snowfall events during those years (five and eight snowfall events respectively). Similar to the images of the Sani region, the Kotisephola Pass region experienced almost no burning during the winter of 1993 and 1996 (Figure A.7). A single large burn scar (greater than 16 000m^2) is observed for 2009, while four large burn scars (>9 000m^2) are observed for 2010 (Figure A.7 and A.8). In 2008, the burn scar extent increased with many greater than 500m^2. Small patchy burn scars (<100m^2) are present in many of the Boolean images, such as 1995, 1996, 1998, 2000, 2006 and 2010. The burn scars were most commonly observed on the slopes towards the north of Kotisephola Pass. The majority of the burn scars are observed along the slopes and very rarely cross the watersheds.

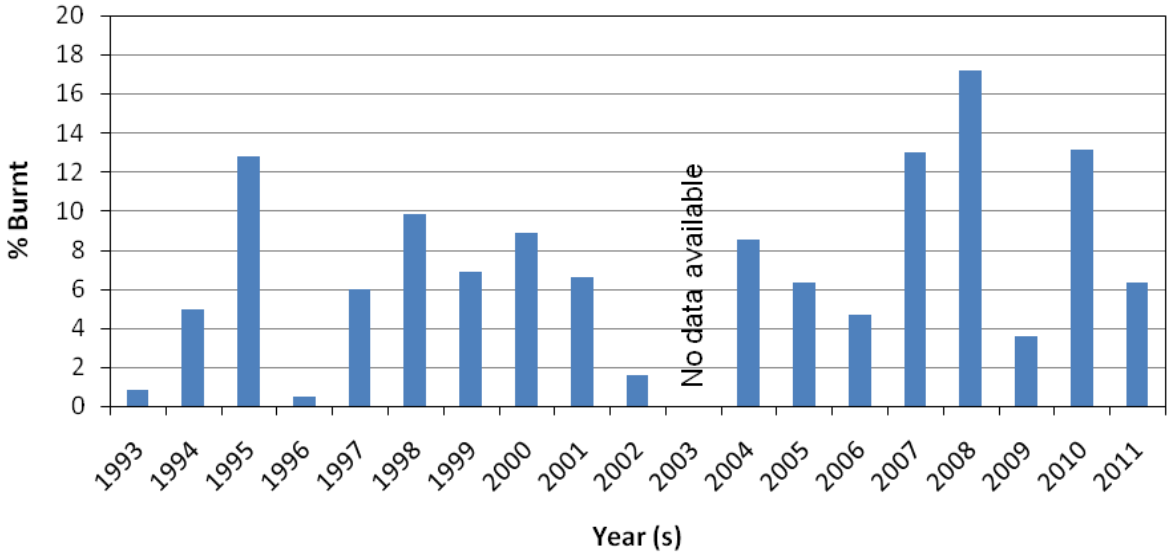


Figure 6.8: The percentage burnt area in the Kotisephola Pass region betwen 1993 and 2011.

6.2.3 Burn frequency map and burn scar data

The size of the observed burn scars was small (<500m^2) when compared to the large scale burning events seen along the escarpment edge (>20 000m^2), especially between the South African and Lesotho border posts, and those observed within the Drakensburg foothills, South Africa. Some of the small (<100m^2) patchy burn scars were observed in the same location on the majority of the images, and thus for many of the studied years. These were identified through the use of aerial photographs and composite remotely sensed images, as well as in the field, as areas containing large boulders, outcrops or exposed damp soil and not burn scars. Yet, these features reflected similar negative values to burnt tussocks which were then reclassed to depict burn scars. Thus, explaining why the NBR classified images showed a slightly higher number of burn scars than observed in the field. These false burn scars were removed from the boolean burn scar images for each year before they were overlayed.

The boolean images for each the Sani region and the Kotisephola Pass region were separately overlayed from 1993 to 2011to produce burn frequency maps (Figure 6.9 and 6.10). To assist with the analysis and selection of study sites, the images were also only overlayed for the 10 years prior to the start of the study (1998 to 2008) (Figures 6.11 and 6.12). As observed in the Boolean images, the marshy areas in the north of the study site have a very low burn frequency (0 to 1 burn per 10 years), possibly due to moisture in these areas preventing burning. It is observed in many of the images that rivers, streams and marshy areas formed the borders or created a mosaic of burn scars, thereby acting as natural fire breaks. The majority of the southwest-facing slopes appear to have only been burnt once between 1993 and 2011 (without considering 2003) and some have not been burnt at all, while the northeast-facing slopes have a burn frequency of between 0.07 and 0.6 burns per annum between 1993 and 2008. Burn scars are not observed along the upper mountain ridges and spurs as these are mainly covered with exposed rock. Wetlands have been burnt once or remain unburnt across the 19 year burn frequency map, despite many within the study valley becoming dry on the surface over the winter months. The flat areas divided by streams and wetlands to the south of the main road have a high burn frequency (between 0.33 and 0.53 burns per annum between 1993 and 2008), and similarly so for the streams and depressions to the southwest of the study area. Large extensive burn scars are observed along the escarpment edge towards the southeast of the study site.

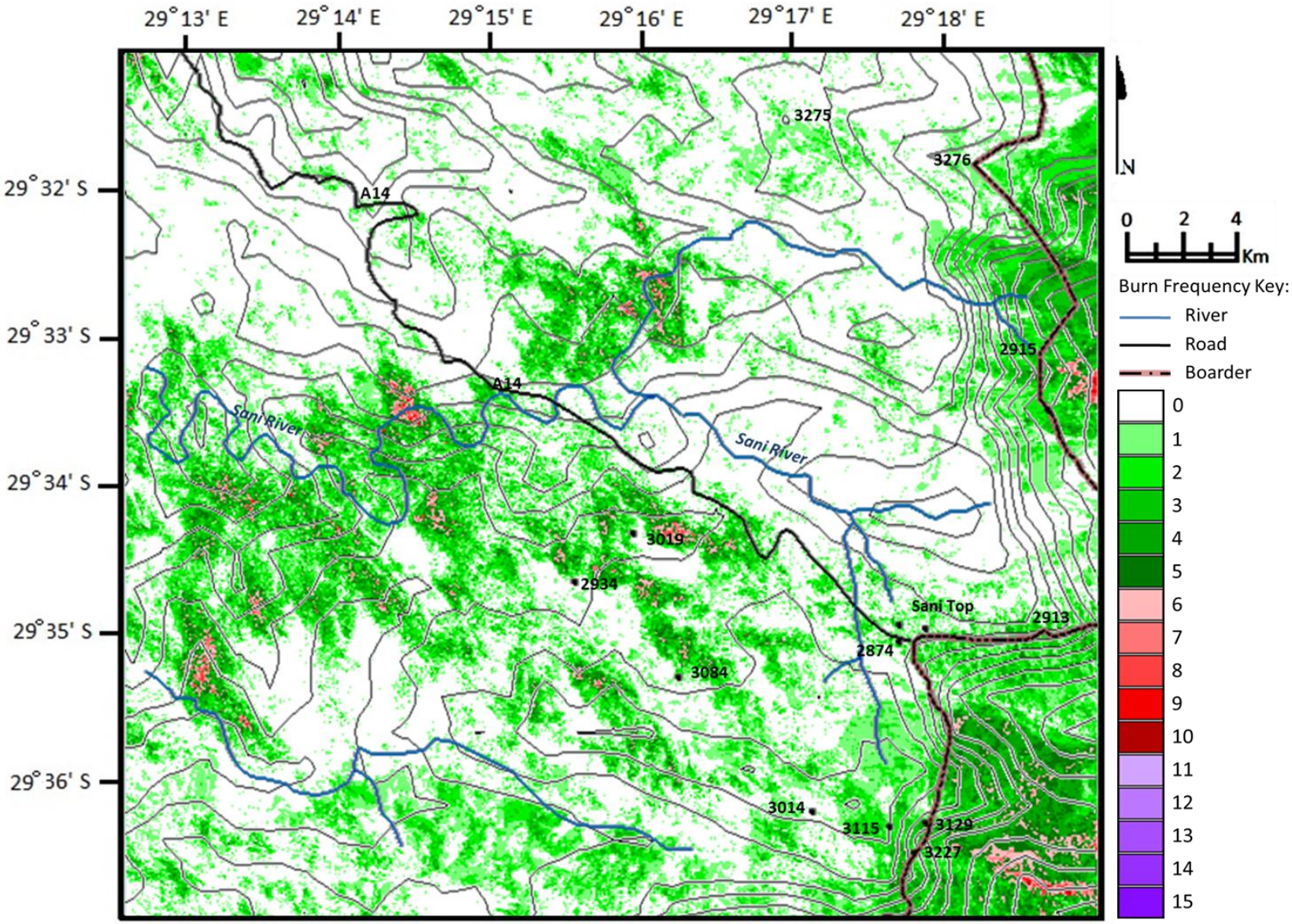


Figure 6.9: Burn frequency map of the Sani Region from 1993 to 2008.

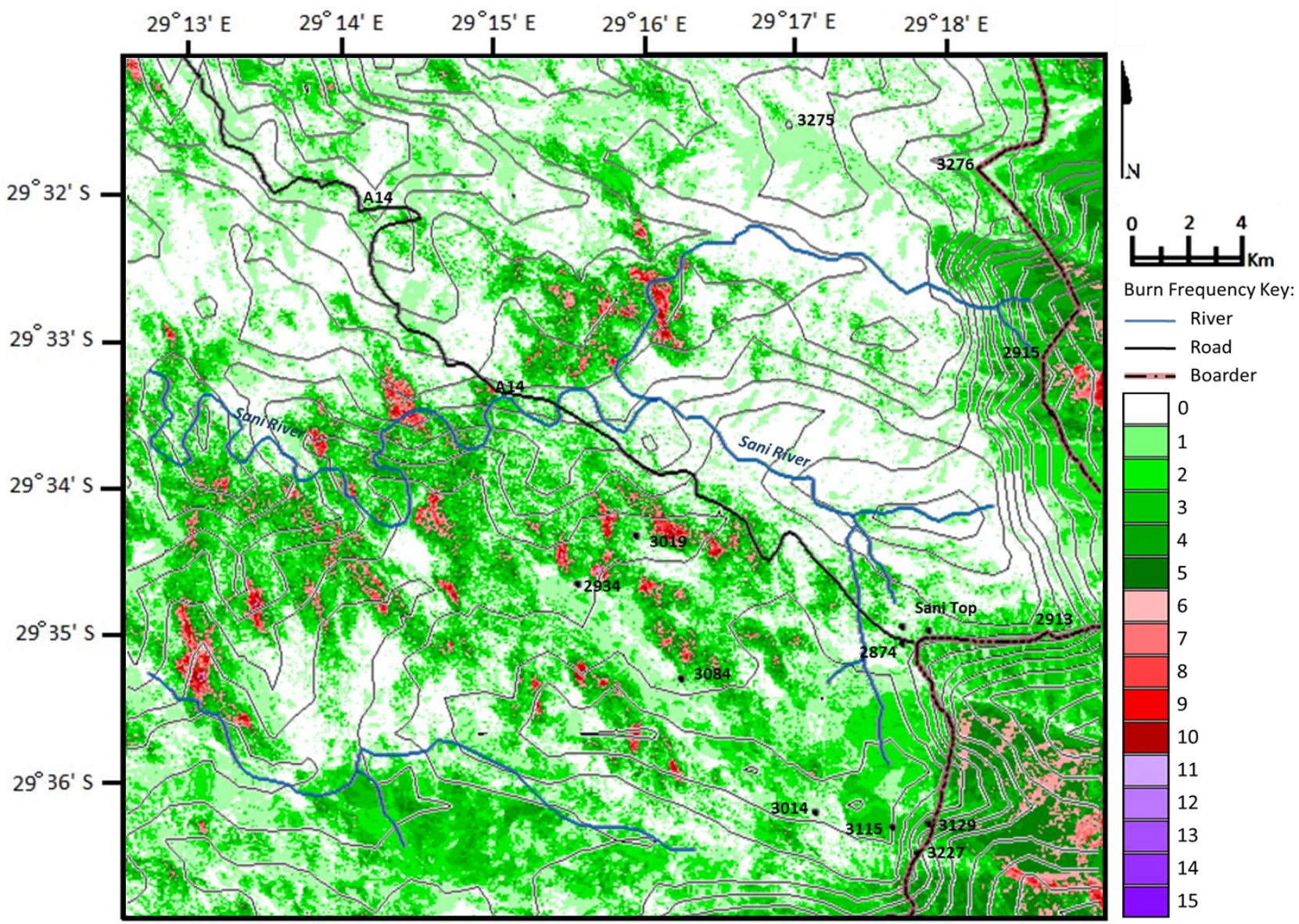


Figure 6.10: Burn frequency map of the Sani Region from 1993 to 2011.

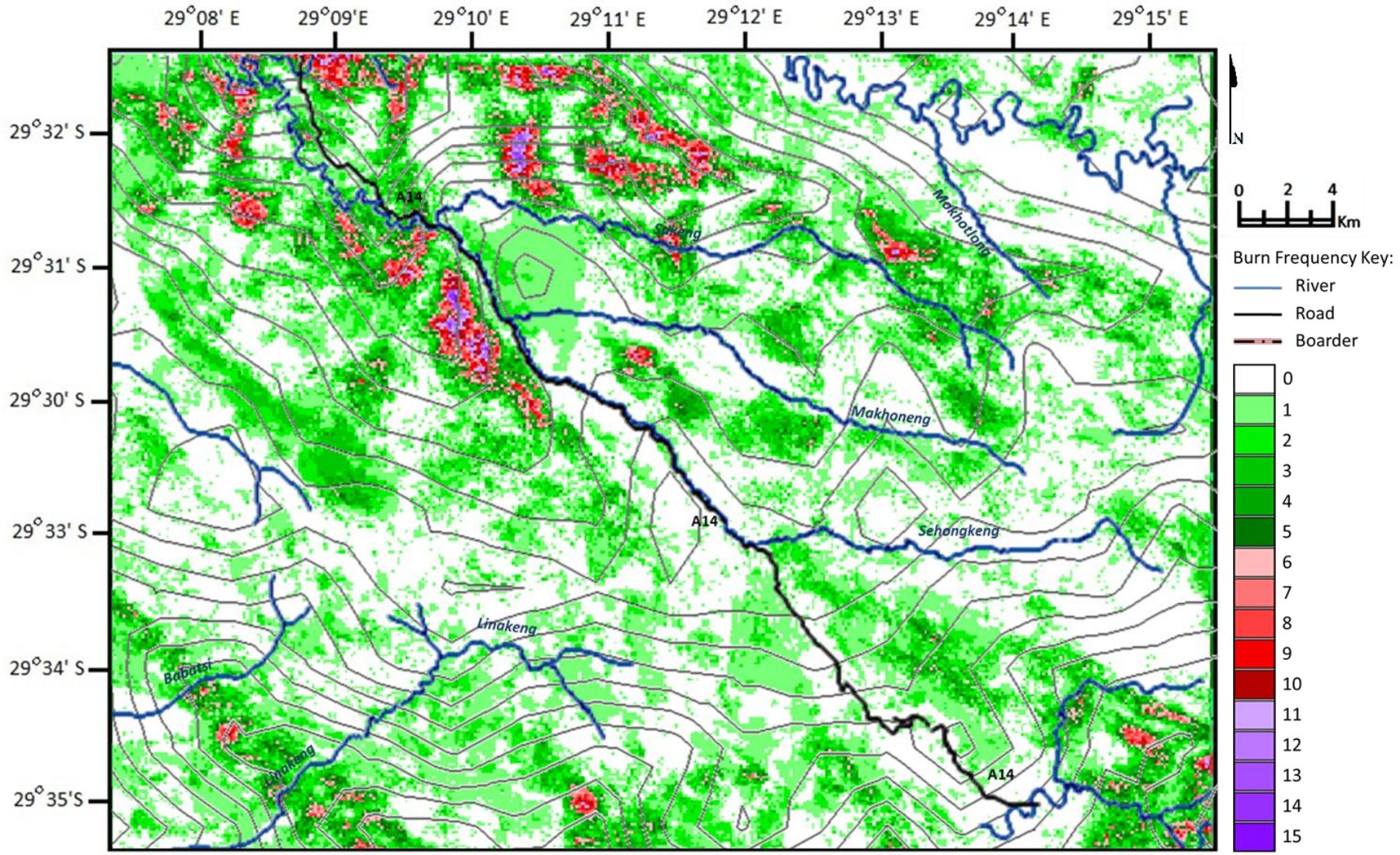


Figure 6.11: Burn frequency map of the Kotisephola Pass Region from 1993 to 2008.

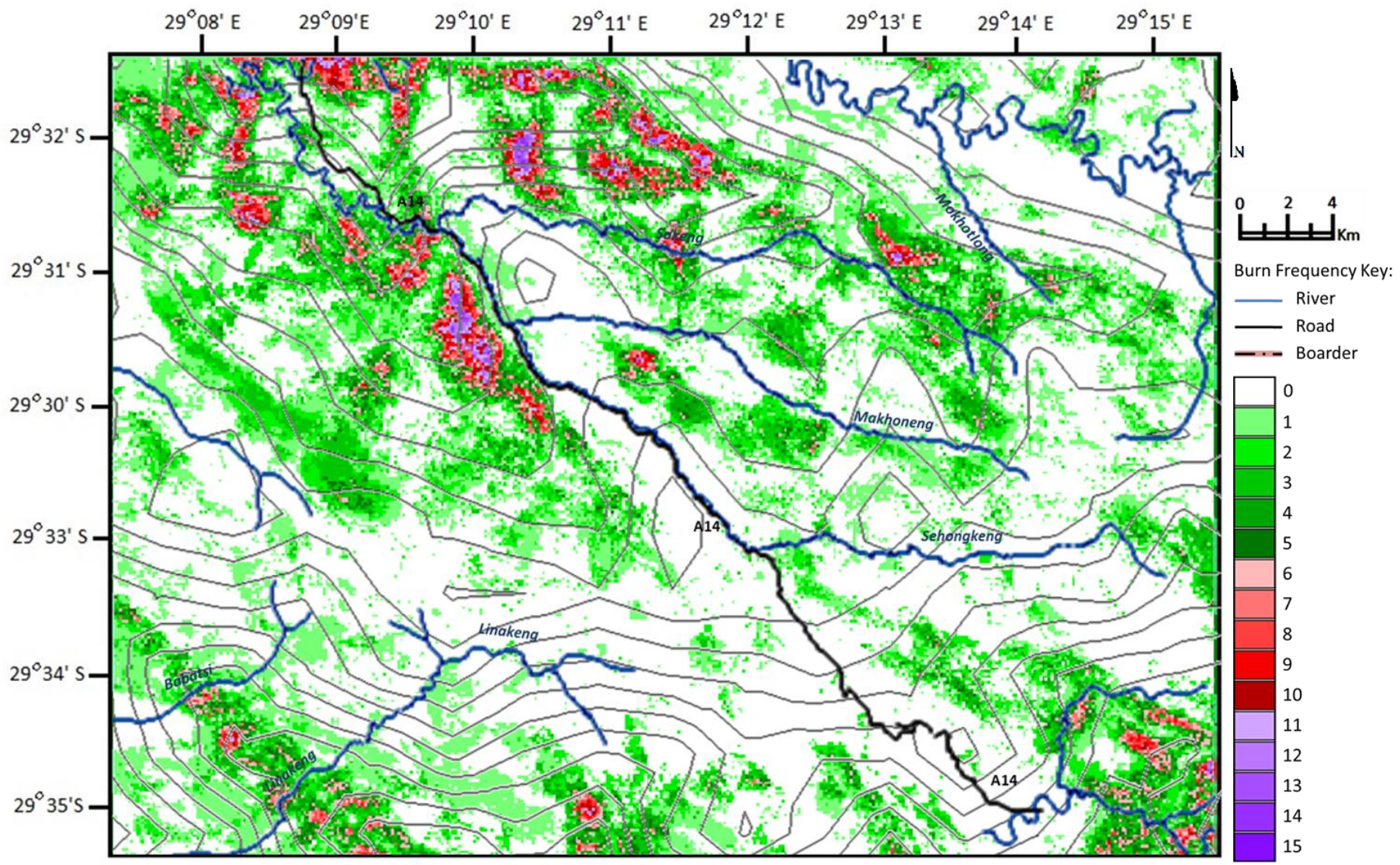


Figure 6.12: Burn frequency map of the Kotisephola Pass Region from 1998 to 2011.

The histogram of the burn frequency for the Sani top region supports the observation that the majority of the area has been burnt at least once between 1993 and 2011 (Figure 6.13). Between, 1993 and 2008, the median burn frequency value is 0.18 burns per annum (1.78 burns per decade) while the standard deviation is 1.29. Areas that have not been burnt between 1998 and 2008 constitute 39.8%, while 27.8% has only been burnt once. When the time scale is increased to between 1993 and 2011, the median burn frequency is 0.19 burns per annum (1.91 burns per decade), and the standard deviation (1.69) increases. Areas that have not been burnt between 1993 and 2011 constitute 28.7%, whilst areas burnt once during the same time period constitute 28.1%. The histogram indicates that widespread burning (>10 000m^2) at frequencies above 0.2 burns per annum do not occur. The Kotisephola Pass region has a median value of 0.06 burns per annum and a standard deviation of 1.51 between 1998 and 2008 (Figure 6.14). Most of the area (80.8%) was not burnt between 1998 and 2008, with 6.6% having been burnt once and a further 13.6% burnt two or more times. A median value of 0.12 burns per annum indicates that the majority of the area was burnt twice or less between 1993 and 2011 (Figure 6.14). The standard deviation of 1.19 confirms this finding. The greatest proportion of the area has not been burnt between 1993 and 2011 (54.2%), while 20.9% has been burnt only once. Widespread burning (>8 000m^2) may not occur more than twice within every second year.

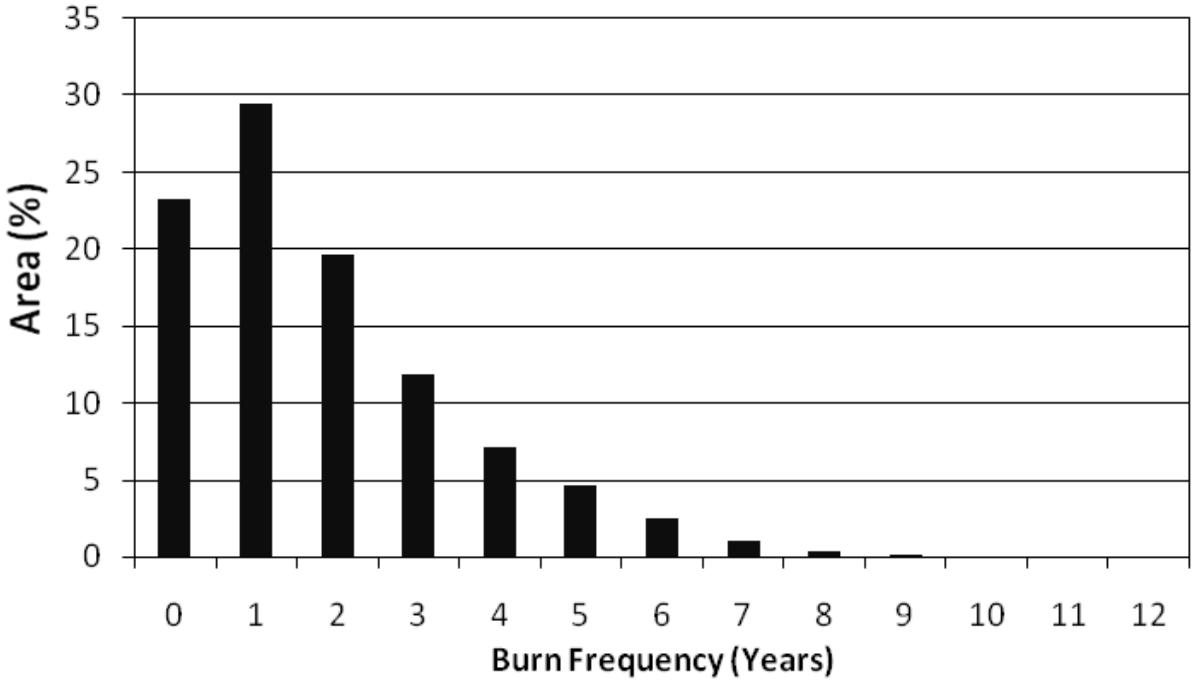


Figure 6.13: Percentage area for each burn frequency between 1993 and 2011 for the Sani region (omitting 2003).

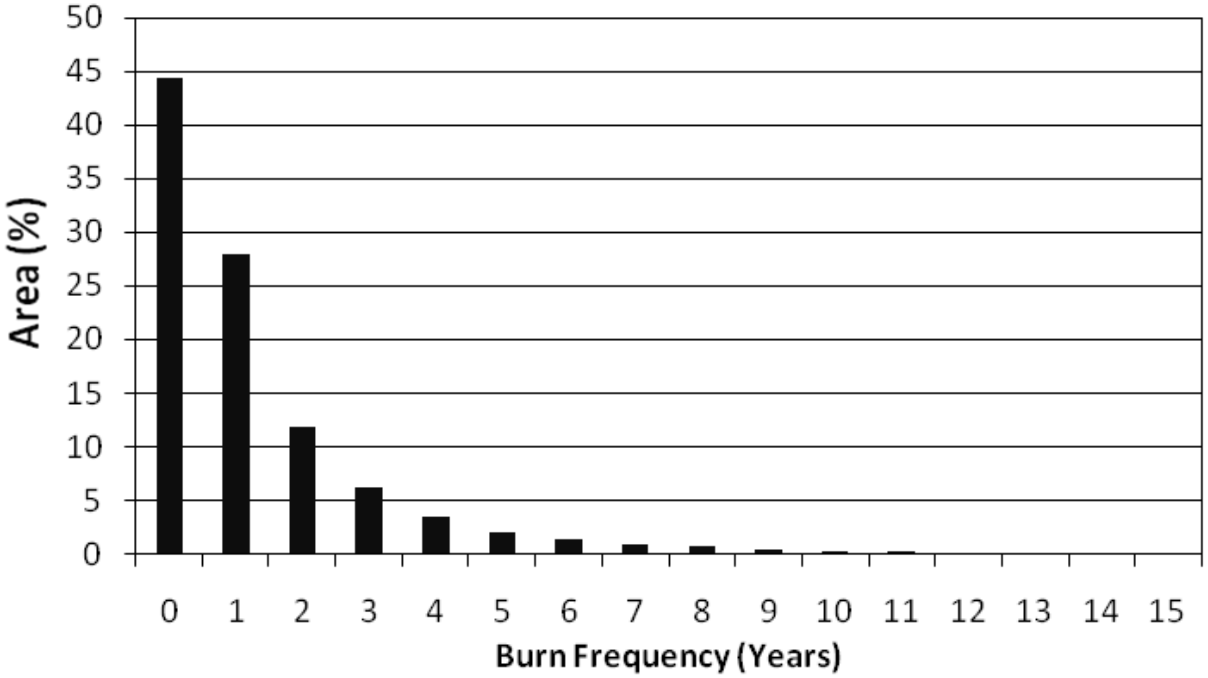


Figure 6.14: Percentage cover of each burn frequency between 1993 and 2011 for the Kotisephola Pass Region (omitting 2003).

The greatest area which was burnt during the 19 year study period is the area burnt only once (6517.4km^2 in the Sani region; 27.9km^2 in the K.P. region) (Table 6.4). Only 0.002km^2 was measured to have been burnt at the highest frequency of 0.8 burns per annum in the Sani region, while 0.033km^2 were measured to have been burnt at a frequency of 0.93 burns per annum in the Kotisephola Pass region during the same period. The highest burn frequency of 0.83 burns per annum is experienced in the K.P. region between 1993 and 2011.

The percentage burnt land increases in each five year period in both total burnt percentage over the five years and the average burnt percentage for both the Sani and Kotisephola Pass regions (Table 6.5). In the Sani region, the average percentage increases by 2.3% between the first and second 5 year period and by a further 3.5% between the second and third 5 year period. Yet, the average burnt area decreases by 4.6% between 2009 and 2011. A similar trend is observed in the Kotisephola Pass region, where the percentage burnt area increases by 1.7% between the first and second 5 year period and by 3.2% between the second and third 5 year period. However, it decreases by 2.3% between 2009 and 2011. Thus, for both the Sani and Kotisephola Pass regions, the burn frequency is increasing and the area remaining unburnt each year is also generally increasing (i.e. the number of burns is becoming more frequent but smaller in extent; Figure 6.15 and Tables 6.6, 6.7, 6.8 and 6.9).

Table 6.4: Area burnt (km^2) for each burn frequency class.

	Area burnt (km^2)			
	Sani Region		K. P. Region	
Class	1993-2008	1993-2011	1993-2008	1993-2011
0	58.072	5163.136	44.086	53.903
1	56.970	6517.442	27.856	20.737
2	36.221	4327.536	11.771	9.575
3	22.576	2616.388	5.599	6.159
4	12.576	1559.270	3.105	3.426
5	8.877	1019.329	2.170	2.102
6	3.906	548.219	1.409	1.427
7	1.558	216.75	0.938	0.949
8	0.965	68.111	0.651	0.696
9	0.486	22.764	0.447	0.480
10	0.214	7.738	0.252	0.333
11	0.035	1.975	0.175	0.203
12	0.002	0.360	0.091	0.126
13		0.090	0.050	0.086
14			0.013	0.033
15				0.002

Table 6.5: Average percentage burnt area in 5 year periods (excluding 2003).

	Sani Region	Kotisephola Pass Region
Years	Total % area burnt	Total % area burnt
1993-1997	37.5	25.0
1998-2002	49.2	33.7
2004-2008	66.8	49.8
2009-2011	26.2	26.2

Table 6.6: The number of burn scars for the Sani study region for each year and percentage within each size category for 1993 to 2002.

Year	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Number of burn scars	170	2250	3130	143	2459	1797	1674	2554	1226	2315
Average burn scar size (km2)	0.004	0.011	0.012	0.003	0.005	0.027	0.008	0.006	0.012	0.016
Maximum burn scar size (km2)	0.068	7.359	11.828	0.083	1.926	16.787	2.934	5.400	5.306	5.997
Minimum burn scar size (km2)	0.006	0.006	0.006	0.005	0.006	0.005	0.006	0.006	0.006	0.006
SD	0.010	0.165	0.223	0.009	0.043	0.515	0.084	0.113	0.167	0.170
Percentage burn scar areas (km2)										
>10	0	0	0.03(1)	0	0	0.11 (2)	0	0	0	0
2-10	0	0.04 (1)	0.03(1)	0	0	0.11 (2)	0.06 (1)	0.04 (1)	0.16 (2)	0.13 (3)
1-2	0	0.08 (2)	0.10(3)	0	0.04 (1)	0.19 (1)	0.06 (1)	0.08 (1)	0.08	0.17 (3)
0.9-1	0	0	0	0	0	0	0	0	0.08 (1)	0.04 (1)
0.8-0.9	0	0	0	0	0	0	0.06 (1)	0	0	0.04 (1)
0.7-0.8	0	0	0.03 (1)	0	0	0	0.06 (1)	0	0.08 (1)	0.04 (1)
0.6-0.7	0	0.04 (1)	0.03 (1)	0	0	0	0	0	0	0
0.5-0.6	0	0.18 (4)	0.10 (3)	0	0	0	0	0	0	0.17 (4)
0.4-0.5	0	0.09 (2)	0.06 (2)	0	0.04 (1)	0.06 (1)	0.06 (1)	0	0	0.13 (3)
0.3-0.4	0	0.13 (3)	0.03 (1)	0	0.08 (2)	0.11 (2)	0.06 (1)	0.08 (2)	0	0.09 (2)
0.2-0.3	0	0.18 (4)	0.32 (10)	0	0.12 (3)	0.39 (7)	0.12 (2)	0.04 (1)	0.16 (2)	0.26 (6)
0.1-0.2	0	0.62 (14)	0.48 (15)	0	0.41 (10)	0.67 (12)	0.48 (8)	0.35 (9)	0.33 (4)	0.73 (17)
0.05-0.1	1.76 (3)	0.53 (12)	1.05 (33)	1.40 (2)	0.49 (12)	1.00 (18)	0.78 (13)	0.63 (16)	0.65 (8)	1.25 (29)
0.01-0.05	7.65 (13)	5.42 (122)	4.63 (145)	6.29 (9)	4.07 (100)	5.29 (95)	4.78 (80)	4.97 (127)	6.20 (76)	9.46 (219)
0.005-0.01	4.71 (8)	4.76 (107)	4.82 (151)	6.99 (10)	4.11 (101)	5.23 (94)	5.68 (95)	4.74 (121)	3.75 (46)	8.38 (194)
0.001-0.005	27.65 (47)	31.51 (709)	29.42 (921)	24.48 (35)	30.38 (747)	30.05 (540)	30.35 (508)	28.97 (740)	28.87 (354)	32.87 (761)
0.0005-0.001	58.24 (99)	56.40 (1269)	58.85 (1842)	60.84 (87)	60.27 (1482)	56.93 (1023)	57.47 (962)	60.14 (1536)	59.62 (731)	46.26 (1071)
0.0001-0.0005	0	0	0	0	0	0	0	0	0	0

*The numbers in brackets for each size category give the actual number of burns

Table 6.7: The number of burn scars for the Sani study region for each year and percentage within each size category for 2003 to 2011.

Year	2003	2004	2005	2006	2007	2008	2009	2010	2011
Number of burn scars	No data	2911	1726	2140	2472	3268	2385	10418	13341
Average burn scar size (km2)	due to	0.006	0.018	0.003	0.019	0.009	0.003	0.001	0.001
Maximum burn scar size (km2)	satellite	5.196	17.862	0.692	12.847	2.809	1.060	2.008	1.147
Minimum burn scar size (km2)	error	0.005	0.005	0.005	0.005	0.005	0.001	0.001	0.001
SD		0.099	0.439	0.019	0.325	0.083	0.030	0.020	0.014
Percentage burn scar areas (km2)									
>10		0	0.06 (1)	0	0.04 (1)	0	0	0	0
2-10		0.03 (1)	0.06 (1)	0	0.08 (2)	0.03 (1)	0	0.01 (1)	0
1-2		0	0.06 (1)	0	0.12 (3)	0.15 (5)	0.04 (1)	0.02 (2)	0.01 (1)
0.9-1		0	0	0	0	0	0	0	0
0.8-0.9		0.03(1)	0	0	0	0.03 (1)	0.04 (1)	0	0
0.7-0.8		0	0	0	0.04 (1)	0.03 (1)	0	0.01 (1)	0
0.6-0.7		0	0	0.05 (1)	0.08 (2)	0	0	0	0.01 (1)
0.5-0.6		0.03 (1)	0	0	0.04 (1)	0.03 (1)	0	0.02 (2)	0
0.4-0.5		0	0	0	0.04 (1)	0.12 (4)	0	0	0.01 (1)
0.3-0.4		0.03 (1)	0.12 (2)	0	0.04 (1)	0.03 (1)	0.04 (1)	0.03 (3)	0.01 (2)
0.2-0.3		0.07 (2)	0.12 (2)	0.05 (1)	0.28 (7)	0.31 (10)	0.08 (2)	0.07 (7)	0.03 (4)
0.1-0.2		0.62 (18)	0.58 (10)	0.33 (7)	0.65 (16)	0.46 (15)	0.29 (7)	0.11 (11)	0.07 (9)
0.05-0.1		0.79 (23)	1.33 (23)	0.98 (21)	0.73 (18)	0.73 (24)	0.42 (10)	0.26 (27)	0.11 (15)
0.01-0.05		4.23 (123)	5.85 (101)	3.50 (75)	5.91 (146)	5.05 (165)	2.56 (61)	2.02 (210)	0.94 (125)
0.005-0.01		3.54 (103)	5.27 (91)	4.63 (99)	4.89 (121)	4.96 (162)	1.80 (43)	2.60 (271)	1.18 (158)
0.001-0.005		29.65 (863)	30.82 (532)	30.51 (653)	29.41 (727)	31.46 (1028)	7.80 (186)	7.45 (776)	6.60 (880)
0.0005-0.001		60.98 (1775)	55.74 (962)	59.95 (1283)	57.61 (1424)	56.61 (1850)	5.70 (136)	6.60 (688)	6.64 (886)
0.0001-0.0005		0	0	0	0	0	81.22 (1937)	80.81 (8419)	84.39 (11259)

*The numbers in brackets for each size category give the actual number of burns

Table 6.8: The number of burn scars for the Kotisephola Pass study region for each year and percentage within each size category for 1993 to 2002.

Year	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Number of burn scars	225	1075	1659	233	1197	1238	976	1165	687	166
Average burn scar size (km2)	0.003	0.004	0.007	0.003	0.005	0.007	0.007	0.007	0.010	0.010
Maximum burn scar size (km2)	0.076	0.359	1.107	0.101	0.407	0.905	1.418	2.068	0.574	0.515
Minimum burn scar size (km2)	0.006	0.006	0.006	0.006	0.006	0.005	0.006	0.006	0.006	0.006
SD	0.009	0.017	0.042	0.009	0.022	0.037	0.049	0.073	0.046	0.049
Percentage burn scar areas (km2)										
>10	0	0	0	0	0	0	0	0	0	0
2-10	0	0	0	0	0	0	0	0.09 (1)	0	0
1-2	0	0	0	0	0	0.08 (1)	0.1 (1)	0	0	0
0.9-1	0	0	0.06 (1)	0	0	0	0	0.09 (1)	0	0
0.8-0.9	0	0	0	0	0	0.06 (1)	0	0.09 (1)	0	0
0.7-0.8	0	0	0	0	0	0	0	0	0	0
0.6-0.7	0	0	0	0	0	0	0	0	0	0
0.5-0.6	0	0	0.12 (2)	0	0	0	0	0	0.15 (1)	0.60 (1)
0.4-0.5	0	0	0	0	0.08 (1)	0.06 (1)	0	0.09 (1)	0.29 (2)	0
0.3-0.4	0	0.09 (1)	0.18 (3)	0	0.17 (2)	0.11 (1)	0	0	0.44 (3)	0
0.2-0.3	0	0.09 (1)	0.30 (5)	0	0.08 (1)	0.39 (5)	0	0.09 (1)	0.44 (3)	1.20 (2)
0.1-0.2	0	0.19 (2)	0.84 (14)	0.43 (1)	0.33 (4)	0.67 (8)	1.02 (10)	0.26 (3)	1.02 (7)	0
0.05-0.1	0.89 (2)	1.12 (12)	1.33 (22)	1.29 (3)	0.84 (10)	1.00 (18)	0.82 (8)	0.69 (8)	1.75 (12)	0.60 (1)
0.01-0.05	6.67 (15)	5.67 (61)	5.61 (93)	4.72 (11)	5.18 (62)	5.29 (91)	6.56 (64)	4.89 (57)	5.24 (36)	7.23 (12)
0.005-0.01	3.56 (8)	7.44 (80)	4.16 (69)	3.86 (9)	6.27 (75)	5.23 (78)	7.07 (69)	5.84 (68)	6.11 (42)	7.83 (13)
0.001-0.005	26.67 (60)	29.21 (314)	28.45 (472)	24.89 (58)	29.82 (357)	30.05 (399)	33.61 (328)	30.82 (359)	29.69 (204)	29.52 (49)
0.0005-0.001	62.22 (140)	56.19 (604)	58.95 (978)	64.81 (151)	57.23 (685)	56.93 (636)	50.82 (496)	57.08 (665)	54.88 (377)	53.01 (88)
0.0001-0.0005	0	0	0	0	0	0	0	0	0	0

*The numbers in brackets for each size category give the actual number of burns

Table 6.9: The number of burn scars for the Kotisephola Pass study region for each year and percentage within each size category for 2003 to 2011.

Year	2003	2004	2005	2006	2007	2008	2009	2010	2011
Number of burn scars	No data	1303	943	939	1331	1952	1354	6472	7248
Average burn scar size (km2)	due to	0.006	0.006	0.005	0.009	0.008	0.003	0.001	0.001
Maximum burn scar size (km2)	satellite	0.489	0.368	0.365	0.925	1.921	1.623	2.008	0.280
Minimum burn scar size (km2)	error	0.005	0.005	0.005	0.005	0.005	0.001	0.001	0.001
SD		0.028	0.026	0.019	0.058	0.067	0.045	0.024	0.005
Percentage burn scar areas (km2)									
>10		0	0	0	0	0	0	0	0
2-10		0	0	0	0	0	0	0.02 (1)	0
1-2		0	0	0	0	0.15 (3)	0.07 (1)	0.02 (1)	0
0.9-1		0	0	0	0.15 (2)	0	0	0	0
0.8-0.9		0	0	0	0	0	0	0.03 (3)	0
0.7-0.8		0	0	0	0.15 (2)	0	0	0	0
0.6-0.7		0	0	0	0	0	0	0	0
0.5-0.6		0	0	0	0	0.10 (2)	0	0.02 (1)	0
0.4-0.5		0.15 (2)	0	0	0.30 (4)	0.05 (1)	0.07 (1)	0.02 (1)	0
0.3-0.4		0	0.21 (2)	0.11 (1)	0.38 (5)	0.26 (5)	0	0.03 (2)	0
0.2-0.3		0.54 (7)	0.32 (3)	0.11 (1)	0.23 (3)	0.26 (5)	0	0.02 (1)	0.03 (2)
0.1-0.2		0.77 (10)	0.85 (8)	0.64 (6)	0.60 (8)	0.36 (7)	0.15 (2)	0.02 (1)	0.01 (1)
0.05-0.1		1.15 (15)	1.48 (14)	1.17 (11)	0.83 (11)	1.13 (22)	0.44 (6)	0.15 (10)	0.06 (4)
0.01-0.05		5.37 (70)	5.41 (51)	4.79 (45)	4.81 (64)	5.74 (112)	1.62 (22)	1.10 (71)	0.94 (68)
0.005-0.01		5.22 (68)	7.53 (71)	5.54 (52)	4.81 (64)	4.82 (94)	1.11 (15)	1.36 (88)	7.88 (571)
0.001-0.005		27.94 (364)	29.06 (274)	33.76 (317)	30.05 (400)	30.28 (591)	6.72 (91)	6.37 (412)	4.29 (311)
0.0005-0.001		58.86 (767)	55.14 (520)	53.89 (506)	57.70 (768)	56.86 (1110)	5.39 (73)	6.55 (424)	1.96 (142)
0.0001-0.0005		0	0	0	0	0	84.42 (1143)	84.32 (5457)	84.84 (6149)

*The numbers in brackets for each size category give the actual number of burns

The increasing number of burn scars is more evident in the Sani region than in the K.P. region (Figure 6.15). The number of burn scars observed in the Sani region for each year studied differs (Table 6.6 and 6.7), with 2008 recording the greatest number of burns (3268 burn scars) using the lower resolution Landsat images (15-30m). The number of burn scars increases when Spot 5 images are used as the greater resolution (10m) allows for smaller burn scars to be captured, hence 2010 and 2011 record 10 418 and 13341 burn scars respectively. However, 2008, 2010 and 2011 did not experience the largest average burn scar size (average burn scar size was 0.009km^2 or less during these three years), rather the largest average burn scar size was measured in 1998 (0.027km^2), while the largest individual burn was measured in 2005 (17.862km^2). Conversely, the least number of burn scars were observed in 1996 (143 burn scars) followed by 1993 (170 burn scars), while the smallest average burn scar size was observed for 1996 (average burn scar size was 0.003km^2) followed closely by 2006 (average burn scar size was 0.003km^2). In 2004, 94% of the total burn scars had areas of 0.01km^2 or less and 1996 recorded 99%. In 2009, 2010 and 2011, the Spot 5 imagery recorded 81%, 81% and 84% of the burn scars with areas less than 500m^2. The resolution of the Landsat images did not allow for burn scars smaller than 500m^2 to be detected, and the confidence in the results decreases below 900m^2 due to pixel size. Through personal observations as well as confirmation through the use of Spot 5 images, it is believed that many more burns would have been recorded if this was possible. Less than 2% of the burn scars recorded throughout the study were greater than 0.1km^2, and less than 0.2% of burn scars exceeded 1km^2.

Using the spot 5 imagery for the Kotisephola Pass region, the greatest number of burn scars observed was 7 248 in 2011 and 6 472 in 2010. Following this, the Landsat imagery found 1952 burn scars in 2008, followed by 1 659 burn scars in 1995 (Table 6.8 and 6.9). The lowest number of burn scars was measured in 2002, with only 166 burn scars. The largest burn scar was measured in 2000 (2.068km^2).The average burn scar size was greatest in 2001 (0.010km^2) and lowest in 2011 (0.001km^2). Similar to the Sani study region, more than 50% of the observed burns were between 500 and 1000m^2 in extent between 1993 and 2008, while between 26% and 34% of the burns were between 1000m^2 and 5000m^2 during the same period. The Spot 5 imagery recorded that 84.4% of the burn scars in 2009 were less than 5m^2, 84.3% in 2010 and 84.8% in 2011.

A Kruskal-Wallis test of the nearest neighbour data indicates that there is no significant differenece between the medians each year for both the Sani region ($p = 1.000$) and the Kotisephola region ($p = 0.312$). This is not surprising given the high variation within years.

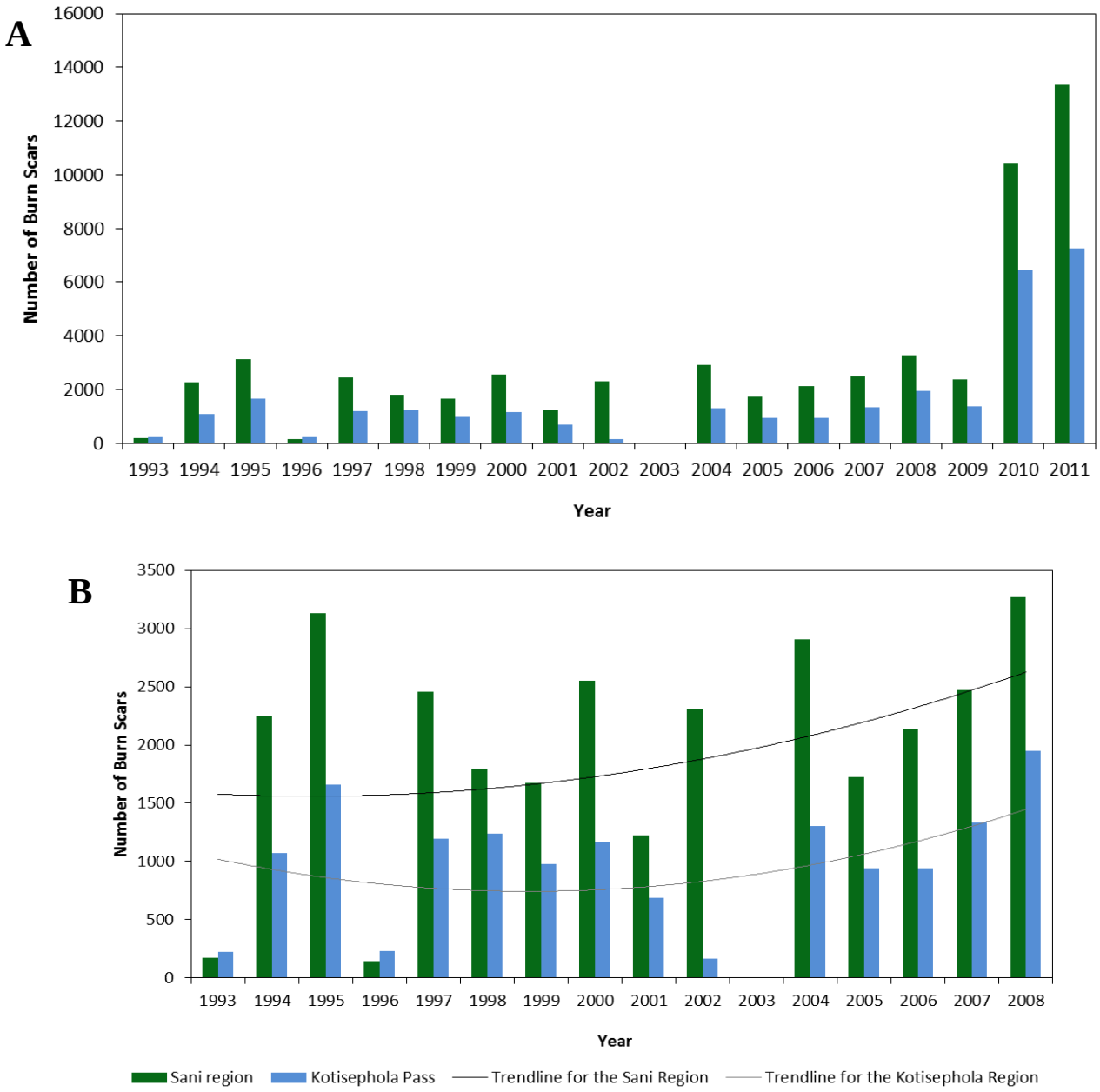


Figure 6.15: (A) Number of burn scars observed each year between 1993 and 2011 (2009 to 2011 using Spot 5 resolution: 10m) and (B) at a smaller scale between 1993 and 2008 (Landsat resolution: 15-30m).

The greatest average distance between burn scars in the Sani region was in 1996 (3 800.1 m), a year of plenty of snow, while the shortest average distance was in 2011 (24.6m; Table 6.10). In the K.P. region, the greatest average distance was recorded in 2002 (3 246.8m), while the shortest average distance was measured in 2010 (29.1m; Table 6.11). These results were expected as they reflect the burn scar density observed in the polygon images for each year. The nearest neighbour ratio is below one for all the years of study, suggesting that clustering of the burn scars is occurring, and is potentially due to the positioning of grazing posts, grazing routes, roads, rivers and villages. The strongly negative z-values suggest that there is high deviation within the data and that the burn scars are clustered in a manner which suggests there is a cause for their distribution. The causes may be the same as those suggested

for clustering but may also include topography. The differences between data values before 2008 and those after 2009 are likely owing to the higher resolution Spot 5 images (10m) than the Landsat images (15-30m). The higher resolution resulted in an increased number of individual burn scars which altered the nearest neighbour ratios and the z-values.

In the Sani study region, the majority (>60%) of the burn scars occurred in flat areas between 1993 and 2008, with 85.1% of the burn scars in 2001 occurring on flat land (Table 6.12; Figure 6.16). This is not surprising as the flat areas account for 61% of the region studied. The lowest percentage of burn scars on flat land is measured in 2009 (41.3%). In many years the number of burn scars recorded was greatest on east-facing slopes (Table 6.12). The east-facing slopes only cover 10% of the study region, yet contain the greatest number of burn scars during most years (north-facing slopes: 8%; west-facing slopes 10%, south-facing slopes 12%). Additionally, the south-facing slopes had a greater percentage of burn scars in 2009, 2010 and 2011 (32.6%, 18.5% and 16.0% respectively) than the other aspects. The north- and west-facing slopes generally accounted for the lowest percentage of the burn scars (5.3% and 5.5% respectively).

Table 6.10: Nearest neighbour distance and analysis of annual burn scar data for the Sani region.

Year	Average Distance (m)	Minimum Distance (m)	Maximum Distance (m)	N.N. ratio	Z-score
1993	161.5	12.4	3477.0	0.302	-18.395
1994	111.4	7.7	1268.3	0.702	-30.468
1995	92.1	2.2	755.8	0.820	-22.782
1996	139.2	18.7	3800.1	0.409	-14.795
1997	102.3	3.2	1019.6	0.738	-26.784
1998	104.2	4.7	859.9	0.744	-25.971
1999	118.1	10.3	1829.7	0.695	-25.477
2000	104.0	7.8	1696.2	0.755	-25.226
2001	124.6	9.7	994.6	0.676	-17.490
2002	167.4	27.8	3246.8	0.456	-14.279
2003	No Data due to satellite error				
2004	96.2	4.2	874.5	0.762	-26.741
2005	115.9	3.3	844.2	0.748	-23.075
2006	103.5	6.7	910.1	0.673	-30.160
2007	94.7	5.7	868.1	0.783	-25.454
2008	89.6	4.7	1131.5	0.795	-25.813
2009	57.3	0.1	229.1	7.4×10^{-5}	-107.431
2010	25.5	5.6	90.8	0.002	-334.835
2011	24.6	7.9	82.4	0.002	-255.561

In the Kotisphola Pass region, the flat areas did not always contain the greatest percentage of the burn scars and only represented 1.4% of the area (Table 6.13; Figure 6.17). The flat areas account for a maximum of 54% of the total number of burn scars in 1994. The flat areas did not account for the greatest proportion of burn scars in 1993 (24.4%), 2001 (21.7%) and 2002 (18.1%), as the greatest percentage was observed on the south-facing slopes. The north-facing slopes generally accounted for the least number of burns with no burns recorded in 1993. The exception occurred in 2000 when both the north-facing and south-facing slopes accounted for the same percentage of the burn scars (29.0%), as well as in 2009 and 2011 (25.1% and 25.9% respectively) when the north-facing aspects accounted for a greater number of burn scars than other aspects.

A weighted Friedman's test indicated that highly significant differences existed between the different aspects in each of the study regions ($p<0.0001$; Table 6.14).

Table 6.11: Nearest neighbour distance and analysis for annual burn scar data for the K.P. region.

Year	Average Distance (m)	Minimum Distance (m)	Maximum Distance (m)	N.N. ratio	Z-score
1993	130.5	14.2	2034.2	0.401	-18.162
1994	99.1	6.5	1340.6	0.667	-22.299
1995	92.3	4.5	1055.5	0.811	-16.535
1996	147.2	29.4	3105.4	0.439	-14.956
1997	95.9	6.6	983.9	0.688	-22.276
1998	105.8	6.3	702.2	0.782	-16.026
1999	113.8	15.9	740.3	0.733	-17.115
2000	107.7	9.2	783.8	0.770	-16.373
2001	124.6	9.7	994.6	0.676	-17.490
2002	167.4	27.8	3246.8	0.456	-14.279
2003	No Data due to satellite error				
2004	107.3	24.3	951.4	951.424	-14.542
2005	115.2	8.7	749.9	0.737	-16.696
2006	106.2	5.5	831.8	0.666	-20.823
2007	100.2	4.0	756.9	0.796	-16.124
2008	93.3	6.4	671.4	0.891	-10.390
2009	46.7	0.2	219.1	5.60×10^{-5}	-79.520
2010	29.1	0.1	72.1	1.13×10^{-4}	-262.758
2011	55.7	0.2	115.6	1.03×10^{-4}	-177.217

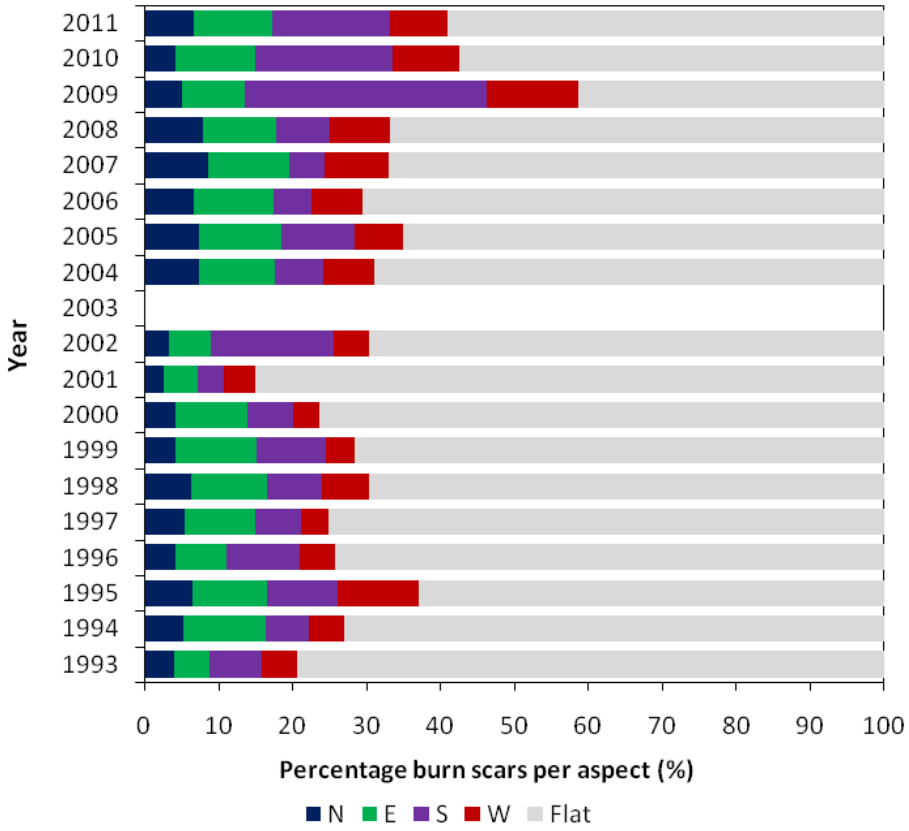


Figure 6.16: Percentage of burnt scars on each slope aspect as well as on flat areas in the Sani region.

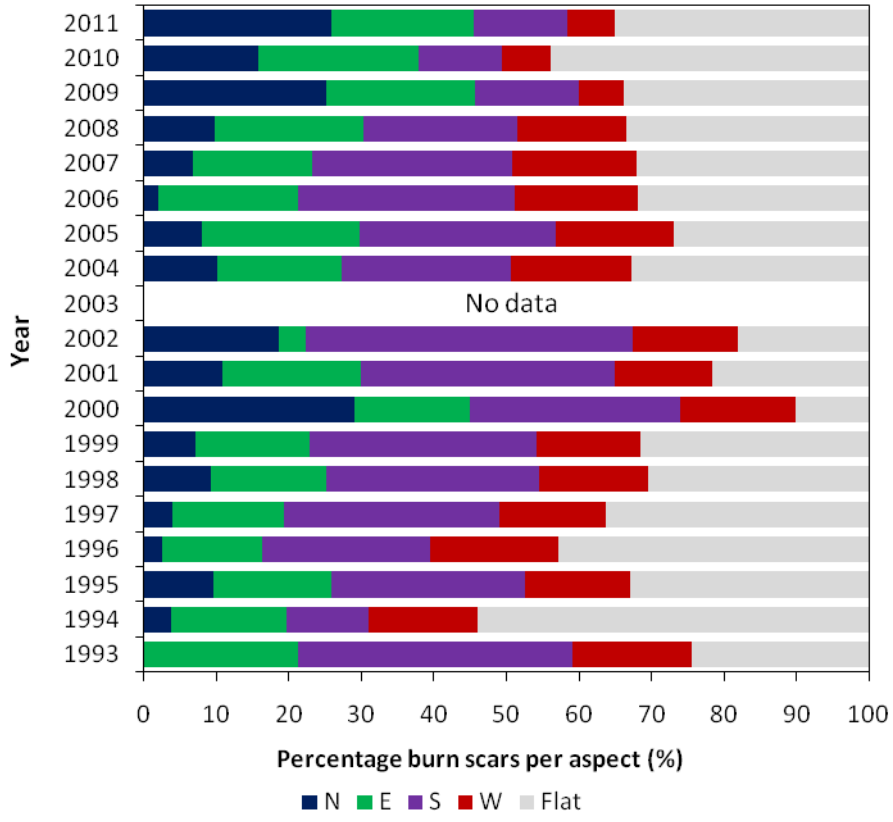


Figure 6.17: Percentage of burn scars on each slope aspect as well as on flat areas in the Kotisephola region.

Table 6.12: The number of burn scars observed for each aspect within the Sani Region.

Year	Total number of burn scars	North-facing			East-facing			South-facing			West-facing			Flat land		
		Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size(km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD
1993	170	7	0.001	0.001	8	0.001	0.000	12	0.001	0.000	8	0.001	0.001	135	0.003	0.008
1994	2250	121	0.004	0.014	247	0.010	0.086	131	0.020	0.117	111	0.006	0.030	1640	0.010	0.122
1995	3130	206	0.006	0.022	315	0.009	0.037	298	0.011	0.045	341	0.008	0.043	1970	0.012	0.214
1996	143	6	0.001	0.001	10	0.001	0.001	14	0.002	0.001	7	0.001	0.002	106	0.004	0.017
1997	2459	135	0.002	0.008	232	0.004	0.023	155	0.018	0.147	89	0.001	0.002	1848	0.004	0.018
1998	1797	115	0.017	0.057	182	0.026	0.134	134	0.044	0.124	114	0.037	0.109	1252	0.023	0.377
1999	1674	71	0.010	0.025	182	0.008	0.053	159	0.011	0.052	65	0.005	0.017	1197	0.006	0.051
2000	2554	107	0.004	0.012	250	0.010	0.090	159	0.011	0.070	87	0.004	0.019	1951	0.005	0.072
2001	1226	33	0.005	0.005	55	0.005	0.010	43	0.010	0.053	52	0.072	0.070	1043	0.009	0.256
2002	2315	78	0.005	0.012	132	0.020	0.037	384	0.008	0.026	108	0.022	0.018	1613	0.007	0.209
2003	No data	due to	satellite	error												
2004	2911	218	0.004	0.015	298	0.009	0.065	186	0.009	0.053	201	0.003	0.011	2008	0.005	0.055
2005	1726	128	0.003	0.007	191	0.022	0.148	173	0.057	0.341	111	0.014	0.054	1123	0.012	0.251
2006	2140	142	0.001	0.001	234	0.004	0.024	106	0.004	0.023	148	0.003	0.012	1510	0.003	0.013
2007	2472	213	0.002	0.009	271	0.014	0.071	117	0.028	0.146	216	0.005	0.024	1655	0.018	0.293
2008	3268	262	0.004	0.012	319	0.008	0.051	238	0.009	0.058	265	0.005	0.025	2184	0.008	0.057
2009	2385	120	0.003	0.035	206	0.001	0.001	777	0.000	0.002	298	0.001	0.004	984	0.000	0.000
2010	10418	445	0.002	0.038	1126	0.007	0.063	1927	0.003	0.036	936	0.006	0.102	5984	0.001	0.032
2011	13341	901	0.001	0.009	1402	0.003	0.036	2133	0.002	0.017	1025	0.003	0.037	7880	0.001	0.017

SD: Standard deviation

Table 6.13: The number of burn scars observed for each aspect within the Kotisephola Pass Region.

Year	Total number of burn scars	North-facing			East-facing			South-facing			West-facing			Flat land		
		Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD	Number of burn scars	Average burn scar size (km^2)	SD
1993	225	0	0	0	48	0.087	0.046	85	0.002	0.004	37	0.003	0.007	55	0.003	0.006
1994	1075	41	0.003	0.009	171	1.007	8.620	122	0.004	0.014	160	0.004	0.015	581	0.004	0.011
1995	1659	160	0.004	0.022	270	0.884	3.737	443	0.007	0.029	239	0.005	0.015	547	0.007	0.041
1996	233	6	0.001	0.001	32	0.126	0.052	54	0.003	0.004	41	0.002	0.002	100	0.003	0.018
1997	1197	48	0.002	0.005	184	0.389	2.332	354	0.006	0.022	176	0.003	0.010	435	0.003	0.011
1998	1238	114	0.006	0.030	198	2.588	13.358	362	0.007	0.026	187	0.005	0.013	377	0.006	0.024
1999	976	69	0.004	0.008	155	0.836	5.341	305	0.007	0.022	139	0.005	0.012	308	0.006	0.046
2000	1165	338	0.006	0.024	186	1.048	9.009	338	0.006	0.024	184	0.005	0.025	119	0.006	0.050
2001	687	75	0.007	0.028	131	0.461	1.038	240	0.011	0.047	92	0.005	0.013	149	0.006	0.022
2002	166	31	0.016	0.073	6	2.016	3.658	75	0.005	0.067	24	0.006	0.031	30	0.009	0.035
2003	No data	due to	satellite	error												
2004	1303	132	0.004	0.015	223	0.905	6.471	305	0.007	0.033	215	0.004	0.016	428	0.005	0.021
2005	943	76	0.003	0.007	205	2.247	14.826	254	0.007	0.024	154	0.004	0.013	254	0.005	0.017
2006	939	19	0.001	0.001	180	0.361	2.416	282	0.005	0.017	158	0.003	0.008	300	0.004	0.013
2007	1331	90	0.002	0.009	219	1.441	7.092	368	0.010	0.040	227	0.006	0.017	427	0.009	0.050
2008	1952	192	0.004	0.012	399	0.819	5.065	413	0.011	0.054	296	0.006	0.021	652	0.007	0.045
2009	1354	340	0.003	0.035	279	0.084	0.050	193	0.015	0.125	83	0.031	0.187	459	0.005	0.071
2010	6472	1027	0.002	0.038	1427	0.680	6.336	743	0.003	0.068	429	0.001	0.014	2846	0.002	0.044
2011	7248	1878	0.001	0.009	1416	0.297	3.567	937	0.001	0.009	478	0.001	0.002	2539	0.001	0.008

SD: Standard deviation

Table 6.14: Weighted Friedman test between different aspects and flat areas in both the Sani study region and the Kotisephola Pass study region.

Study Region	N	Df	Chi-square	P-value
Sani Region	56379	4	23.778	<0.0001†
Kotisephola Pass	30043	4	38.622	<0.0001†

*P<0.005; **P<0.001; ***P<0.0005; †P<0.0001.

6.2.4 Selection of study sites based on burn frequency

Aerial photographs of the study site were reclassified to indicate the presence of *Merxmuellera* tussock communities (Figure 6.18). The community positions were confirmed in the field and the reclassification was indicated to be a slight (9.2%) overestimation of community extent in the valley bottom and near the banks of the Sani river tributarys. Similarly, the species of tussock were identified in the field and confirmed through inflorescence samples returned to the lab and placed under a light microscope.

The study sites used in the baseline and monitoring studies were selected using the burn frequency map and the *Merxmuellera* distribution map to show varying altitude, aspect, slope and burn frequency (Figure 3.3, 3.4 and Table 6.15). The Sani East (S.E.) and West (S.S.) sites are at an altitude of approximately 2 904m a.s.l.; both of which are situated on gentle slopes and the sites are east-facing and south-facing respectively. Both sites have patchy fire frequencies ranging from 0.1 to 0.9 burning events per annum between 1997 and 2007, when they were last burnt. The Sani South site is in close proximity to a path to the north and approximately 380m from the road, yet only livestock were observed within or near the study site (Figure 6.19). The Sani East and Sani West sites are situated in an area with a history of frequent patchy burns, and a mosaic of between 0.11 to 0.26 burns per annum between 1993 and 2011, yet the sites chosen were selected to have experienced 0.26 burns per annum between 1993 and 2011.

The Valley Slope North (V.S.N.) and East (V.S.E.) sites are situated at an average altitude of 2 962m a.s.l. and are east-southeast-facing and north-facing respectively, and on relatively steep slopes (between 30^{o} and 45^{o}). Both have a burn frequency of0.4 burns per annum (between 1997 and 2007) and were last burnt in 2006 (Figure 6.20). The Saddle Stream (S.S.) site is positioned near a saddle pass and has an average altitude of 2 981m a.s.l, and is north-facing (Figure 6.21). This area has only been burnt once during the last ten years (between 1997 and 2007) and was

Table 6.15: Aspect, altitude, frequency and the time since last burnt for each study site.

Site	Aspect	Altitude	Burn Frequency (1993-2008)	Last Burnt
Sani East	E	2896	5	2007
Sani South	S	2896	5	2007
Valley Slope North	NW	2975	6	2007
Valley Slope East	NE	2975	6	2007
Saddle Stream	E	2984	1	1999
Hodgson's East	NE	2962	2	2005
Hodgson's West	SW	2962	2	2005
Valley Floor	N	2912	3	2000
Mshweshwe's Finger Upper	S	2940	2	2005
Mshweshwe's Finger Lower	S	2910	2	2005
Apostles Plateau	Flat	2892	1	2005
Kotisephola Pass	N	3230	3	2007

*Altitude and burn frequency calculated from the center of each site

last burnt in 1998. The slope is steep (>45^{o}) and is situated less than 100m from a grazing post, yet does not have any paths through or near the site. The average altitude of the Hodgson's East (H.E.) and West (H.W.) sites is 3014m a.s.l., are northeast-facing and southwest-facing respectively and are separated by a stream (Figure 6.22). The burn frequency map indicates a frequency of 0.2 burns per annum (1997 to 2008), and was last burnt in 2005. The Valley Floor (V.F.) site is situated on the banks of a stream at an altitude of 2 912m a.s.l., has a burn frequency of 0.3 burns per annum (1997 to 2008) and was last burnt in 2000 (Figure 6.23 top). The Valley Floor site is north-facing and is relatively sheltered from the wind (personal observation). The south-facing Mshweshwe's Finger (M.F.) site was divided into an upper site which has an average altitude of 2 940m a.s.l. and a lower site with an average altitude of 2910m a.s.l. (Figure 6.23 bottom). Both sites were last burnt in 2005 and have a burn frequency of 0.2 burns per annum (1997 to 2008). The Apostles Plateau (A.P.) site is flat with an average altitude of 2 892m a.s.l. (Figure 6.24 top) and was burnt once between 1993 and 2008, and last burnt in 2005. The Kotisephola Pass site was added as this site is positioned at a considerably higher altitude, averaging 3230m a.s.l. (Figure 6.24 bottom and figure 3.3). This site has a burn frequency of 0.2 burns per annum (1998 to 2008) and was last burnt in 2007. The study site is located in a valley head, 143m to the east of the Sani-Mokhotlong road which provided access to the study site.

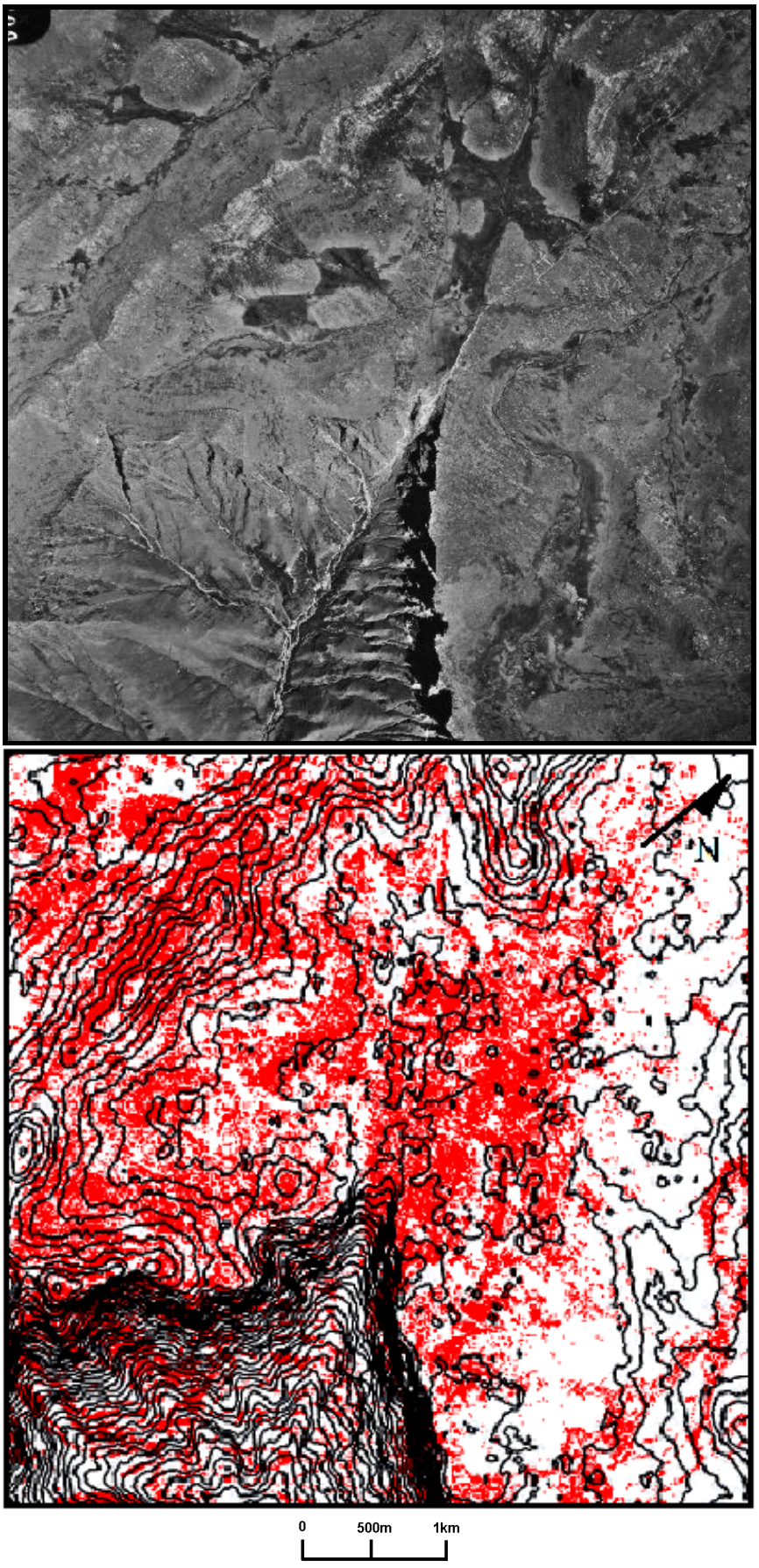


Figure 6.18: Aerial photograph of the Sani region (top) and reclassified image showing the presence of *Merxmuellera* communities (bottom). Red: presence of *Merxmuellera* communities.



Figure 6.19: Sani East site (top) and Sani South site (bottom).



Figure 6.20: The Valley Slope North site (top) and the Valley Slope East site (bottom).

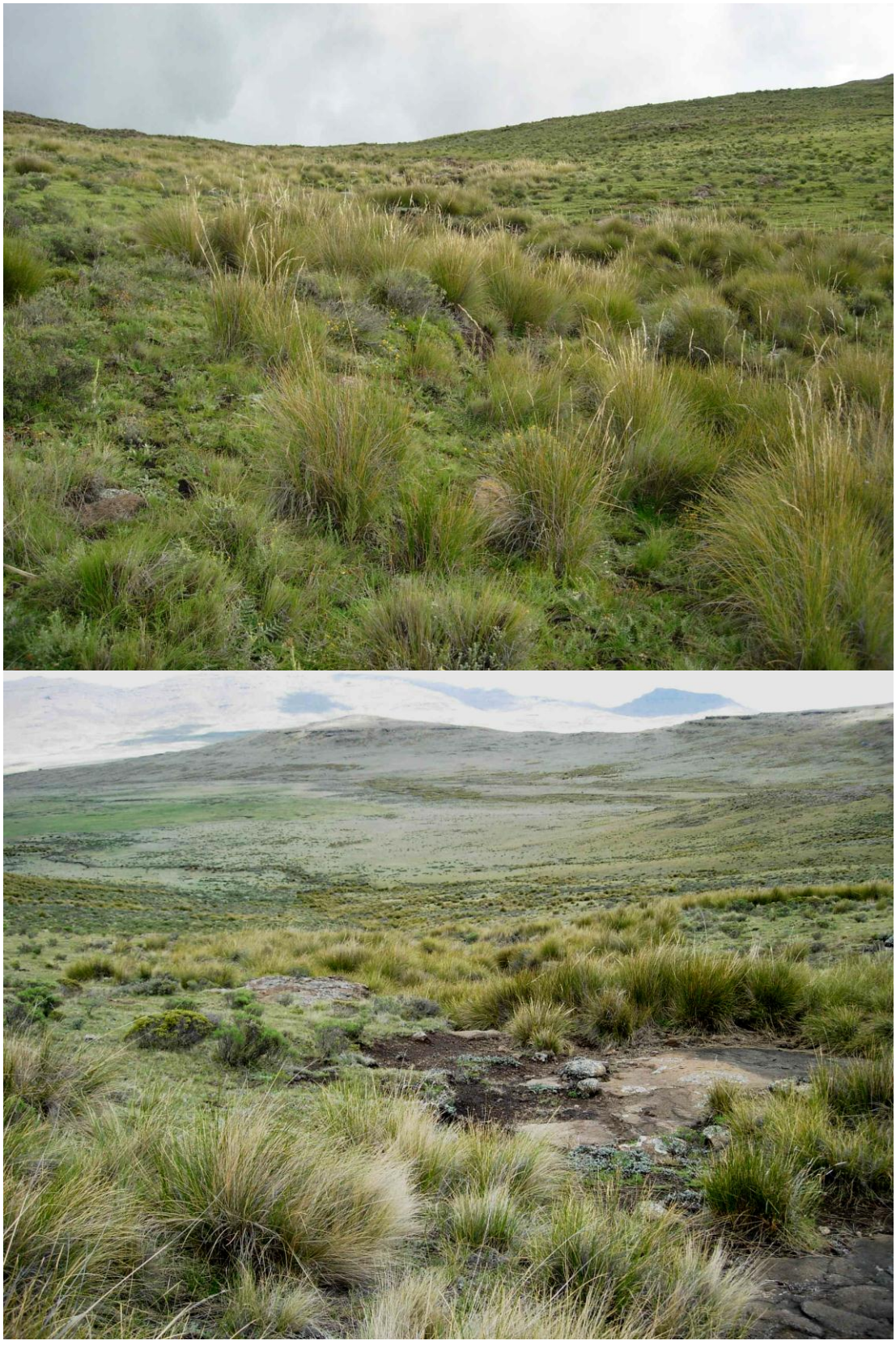


Figure 6.21: Saddle Stream site looking upwards (top) and downwards (bottom).



Figure 6.22: Hodgson's East (top) and Hodgson's West (bottom).



Figure 6.23: Valley Floor site (top) and Mshweshwe's Finger sites (upper and lower) from across the valley (bottom).



Figure 6.24: The Apostles Plateau (top) and Kotisephola Pass sites (bottom).

6.3 Discussion

Many features must be taken into account when analyzing burn scars at a landscape level, such as scale, standardization, pixel size and cost (Key and Benson, 2006). The scale of the landscape is important as this is directly linked to the detail of the information that is obtained. The use of Landsat and Spot 5 images has permitted the production of a relatively high resolution (i.e.15-30m) burn frequency map. In terms of standardization, neither the intensity nor severity of the burn scars were considered, and it was assumed that each burn event resulted in defoliation of the vegetation. The diagrammatic results of using the NDVI and NBR methods to identify burn scars within the study region support the literature. The NBR was better suited to accurately and clearly indicate the presence of burn scars, and this supports the findings of researchers such as Kushla and Ripple (1998), Bobbe *et al.* (2003), Epting *et al.* (2005), Hudak *et al*. (2006) and Escuin *et al.* (2008). Thus, the NBR was chosen for identification of burn scars for this study. The NBR produces suitable results through the use of the near-infrared and the mid-infrared bands, as dry grassland has a high reflectance which lowers considerably when an area is burnt (Pereira, 2003). It has been indicated that the hyperspectral NBR provides more accurate burn scar identification through 5m resolution as opposed to 15-30m resolution Landsat images (Robichaud *et al*., 2007), thus producing more accurate results for both the frequency map and the monitoring study. Despite the lower resolution, Landsat was specifically used in this study owing to its accessibility and affordability.

The size of the observed burn scars in the Sani region were small (on average less than 0.05km^2) when compared to the large scale burning observed along the escarpment slopes (generally greater than 2km^2) and those within South Africa. Fires in Lesotho may cater for small grazing herds, warmth and pest control (rodents/jackals) as opposed to large landscape scale fires to the east of the escarpment which are associated with firebreaks and rotational fire management programs. Despite the burn scars being small (<0.5km^2), many burn scars were observed as opposed to only a few large scale burn scars. The large number of small burn scars may be an indication of the burning of tussocks by herdsmen or the dispersed nature of the communities, and limited or low lying inter-tussock vegetation. However, the NBR has been demonstrated to potentially overestimate the number of burn scars as a result of underestimating low-severity burn scars (Srivastava *et al.*, 2012). The majority of the burn scars do not present a uniform shape, with many burn scars indicating rough edges. This is believed to be the result of inter-tussock spaces and the patchy location of woody vegetation. The majority of the small burn scars (less than 0.001km^2) are oval in shape due to the shape of the burnt communities.

South-facing slopes may preserve winter snow for several weeks while snow on the north-facing slopes generally melts after a few days (Guillarmod, 1971; Morris *et al.*, 1989; Mulder and Grab, 2002; Grab *et al.*, 2009). The snow melt provides the moisture necessary for the growth of many plants. The slow melting of snow on the south-facing slopes would increase the moisture content of the tussock grasses, perhaps making *M. drakensbergensis* slightly more palatable, or alternatively increasing the palatability of other less grazed species. This would decrease the need for burning on south-facing slopes and possibly explain the results observed. North-facing slopes are exposed to sizable diurnal temperature changes (Guillarmod, 1971; Granger and Schulze, 1977) and desiccate more rapidly than south-facing slopes. It also suggests less vegetation and more bare ground may be observed on the north-facing slopes when compared to the south-facing slopes.

Disturbance of alpine ecosystems depends largely on the topography of an area (Crawford, 2008). Fire is restrained by roads, deep gullies, streams, rivers and large areas of bare ground (Guillarmod, 1971; Whelan, 1995). In the current study, borders of many burn scars are observed to be roads, streams, bare ground and marshes. The grass cover is for the most part also short, contributing less height to the flames, thereby preventing flames effectively moving up steep slopes (Guillarmod, 1971). This may explain why the fire frequency decreases on the steeper slopes of the valley sides and increases in the valleys. The steeper sides are also less accessible to livestock, and thus are less frequently burnt to provide greener fodder. Similarly, fewer herdsmen are found on the steeper slopes, guarding their herds and flocks and thus fewer fires may be lit in these areas to provide warmth. Differences in species distribution may also have an influence on the results obtained for south- and north-facing slopes. *Themeda triandra* generally grows on the sunnier north-facing slopes, whilst *Festuca caprina* is generally located on the cooler, less sunny south-facing slopes (Guillarmod, 1971; Morris *et al.*, 1989; Pers. obs.). Aspect not only has an impact on tussock grasses but also the other vegetation occurring on the slopes, with species requiring cooler, moister conditions on the south-facing slopes. However, under intense grazing, this pattern is often disturbed by the invasion of other species.

A study in northern Australia found that areas over 500m from settlements were burnt more regularly (Elliott *et al.*, 2009). This was concluded to be the result of settlements depressing fire frequency for safety reasons, and may also be the result of a lack of ability to control large scale wildfires, and thus merely limits the spread of fire near settlements (Elliott *et al.*, 2009). In the current study, the Sani top village and the border post are situated in an area which has not been

burnt during the last 19 years, supporting this theory. In addition, a cattle post on the northeast-facing slopes of the study valley is situated in an area with a low (and patchy) burn frequency (mostly 1burn per 19 years but small areas of 4 burns per 19 years were observed). Perhaps the herdsmen burn the area in and around their pens to increase the amount of grazing available to the livestock, especially towards the end of the winter season. Herdsmen in charge of attending to livestock overnight may light small fires to keep both themselves and their livestock warm.

The use of satellite images has limitations such as the image dates available and resolution. Cloud cover may be a limiting factor (Pereira, 2003; Kerr and Ostrovsky, 2003; Minchella *et al.*, 2009; Mohler and Goodin, 2010), as was the case along the escarpment edge. Snow cover hampered the choice of some of the available images (e.g. 1996), and in some cases delayed the available dates to be closer to the onset of the wet season, making it more difficult for burn scars to be identified. Another obstacle occasionally encountered was that of little differentiation between burnt and unburnt areas; as also discussed in a study on burn scars from Riley County, Kansas, by Mohler and Goodin (2010). This limitation was resolved through field-based observations in combination with using aerial and satellite images to verify burn scars and bare ground. The accuracy of the burn frequency maps could have been increased through greater access to aerial images for the GIS study period. This could have assisted in the validation of the images and improved spring burn representation, thus increasing the overall accuracy of burn scar counts.

Burnt areas typically appear black in images taken soon after burning, yet as the time since last burnt increases, so the reflectivity changes. An increased proportion of bare ground and a higher reflectance was observed to be characteristic of older burns, as also reported by Dubinin *et al*. (2010). Thus, the lag time between burning and when the satellite image is taken can influence the burn scars detected using the NBR and dNBR methods (Veraverbeke *et al.*, 2010). The classification of burn scars was adjusted (following that of Justice *et al*., 1997), however, the results may suggest an under- estimation of the size of older burnt areas in each annual image. This is perhaps especially true where the burnt area was situated in already slightly sparse or rocky conditions. The near infrared band is highly sensitive to areas exceeding 13 days since burning in arid grassland, but the ability to identify burn scars was reduced after one month (Trigg and Flasse, 2000). More recently, high reflectance values were indicated to allow for burn scar detection to occur two to three years after burning (Dubinin *et al*., 2010), which may suggest that the same burn scars are detected in more than one year and thus increase the burn frequency.

However, the reclassing values and successive image checking allowed the older burns to be eliminated to a large extent, and thus reduce their likely inclusion in successive years. However, this method does reduce the inclusion of less likely spring (October) burns.

The mean burn frequency (0.17 burns per annum) for the Sani top region suggests that the region has been burnt three times or less between 1993 and 2008. This figure is not surprising given the possibility of runaway fires and the reluctance of villagers and herdsmen to burn large areas (personal observation). The mean burn frequency for the Kotisephola Pass region suggests that the majority of the area has been burnt twice or less between 1993 and 2008. The relatively high frequency burns occasionally observed (>0.2 burns per annum), tend to be patchy and decrease in area with increasing frequency. The high number of observed burns differs from the results obtained by Archibald *et al.* (2010) for Lesotho due to the finer resolution of the Landsat images used in the current study, compared to using courser MODIS images. It is expected that with even finer resolution at scales (<10m^2), the data obtained may indicate higher burn frequency than thus far recorded.

6.4 Summary

This chapter examined the burn history of the Sani and Kotisephola Pass regions through satellite imagery between 1993 and 2011. The following points are the main findings from this study:

1. The NBR proved a better indicator for burn scars than the NDVI, especially as the images were reclassified into boolean images.
2. Burnt areas represent less than 30% of the area in the Sani region and 20% of the Kotisephola Pass region each year (omitting 2003).
3. The average size of the observed burn scars within Lesotho is small (<0.05km^2) when compared to the large scale burning events along the escarpment edge (>2km^2), especially between the South African and Lesotho border posts, and those observed within the Drakensburg foothills, South Africa.
4. 29% of the Sani and 54% Kotisephola Pass regions have not been burnt between 1993 and 2011, while 28% and 21% have been burnt once only during the same time period respectively.
5. The burn frequency and the number of burn scars are observed to be increasing and the area remaining unburnt each year is also generally increasing (i.e. the number of burns is increasing but are smaller in extent).

6. The majority of the burnt areas each year measure less than 1000m^2 in extent with more than 80% of the burn scars each year smaller than 5000m^2, and very few burn scars measuring greater than 0.1km^2 during the study period.
7. In the Sani study region, the majority (>60%) of the burn scars occurred in flat areas between 1993 and 2008, while the north- (5%) and west-facing (6%) slopes generally accounted for the lowest percentage of the burn scars. In the Kotisphola Pass region, the flat areas did not always contain the greatest percentage of the burn scars. The greatest percentage was observed on south-facing slopes, except in 2000 when both the north-facing and south-facing slopes accounted for 29% of the burn scars, as well as in 2009 and 2011 (25% and 26% respectively) when the north-facing aspects accounted for a greater number of burn scars.

Chapter 7:

Tussock Shape Study



CHAPTER 7:

Tussock Shape Study

Tussock grasses are commonly defined as being grasses whose structure forms dense bunches with 'a tight mass of buds at 1-3 cm below the surface' which often hold soil between the stems (the crown); a formation that protects them from death by fire (Cremer, 2004, p17). Yet, the *M. drakensbergensis* individuals within the study regions have different shapes or structures. Differences in tussock structure were first noted by Cole and Wilkins (1958), who distinguished three different forms: normal, hollow center and clump edge. These authors believed that the hollow center (or ring shape) and clump edge depicted deterioration, with the normal form representing the initial structure. Subsequently, Gatsuk *et al*. (1980) identified eight stages, each with their own characteristics: seed, seedling, juvenile, immature, virginile, reproductive (young, mature and old), subsenile and senile. However, they were unsure of the causes in change of shape, whether they were environmental or merely a natural growth pattern, but argued that shape changed with age.

Several other explanations have been proposed for tussock shape changes, some of which may be true to a particular setting. For example, the structure is said to influence environmental factors promoting growth (e.g. soil moisture and soil nutrients) (Ravi *et al*., 2008). Growth from seed, rather than vegetative propagation, has also been proposed as a potential cause of deterioration (Lewis *et al.*, 2001). Deposition of sediment within centers due to the filtering and trapping abilities of tussock structure was first suggested by Ravi *et al*. (2008), whilst grazing exclusion, overgrazing and over-burning are believed to accelerate the fragmentation of the tussock (Cole and Wilkins, 1958; Feldman *et al.*, 2007). It is thus possible that by examining the shape of tussocks, a relationship between burn frequency and shape may be detected. This chapter thus aims to determine whether there is a difference between the shapes at each study site and how these relate to recent burn frequencies. Two sites burnt during the course of this study were further investigated to determine whether shape was altered with burning over relatively short (2 years) time scales. The relationship between tussock reproduction and shape will be briefly examined and discussed. Finally, the influence of small flowering plants growing within individuals and wind will be briefly discussed as they may explain results obtained later in the study. Tussock shape forms a potential and valuable indicator of age, mortality, reproduction rates and consequent population survival (Brzosko, 2001).

7.1 Methodology

Differences in tussock structure were first noted by Cole and Wilkins (1958), who distinguished three different forms: normal, hollow center and clump edge. These authors believed that the hollow center (or ring) and clump edge forms depicted deterioration, with the normal form representing the initial structure. Overgrazing and over-burning are believed to accelerate this deterioration towards the death of the tussock as a single tuft (Cole and Wilkins, 1958). It was thus felt that examining the shape of *M. drakensbergensis* tussocks may indicate a relationship between burn frequency and shape. For the purposes of this study, tussock shape has been divided into structure and height.

The study was undertaken in early October 2009 when the structures of individuals within each of the *M. drakensbergensis* communities forming the study sites were noted, both within the study region and at Kotisephola Pass. The form classes developed by Cole and Wilkins (1958) were used due to their simplicity and ease (see Figure 2.3, Chapter 2.6). The eight classes identified by Gatsuk *et al.* (1980) were not used as they present many different options, many of which are difficult to distinguish between, especially in the winter months when the tussocks are not reproducing. Each individual *M. drakensbergensis* tussock was assigned a label (e.g. 'A', 'B') according to its current structure at the time fieldwork was undertaken. The individuals of each site were classified as either being type 'A', 'AB','B' or 'C' (Figure 7.1). Type 'A' was observed to have no empty or dead spaces on the crown and is roughly circular in shape. A hollow or dead section, not necessarily round, within the central parts of a tussock were termed type 'B'. Type 'C' was indicated by a large hollow section and growth limited to clusters of tillers in varying positions along the edges. Different morphological varieties of each type were observed, such as a crescent shaped type 'C', and were classified according to the most appropriate type. Some type 'C' individuals were difficult to identify as the original circular structure was large (approximately >2m) and the crown was no longer present. In these cases, clusters of the original tussock were identified to represent type 'A' as vegetative reproduction through age had occurred. The current study introduced a fourth form, as differentiating between type 'A' and type 'B' was not always conclusive due to accumulated leaf litter; thus a type 'AB' was assigned to these individuals. These loosely corresponded to Gatsuk *et al.*'s (1980) forms; 'A' represented the juvenile and young reproductive forms, 'B' represented the mature reproductive form and 'C' represented the old reproductive, subsenile and senile forms.

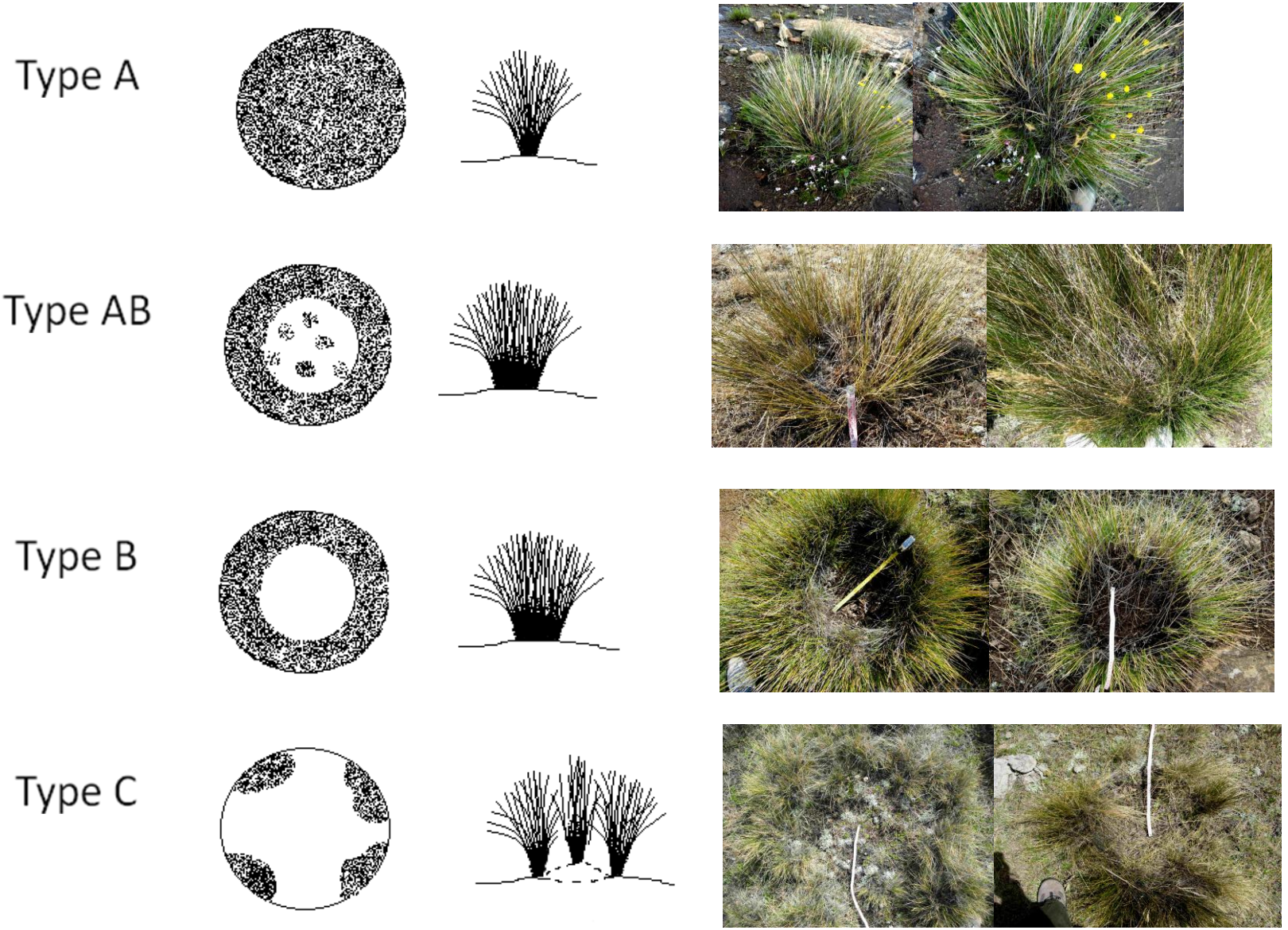


Figure 7.1: Diagrams and photographs depicting the tussock types identified. In the aerial view, shaded sections indicate living leaves.

Seedlings were not included as many are difficult to distinguish from the surrounding shorter sod grasses, yet juveniles were included as soon as they could be identified as the study species. As with other studies, only *M. drakensbergensis* was used to maintain continuity as well as eliminate any different growth forms and strategies of different species. All twelve sites were included within this study but the M.F. site was not divided into an upper and lower site until later. Fifty individuals were recorded for each site, except at site S.E. as only five individuals were present at site S.E. and thus only five types were recorded. Classifications were made for individuals throughout each of the study site communities: from the community periphery to the centre. In the smaller sites, all the individuals were recorded while in the larger sites a grid was formed using 50m tape measures. The grid was scaled to fit across the community (e.g. in the case of site V.S.N the grid was 0.5m by 0.5m) with fifty intersections occurring within the community or along the periphery. The individuals nearest each intersection were recorded.

The collected data were entered into Microsoft Excel for Windows 7. Type 'A' was represented by a value of 1, type 'AB' by a value of 2, 'B' by a value of 3 and 'C' by a value of 4. Totals and percentages for each tussock type were calculated for each site and presented as pie charts. The shape data could then be used to determine whether any relationship existed with the burn frequency of each site and the time since each site was last burnt. Both the burn frequency and the time since each site was last burnt were obtained from the Burn frequency maps produced using remote sensing in Chapter 6. The presence of a relationship was observed using a two-tailed Pearson's product moment correlation in SPSS version 21, after the data was observed to meet the required assumptions (e.g. normal distribution). Altitude and aspect may also play a role in the structural shape types observed and thus the presence of any relationships were also briefly observed (also using Pearson's correlations and SPSS version 21) and discussed. This would potentially indicate whether the *M. drakensbergensis* tussocks were aging at an accelerated rate at sites with high burn frequencies, and similarly, at sites with long time intervals between burning. The structural shape type data were also compared to the average number of inflorescences observed to examine the role which shape type may play in reproduction. The results were also compared to the favoured grazing routes determined within the grazing study. During the course of the fieldwork it was noticed that many small flowering plants were growing within the *M. drakensbergesis* individuals. These were noted, photographed and identified as they may play a role in the structural progression suggested by Cole and Wilkins (1958) and Gatsuk *et al.* (1980). The main findings are briefly presented and discussed as they may explain the results obtained but are not the focus of this study.

Site V.S.E. and the A.P. site were both partially burnt during 2009 (site V.S.E. in late winter and the A.P. site in early spring) and the burnt tussocks included in this study. They were recorded using the same structure classification as used with the unburnt tussocks, but the records were kept separate. The structures of all the burnt individuals from each of the burnt areas were recorded. Thus, the burnt individuals measured were from within the burnt area and along the periphery. Structures of the burnt tussocks prior to burning were noted during August 2009 as additional information for the monitoring study (Chapter 9). This information proved useful as it provided evidence of the structure of the burnt individuals before burning had occurred. Thus, structural changes observed one month after burning, in October 2009, maybe attributed to the burning event or post-burning conditions. Repeated noting of the individual tussock structures from the burnt sites was performed three months after burning during December 2009 and in the case of the A.P. site, five months after burning during February 2010. A repeated measures ANOVA was performed to determine whether the two burnt sites responded in a similar manner with time. Mauchly's test of sphericity was performed and the conditions of sphericity were met as well as the assumptions for an ANOVA (e.g. normal distribution, equal variances). Additionally, a Pearson product moment correlation was undertaken to indicate whether a relationship was evident between the percentage shape types and time.

The dominant wind direction may influence the shape of the tussocks and may explain some of the results obtained later in the study. Thus, the *M. drakensbergensis* shape was also briefly observed through the average height of the measured individuals. Each individual was divided into four sections according to compass directionality (i.e. north, east, west and south). The average leaf heights were calculated for each section by measuring an edge tiller, radial midpoint tiller and a central tiller. The measurements were then averaged to reach a single value for each aspect. The values were tabulated and the variation calculated. A Friedman test for related samples was performed using SPSS version 21 to determine whether differences existed between the average heights of each aspect. This test was chosen as the data for the east and south sections of the individuals did not follow a normal distribution and the measurements for each aspect were taken from the same individuals (Townend, 2002; Dytham, 2011). The average height is discussed in greater depth in the following chapter (Chapter 8). Observations were also made regarding the greatest crown diameter and the position of accumulated matter such as hail after an afternoon storm event.

7.2 Results of the Shape Study

The most common tussock structural type observed was type ‘A’ (>46% for most sites) for all sites except sites S.E., A.P. and the site at Kotisephola Pass. Type ‘AB’ was the next most common type found at the sites (>30% except at sites S.Str., H.E., H.W., V.F. and K.P.). The S.E. site contained the greatest percentage of type ‘AB’ individuals (40%). Less than 9% of type ‘B’ was found at most sites, except sites S.E., A.P. and K.P. The most variation was observed in the occurrence of type ‘C’. No type ‘C’ individuals were observed at sites S.E. and V.S.N., while more than 20% was observed at the A.P. site and at Kotisephola Pass.

Pearson’s product moment correlations were used to determine whether the burn frequency and the time since last burnt were the drivers of the proportioning of structural shape types (Table 7.1). No significant linear relationships were present when the shape types were compared to the burn frequency ($p > 0.1$). The time since each site was last burnt was not quite significantly related to the percentage of type ‘B’ individuals observed at each site ($p = 0.06$). The relationship is indicated to be indirect through a negative correlation coefficient (r). Altitude was significantly correlated to the percentage of type ‘AB’ individuals present ($p = 0.02$). This relationship was also observed to be indirect. No relationships were observed between aspect and the average number of inflorescences and the structural shape types ($p>0.1$).

Small flowering plants were observed to grow within the central regions of *M. drakensbergensis* individuals or along the periphery (Figure 7.3). These species were identified and three species found to be endemic to the Eastern Mountain Region (Table 7.2). The majority of the smaller plants were observed within type ‘B’ and type ‘C’ individuals. However, some were also noted within type ‘AB’ individuals (Figure 7.4). Almost no invasion of type ‘A’ individuals as observed. Burnt sites V.S.E. (62%) and A.P. (70%) experienced the greatest percentages of flowering plants within tussocks. *Oxalis depressa* and *Trifolium burchellianum* were common species found within the burnt remains of *M. drakensbergensis* tussocks at site V.S.E. *Rhodohypoxis baurii* were also a common site and were found on 43.3% of tussocks on the Apostles plateau, after burning and after rainfall. *Helichrysum* species (e.g. *Helichrysum adenocarpum*) were common to all sites, as was *Alchemilla colura*, especially at wetter sites.

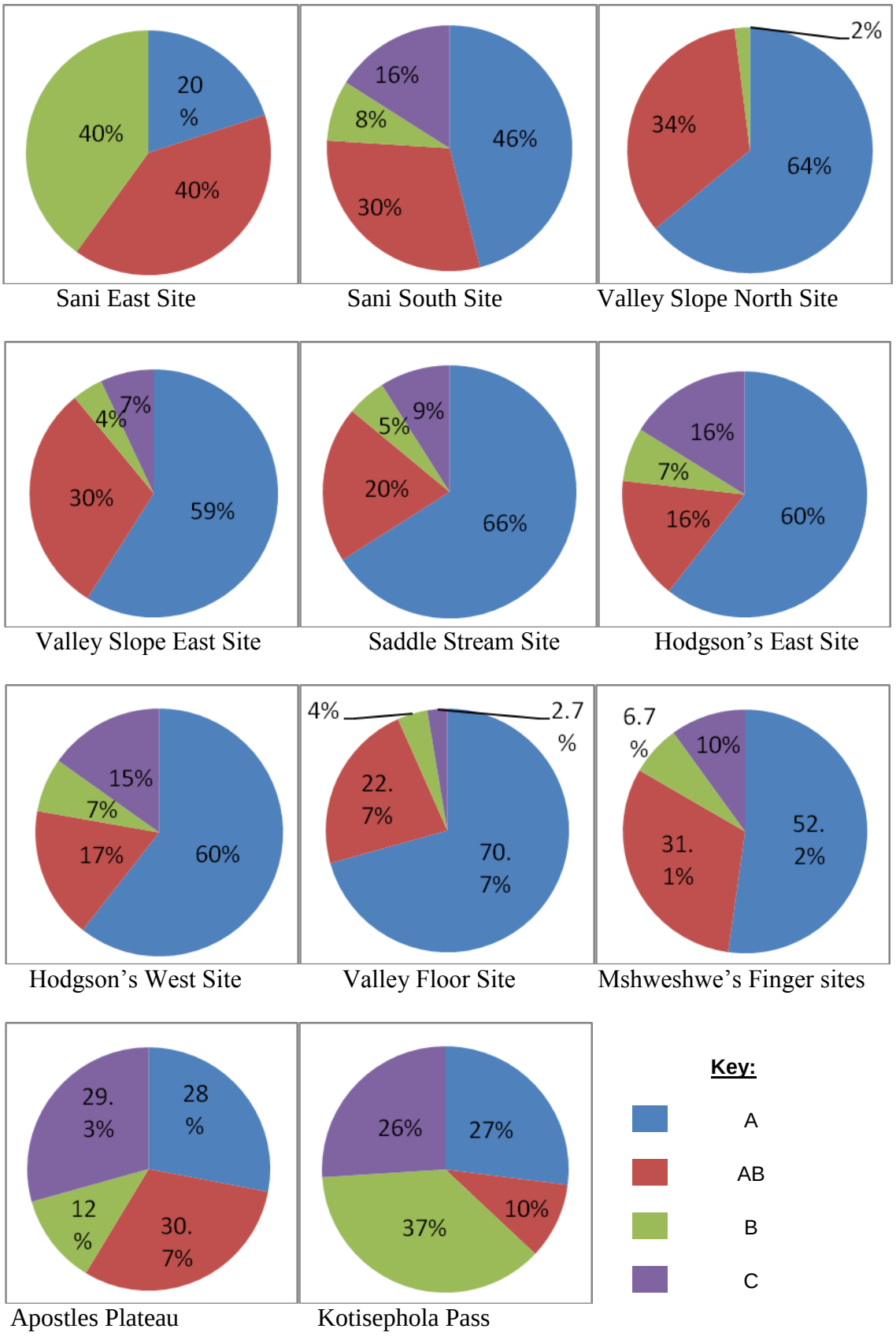


Figure 7.2: The percentages of each shape type found at each site.

Table 7.1: Correlation coefficients (r), p-values and significance as the results of Pearson correlations using burn frequency and time since the last burn as well as altitude, aspect and average inflorescence against shape (n = 12).

Variable	Shape	r	P-value
Burn frequency	A	-0.436	0.157
	AB	0.184	0.568
	B	0.495	0.102
	C	-0.068	0.833
Time since last burnt	A	0.499	0.099*
	AB	0.169	0.599
	B	-0.559	0.059*
	C	-0.281	0.376
Altitude	A	-0.170	0.598
	AB	-0.647	0.023**
	B	0.442	0.151
	C	0.336	0.286
Aspect	A	0.128	0.708
	AB	0.305	0.361
	B	-0.468	0.146
	C	0.161	0.635
Inflorescences (Average)	A	0.113	0.740
	AB	0.273	0.417
	B	-0.111	0.744
	C	-0.312	0.350

r: correlation coefficient; P-value: significance value.
* P<0.10; **P<0.05; ***P<0.01

The percentage of each structural shape type occurring in each of the burnt tussocks at sites V.S.E. and on the Apostles plateau was recorded and the results are presented in Figures 7.5 and 7.6. Examples of burnt tussocks are shown in Figures 7.7 and 7.8. After burning, an increase in type 'C' tussocks was observed when compared to the percentages of shape types before burning. Yet, three months after burning at site V.S.E. and site S.E., the percentage of type 'C' tussocks had decreased.

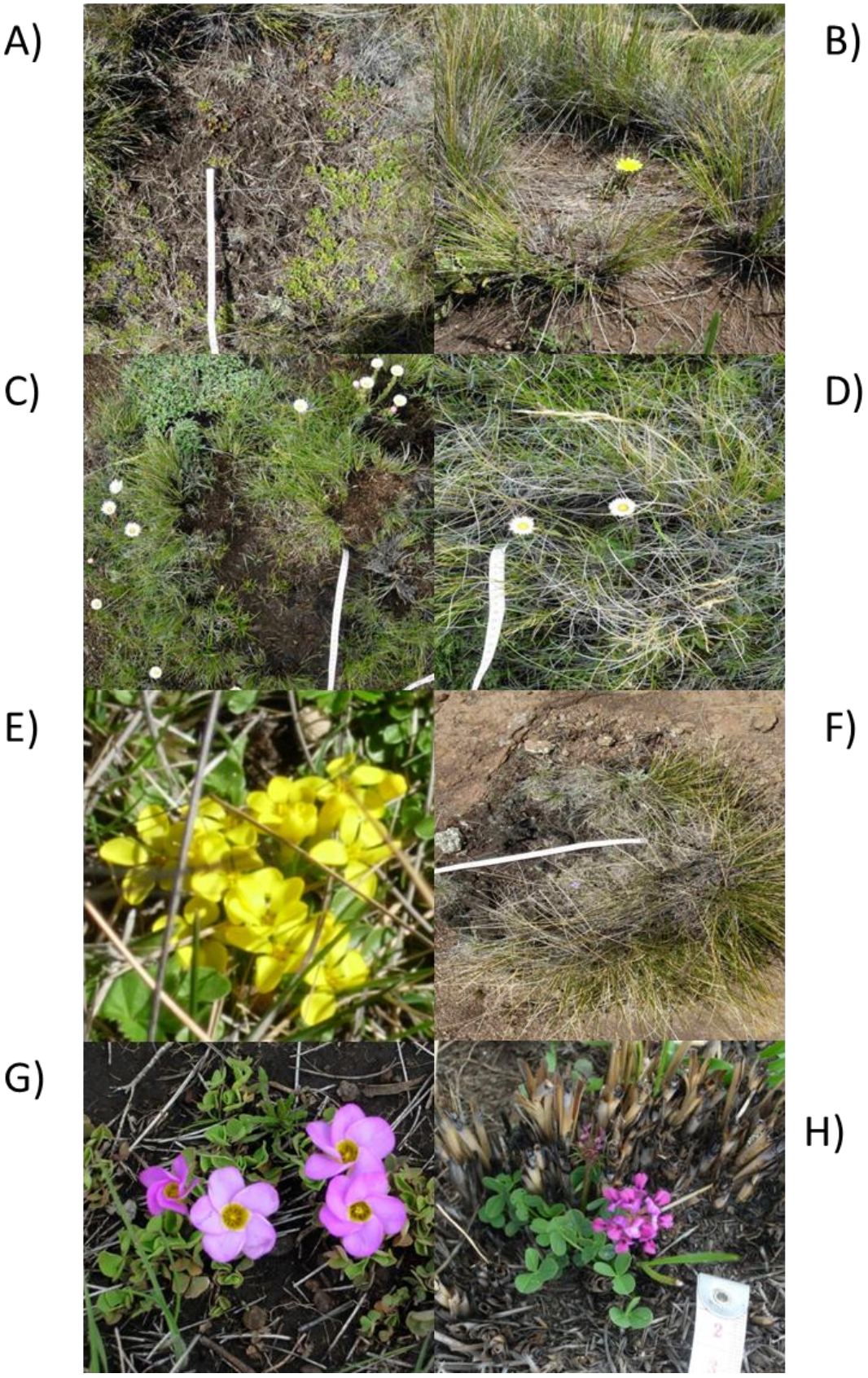


Figure 7.3: A) *Alchemilla colura*, B) *Gazania krebsiana*, C) *Helichrysum albo-brunneum*, D) *Helichrysum bellum*, E) *Sebaea morlothi*, F) *Morea alpine*, G) *Oxalis depressa* and H) *Trifolium burchellianum*.

Table 7.2: Examples of species found within the ring centers and along the edges of tussocks.

Species
*Alchemilla colure**
*Berkheya multijuga**
*Cotula socialis**
Craterocapsa tarsodes
Gazania krebsiana
Helichrysum adenocarpum
Helichrysum albo-brunneum
Helichrysum bellum
*Moraea alpine**
Oxalis depressa
Rhodohypoxis baurii
*Sebaea marlothii**
Trifolium burchellianum

* Species is endemic to the Eastern Mountain Region (EMR)



Figure 7.4: Type 'AB' tussock with a *Helichrysum* sp. and *Rhodohypoxis baurii* emerging from the central and edges. *Craterocapsa tarsodes* can be seen at the base of the tussock.

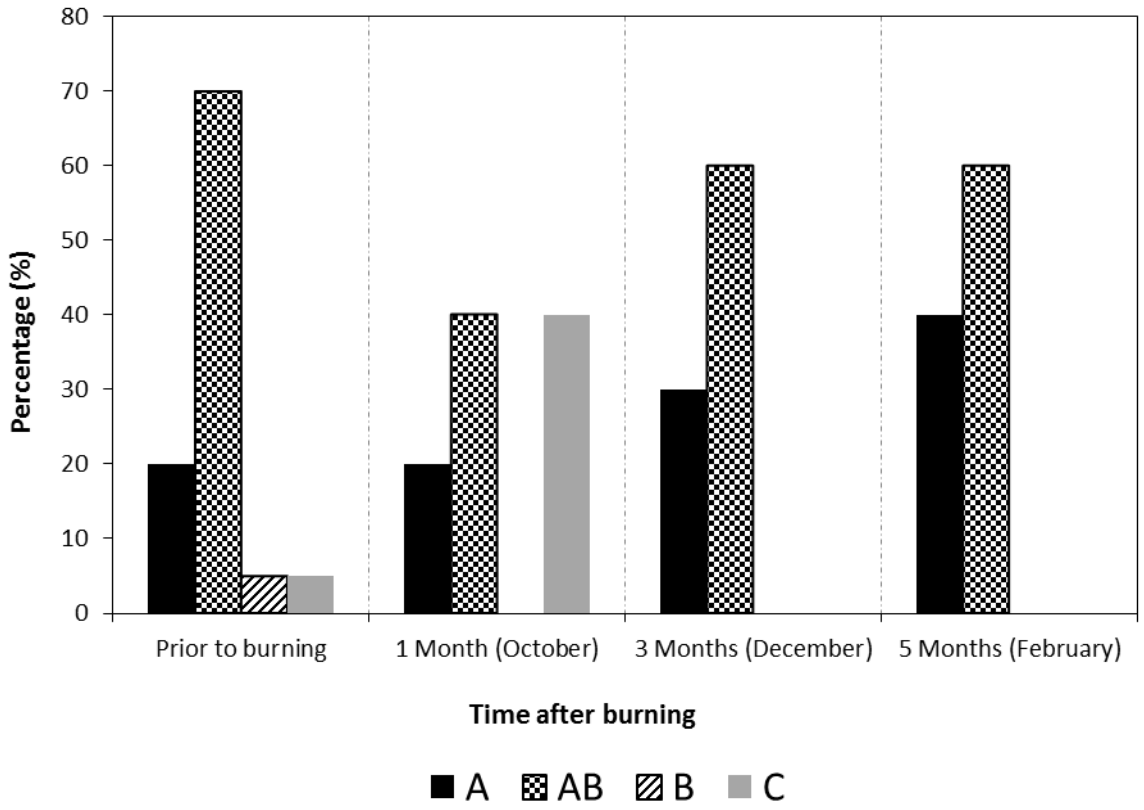


Figure 7.5: Percentage of each shape type (A, AB, B and C) for the same individuals within the burnt areas at site V.S.E. over a three month period.

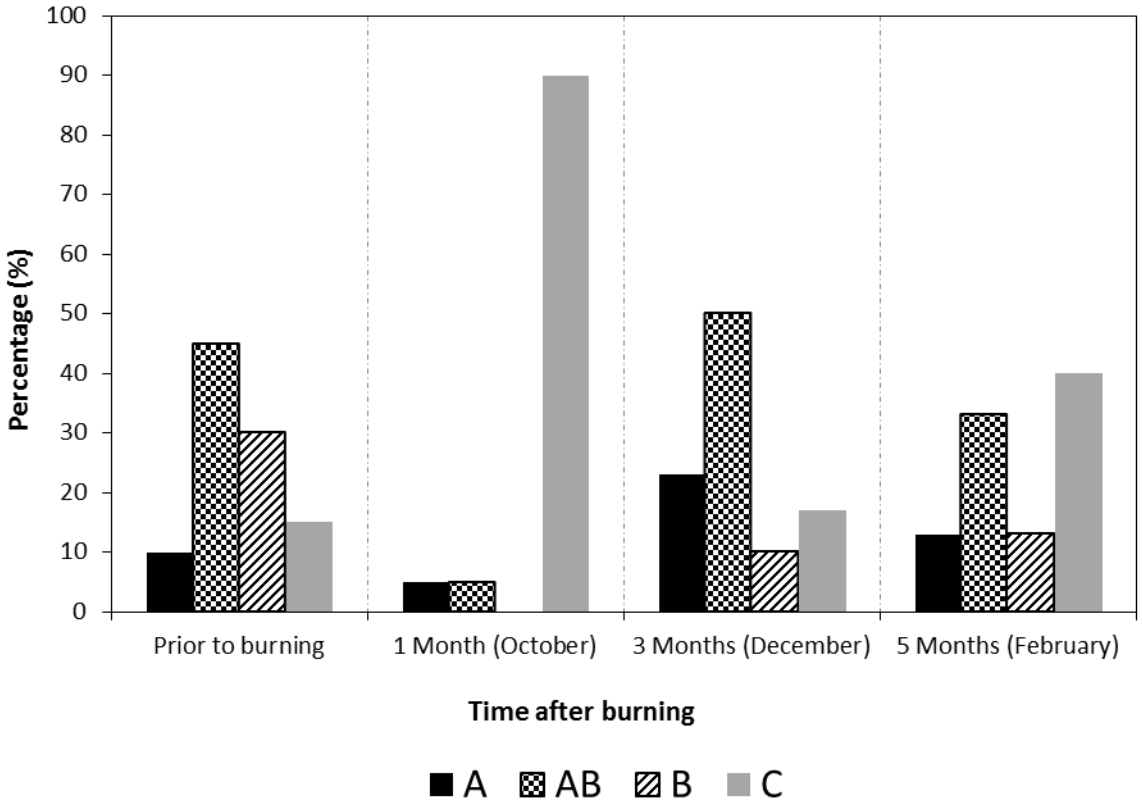


Figure 7.6: Percentage of each shape type (A, AB, B and C) for the same individuals within the burnt areas at the A.P. site over a five month period.



Figure 7.7: Examples of burnt tussocks at site V.S.E. Tape measure for scale (width 1.2cm).

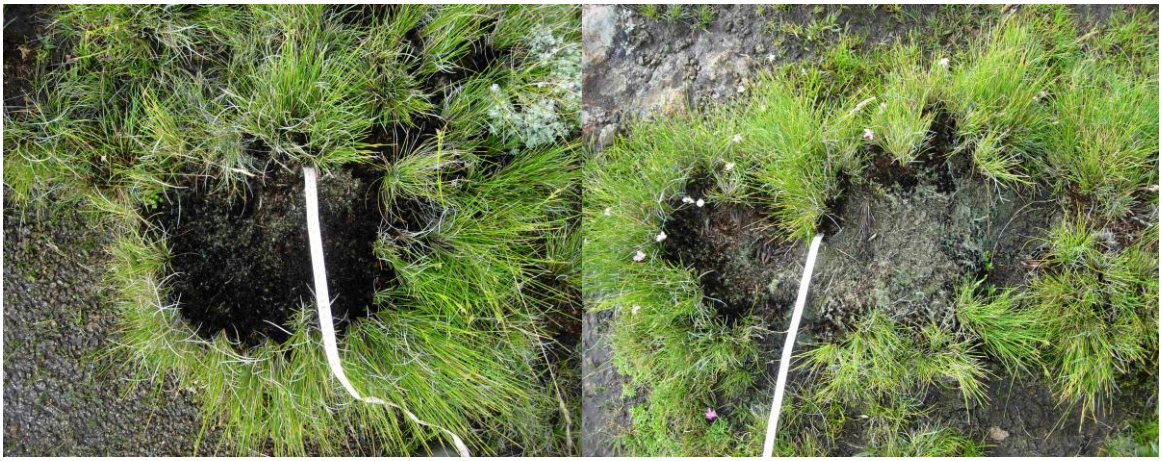


Figure 7.8: Examples of burnt tussocks at the A.P. site. Tape measure for scale (width 1.2cm).

A repeated measures ANOVA was performed to determine whether the sites responded in a similar manner with increasing time since burnt. The results indicated that there was no significant difference between the responses of the V.S.E. and A.P. sites, Wilks' Lambda = 1.0, $F(1, 15) = 0.01$, $p = 0.92$. Pearson's product moment correlations were undertaken to observe the presence of a relationship between shape types and increasing time since being burnt, as well as all shape types and time (Table 7.3). The results indicated no significant relationships between structural shape types and time ($p<0.1$ for all correlations).

Table 7.3: Pearson's product moment correlations indicating the presence of a relationship between shape type and time, as well as sites and time.

Variable	Shape	r	P-value
Time	A	0.517	0.189
	AB	-0.003	0.994
	B	-0.318	0.442
	C	-0.104	0.806
Variable	**Site**	**r**	**P-value**
Time	V.S.E.	-0.012	0.966
	A.P.	0.000	0.999

r: correlation coefficient; P-value: significance value.
* P<0.10; **P<0.05; ***P<0.01

The role played by wind and aspect in determining *M. drakensbergensis* tussock shape were observed through the differences in tiller height at different tussock aspects (Table 7.4). However, grazing cannot be discounted, thus the results are an indication of vertical shape rather than the roles played by wind and aspect. The average leaf heights across all sites were observed to be greater on the east and south sides of the tussocks, while the average leaf heights were shorter on the northern and western sides. The exception is observed in January 2009 when the western sides were tallest and the eastern sides were shortest. The majority of the *M. drakensbergensis* tussocks were shortest on the western sides and tallest on the southern sides. The Friedman's test found significant differences between at least two of the aspect medians ($p = 0.000$). Significant differences were not observed between the northerly and westerly aspects, as well as between the southerly and easterly aspects ($p>0.1$; Table 7.5). However, wind was

observed to play a role in *M. drakensbergensis* shape. The greatest basal diameters were observed to correlate to the dominant wind direction (Figure 7.9). Similarly, hail was observed to accumulate on the northwesterly side of the individuals of site S.E. after a storm event in which the wind was blowing in the dominant northwesterly direction (Figure 7.10).

Table 7.4: Average peripheral leaf height across all sites for each aspect with mean variance.

Year	Month	North: Average Height (cm)	East: Average Height (cm)	South: Average Height (cm)	West: Average Height (cm)
2009	**January**	29.8 ± 9.8	26.7 ± 12.8†	29.3 ± 12.6	30.8 ± 11.8*
	March	33.3 ± 9.4	36.9 ± 7.3	37.0 ± 9.7*	32.3 ± 6.5†
	May	34.0 ± 8.4	35.1 ± 6.2	38.0 ± 7.3*	33.4 ± 7.7†
	August	26.5 ± 3.2	33.4 ± 9.0*	27.3 ± 8.1	25.7 ± 2.5†
	October	29.4 ± 6.8†	35.0 ± 9.5*	34.4 ± 9.5	31.0 ± 9.1
	December	34.4 ± 7.9†	36.0 ± 5.1	37.8 ± 9.0*	34.4 ± 8.4†
2010	**February**	34.2 ± 9.2†	36.7 ± 8.2*	36.1 ± 9.7	36.0 ± 10.0
	April	34.0 ± 9.3†	38.1 ± 7.3*	36.4 ± 9.4	36.2 ± 11.4
	June	34.8 ± 8.6†	38.3 ± 7.9*	37.3 ± 9.2	36.3 ± 11.2
	August	36.2 ± 8.1	37.1 ± 10.3	37.7 ± 8.9*	35.8 ± 9.4†
	October	37.6 ± 7.5	38.0 ± 10.5	40.3 ± 9.1*	36.9 ± 9.4†
	December	39.4 ± 8.7	40.3 ± 11.6	41.7 ± 10.7*	39.0 ± 9.8†
2011	**March**	37.3 ± 11.1†	38.1 ± 13.2	40.7 ± 12.4*	37.7 ± 11.7
	May	36.0 ± 9.3	38.9 ± 8.4*	37.3 ± 7.9	34.4 ± 7.8†
	August	34.4 ± 8.3	35.6 ± 8.3	36.1 ± 7.5*	33.1 ± 7.4†
	October	37.0 ± 8.8	37.7 ± 8.3	38.5 ± 8.1*	35.9 ± 7.6†
	December	38.1 ± 9.8	38.0 ± 10.2	38.9 ± 10.6*	37.4 ± 8.8†

* Tallest average height; † Shortest average height

Table 7.5: Significance values as a result of the Friedman test comparing average height from each of the four aspects (north, south, east and west).

Sample comparison	Significance	Adjusted significance
North - East	0.002***	0.011**
East- South	0.425	1.000
South - West	0.000***	0.000***
West - North	0.507	1.000
North - South	0.000***	0.001***
West – East	0.000***	0.001***

* P<0.10; **P<0.05; ***P<0.01

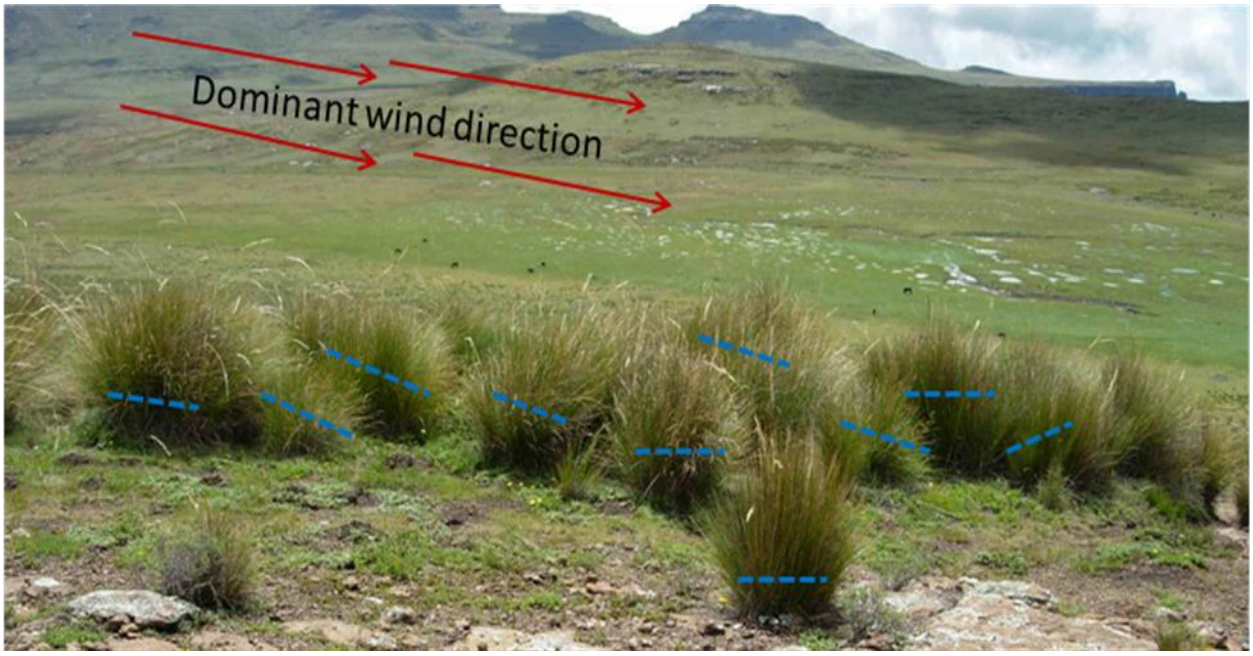


Figure 7.9: Photograph taken at site V.S.N. indicating the dominant wind direction (red arrows) and the widest tussock width (blue dashed lines).



Figure 7.10: A tussock from site S.E. following a hail storm where hail can be seen on the northwesterly side of the tussock and small plants are visible on the southeasterly side. The tussock also appears denser with longer leaves on the southeastern side.

7.3 Discussion

Cole and Wilkins (1958) suggested that the ring and clump edge shapes depict deterioration, with the normal form representing the initial structure. Considering this, the results suggest that the A.P. and K.P. sites are greatly degraded due to the percentage of type ‘B’ and ‘C’ tussocks present. Other sites show some degradation, whilst sites V.S.N. and V.F. are the least degraded. However, if the more recent argument suggested by Gatsuk *et al*. (1980) and Derner *et al*. (2004) are taken into account, a different assumption could be made. The idea suggests that a series of progressive stages is followed with increasing age of each individual. Thus, sites S.E., A.P. and K. P. sites contain older individuals. The roughly even percentages of each type suggest that these three populations are stable. The shapes of tussocks are a good indication of age and thus an indicator of the reproduction and mortality rates (Brzosko, 2001). This suggests that the three sites which have a greater percentage of older individuals have a high mortality rate and a low reproduction rate. Further, this suggests that those populations may be regressing and may not be present within a few years. Conversely, this theory may suggest that those sites with high percentages of type ‘A’ tussocks have a high reproduction rate. Combined with a low percentage of type ‘B’ and ‘C’ tussocks, indicating mortality rates, this would indicate growing populations. This was observed for most sites, excluding sites S.E., the A.P. and K.P. If both of these theories are combined, the assumption is that the stages presented by Gatsuk *et al.* (1980) occur at an increased rate when stresses promoting degradation are applied. This would suggest that sites S.E., the Apostles plateau and Kotisephola Pass experience the most influential factors acting to degrade the tussock structure.

The burn frequency and time since the last burnt may be considered stress factors when the burn frequency is too high to allow for adequate recovery or too low allowing for suffocation by leaf litter. Thus, the lack of significant results of the correlation when shape type was compared to burn frequency was surprising. Burning has been shown to influence the shape through higher temperatures experienced in the crown (Wright, 1974). The high temperatures are able to injure potential growth points, preventing central growth and thus leading to the hollows which are sometimes referred to as “Monk’s tonsure-like gaps” (e.g. Lewis *et al*., 2001) or even as ‘rings’ (e.g. Wikberg and Mucina, 2003). This is supported by the theory that central tissues are known to be killed first (Wright, 1971; Peláez *et al*., 1997). Thus, it was expected that sites with a higher burn frequency would have a higher proportion of type ‘B’ and ‘C’ tussock. This was observed when examining the raw data, but was not statistically significant ($P > 0.1$).

The time since each site was last burnt was not quite significantly related to the percentage of type ‘B’ individuals ($p = 0.06$). The relationship, indicated by the correlation coefficient, is indirect. Hence, as the time since the site was last burnt increases, the percentage of type ‘B’ individuals will decrease. The raw data suggested greater percentages of type ‘A’ at sites which have experienced nine or more years since the last burn but was observed to be not significant ($p = 0.1$). The relationship did suggest that as the time since burning increases, so the percentage of type ‘A’ individuals will increase. The central tissues may suffer the greatest amounts of damage due to accumulation of litter and longer smoldering times. The resulting increased recovery times or the further degradation of central tissues may explain the decrease in type ‘B’ individuals with increasing time.

The two recently burnt sites indicated that they respond similarly with increasing time since last burnt, on a short time scale (five months). This suggests that the regrowth follows a similar pattern regardlessof the site location. However, the changes with structural shape types did not indicate the prescence of a relationship with increasing time on a short time scale ($p>0.1$). Combined with the relatively longer term results (the time since each site was last burnt), this may suggest that the relationship between the time since last burnt and structural shape type increases with increasing time. Despite the lack of significant results at the short time scale (five months) the raw data was observed for the presence of trends. This was done as two burnt sites is a very small sample and the significance may alter with a larger sample size, potentially increasing the significance of observed trends.

It has been suggested that the ring form (type ‘B’) may be hidden by dead plant matter until burnt, and that the gaps are as a result of non-living tissue before the disturbance, with burning or overgrazing merely allowing it to be seen (Lewis *et al.*, 2001; Feldman *et al*., 2007). This could explain the increase in observed types ‘B’ and ‘C’ after burning, as was observed at the A.P. site and at site V.S.E. Burn severity differs across individuals and may result in different sprouting patterns. Growth after burning was most frequently observed along the tussock edges, with small tufts of growth in the centers of some individuals (Figure 7.5 and 7.6). The lack of regrowth in the centers may be the result of a lack of heat dissipation as well as very high temperatures resulting from the accumulation of organic matter and a high density of tillers resulting in long smoldering durations (Wright, 1971; Peláez *et al*., 1997; Paysen *et al.*, 2000; Bruhjell and Tegart, 2001). Centers were observed to be easily loosened and broken after smoldering for half an hour

or more. Thus, it is believed that burning promotes the development of central sediment filled rings, through the death and degradation of crown centers.

With increasing time since burning of the burnt sites, the percentages of type ‘A’ and especially type ‘AB’ tussocks, became more prevalent. This confirms that new growth emerging from the crown begins after a longer recovery period than that of the outer edges. The high density of the crown may be responsible for this as meristematic tissues may be hidden within the dense mass of burnt tiller bases. Time would be required for new tillers to reach the surface; whilst at the periphery new shoots have more space to emerge. Tillers emerging from the outer edges were taller than those emerging from the centers (see Figure 7.1), as has been reported in Oklahoma by Volesky (1994). This may simply be due to the emergence of edge tillers before those of central tillers, and thus have a longer time for growth. Alternatively, the outer tillers have a faster growth rate than central tissues, which will be discussed in more detail in the monitoring study (Chapter 8). If the growth rates are different, the result would be a further increase in the rate of structural change. Similarly, some tussocks may not generate new growth in the center, as was observed with some tussocks at both burnt sites, which may be a result of damage to the crown. The result would be a rapid progression from either type ‘A’ or type ‘AB’ to type ‘B’ or ‘C’ tussocks. In accordance with Gatsuk *et al.* (1980), this would mean a faster aging of the tussock. Whether this would decrease the life expectancy of the individual tussock is unknown and would require further research.

The local environment and burning can influence the shape of tussock grasses, as was found for *Festuca varia* (Haenke.) in the northwest Caucasus, Russia (Pokarzhevskaya, 1998). Round unordered tussocks were found on gentle slopes and oblong shaped tussocks were found in rows on steeper slopes. Round tussocks were observed to have even litter around their bases and new shoots distributed evenly (Pokarzhevskaya, 1998). This suggests that round tussocks would undergo uniform burning around the base and produce evenly distributed new shoots after burning. Oblong tussocks found on slopes have bare patches on the upslope portion, new shoots in the upper and middle portions and litter found on the down-slope side of tussocks (Pokarzhevskaya, 1998). Burning may therefore be uneven and the distribution of new shoots after burning may be concentrated on the upper side. Regrowth after burning tends to be more erect, reducing the plant cover between tussocks (Daubenmire, 1968). The result would be increased erosion within inter-tussock spaces, which in turn would negatively influence the outer edges of the tussocks. In the current study, the more erect shape of new tillers is not significant,

as many new tillers were produced at angles from the sides of the burnt crown. The lack of vegetation cover was observed to increase erosion, whilst undercutting of some *M. drakensbergensis* tussocks was also observed (personal observation).

The degradation of tussocks, either through structural age state or by environmental factors (fires, grazing, wind etc.) may influence the reproductive ability of these grasses. The later the age stage and the younger the tussock, the less likely they are to produce inflorescences and seeds (Gatsuk *et al*., 1980; Derner *et al*., 2004). The reproductive stages are the young (large type 'A'), mature and old reproductive stages (type 'AB') (Figure 7.1). Therefore, the greater the percentage of type 'A' and 'B' tussock, the greater the number of inflorescences (and seed) expected. There is no significant relationship between the number of inflorescences at each site and the percentage of each shape type. Each tiller has the ability to produce an inflorescence and inflorescence production is dependent on environmental conditions. Assuming tillering is occurring to replace dead tillers each year, the number of tillers should remain fairly constant, resulting in few changes in inflorescence number with slow shape changes. However, the results may be slightly inaccurate as this merely compares the percentage of each type and the average inflorescences. It does not examine the same individual and its own inflorescences, and thus the results may differ slightly from the averages which were used. This brief study does also not take into account the effects of wind, hail, snow, livestock damage and grazing to the inflorescence stalks. Stalks may be broken, thereby reducing the number of inflorescences present. This in turn would have led to inaccurate measurements and thus inaccurate results.

Greater annual growth rates and size of ramet individuals along the edges of sedges may be attributed to more available space (Wikberg and Svensson, 2003). Thus, these larger ramets are able to vegetatively reproduce more effectively, producing new ramets where space and resources are more available. Similarly, Gatsuk *et al*.'s (1980) theory suggests that degradation of a tussock can result in several new tussocks. Burning coupled with a high percentage of type 'C' tussocks has great potential for reproduction through vegetative means. Thus, burning may speed up the process of reproduction through fragmentation and severing links between already fragmented individuals. This would result in an increase in type 'A' individuals after burning, as was observed.

Altitude was significantly negatively correlated to the percentage of type 'AB' individuals ($p = 0.02$). The relationship suggests that the amount of loose material collecting to form a type 'AB'

decreases with increasing altitude. This finding was surprising as the majority of grazing occurs in the valley base (i.e. at a lower altitude) and grazing may limit leaf litter accumulation preventing the formation of an 'AB' type. Rather, the decrease in the percentage of type 'AB' individuals may be attributed to potentially slower growth rates at higher altitudes and increasing wind strength (due to increasing exposure), resulting in less accumulation of litter. However, the actions of grazing may still playa role. Grazing of tussocks may assist fragmentation or subdivision of individual tussocks, thus increasing the total plant density of a landscape (Butler and Briske, 1988; Oliva *et al.*, 2005). As well as the defoliation effects of grazing, livestock may lie on and flatten individuals, as was observed during this study. Similarly, larger tussocks show a more rapid rate of subdivision than smaller individuals, as observed in Argentina (Oliva *et al.*, 2005). Differences in *M. drakensbergensis* tussock size were measured within the study region; site V.F. has the largest all-round tussocks whilst site S.E. and S.S. measured the smallest (see Chapter 8). On further investigation, the largest individuals are associated with areas of least grazing while the smallest individuals were observed in areas of higher grazing intensities, which conforms with the results obtained by Oliva *et al.* (2005).

In several cases, different species were observed in the center, especially within the A.P. site. Plants were observed in the centers of hollow (type 'B') and partial (type 'C') individuals (Table 7.4 and Figure 7.3) and occasionally type 'AB' tussock (Figure 7.8). All species which were found within tussocks reproduce through seed, which suggests that wind and the trapping effects of tussock leaves may potentially be responsible. Alternatively, conditions within the central regions of individuals may be better for seedling establishment and growth, such as increased light, increased soil temperatures and increased nutrient availability (Vogl, 1974; Vermeire *et al.*, 2005). Their survival and success in flowering within *M. drakensbergensis* tussocks suggests that resource depletion, as a cause of central hollows, may not be occurring. However, the resources stored within the crown and the roots may provide the perfect site for growth of the plants occurring within the centers (Ryel *et al.*, 1994), and they may be responsible for accelerated resource depletion within the centers, as observed in Utah, U.S.A. Yet, light availability needs to be adequate before species can germinate and grow successfully within the centers, as the centers are the most shaded areas within the tussocks (Ryel *et al.*, 1994; Monteiro *et al.*, 2011). This implies that tiller numbers within the centers must be reduced before other species begin to invade, thus burnt tussocks may provide even more suitable habitats for other species.

Sites which experienced late winter (site V.S.E.) and early spring (Apostles plateau site) burns had the highest percentage (62% and 70% respectively) of flowering plants within their centers. *Oxalis depressa, Berkheya multijuga* and *Trifolium burchellianum* were commonly observed within the burnt tussocks at site V.S.E., along with *Rhodohypoxis baurii* which were found on 43.3% of tussocks on the Apostles plateau, after burning and after rainfall. The difference may potentially be due to soil moisture as the Apostles plateau was saturated at the time of observation, while site V.S.E. is situated on a slope and was merely damp. The burnt tussocks near site S.S. were colonized slower than the other burnt sites, perhaps because it had relatively lower soil moisture content than the other sites after burning. Kotisephola Pass burnt tussocks were not colonized in the centers after burning. *Helichrysum* species and small *Erica* shrubs appear in inter-tussock spaces in the growing season following burning but these did not colonize the tussock centres.

The results suggest that a factor not yet considered, or a combination of factors, is influencing the percentage of individuals of each shape type present at each site. Wikberg and Mucina (2002) suggest that the deposition of substances inhibiting growth, or even natural form progression may be responsible. Research has suggested that sediment deposition and a resulting decrease in growth in the center of clumps was responsible for the change in form (Ravi *et al*., 2008). The wind speeds and their duration occurring in the Sani region exceed the required thresholds (approximately 5.4 to 5.5ms^{-1} for dry soil on the North American Great Plains; Stout, 2007) for sediment transport, especially during the winter months (Grab, 2010). Within the study region, sediment was observed within the centres to increase with the prominenance of the hollow or ring (personal observations). Tussocks forming rings (see Figure 7.1 and Figure 7.3) were observed to have elevated centers, which conform to the literature (Lewis *et al*., 2001). The sediment gathering in the central regions may bury and prevent the growth and/or survival of the central tillers, as observed in the northern Chihuahuan Desert, New Mexico (Ravi *et al*., 2010). The accumulation of fine sediment within the central regions also alters the water availability, with runoff increasing within the center towards the edges (Ravi *et al*., 2007; Ravi *et al*., 2010). Hence the periphery becomes more efficient with regards to water infiltration and water uptake (Ravi *et al*., 2007, 2010). Additionally, seed deposited in the central regions along with sediment may begin to grow within the sheltered environment, accelerating the outward tussock expansion. Yet, as the tussock ring expands the central region becomes less protected by wind and may result in sediment removal (Ravi *et al.*, 2008). These observations may help to explain the results obtained later in the study.

It is also thought that the degradation of the centers is more likely due to competition for light, water and resources with outer tillers (Wan and Sosebee, 2000; Sheffer *et al.*, 2003). Root confinement in the center is also believed to play a role, especially in promoting outward expansion (Sheffer *et al.*, 2003). Ash and dead plant matter collected within these centers confirmed the findings of Lewis *et al.* (2001). This would increase the organic and nutrient contents, making them more supportive of growth than the inter-tussock spaces. However, negative-plant soil feedback has been proposed as another cause of the ring formation (Cartení *et al*., 2012). The negative plant soil feedback suggests that accumulated litter in the central regions, its decay and microbal activity may release toxins into the soil, reducing plant matter within the centers (Cartení *et al*., 2012). However, the soil in the centers was often observed to be colonized by small flowering plants and in some places grasses, suggesting that this is not the case in the Sani region. The growth of colonizing plants may potentially increase the hollow and assist in outward expansion, despite the possible increased nutrient content of the center gaps (Table 7.2). In many individuals, the dense dead central tissues posed an obstacle to inward growth. The hard dense centers were observed to take a long time to deteriorate, a process which was observed to be expedited by burning.

The dominant wind direction may influence the shape of the tussocks, causing the crown to become increasingly ellipsoid or asymmetrical as opposed to round (Wilson, 1959). Slope is perhaps more influential on tussock shape as found by Sanchez and Puigdefabregas (1994). Tussocks have been observed to be aligned with the contours of a slope (Sanchez and Puigdefabregas, 1994), and this may be the case in this study. At least two medians from the average heights for each aspect were significantly different ($p = 0.000$). The *M. drakensbergensis* tussocks in Figure 7.9 appear to be both influenced by the dominant wind direction and the slope. However, this requires a more in depth investigation (on both slopes and flat areas) before it may be confirmed. Tiller height may also be another indicator with the leaves appearing longer on the leeward side of the tussock (Wilson, 1959). In the study region the dominant wind direction is northwesterly (Sene *et al.*, 1998; Grab, 2010; see Chapter 4), hence the tiller heights may be greater on the southeastern sides of the tussocks. No significant differences were observed between the northerly and westerly aspects, as well as between the southerly and easterly aspects ($p>0.1$; see Table 7.5). This suggests that the dominant wind direction (northwesterly) may act to unify the average heights. The wind speeds were observed to gust at greater speeds in 2010 than in 2011 (maximum of $19.42ms^{-1}$ in May 2010 and $19.29ms^{-1}$ in June 2010) and was reflected in the average heights. The average leaf height from the peripheral tillers was observed to be

slightly greater on the east and south sides of the tussocks (Table 7.4). Conversely, the north and west tussock tiller average heights were shorter on the north and west sides of the studied tussocks (e.g. Figure 7.10). The only exception was observed in May 2009 and may be the result of increased grazing activity during this month.

7.4 Summary

Tussock shape alters with time and may be accelerated by burning and grazing activities. This study aimed to determine whether this was taking place within the study region and the main findings are as follows:

1. Differences were observed in the proportions of different shapes at each site suggesting differing *M. drakensbergensis* community ages and different factors, or factors of variable levels of influence, acting on the tussocks to alter the shapes.
2. The time since each site was last burnt was not quite significantly correlated to the percentage of type 'B' individuals, and may be attributed to delayed recovery of the central tissues.There were no significant relationships between shape and burn frequency.
3. Shortly after burning, increases in the percentage of type 'C' tussocks were observed yet decreased three months after burning. This was attributed to slower recovery of the central tissues likely to have experienced the most damage from burning.
4. The growth of small flowering plants (e.g. *Helichrysum* sp. and *Rhodohypoxis baurii*) and grasses within the ring structure was observed. This was also suggested to accelerate the rate of progression through the age states.
5. The accumulation of leaf litter within tussock structures was indirectly correlated to altitude. Potential slowing of growth rates and increased wind strength at high altitudes were indicated as potential causes.
6. No significant relationship was observed between shape and inflorescence number.
7. Average leaf height was greater on the south and eastern sides of the tussocks as opposed to the shorter average heights recorded on the north and western sides of the tussocks. This was attributed to the dominant wind direction (northwesterly) and its effects.
8. Soil movement via wind and trapping by the tussock allowing soil to accumulate in the central regions was confirmed as a possible cause of increased progression through the age states proposed by Gatsuk *et al.* (1980), while degradation due to toxicity resulting from decomposition and microbal action was rejected.

Chapter 8:

Annual Growth and Inflorescence Study



CHAPTER 8:

Annual Growth and Inflorescence Study

The baseline study aims to compare *M. drakensbergensis* communities forming the different sites with differing burn frequencies, which has the potential to indicate which communities are surviving at their optimum. The study also aims to determine whether a relationship exists between characteristics of *M. drakensbergensis* and the burn frequency and the time since each site was last burnt. Five main features were outlined for comparison: basal circumference, average height, basal area, standing biomass and the number of inflorescences produced. Basal circumference was measured as it may be used as an indicator of grazing intensity (e.g. Cole and Wilkins, 1958), hydrological stability (e.g. Morris *et al.*, 1989) and evaluating the potential for tussock damage and recovery (Wright and Klemmedson, 1965). Average growth was indicated through average height reached by the tillers collectively. Basal area is considered to be a more stable indicator than height as an indicator of survival, and thus was included. Standing biomass gives an indication of the dry weight of the study species within each site. Each of these factors will provide an indication of whether tussock communities at each of the sites are surviving at an optimum and whether differences exist between the communities at different locations. The results of the annual growth and inflorescence study are discussed in parts, namely; basal circumference, average height, living basal area, standing biomass, and the influence of burn frequency and time since the site was last burnt, are also included.

8.1 Methodology

A single study was carried out in early January 2009 as winter burns would have been completed and leaf growth sufficiently underway, given that it was mid growing season. Midsummer is also an appropriate time to measure the tussocks as more than 80% of the seasonal growth would have already taken place and the biomass expected to be at its peak (Gibson, 2009). The GIS map was used to select areas of different post-burn ages, within the study region. These sites were also selected on the basis that they needed to contain a representative sample of the dominant species found in the area, be of uniform habitat, and as homogenous as possible (Mueller-Dombois and Ellenberg, 2002). Each site spanned a single *M. drakensbergensis* community, thus the area covered by each site was determined by the community size and ranged in extent from 1.4m^2 to 0.1km^2. From these areas, measurements were innitialy collected using the random quadrat method as this method was best suited to the study region. The quadrat size used was 1m x 1m as

this was tested in the field to be the most appropriate size given the typical size of *M. drakensbergensis*. A minimum of 15 measurements and a maximum of 30 measurements of each variable studied were collected from each study site. The number of measurements made depended on the size of the site, weather conditions at the time, as well as the time duration of each fieldtrip. Site S.E. was the only exception as this site contained a small community comprising of five individuals, thus each individual was measured. The locations of the tufts were recorded using sketches and photographs. This enabled subsequent measurements to be undertaken using the same tufts. Measurement of basal circumference, average height, basal area and standing biomass were undertaken and the methods pertaining to each variable discussed in greater detail in the following paragraphs.

Basal measurements of tussocks may indicate grazing intensities, as has been found in Idaho Fescue (Cole and Wilkins, 1958). Grazing may increase tillering, resulting in increased basal area (Gibson, 2009). In addition, basal area is a good method for assessing the hydrological stability within an area (Morris *et al.*, 1989). Although many studies measure basal diameter (e.g. Wright and Klemmedson, 1965; Thurow *et al.*, 1988; Lee *et al.*, 1988, 1993; Brzosko, 2001), recording crown circumference has been recommended by Morgan and Lunt (1999), given the difficulty when measuring diameter due to the irregular shape of the tussocks. The basal circumference was measured using a flexible tape measure wrapped around the base of the crown approximately 1cm above the soil surface. Size is important to this study as sprouting ability is size dependent with the weakest ability found in smaller grasses (Bond and Van Wilgen, 1996). Thus, basal diameter provides a useful tool for evaluating the susceptibility of the tuft to damage and ability to recover (Wright and Klemmedson, 1965). For the same reason, average height was measured, as this provided information on the level of tussock recovery since burning. Average leaf height was defined as being the average height reached by the tillers collectively. Leaf height has been used in the past to indicate grass growth (Cole and Wilkins, 1958). Maximum height of the tillers within a tussock has been used by Lee *et al.* (1993) as a means of measuring the height. However, this method does not indicate the height of the majority of tillers. The average height was measured in centimeters to the nearest millimeter, as recommended by Guevara *et al.* (2002). Leaf blades were straightened out from half of the tuft, then measured and the average height calculated.

Tussock basal area is more stable than aerial cover (Hanson, 1961) and is influenced by longer term changes. This made it ideal for use within this study of post-fire recovery as it indicates

longer-term effects of burning, as well as the extent of the area covered in tussocks. Measurements of tuft circumference and height (and later leaf length) were measured in millimeters to improve accuracy. The percentage basal area of living and dead matter within the crown was observed in cm^2 with the aid of a string grid, placed above a tuft. The basal area was calculated in tall grasses by parting the leaves to reveal the top of the crown. The living proportion of the crown was calculated according to Wright and Klemmedson's (1965) method of measuring the basal area of the total crown minus that of the non-living proportion. This was also performed using the string grid in cm^2, and determines the proportion of living crown, and consequently the amount of possible vegetative sprouting that may occur after burning.

Standing biomass was calculated using the dry-weight-rank method as described by 't Mannetjie and Haydock (1963) and later by Ratliff and Frost (1990). This is a relatively quick and accurate method compared to commonly used harvest methods ('tMannetjie and Haydock, 1963; Ratliff and Frost, 1990; Dowhower *et al.*, 2001). The dry-weight-rank method is best suited in old, diverse grasslands containing species growing in patches (Neuteboom *et al.*, 1998). Although this method has not been applied in Lesotho, it has been tested in the Mpumalanga highveld where it was proven accurate, easy to use and cost effective (Barnes *et al.*, 1982). The dry-weight-rank method has also been successfully used on tussock grasses in northern Australia (Ash and McIvor, 1998). Quadrat size apparently has little influence on results but large quadrats may result in overestimations while smaller quadrats usually provide best results (Neuteboom *et al.*, 1998). Quadrats 1m x 1m were used as this size ensured that at least three species occurred in most quadrats, as suggested by Barnes *et al.* (1982). Estimates of the contributing dry-weight of species found within the quadrats were used to rank the dominant species into first, second and third place ('t Mannetjie and Haydock, 1963; Ratliff and Frost, 1990). This was performed for 30 quadrats at each site. As each site was limited in spatial extent, the recommended 100 quadrats (Barnes *et al*., 1982) could not be achieved. These were then tabulated to provide proportions which were multiplied by 70.19, 21.08 and 8.73 respectively ('tMannetjie and Haydock, 1963). The result was a dry-weight percentage of each species within each study site. New multipliers, as was developed by Jones and Hargreaves (1979), were not used as the difference is marginal. Similarly, correction of species occurring in clumps as developed by Haydock and Shaw (1975) were not used as it was these clumps which were the focus of the study. However, 'cumulative ranking' was used, as described by Ratliff and Frost (1990), given that *M. drakensbergensis* was consistently dominant at many sites. The baseline study was undertaken for three years (2009-2011) and supplemented with new sites each year, such as at Kotisephola Pass (December 2009)

which was at a higher altitude (3200m a.s.l.). To record the measurements and ensure no missing measurements, field sheets were designed; these can be seen in the appendices (Figure A.9).

8.1.1 Data Analysis

Any significant differences between the measured variables (basal circumference, average height, living basal area) from each of the sites would potentially indicate the presence of different magnitudes of influence acting on each site. This may suggest that the burn frequency and the time since last burnt have an influence on the obtained measurements of the variables. Using IBM SPSS version 21, a one-way analysis of variance (ANOVA) was used to determine whether the differences between the sites were significant each year using the raw data (Howell, 2004). The standard null hypothesis used was that the data from each site had the same mean (Townend, 2008; Dytham, 2011). Before the ANOVA was undertaken, the appropriate assumption tests were carried out: independence, homogeneity of variance (Levene's test using medians), normal distribution (Shapiro-Wilk test) and the non-presence of outliers (Puri, 2002; Townend, 2008). Where the conditions were not met, the data were appropriately corrected before the analysis was undertaken. The use of the medians to test for the homogeneity of variance occasionally meant that the Levene's test using means was significant, thus as a precaution the Brown-Forsythe and Welch procedures were performed. In order to limit the possibility of Type 1 error due to the large number of different comparisons being made, Tukey's HSD post-hoc test was performed for significant results at an Alpha value of 0.01. The measured basal circumference, average height, living basal area and standing biomass were tabulated to indicate the results and allow for any trends to be observed. Averages were used as individual data presented a large data set from which it was difficult to compare sites under different conditions. The standard deviation was calculated for each set of data to indicate the internal variability at each site.

A generalized linear model (GLM) was used to determine the best predictors of the measured variables (average height, basal circumference, living basal area and standing biomass). GLMs examine the ability of predictor variables to explain a dependent data set (Guisan *et al.*, 2002). The strength of the relationship and the contributions made by each predictor variable may also be determined (Guisan *et al.*, 2002). GLMs are able to analyze data which fall into any of several different distribution patterns and thus are useful for ecological data sets (Guisan *et al.*, 2002). The covariates used were the burn frequency values of each site and the time since each site was last burnt (these values were calculated in Chapter 6). The measured variables were explored and

the outliers were scaled down to reduce their influence, thus meeting the required assumptions. As determined for the ANOVA tests, the distribution was observed to be normal. The corrected data for each variable (average height, basal circumference, living basal area and standing biomass) were entered into STATA version 11. The multivariate GLM was performed, through entering the data as the dependent variable, and both the burn frequency and time since each site was last burnt as the independent variables for each year. The canonical link function used was identity. The GLM then performed a multiple regression as this allows for the presence of a linear relationship to be observed between a single variable and two or more independent variables (Townend, 2002). The generated results were tabulated with significant relationships indicated. Multicollinearity was not observed for any data sets due to the low number of predictor variables used. Scatterplots were generated to observe whether polynomial relationships were present and, using a polynomial regression, were found to be significant. In the case that they were, the data was presented and the scatterplots placed in the appendices.

8.1.2 Inflorescences

The number of inflorescences present on each measured tussock was included within the baseline study, and records taken during various field visits between 2009 and 2011. *M. drakensbergensis* is known to flower between October and March, thus the baseline study was conducted during early January, which represents the mid flowering season. Only the inflorescences present were recorded and not the number of inflorescence stalks as the stalks appear similar to the leaves leading to inaccurate results. Missing inflorescences may have been ingested by grazers or damaged through other abrasive agents. If the inflorescences were not fully matured before falling, their ability to successfully germinate may be significantly reduced. Thus, only the inflorescences found on the tussocks were recorded. Emerging or older inflorescences were not discriminated against, and were included within the study. It was assumed that the young (still green and unopened) inflorescences would mature with time and that the older inflorescences had already dispersed their seed. It is important to know the number of potential seedlings which may be able to germinate to prevent a potential decline in the species after an extreme defoliation event. However, the actual number of seeds produced by each inflorescence was not recorded due to time constraints. It was believed that the number of inflorescences would be a good representation of the number of seed produced during each growing season. The average number of inflorescences was calculated for each site.

8.1.3 Inflorescence data analysis

Significant differences between the inflorescence numbers observed at each site may suggest that the burn frequency and the time since each site was last burnt play a role in influencing seed production. This was most simply tested using an ANOVA, hence the assumption tests for independence, homogeneity of variance (Levene's test using medians), normal distribution (Shapiro-Wilk test) and the non-presence of outliers were undertaken. The results did not indicate that the data were distributed normally or had homogeneity of variances. Transforming the data failed to generate a normal distribution. Thus, the non-parametric alternative to a one-way ANOVA, the independent-samples Kruskal-Wallis test was performed using IBM SPSS version 21 (Townend, 2002; van Emden, 2008; Dytham, 2011). The possibility of Type I error is reduced using this method (Dytham, 2011). The null hypothesis (H_0) used was that the inflorescences from each site had the same median, and the alternative hypothesis (H_1) was that the medians were different between sites. The inflorescence data were tabulated to allow for any trends to be observed. Averages were used for the purpose of the table as individual data presented a large data set from which it was difficult to compare sites under different conditions. The standard deviation was calculated for each set of data to show trends in the data not shown by the average values.

Multivariate GLMs were used to ascertain whether burn frequency and the time since each site was burnt were good predictors of the number of inflorescences produced each year. The raw data was used as opposed to average numbers of inflorescence for each site. The GLMs were able to indicate the significance of the relationship as well as the role played by each variable (Guisan *et al.*, 2002). The GLMs were performed using STATA version 11 for each year of study. This allowed for differences in inflorescence production between the years to be included and discussed in greater detail.

8.1.4 Pearson's product moment correlations

In order to determine whether a relationship exists between the characteristics of *M. drakensbergensis* (i.e. basal circumference, average height, living basal area, standing biomass and inflorescences) Pearson's product moment correlations were performed using SPSS version 21. Pearson's correlations calculate whether a linear relationship exists between two variables and indicates the strength and direction of this relationship (Townend, 2002; Dytham, 2011). The raw data met the required assumptions (e.g. normal distributions); equal variances are not

required for a Pearson's correlation and therefore were not tested (Townend, 2002). The results of the correlations were tabulated and significant relationships indicated.

8.2 Results of the annual growth and inflorescence study

8.2.1 Results of the ANOVAs

The basal circumference measurements from each site were significantly different in 2009 (p = 0.015), 2010 (p = 0.007) and 2011 (p = 0.001) (Table8.1). Significant differences were observed between the average height measurements of the sites were also observed in 2009 (p< 0.001), 2010 (p< 0.001) and 2011 (p< 0.001). The high SS and MS values were attributed to the large variances within the data. The average height range was greatest at the V.F. site in 2009 when

Table 8.1: The results of the ANOVA tests: testing for differences for each tussock variable between sites.

Variable	Year		Df	SS	MS	F	P-value
Basal circumference	2009	Between groups	10	70801.915	7080.192	2.306	0.015*
		Within groups	144	442077.719	3069.984		
	2010	Between groups	11	67255.404	6114.128	2.488	0.007**
		Within groups	163	400577.674	2457.532		
	2011	Between groups	11	85271.029	7751.912	3.212	0.001†
		Within groups	148	357239.428	2413.780		
Average height	2009	Between groups	10	33987.377	3398.738	11.898	0.000†
		Within groups	108	30851.602	285.663		
	2010	Between groups	11	4957.702	450.700	3.997	0.000†
		Within groups	141	15899.018	112.759		
	2011	Between groups	11	7252.814	659.347	7.538	0.000†
		Within groups	142	12421.191	87.473		
Living basal area	2009	Between groups	10	7730.200	773.020	3.035	0.002*
		Within groups	99	25214.200	254.689		
	2010	Between groups	11	2964.875	269.534	1.506	0.135
		Within groups	143	25598.867	179.013		
	2011	Between groups	11	2581.843	234.713	2.976	0.001†
		Within groups	157	12382.217	78.868		

SS: sum of squares; Df: degrees of freedom; MS: mean square; F: the ratio of MS_{group} to MS_{error}; P: significance value.
*P<0.05; **P<0.01; ***P<0.005; †P<0.001

the height ranged between 31.9cm and 159.5cm. Statistically significant differences were observed between the living basal area measurements in 2009 ($p = 0.002$) and 2011 ($p = 0.001$). However, the difference was not significant in 2010 ($p = 0.135$). Tukeys post-hoc tests indicated that the Valley Floor site (V.F.) differed significantly from the majority of other sites regardless of the variable concerned. Sites with the same burn frequency (e.g. site S.S. and S.E.; V.S.N. and V.S.E; H.E. and H.W.) were generally not significantly different from each other. This adds support for the possibility that burn frequency is able to influency the growth of the *M. drakessbergensis* tussocks.

8.2.2 Basal Circumference

The mean basal circumference measurements confirm differences between the sites (Table 8.2). In 2009, site V.F. (154.0cm) has the greatest basal circumference, followed by site V.S.E. (126.7cm). Site V.F. retains the greatest basal circumference in 2010 and 2011 (150.65cm and 154.1cm respectively), followed by the upper M.F. site (125.5cm and 127.9cm respectively). No sites had basal circumferences less than 100cm, except for sites S.E., H.E. (82.8) and H.W. (92.8cm). The smallest average basal circumference was observed for the S.E. site (60.4cm), but is not surprising as only five individuals make up this community and one is a small individual (15.3cm basal circumference). In 2010, a Kotisephola Pass site was added and maintained an average basal circumference lower than 100cm (87.4cm).

Standard deviation is a good indication of the amount of variance within a data set (Rumsey, 2003) and is reflected in the coefficient of variance but is more comparable. The mean standard deviation of basal circumference across all sites was 50.4cm in 2009 and rose to 56.7cm in 2010, but then fell to 25.4cm in 2011. Comparatively low standard deviations were observed for sites S.E., V.S.N., V.S.E. and H.E. for both 2009 and 2010. This implies that circumferences were similar between individuals at each of these sites. In 2011, the Apostles plateau site had the lowest standard deviation at 16.2cm, while the highest standard deviation was recorded for the V.F. site (94.5cm). A high standard deviation implies high variation in basal diameters between individuals within this site. The highest standard deviation was recorded at site S.S. (89.1cm) during the previous year (2010).

The greatest mean growth between 2009 and 2010 was observed for the S.Str. site as a growth in basal circumference of 3.1cm/year. Positive growth was observed at sites S.E. (0.7cm/year), S.S.

Table 8.2: Statistical results for the measurements of basal circumference (2009-2011).

	Year	Sites												
		S.E.	**S.S.**	**V.S.N.**	**V.S.E.**	**S.Str.**	**H.E.**	**H.W.**	**V.F.**	**M.F. Up.**	**M.F. Lo.**	**A.P.**	**K.P.**	**All**
Basal circumference														
Mean (cm)	2009	60.4	113.0	106.0	126.8	114.0	82.8	92.8	154.3	122.7	122.2	102.5		108.9
n		5	15	15	15	15	15	15	15	15	15	15		15
SD		31.5	74.9	28.3	28.4	68.8	35.0	41.1	66.1	65.9	57.9	56.4		50.4
Highest value (cm)		97.8	309.1	146.9	172.9	263.5	138.0	189.9	266.0	243.9	242.0	328.0		328.0
Lowest value (cm)		13.9	28.1	59.9	79.6	39.0	31.2	45.2	49.6	42.0	42.4	29.4		13.9
Mean (cm)	2010	61.1	115.7	107.9	125	117.1	85.6	90.4	150.7	126	125	105.6	87.4	108.1
n		5	15	15	15	15	15	15	15	15	15	15	15	15
SD		32.2	89.1	27.3	35.8	79.5	27.5	37.3	32.0	53.4	44.8	87.4	55.6	56.7
Highest value (cm)		99.7	355.8	149.1	173.1	336.4	157.3	167.0	178.9	209.6	205.9	382.4	269.9	382.4
Lowest value (cm)		15.2	37.4	54.8	69.4	34.8	48.1	35.8	93.2	42.2	75.3	40.1	36.5	15.2
Mean (cm)	2011	61.5	116	109.9	128	116.3	85	88.5	154.1	128	127.6	101.5	84.2	108.4
n		4	15	15	15	15	15	15	15	15	15	15	15	15
SD		9.5	34.0	35.2	40.6	55.8	22.8	45.4	94.5	44.0	86.2	16.2	20.2	25.4
Highest value (cm)		68.0	193.2	175.9	177.2	273.8	116.8	219.4	350.0	203.1	330.1	122.6	132.3	350.0
Lowest value (cm)		16.8	79.6	53.2	71.8	41.6	41.7	47.8	54.1	58.1	32.4	89.4	47.1	16.8

(2.7cm/year), V.S.N. (1.9cm/year), H.E. (2.8cm/year) and at M.F. upper and lower (2.8cm/year). A decrease in basal circumference was recorded at site V.S.E. (-1.5cm/year), site H.W. (-2.4cm/year) and the greatest loss of 3.4cm/year at the V.F. site. Between 2010 and 2011, the highest average growth was recorded at the V.F. site (3.5cm/year). Growth was also recorded at sites S.E. (0.4cm/year), S.S. (0.3cm/year), V.S.N. (2.0cm/year), V.S.E. (2.41cm/year), M.F. upper (2.4cm/year) and M.F. lower (2.6cm/year). During this time, a large decline in average basal circumference was observed at the A.P. site where 4.0cm/year was lost. Decreases were also experienced at sites S.Str. (-0.8cm/year), H.E. (-0.6cm/year), H.W. (-1.9cm/year) and the K.P. site (-3.19cm/year).

8.2.3 Average Height

The average height across the sites is 41.6cm for 2009, drops to 38.3cm in the 2010 and drops again to 36.8cm in 2011 (Table 8.3). Sites V.F. (75.0cm) and H.W. (61.0cm) indicate the highest average height in the same year. The highest values for 2010 were recorded at site H.W. (51.1cm) and the V.F. site (49.1cm). In 2011, the highest average height was again recorded at the V.F. site (51.5cm), while the lowest average height was recorded for site S.S. (28.5cm). The highest individual average was recorded at the V.F. site (169.5cm) in 2009, but dropped to 126.8cm in 2010 for an individual at K.P. The smallest individual may be either a seedling or heavily grazed individual, and thus is not considered important within this section of the study; rather focus is given to site averages. Between 2009 and 2010, the greatest average growth of 12.0cm was recorded at the V.S.N. site. In contrast, the V.F. site experienced a dramatic loss of 25.9cm during the year, and site V.S.E. a loss of 16.4cm. The standard deviation for all individuals is 13.1cm for 2009 and drops to 8.6cm in 2010. However, large decreases in standard deviation were observed between the 2009 and 2010 measurements at sites S.S., V.S.E., H.W. and at the V.F. site. The standard deviation only increased marginally at sites V.S.N. (from 5.2cm to 6.5cm) and S.Str. (from 8.0cm to 9.0cm). The greatest positive change between 2010 and 2011 in growth was observed at the V.S.E. (9.3cm), and was lowest at the lower M.F. site (1.0cm). Negative growth was recorded at sites V.S.N. (3.0cm), S.Str. (4.3cm), M.F. upper (4.2cm), K. P. (3.6cm), and H.E. (17.6cm).

Table 8.3: Statistical results for the measurements of average height (2009-2011).

	Year	Sites												
		S.E.	**S.S.**	**V.S.N.**	**V.S.E.**	**S.Str.**	**H.E.**	**H.W.**	**V.F.**	**M.F. Up.**	**M.F. Lo.**	**A.P.**	**K.P.**	**All**
Mean (cm)	2009	31.2	29.1	29.3	53.7	34.3	32.6	61.0	75.0	41.4		28.3		41.6
n		4	15	15	15	15	15	15	15	15		15		15
SD		8.1	14.9	5.2	15.1	8.0	12.8	22.0	28.7	9.5		6.5		16.2
Highest value (cm)		42.7	65.0	36.0	76.4	46.1	54.2	98.2	169.5	59.2		38.7		169.5
Lowest value (cm)		16.5	15.0	21.5	25.7	18.2	12.5	29.7	33.7	28.1		20.4		12.5
Mean (cm)	2010	35.0	27.0	41.3	37.3	42.3	35.6	51.1	49.1	34.8	28.7	28.8	38.6	38.3
n		4	15	15	15	15	15	15	15	15	15	15	15	15
SD		7.3	7.3	6.5	6.5	9.0	6.9	7.7	11.4	6.5	1.8	4.9	22.1	8.6
Highest value (cm)		42.3	41.9	53.0	52.0	61.6	47.6	51.7	65.1	40.6	30.8	36.0	126.8	126.8
Lowest value (cm)		25.1	16.8	26.3	24.9	26.6	20.0	24.7	30.3	28.7	25.6	21.1	19.0	16.8
Mean (cm)	2011	36.6	28.5	38.3	46.6	38.0	40.8	33.5	51.5	30.6	29.7	32.5	35.0	36.8
n		4	15	15	15	15	15	15	15	15	15	15	15	15
SD		8.3	5.9	10.9	11.4	13.6	8.2	6.8	13.6	4.3	4.3	5.3	8.1	6.9
Highest value (cm)		42.4	42.6	57.0	58.5	55.6	53.2	46.2	68.2	38.7	37.2	39.3	52.6	68.2
Lowest value (cm)		24.7	18.8	13.4	22.7	12.7	27.8	22.6	21.9	24.4	21.2	25.1	19.1	12.7

8.2.4 Living Basal Area (LBA)

The mean proportion of living basal area across all the sites was 60% in 2009, 71.5% in 2010 and 70.7% in 2011 (Table 8.4). The general trend indicates increases in LBA for all sites, however not all sites indicate the same rate of increase. The basal area of site H.E. increased by 27% between 2009 and 2010, whilst site S.Str. similarly increased by 25%. The lowest increase was recorded at the Apostles plateau, where LBA only increased by 1%. Lower increases were generally observed between 2010 and 2011, with the greatest increase of 4% at site H.W. The lowest LBA increases were observed at the M.F. sites, where the lower site recorded a 0.1% growth and no growth at the upper site. Losses in living basal area between 2009 and 2010 wererecorded at site S.S. (3.3%), S.S. (0.8%), V.S.N. (2%), V.S.E. (2%), 3 (1.1%) and at site H.E. (2%). Losses between 2010 and 2011 were recorded at sites S.E. (0.3%), S.S. (1.6%), M.F. upper (5%), M.F. lower (5%) and at the A.P. site (2.7%).The lowest percentage of LBA was observed for site S.Str. (47%) in 2009, while the highest was 81% for the V.F. site in 2010 and 2011. The standard deviation across all sites drops from 15.8% in 2009 to 12.2% in 2010, and then drops further to 3.9% in 2011. Increases occurred for sites S.S. (from 15.7% to 19.3%), H.W. (from 13.8% to 22.8%) and M.F. sites (from 11.6% to 14.5%). The highest standard deviation was observed for site H.E. in 2009 (24.7%), while the lowest was 6.6% for the V.F. site in 2010.

8.2.5 Standing Biomass

The standing biomass presented in this section is only that for the *M. drakensbergensis* communies present at each site, and does not include other species. The biomass increased by 2.5% across all sites between 2010 and 2011, but decreases by 0.7% between 2009 and 2011. The percentage standing biomass for *M. drakensbergensis* was lowest at site S.E. (1.2% in 2009; 4.3% in 2010; 4.1% in 2011) as was expected due to the low number of individual tussocks (Figure 8.1). Most standing biomass measurements were greater than 30%. The highest percentage occurred at site S.Str. in 2009 (58.4%) and 2010 (50.1%), but in 2011 site S.Str. (53.1%) was exceeded by site V.S.N. (54.8%), which increased consistently throughout the study. Between 2009 and 2010, increases in standing biomass were recorded for sites S.S. (1.1%), S.E. (3.1%), V.F. (4.2%) and A.P. (5.5%), while the greatest increase was 8.2% at site V.S.N. Decreases in the dry weight of *M. drakensbergensis* were recorded for sites V.S.E. (3.1%), H.W. (5%) and M.F. (3.4%). The most significant decreases were observed for sites

Table 8.4: Statistical results for the measurements of living basal area (2009-2011).

	Year	Sites												
		S.E.	**S.S.**	**V.S.N.**	**V.S.E.**	**S.Str.**	**H.E.**	**H.W.**	**V.F.**	**M.F. Up.**	**M.F. Lo.**	**A.P.**	**K.P.**	**All**
Mean (%)	2009	54.2	62.2	67.0	67.0	47.0	49.0	59.5	74.0	53.0		67.5		60.0
n		4	15	15	15	15	15	15	15	15		15		15
SD		16.1	15.7	21.0	11.6	17.3	24.7	13.8	16.0	11.6		10.3		9.0
Highest value (%)		70	88	90	80	80	80	85	95	70		85		95
Lowest value (%)		20	31	30	40	20	10	40	50	30		55		10
Mean (%)	2010	72.0	70.7	73.3	72.7	72.0	76.0	63.0	81.0	68.5	67.9	68.5	68.3	71.5
n		4	15	15	15	15	15	15	15	15	15	15	15	15
SD		14.4	19.3	7.7	9.4	11.5	6.9	22.8	6.6	10.0	18.5	6.7	14.6	4.7
Highest value (%)		85	90	85	85	90	85	85	85	80	85	80	85	90
Lowest value (%)		50	35	55	45	50	65	15	65	45	30	55	35	15
Mean (cm)	2011	68.8	69.9	71.3	70.7	70.9	74.0	67.0	81.7	68.5	68.0	69.5	68.3	70.7
n		4	15	15	15	15	15	15	15	15	15	15	15	15
SD		4.8	8.8	9.3	7.5	8.7	8.7	7.0	9.2	10.0	10.7	8.2	8.4	3.9
Highest value (cm)		75	85	85	80	80	85	80	95	85	85	85	85	95
Lowest value (cm)		65	55	60	60	60	60	55	65	55	45	55	60	45

S.Str. and H.E., with decreases of 8.3% and 8.1% respectively. The change in standing biomass between 2010 and 2011 was greatest at site H.E. (14.1%) and lowest at site V.F. (1.7%).

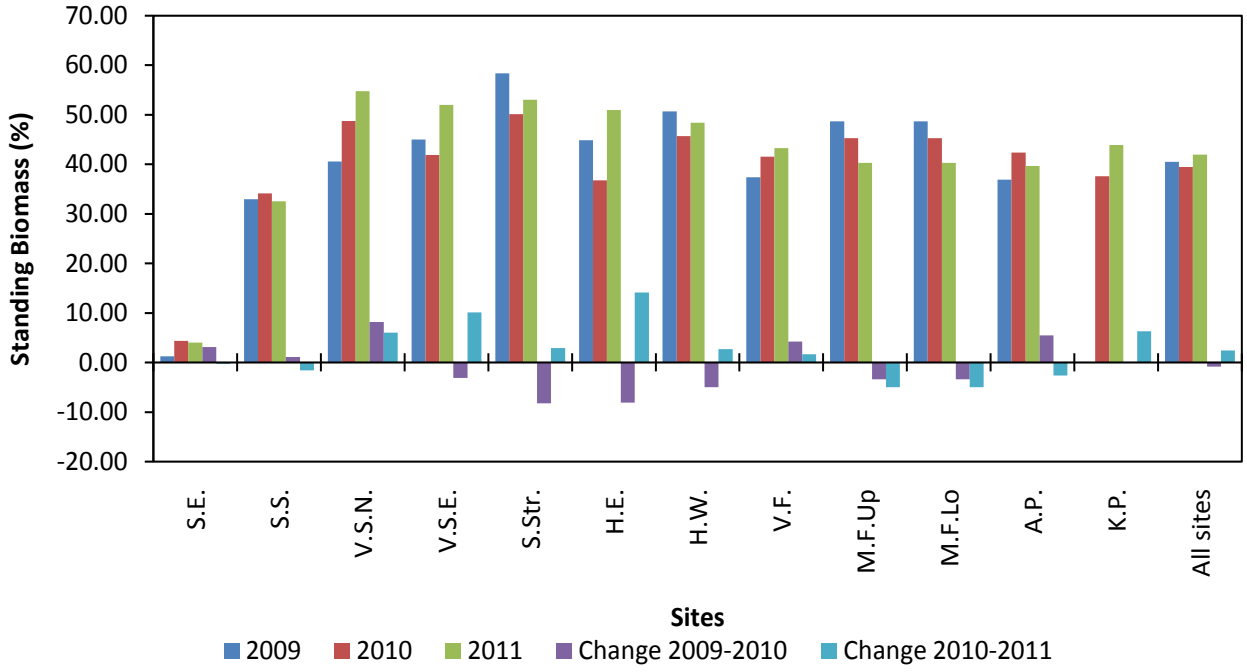


Figure 8.1: The standing biomass of *M. drakensbergensis* communities between 2009 and 2011.

8.2.6 Effects of burn frequency and the time since last burnt

The data were analyzed using GLM multiple regressions, using the measured variables (e.g. basal circumference, average height and living basal area), burn frequency and time since last burnt (Table 8.5). The GLMs using basal circumference as the dependent variable yielded a poor fit (R^2 = 12%) for all three years, but the overall relationship was extremely significant (F_{174} = 9.85, $p < 0.0001$ in 2009; F_{189} = 11.66, $p < 0.0001$ in 2010; F_{189} = 10.99, $p < 0.0001$ in 2011). The time since each site was last burnt ($t = 4.33$, $p < 0.0001$ in 2009; $t = 4.64$, $p < 0.0001$ in 2010; $t = 4.64$, $p < 0.0001$ in 2011) was more significantly associated than the burn frequency ($t = 2.92$, $p = 0.004$ in 2009; $t = 2.88$, $p = 0.005$ in 2010; $t = 3.18$, $p = 0.002$ in 2011), but both were very significant in the model. The burn frequency and time since each site was last burnt was an increasingly poor fit to average height for all three years of study (R^2 = 9% in 2009, 6% in 2010 and 4% in 2011). Despite this, the overall relationship was significant for all three years ($p = 0.01$ in 2009; $p = 0.01$ in 2010; $p = 0.02$ in 2011). There was no trend in contribution of each of the variables across the three years. The GLMs using living basal area were also a poor fit for all three years (R^2 = <10%), with the overall relationship very significant in 2009 ($p = 0.009$) and significant in 2011 ($p = 0.03$). Both the burn frequency and the time since last burnt contribute evenly in 2010 and

2011, yet only significantly in 2011 (p = 0.01). The standing biomass was a good fit to the burn frequency and time since each site was last burnt in 2009 (R^2 = 63%). However, the overall relationship is not quite significant in 2009 (F_{143} = 5.93, p = 0.03) and not significant in 2010 (F_{189} = 2.93, p >0.1) and 2011 (F_{189} = 1.36, p >0.1). The goodness of fit decreased in 2010 (R^2 = 42%) and in 2011 the independent variables only explained 25% of the standing biomass variation. The full equations are visible in the appendices (Table A.3).

Table 8.5: Generalized linear model multiple regression results depicting the influence of burn frequency and the time since last burnt (TSLB) on the measured variables.

Variable	Year	Factors	Coefficient (b)	R^2	SE	Significance (*p*-value)
Basal circumference	2009	Burn frequency	14.669	0.115	5.026	<0.0001†
		TSLB	9.114		2.104	
	2010	Burn frequency	13.006	0.118	4.515	<0.0001†
		TSLB	8.401		1.812	
	2011	Burn frequency	12.167	0.123	3.822	<0.0001†
		TSLB	7.264		1.565	
Average height	2009	Burn frequency	3.941	0.080	2.126	0.010**
		TSLB	2.697		0.882	
	2010	Burn frequency	1.326	0.061	0.439	0.009**
		TSLB	1.399		1.057	
	2011	Burn frequency	1.077	0.046	0.414	0.030*
		TSLB	2.411		1.007	
Living basal area	2009	Burn frequency	5.209	0.085	1.802	0.009**
		TSLB	1.197		0.780	
	2010	Burn frequency	2.151	0.020	1.267	0.211
		TSLB	0.864		0.519	
	2011	Burn frequency	2.188	0.042	0.862	0.029*
		TSLB	0.890		0.349	
Standing biomass	2009	Burn frequency	-6.491	0.629	3.002	0.031
		TSLB	0.172		1.332	
	2010	Burn frequency	-2.676	0.423	2.827	0.111
		TSLB	0.859		1.207	
	2011	Burn frequency	-4.279	0.252	3.859	0.313
		TSLB	-0.111		1.647	

*P<0.05; **P<0.01; ***P<0.001; †P<0.0001.

The direct relationship indicated by the coefficient with regards to burn frequency and average height was surprising and thus, scatter plots were produced. These indicated polynomial relationships for all three years of study (Table 8.6, Table A.4 and Figures A.10 to A.12). When a quadratic polynomial regression curve was fitted, the relationship between average height and burn frequency was not quite significant in 2010 ($F_{2, 150} = 2.82$, $p = 0.063$), but extremely significant in 2009 and 2011 (in 2009: $F_{2, 111} = 10.38$, $p<0.0001$; in 2011: $F_{2, 151} = 10.96$, $p<0.0001$). The quadratic trendlines indicated that the greatest average height was observed at a burn frequency of three years/decade. Similarly, the time since each site was last burnt also indicated polynomial relationships when compared to average height. The quadratic polynomial regressions were extremely significant in 2009 ($F_{2, 111} = 11.08$, $p<0.0001$) and 2011 ($F_{2, 151} = 12.43$, $p<0.0001$), and very significant in 2010 ($F_{2, 151} = 5.69$, $p = 0.004$). The equation indicated that the greatest average heights were observed at 7, 8 and 9.5 years after burning in 2009, 2010 and 2011 respectively. A polynomial regression comparing living basal area and burn frequency was extremely significant in 2009 ($F_{2, 111} = 9.61$, $p<0.0001$; Table 8.6, Table A.6 and Figures A.12 to A.15). The equation reached a maximum at a burn frequency of 3.5 years/decade. Similarly, when the time since last burnt was used the relationship was very significant in 2009 ($F_{2, 111} = 5.75$, $p = 0.004$) and significant in 2011($F_{2, 166} = 4.64$, $p = 0.011$). The quadratic equation reaches a maximum at 7 years since last burnt in 2009 and 9 years in 2011.

Table 8.6: Quadratic polynomial regression results for average height under differing burn frequencies and times since last burnt (TSLB).

Variable	Factor	Year	Coefficients b1	Coefficients b2	R^2	Constant	Significance (P-value)
Average height	Burn frequency	2009	27.723	-5.037	0.158	15.639	0.000†
		2010	6.282	-1.246	0.036	30.643	0.063
		2011	15.188	-2.581	0.127	19.886	0.000†
	TSLB	2009	15.244	-0.990	0.166	-2.679	0.000†
		2010	3.790	-0.197	0.071	22.071	0.004
		2011	9.421	-0.521	0.141	1.252	0.000†
Living basal area	Burn frequency	2009	20.797	-3.043	0.152	31.495	0.000†
		2010	5.326	-0.856	0.012	64.977	0.391
		2011	6.391	-1.053	0.033	63.487	0.063
	TSLB	2009	8.331	-6.649	0.097	40.145	0.004**
		2010	3.426	-0.211	0.017	60.298	0.267
		2011	5.011	-0.273	0.053	51.560	0.011*

*P<0.05; **P<0.01; ***P<0.001; †P<0.0001.

8.2.7 Inflorescence results

A Kruskal-Wallis test rejected the null hypothesis that the distribution of inflorescences was similar between sites ($p<0.05$ for all three years; Table 8.7). Thus, the alternative hypothesis was accepted that the medians were different between sites. Inflorescences were observed on most *Merxmuellera* tussocks at each site, however, the number observed for each individual differed (Figure 8.2 and 8.3). Overall, a greater percentage of individuals experienced no inflorescences in 2009 (16.5%) than 2010 (14.1%), and fewer inflorescences were observed in 2009 (mean = 9.9) than in 2010 (mean = 11.8) and in 2011 (mean = 13.1) (Table 8.8). Exceptions were observed for sites S.E., V.S.N. and V.S.E. where the average number of inflorescences dropped by 5.0, 4.4 and 18.7 inflorescences respectively between 2009 and 2010 (Figure 8.3). Site V.F. indicated the largest gain between 2009 and 2010, with an average increase of 20.6 inflorescences. An average increase of 7 inflorescences was measured for site S.S., whilst the remaining sites increased by less than 5 inflorescences between the 2009 and 2010 growing seasons. In 2011, the greatest increases were observed for site V.S.E. (10.5), despite a drop the previous growing season. Reduced numbers of inflorescences were observed at some sites: site S.Str. (4.4), upper M.F. site (4.4) and the A.P. site (1.5). The highest average number of inflorescences was observed for site V.S.E. (mean = 26.0 inflorescences) in 2009, site V.F. (mean = 33.1 inflorescences) in 2010 and again in 2011 (mean = 33.1 inflorescences). The lowest average number of inflorescences were observed for site S.Str. (mean = 2.2 inflorescences) in 2009, S.E. (mean = 4.5 inflorescences) in 2010 and 2011 (mean = 4.5 inflorescences).

Table 8.7: Results of the Kruskal-Wallis test for significant differences between the inflorescences observed at each of the sites (n = 225 in 2009; n = 255 in 2010 and 2011).

	n	df	χ^2	p-value
Inflorescences:				
2009	225	10	29.106	0.001†
2010	255	11	22.941	0.018*
2011	255	11	19.818	0.048*

Df: degrees of freedom
*P<0.05; **P<0.01; ***P<0.005; †P<0.001

Table 8.8: Statistical results for the inflorescence data (n = 30, 2009-2011).

Year	Site S.E.	Site S.S.	Site V.S.N.	Site V.S.E.	Site S.Str.	Site H.E.	Site H.W.	Site V.F.	Site M.F. Upper	Site M.F. Lower	Site A.P.	K. Pass	All Sites
2009													
Mean	9.5	4.2	12.7	25.7	2.2	9.4	4.2	12.5	9.3		8.7		9.8
Standard Deviation	10.08	8.02	12.23	19.33	2.35	8.76	3.99	19.19	8.43		6.43		6.69
Maximum	23	37	35	57	6	37	14	78	28		20		78
Minimum	0	0	0	5	0	0	0	0	0		0		0
% not flowering	25.0	15.0	30.0	0.0	30.0	15.0	15.0	15.0	13.3		6.7		16.5
2010													
Mean	4.5	11.3	8.3	7.0	17.2	9.8	8.9	33.1	11.0	10.1	11.0	9.4	11.1
Standard Deviation	3.11	9.78	6.21	3.81	12.86	1.78	9.58	30.83	8.27	22.19	7.57	7.05	7.72
Maximum	7	36	21	14	43	43	37	109	24	88	20	22	109
Minimum	0	1	0	3	1	0	0	0	1	0	2	0	0
% not flowering	25.0	0.0	13.3	0.0	0.0	6.7	6.7	6.7	0.0	26.7	0.0	6.7	14.1
2011													
Mean	7.0	13.4	11.2	17.5	12.8	10.1	9.3	28.7	11.9	14.6	9.5	11.7	13.1
Standard Deviation	5.10	11.30	5.64	12.98	11.28	8.02	6.95	24.27	8.48	22.35	8.21	11.65	11.35
Maximum	12	46	21	57	43	32	21	80	31	91	24	33	91
Minimum	0	0	2	1	2	0	0	0	1	0	0	0	0
% not flowering	25.0	10.0	0.0	0.0	0.0	5.0	15.0	5.0	0.0	20.0	15.0	10.0	8.8



Figure 8.2: *M. drakensbergensis* tussock from site V.S.E. showing the inflorescences.

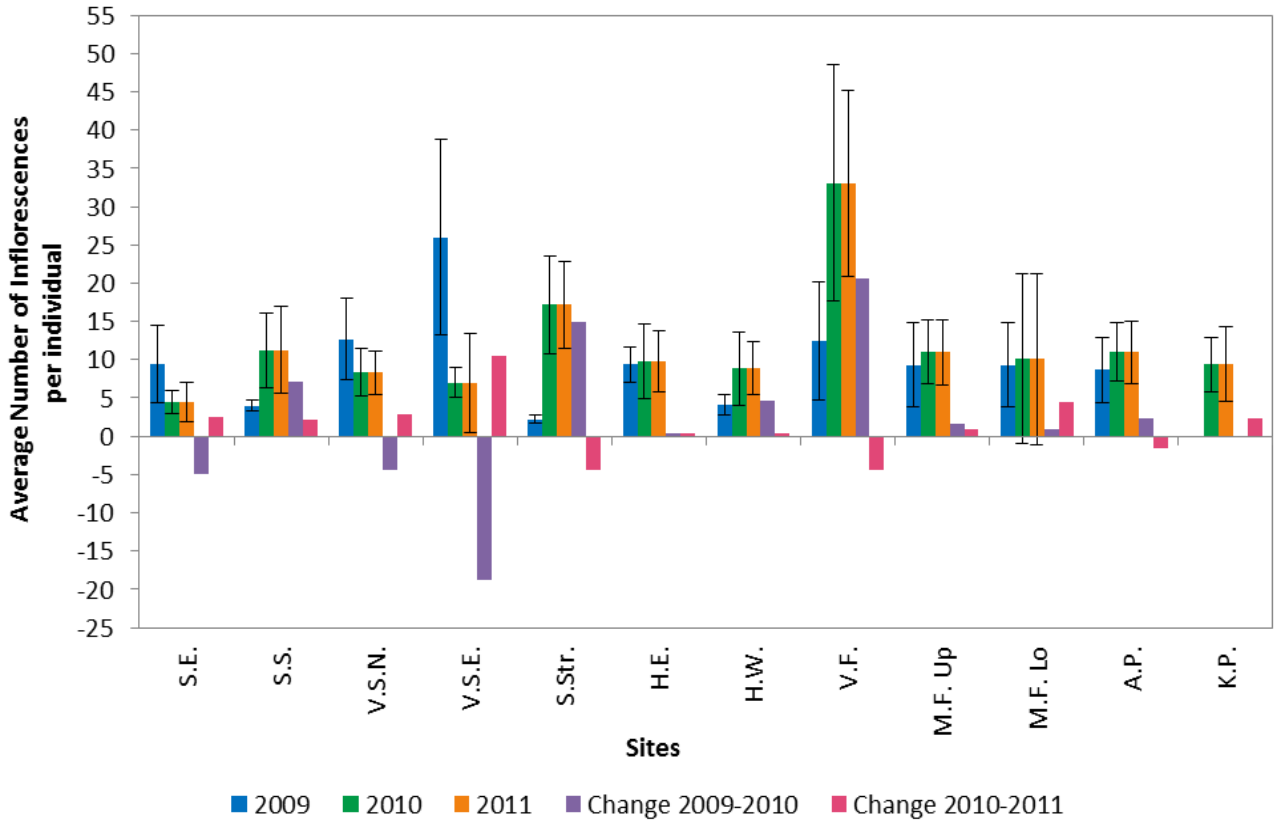


Figure 8.3: The average number and annual change of inflorescences per individual *M. drakensbergensis* tussock for each site, with standard deviation.

The standard deviation across all sites was 6.69 inflorescences in 2009, 7.72 in 2010 and 11.35 in 2011. The S.Str. site had an average of 2.2 inflorescences and the lowest standard deviation (2.35 inflorescences) in 2009. In contrast, site V.S.E. had a high standard deviation (19.33 inflorescences) in 2009 and the high variation is reflected in the maximum (57 inflorescences) and the minimum (5 inflorescences) number of inflorescences. The V.F. site followed a similar trend with a high deviation (19.19 inflorescences) values in 2009. However, in this case the mean number of inflorescences was lower (12.5 inflorescneces) than that of the other sites. In 2010, the highest standard deviation (30.8 inflorescences) was observed at site V.F., as was the case in 2009, yet, this time the lowest standard deviation was observed at site H.E. (1.78 inflorescences). However, this was not reflected in the minimum (0 inflorescences) and maximum (43 inflorescences) values.

A GLM was performed to establish whether an association existed between the number of inflorescences observed and both the burn frequency of each site and the time since each site was last burnt (Table 8.9, Table A.2 in the appendices). The multiple regression was a poor fit (R^2 = 6% in 2009 and 2010, 10% in 2011), but the overall relationship increased in significance during the three years of study. The relationship was very significant in 2009 ($F_{224} = 5.47$, $p = 0.005$) and 2010 ($F_{254} = 6.14$, $p = 0.003$). The overall relationship was extremely significant in 2011 ($F_{254} = 12.73$, $p < 0.0001$). In 2009, only the burn frequency was very significantly associated to the number of inflorescences ($t = 3.23$, $p = 0.002$). The following year, only the time since each site was last burnt was very significantly associated ($t = 3.14$, $p = 0.002$). In 2011, the time since each site was last burnt ($t = 3.48$, $p<0.001$) was more significantly associated than burn frequency ($t = 5.00$, $p<0.0001$), but both were very significant in the model.

Table 8.9: Generalized linear model multiple regression results depicting the influence of burn frequency and the time since last burnt on the number of inflorescences produced.

Variable	Year	Factors	Coeficient (b)	R^2	SE	Significance (P-value)
Inflorescence	2009	Burn frequency	2.765	0.060	0.856	0.005**
		TSLB	0.671		0.362	
	2010	Burn frequency	1.211	0.056	0.950	0.003**
		TSLB	1.241		0.396	
	2011	Burn frequency	3.316	0.103	0.954	<0.0001 †
		TSLB	1.926		0.385	

*P<0.05; **P<0.01; ***P<0.001; †P<0.0001

8.2.8 Pearson's product moment correlations

Pearson's product moment correlations were performed to determine the presence, strength and direction of linear relationships between the measured variables (Table 8.10). In 2009, average height ($p = 0.048$) and the standing biomass ($p = 0.033$) were significantly correlated to the number of inflorescences produced. The average height of *M.drakensbergensis* individuals correlated extremely significantly to the basal circumference in 2009 ($p<0.0001$). Average height also indicated a significant relationship with the percentage living basal area ($p = 0.02$). Living basal area significantly correlated to the standing biomass ($p = 0.024$). The relationships took on an indirect form when the number of inflorescences were correlated to the standing biomass ($r = -0.162$), and then the living basal area percentages were correlated to the standing biomass ($r = -0.022$). The following year (2010), average height was not significantly correlated to the number of inflorescences produced ($p = 0.05$). Yet, the average height was extremely significantly related to the living basal area data ($p<0.0001$). Standing biomass indicated the presence of significant relationships when correlated to average height and basal circumference ($p<0.05$). All the relationships were observed to have positive correlation coefficients. In 2011, average height was again significantly related to the number of inflorescences produced ($p = 0.024$). As in 2009, an extremely significant relationship was observed between average height and basal circumference ($p<0.0001$). A significant relationship was also found between living basal area and average height ($p = 0.023$). The relationship between average height and standing biomass increased in strength and was very significant in 2011 ($p = 0.002$). Again, the correlation coefficients indicated the presence of direct relationships between all the variables.

8.3 Discussion

The ANOVA confirms that significant differences existed between the sites for all the measured variables at a 95% confidence level (Table 8.1). The only exception was living basal area in 2010 ($p = 0.135$). The differences between the sites suggest that the different sites are subject to varying levels of influential environmental factors, such as burn frequency, time since each site was last burnt, soil moisture, temperature, wind, grazing etc.; these are discussed in connection with the correlations which were performed. The lack of a significant difference betweent the sites for the living basal area measurements in 2010 is not surprising due to the low standard deviation around the mean (Table 8.2). If the differences between sites are caused by variation in influencing factors, this would suggest that living basal area was influenced by a factor affecting all the sites similarly. The results of the ANOVA suggest that the differences are either internal

Table 8.10: Pearson's product moment correlations between the variables for 2009 and 2010.

2009		Inflorescences	Basal circumference	Average height	Living basal area
Basal circumference	Coefficient	0.077			
	P-value	0.380			
	N	131			
Average height	Coefficient	0.186	0.411		
	P-value	0.048	0.000		
	N	114	114		
Living basal area	Coefficient	0.089	0.143	0.227	
	P-value	0.365	0.145	0.020	
	N	105	105	104	
Standing biomass	Coefficient	-0.162	0.101	0.028	-0.220
	P-value	0.033	0.210	0.768	0.024
	N	173	155	114	105

2010		Inflorescences	Basal circumference	Average height	Living basal area
Basal circumference	Coefficient	0.148			
	P-value	0.069			
	N	153			
Average height	Coefficient	0.17	0.144		
	P-value	0.050	0.076		
	N	134	153		
Living basal area	Coefficient	0.068	-0.091	0.316	
	P-value	0.409	0.273	0.000	
	N	149	147	135	
Standing biomass	Coefficient	0.071	0.187	0.187	-0.037
	P-value	0.304	0.013	0.020	0.648
	N	210	177	153	155

2011		Inflorescences	Basal circumference	Average height	Living basal area
Basal circumference	Coefficient	0.047			
	P-value	0.555			
	N	159			
Average height	Coefficient	0.182	0.294		
	P-value	0.024	0.000		
	N	154	154		
Living basal area	Coefficient	0.147	0.015	0.184	
	P-value	0.056	0.850	0.023	
	N	169	154	154	
Standing biomass	Coefficient	0.057	0.041	0.252	0.097
	P-value	0.399	0.611	0.002	0.212
	N	224	160	154	169

*P<0.05; **P<0.01; ***P<0.001; †P<0.0001

conditions operating at varying levels at each site, or external factors which differ in intensity at each of the sites.

8.3.1 Basal circumference

During this study, an overall increase in basal circumference was recorded for most sites between 2009 and 2011. Basal circumference increases are the result of tiller production along the outer edges of the crown, balanced with a low rate of tiller deaths (Tsuyuzaki and Tsujii, 1992; Oliva *et al.*, 2005). Increasing circumference may be an indicator of outward expansion, and thus, transition through the growth stages as set out by Cole and Wilkins (1958), Gatsuk *et al*. (1980) and Rose and Platt (1990). Thus, the greater the rate of expansion, the greater the possibility that the *M.drakensbergensis* individuals within that site are experiencing shorter lifespans, and reproducing at a rapid rate through fragmentation. The highest recorded growth was measured at site S.Str. (3.1cm/year) and the A.P. site (3.1cm/year) between 2009 and 2010, and at site V.F. (3.5cm/year) between 2010 and 2011. There is uncertainty surrounding the causes of increasing basal circumference and many different factors have been proposed as a result of internal and external tussock conditions (e.g. Sanchez and Puigdefabregas, 1994; Danin and Orshan, 1995; Wan and Sosebee, 2000; Lewis *et al*., 2001; Feldman *et al*., 2007; Ravi *et al*., 2008), such as resource availability and room for expansion. However, more is known about tiller production, which is the foundation for increasing tussock diameter.

Surprisingly, the growth between 2010 and 2011 indicates a high number of sites experiencing losses, resulting in a low average growth for the year (3.35cm). Basal measurements, such as used in this study, were found to be useful indicators of grazing intensities in Idaho Fescue (Cole and Wilkins, 1958). More recently, grazing was observed to play a role in the size of tussocks, as found in a Patagonian study which demonstrated that the higher the grazing intensity, the smaller the basal area and general size (Oliva *et al*., 2005). An average measurement of 300cm^2 per plant was recorded for ungrazed populations while those exposed to heavy grazing only measured 167cm^2 (Oliva *et al*., 2005). The release of growth buds at the bases of tillers occurs as a result of grazing and stimulates the production of new tillers (Gibson, 2009), thus, basal diameter may also be positively influenced by grazing. Oliva *et al*. (2005) observed that reproduction through vegetative means was significantly greater when tussocks were grazed intensively, which suggests that basal circumference may not be a good indication of movement through the tussock life stages.

The variation decreases with time across most sites, except for those in close proximity to 'flowing water', such as site V.F. where the variation increases with time. The slow growth rate of site S.E. (3.8cm in 2011) may be explained by the high exposure of these individuals to wind, snow, hail and grazing. Each of these factors is likely to result in a reduced tiller length or a reduced rate of growth. Comparably, site S.E. was also consistently one of the driest (soil moisture in May was 21.7% in 2009 and 14.6% in 2011 samples) and rockiest sites studied, which would act to slow growth and tiller production further. Site V.F. had the largest basal circumference in January 2009 and 2011, and in 2010 had the second largest. In addition, this site also experienced the fastest growth rate between 2010 and 2011 (3.4cm). This site is east-facing and situated on the banks of a non-perennial stream, a factor increasing the soil moisture (see soil moisture probe results, Figure 4.11). Site H.W. had a consistently higher average basal circumference than site H.E., and the soil moisture was slightly lower at site H.W. (26.5% in 2009 and 28.8% in 2011) than at site H.E. (31.5% in 2009 and 47.6% in 2011 soil samples) when measured in May each year, perhaps the result of a slightly steeper slope. Grazing livestock also occurred less frequently in the *M. drakensbergensis* community of site H.E. than at site H.W., although both cattle and goats were grazing at both sites during the study. The A.P. site also experienced a fast growth rate during the year. The high growth rates of basal circumference suggest that factors necessary for optimum tillering are available at this site and others with high growth rates. Both of these sites, as well as sites V.F. and S.E., are situated on flatter gradients, suggesting that soil moisture may be preserved for longer than on the slope. Runoff from slopes above these sites would perhaps bring with it minerals and nutrients from the slopes above, suggesting higher soil nutritional contents at these sites.

The mean standard deviation values suggest that the basal circumferences of individual *M. drakensbergensis* tussocks did not all increase at the same rate within a site. The variation may be the result of differences in individual tussock microclimates, or differences in exposure to potentially harmful elements within their environment (such as the movement of livestock, wind etc.), as individuals within communities will be sheltered by those along the periphery. However, this was not the case for all sites, as comparatively low standard deviations were observed for sites S.E., V.S.N., V.S.E., H.E. for all years, suggesting a small extent of uniformity in the individuals of each site. This implies that circumferences were similar between individuals at each of these sites and increased at similar rates. Sites V.S.N. and V.S.E. both have the same time period since they were last burnt (7 years in 2011), and the low variation within the data may suggest that since being burnt, the measured tussocks have continued to grow at similar

rates. The differences in the growth rates since the last burning event could be a function of aspect and associated microclimates, as gradient and altitude were similar for both of these sites. The low standard deviation measured for site H.E. is interesting as site H.W. experienced the greatest rate of growth during the same period and had higher standard deviations. Both sites share similar altitudes, gradient, burn frequencies and time since the last burn. The only differences were aspect and, potentially, grazing frequency. The standard deviations for each site increased greatly between the years at sites S.Str., M.F. sites and the A.P. site. In all three of these sites *M. drakensbergensis* tussocks are separated by spaces often filled with small flowering plants, shrubs or invasive species (e.g. *Pentzia cooperi*). The result is an unequal exposure of individuals within these sites to influencing factors on outward expansion.

The GLMs performed indicate that the burn frequency and the time since each site was last burnt account for only 12% of the variation in the basal circumference data set. However, the overall relationship was extremely significant ($p < 0.0001$), indicating that the GLM is a good predictor of basal circumference. The time since each site was last burnt was more significantly associated with basal circumference than the burn frequency for all three years. There is a significant positive relationship between the time since each site was last burnt and the basal circumference values (i.e. as the time since burning increases so does the basal circumference). Similarly, but to a slightly lesser degree the burn frequency and the basal circumference values are positively related (i.e. as the burn frequency increases so does the basal circumference).

Cole and Wilkins (1958) presented evidence that prior to death from old age, tussocks have leaves surrounding dead centers or leaves situated in clumps on the edge. These authors further suggest that defoliation will accelerate the rate at which tussocks take on these two growth forms. Thus, increases in basal circumference linked to increases in burn frequency may indicate that the progression of age shape changes increases under increasing burn frequency. The results are contradicted by the findings of Snyman (2004b; 2005), who found that burning decreases basal cover in a semi-arid grassland near Bloemfontein, South Africa. However, the species and the location of study were different to those in the current study. This may also suggest that other environemental variables such as aspect and altitude may be stronger influences than that of burn frequency. Temperature and soil moisture are able to influence the rate and amount of tillering, and thus basal circumference, through the metabolic processes of the plants and the storage and use of resources within the plants (Gibson, 2009). Direct relationships are known to occur between rainfall and basal cover, with basal cover increasing as rainfall increases, and vice versa (Le Roux and Morris, 1977).

The diect relationship between the basal circumference and the time since each site was last burnt suggests that despite being burnt, increases in basal area follow normal growth and expansion patterns of those not burnt. The decreasing b-values (coefficients) indicate that the relationship decreases with increasing time, as was expected. West (1965, in Daubenmire, 1968), and Morgan and Lunt (1999) found that basal area decreased as litter accumulated with increasing time since burning. Manson *et al*. (2007) report a similar finding from a montane grassland at Cathedral Peak, South Africa. However, these studies were undertaken at lower altitudes and although investigated in grassland, also included other vegetation types. The results of the current study do not support that this is occurring at the higher Lesotho altitudes, possibly due to the slow growth rate.

8.3.2 Average height

According to Ellis (1981), Schmitz (1984), Killick (1990) and Pooley (2003), *M. drakensbergensis* may reach a maximum height of 1m, a height reached by only a few individuals at site V.F. in 2009 and a few at Kotisephola Pass in 2010. Across all the sites, the average height was 41.6cm, well below 1m. The success of *M. drakensbergensis* growth at some sites (V.F. and H.W.) suggests that the study valley is not unsuitable towards the tussocks reaching their maximum height, yet with no sites averaging close to 100cm, it may suggest that intensive grazing and/or burning are factors impacting negatively on grass height. The reduced average height across all sites from 2009 to 2011 may be owing to slow growth due to one or more factors including longer term climatic conditions than the data presented in this study and/or the continuous effects of livestock grazing and trampling, wind damage, hail, burning and disease. The temperature, precipitation and solar data gathered for this study period do not explain the stunted height.

The shortest individuals were observed at the same sites for two years: the A.P. (28.3cm in 2009; 28.8cm in 2010; 32.5cm in 2011) and S.S. site (29.1cm in 2009; 27.0cm in 2010; 28.5cm in 2011), and may be related to slow growth rates. The low growth rate observed at the A.P. site (0.6cm) suggests that grazing may have occurred for a longer duration of time, or at a greater intensity, prior to measuring, than that observed at site S.S., as the average at site S.S. still indicates a loss in leaf length. Between 2010 and 2011, the growth rates for both sites were positive (1.5cm at site S.S. and 3.7cm at the A.P. site) and are reflected in increased average heights.

The large number of sites with negative growth brought the average growth across all sites down to a loss in height of 3.3cm between 2009 and 2010. The loss in average height continues with a large number of sites experiencing negative growth between 2010 and 2011 resulting in a general decrease in average height of 0.7cm. As burning events and the tussocks affected have been excluded from this study, grazing is the most likely factor that can account for the loss of tussock height. Grazing is able to stunt the growth of tillers after grazing, resulting in temporarily shorter tillers, such as observed on the Murchison Mountains, New Zealand (Lee *et al*., 1988; Lee *et al*., 2000). This is especially true if grazing takes place towards the start of the growing season when stored resources are low (Trlica, 1999). Similarly, defoliation of too many leaves may draw too heavily on resources and result in a slower growth rate (Trlica, 1999). The New Zealand alpine tussock, *Chionochloa pallens*, has been observed to take approximately 15 years to recover to pre-defoliation heights after severe grazing (Lee *et al.*, 1988). It is suggested that severe grazing may slow growth and thus provide an explanation for some of the decreases in height observed (Wolfson, 1999). However, other factors also need to be considered such as the abrasive action of wind, hail and trampling, as well as disease and anthropogenic defoliation. Human use of *Merxmellera* tussocks in Lesotho usually involves the removal of entire plants or cutting off tillers at the crown, which is then used for thatching and manufacturing hats and brooms. As the studied tussocks were unevenly defoliated, it makes human interference an unlikely factor. Similarly, disease is unlikely and was not observed in any of the *Merxmuellera* communities. The action of hail and wind may break tillers but the extent to which this occurs is unknown. In 2010, the Sani region experienced stronger gusting winds between May and November than those of 2011 (Table 4.4). However, it is unlikely that wind is the sole cause of a loss of 25.9cm at site V.F., which is sheltered from the effects of wind.

The variation across all sites drops between 2009, 2010 and 2011 as indicated by the standard deviations (16.2cm, 8.6cm and 6.9cm respectively). This reflects the large decreases in standard deviation for sites S.S., V.S.E., H.W. and site V.F. Decreases in average height were also recorded for these sites, suggesting that the decrease in average height of the individual tussocks acted to bring most of the communities to a similar height. None of the sites decrease in height with an increase in standard deviation, indicating these two properties are linked. Both burning and grazing have similar defoliation effects and have the potential to reduce the studied tussocks to similar heights. However, burning, which did occur at some of these sites (S.S. and V.S.E.), was studied separately, and so does not explain the reduced variability in *M. drakensbergensis* height. Grazing may be a potential and likely alternative cause. Under moderate grazing it is

likely that the majority of the tussocks within a site will be grazed, thus decreasing the variability of height. The removal of grazing would allow for normal growth to occur, only impeded by variation in elements required for growth and exposure to harmful factors such as wind and hail. The results here suggest that the variation in micro-climate is significant enough to result in an increased variability of tussock heights. However, the climate data used in this study were not detailed enough (i.e. not site specific) to indicate whether this was a possible cause.

The GLMs indicate that the burn frequency and the time since each site was last burnt are an increasingly poor fit to the average *M. drakensbergensis* height across the years of study (R^2<9%). This may suggest that the effects of burning decrease with increasing time. Yet, the burn frequency and the time since last burnt do contribute significantly to the overall relationship for all three years of study ($p = 0.01$ in 2009 and 2010, $p = 0.02$ in 2011). However, there is no observable trend in the contributions made by each of the variables across the study period. This is perhaps the result of variation within grazing patterns each year. The coefficients (b-values) indicate that both relationships are direct. This implies that increases in average hight are accompanied by increases in the time since each site was burnt and is consistent with the literature. However, decreases are not uncommon as the tillers of some species are unable to tolerate shading from dead plant matter (Everson *et al.*, 1988a; O'Conner and Bredenkamp, 1997). Morgan and Lunt (1999) and Zimmermann *et al.* (2010) found that if left for longer periods (11 years without fire, in temperate Australian grassland and seven years in South African grassland), the aerial organs of the tussocks 'collapse'. Sites S.Str., M.F. and V.F. were last burnt at least 11, 11 and 9 years respectively, when the first fieldwork was undertaken in January 2009. As the tussocks in this study are not the same species as used by Morgan and Lunt (1999) or Zimmermann *et al*. (2010), the time frames of 11 or seven years to collapse may not apply here, especially as growth rates may be slow. Gitay *et al.* (1992) found that the time since a tussock was last burnt could not be distinguished using any tussock or tiller characteristics. This supports the lack of fit between the average height and the time since last burnt. However, it is known that it takes several years for a tussock or tussock community to recover to its pre-fire size (e.g. 15-20 years for grasslands in eastern Otago; Gitay *et al*., 1992).

The coefficients (b-values) also indicate that the average height increases with increasing burn frequency. This is not consistent with the literature; decreases in average height are associated with a decreased recovery time between burn events (Hofstede *et al.*,1995; Whelan, 1995; Yeates and Lee, 1997; Wahren *et al*., 2001a). The average height of tillers may remain reduced

for several years after burning (Yeates and Lee, 1997), thus the shorter the recovery period between burns, the shorter the tillers. The shortening of tall-grasslands to short-grasslands depicts the destruction of biomass as a result of burning and grazing (O'Connor *et al.*, 1997).

8.3.3 Living Basal area

A general increase in living basal area was observed across all sites (mean increase = 11.4% across all sites). An increase in living tissues would have manifested in the form of new tillers, which may be associated with the progression of change of shape with age (Gatsuk *et al.*, 1980; Derner *et al.*, 2004). An outward expansion of tussocks is an accepted theory and this is perhaps what has increased the living basal area of the studied tussocks. The growth of type 'A' tussocks into larger forms of type 'A' or progression into type 'AB' would increase the living basal area observed, thus explaining the increase in living basal area.

The proportion of living basal area is directly related to the shape and age state of the tussock (see Chapter 9). It is assumed that type 'A' tussocks will have near complete living basal area percentages, while the hollow type 'B' tussocks will have between ca. 40% and 60% living basal cover. This finding corresponds to 56% of live tillers observed for *Festuca orthophylla* in the Bolivian Altiplano (Monteiro *et al.*, 2011). Fragmented tussocks such as those termed type 'C' tussocks will have low (>40%) living basal areas. This proved true for all sites when using the 2010 mean basal area percentages (see Figure 9.2, Chapter 9). The 2009 mean values followed the same trend but percentages were generally slightly lower than in the 2010 study. Thus, tussock shape and percentage living basal area seem related. The majority of *M. drakensbergensis* tussocks within the sites are type 'A' and 'AB', thus losses due to change in age state may be slow or countered by outward growth. The slightly lower standing biomass may also be a result of a potentially slightly longer growing season as grazing livestock may be present for longer periods (see Chapter 5). The length of the growing season may vary due to variation in the onset of spring growth and winter dormancy (Morris *et al.*, 1989); the biomass will thus vary with growing season. The growing season was observed to only start in November at Sani top during the summer of 1988/1989 (Morris *et al.*, 1989). However, greening of the Sani region occurs in early October, in response to seasonal rainfall and warming. Low intensity grazing may also cause an increase in tiller production (Oliva *et al*., 2005; Patton *et al*., 2007), increasing the size and density of the living area of the tussock crown.

Sites H.E. and S.Str. indicate the greatest increase in living basal area between 2009 and 2010 (27% and 25% respectively). Both sites are located near streams, yet if this was the cause, site H.W. and site V.F. would also have high increases in living basal area. However, in the following year (2009) both site H.W. and V.F. had higher percentages of living basal area in (59.5% and 74.0% respectively) than those at sites H.E. and 3 (49.0% and 47.0% respectively). Site S.E., the driest site (21.7% in 2009; 14.6% in 2011 soil samples), also had a high increase in living basal area (17.8%), suggesting that moisture may not be the cause and is supported by the 2011 data.

The standard deviation across all sites drops from 15.8% in 2009 to 12.2% in 2010, and decreases further in 2011 to 3.9%. This may be the result of an increased time since each site was last burnt, allowing more time for the measured tussocks to recover. Tussocks do not experience equal damage during a fire but are dependent on the litter surrounding each tussock, soil content, crown moisture, wind etc., and thus may require different recovery times. Burning may also be patchy as some herdsmen may only burn a few clumps resulting in different burn histories within the same site. Thus, with time since burning, the variation in results should decrease slightly. However, it is questionable whether this can account for the measured decrease in the standard deviation. Within a community, the tussocks do not all share the exact same life stage, and the amount and location of living tissue will vary with tussock shape and life stage. The standard deviation may be an indication of the variation of life stages within each site. The greater the percentage of living tissue, the lower the age-state or the more normal the structure (i.e. type A), according to Cole and Wilkins (1958) and Gatsuk *et al.* (1980). Thus, the higher the percentage of living basal area, the younger the community. Yet, when this was briefly tested using a Pearson correlation coefficient and with findings from Chapter 9, the results were not significant ($r = 0.0823$ and $p = 0.8212$ for 2009; $r = 0.3878$ and $p = 0.2129$ in 2010).

The GLM analyses indicated that the burn frequency and the time since each site was last burnt are a poor fit to the percentage living basal area data for all three years (R^2 <10%). However, the overall combined relationship was significant in 2009 ($p = 0.009$) and 2011 ($p = 0.01$). Both the burn frequency and the time since each site was last burnt contribute significantly and evenly to the living basal area data in 2011. The results of the GLMs suggest that the living basal area is poorly and not consistently associated to burn frequency and the time since burning. The coefficients indicate that as the burn frequency increases, so the living basal area will also increase. This supports the literature in that burning may stimulate growth through the removal

of litter and increased water availability, light, nutrients and water (Mark, 1994; Bond and Van Wilgen, 1996; Morgan, 1999). However, this may be more effective for smaller individuals rather than for a community as a whole (Snyman, 2004a). Larger tussocks are able to accumulate more fuel, allowing them to burn for longer periods and at hotter temperatures, resulting in more damage to tissues and thus to living basal area (Miller, 2000; Peláez *et al*., 1999; Wright and Bailey, 1982). In contrast, other studies have reported long recovery periods (approximately 14 years) for larger tussocks to recover to pre-burn sizes (e.g. South Island, New Zealand by Payton and Mark, 1979). Thus, the more frequent the burning, the less recovery time is available for the tussocks to resume a pre-burn biomass, implying that the tussocks may be unable to recover their reserves before being burnt again, thus reducing the capacity to re-grow.

The coefficients (b-values) indicate that as the time since burning increases the basal area will also increase. However, the increases are small: for every year following burning the basal area increased by approximately 1%. This is not surprising as the meristematic tissues destroyed during a burning event are often not regenerated, thus not influencing the changes in living basal area with time. The research by Morgan and Lunt (1999) in Australian grasslands suggests that a decrease in tiller numbers and thus a loss in living basal area may be due to suffocation if the times since the area was last burnt are too great. The lack of a decrease in living basal area suggests that this is not the case within the study site. Further, the slower growth rate would result in a slower change in the growth stages (e.g. Gatsuk *et al*., 1980).

8.3.4 Standing biomass

The dry-weight-rank method proposed by ‘t Mannetjie and Haydock (1963) proved to be a quick and easy method to use in order to determine standing biomass. Similarly, Neuteboom *et al.* (1998) and Dowhower *et al.* (2001) found this method easier and faster, with more agreement achieved between the researcher and occasional aides using ranks, rather than estimates of cover. The method was also significantly less destructive to the *Merxmuellera* communities than the harvest method. Similar to the Tobosagrass (*Pleuraphis mutica* Buckl.) in Texas, *Merxmuellera* species carry some standing biomass from the previous year over to the following year, usually in the form of dead plant matter (Ansley *et al*., 2008). Consequently, biomass assessments performed by the dry-weight-rank method may be slightly over estimated. This method also required practice and continuity between sites, which was difficult to maintain across years.

The standing biomass of all the measured *M. drakensbergensis* communities was greater than 30% (apart from site S.E. in which a low result was expected each year due to the low number of individuals). The result is unexpected considering that the standing biomass was calculated within *Merxmuellera* communities. The inter-tussock spaces may be up to 100cm in some cases and many are occupied with other types of vegetation such as various *Helichrysum* species. However, this result is not surprising when taking Herbst and Robert's (1974) study into account, where *M. disticha* (then *Danthonia disticha*) comprised 27.9% of the area surrounding the Madibamatso (Oxbow) dam in the Lesotho highlands. *M. drakensbergensis* was a less dominant species, only occupying 3.7% (Herbst and Roberts, 1974). In a later study, *M. disticha* accounted for a maximum of 22.7% of the total vegetation, while *M. drakensbergensis* accounted for a maximum of 4.7% (Morris *et al*., 1993). However, neither of these studies focused solely on *Merxmuellera* communities. In the study region, the abundance of these species appears to be reversed, in which case the results are highly plausible. The highest percentage (58.4% in 2009; 50.1% in 2010; 53.1% in 2011) occurred at site S.Str. where low species diversity and the smallest inter-tussock spaces were observed.

Increases in standing biomass between 2009 and 2010 were recorded for sites S.E. (3.1%), V.F. (4.2%) and A.P. (5.5%), while the greatest increase was 8.2% at site V.S.N. In 2011, increases were observed at sites V.S.N. (6.1%), V.S.E. (10.1%), S.Str. (2.9%), H.E. (14.1%), H.W. (2.7%), V.F. (1.7%) and the K.P. site (6.3%). Increases in the standing biomass of the measured tussocks suggest an increase in the number of individuals and/or in the size of existing individuals. Site S.E. indicated an increase in basal circumference, average height and living basal area as well as in the number of individuals. The increase at site V.F. provides a surprising result as it did not indicate increases in basal circumference and average height. Personal observations did not support the theory that the number of individuals increased significantly. However, the living basal area did increase and may be a reflection of the progressive change in age state.

Decreases in *M. drakensbergensis* community standing biomass were observed at some sites during the course of the study. In 2011, a slight loss in standing biomass was observed (1.6%), but was linked to an overall decrease in living basal area. The most significant decreases were observed for sites S.Str. and H.E. (8.3% and 8.1% respectively) between 2009 and 2010, with smaller decreases also experienced at sites V.S.E. (3.1%), H.W. (5%) and M.F. (3.4%). The average decrease across all sites was 0.7%. The following year, 2011, was more positive with most sites indicating increases in standing biomass, but losses were still recorded for some sites

(e.g. S.E., the A.P. and the M.F. sites). The decreases, although small, may be the result of defoliation in the form of grazing. In New Zealand, clipping, used to simulate grazing, was observed to decrease the biomass of tussocks (Lee *et al.,* 1988). Additionally, *C. pallens* took nearly 30 years to recover its biomass after a single severe defoliation event (Lee *et al*., 2000). Although *Merxmullera* tussocks are not highly palatable, the surrounding sod grasses are frequently overgrazed and livestock have been observed to graze on tussocks. The decrease in standing biomass suggests that grazing at the current rate may not be appropriate to sustain the growth and maintenance of *Merxmuellera* tussocks in the long term. Measurements of biomass in the contect of this study are measurements of the standing crop, the measurement of biomass at a point in time (Martin and Hine, 2012). The decreases observed are thus reductions in the standing crop and may not be reflections of the productivity of *M. drakensbergensis* communities and individuals. Thus, despite consistent productivity, single severe defoliation events or consistent low levels of grazing may reduce the resulting standing crop measured. This suggests that the tussocks are not able to recover to the same biomass as recorded in previous years. Other causes may also be possible, such as the introduction of invasive species (e.g. *P. cooperii and C. cilliata*). These species are able to increase the competition for already limited resources and may be better adapted for faster growth within the alpine areas. The decrease in biomass may also be the result of a buildup of too much biomass, particularly leaf litter which is able to smother emerging new tillers. Some areas of the grassland had not been substantially burned for 14 years by the end of this study (2012) and this has led to increasing leaf litter, perhaps acting to smother new tillers and reduce the measured biomass. The findings of Briggs and Knapp (1995) support this observation, as they found that regularly burnt plots had a higher biomass than consistently unburnt plots.

The GLM analyses indicated that the burn frequency and the time since each site was last burnt were a good fit to the standing biomass data, but the fit decreases with time (R^2 = 63% in 2009, 42% in 2010 and 26% in 2011). This suggests the influence of burning decreases with increasing time. Supporting this, the overall relationship was only significant in 2009 ($p = 0.03$). Singularly, neither burn frequency nor the times since burnt are significant predictors of the standing biomass. This suggests that the role played by grazing in influencing the standing biomass of *M. drakensbergensis* may be greater than that of burning in the study valley. All three coefficients (b-values) were negative, indicating an indirect relationship in which the dry weight percentage of *M. drakenbergensis* increases when the burn frequency decreases. The survival of a tussock is often dependent on its size, which is dependent on the time available between burning events

(Bond and van Wilgen, 1996). The greater the time available for growth, the larger the tussocks are able to become and thus are better able to withstand burning. However, in cool environments such as at high altitudes in Lesotho, growth is slow and thus the longer the time available for growth between burns, the greater the biomass will be. The findings suggest that as time progresses, the relationship between standing biomass and burn frequency decreases. This is not surprising as the literature suggests that as time after burning increases, so the vegetation recovers to its pre-burn 'climax' state (e.g. Whelan, 1995).

The standing biomass is indicated to increase as the time since each burning event increases. Burning is a form of defoliation where tillers may be removed, thus reducing the standing biomass of *M. drakensbergensis*. As the crown resprouts, the biomass increases until the tussock reaches a maximum or another defoliation event occurs (Bond and van Wilgen, 1996). An initial increase in productivity frequently occurs in grasses and growth slows as time since burning increases (Daubenmire, 1968; Whelan, 1995). After burning, biomass requires the longest time to recover of all tussock factors (Snyman, 2004a), especially in alpine areas due to slow growth rates (Mark, 1994). The biomass of alpine tussock *C. rigida* has been observed to require more than 14 years to recover to pre-burn levels on South Island, New Zealand (Payton and Mark, 1979). Burning may also slightly reduce the numbers of individuals, thus negatively affecting standing biomass in the growing season following burning, until individuals may be replaced (Bond and van Wilgen, 1996), thus concurring with the literature.

If the time since the area was last burnt is too great, the number of tillers (and biomass) may decrease. This finding was attributed to 'collapse' of the tussocks as a result of smothering of new shoots, with the amount of dead litter observed to increase at the same time (Morgan and Lunt, 1999; Zimmermann *et al*., 2010). Using the dry-weight-rank method may result in an over estimation of the living biomass due to the presence of dead matter surrounding the individuals. However, through observation no sites contained great amounts of leaf litter. This suggests that the time since the area was last burnt had not reached a critical point for the studied tussocks. Differences in biomass, as well as in N, K, Ca and P, were observed in green shoots at different positions by O'Connor *et al.* (1997) in New Zealand, which could not be explained by soil or species differences. The study suggested that differences in fire frequency, topography, history and localized grazing explained the results. Thus, burning may have long term influences on biomass variation within an area. Consequently, biomass can be used as an indicator of recovery after burning as well as an indication of the intensity and the rate of spread of the next fire.

8.4 Discussion of the Inflorescences Results

During the course of a year, the hills and valleys of the Lesotho highlands are transformed from the dull brown of winter, to the lush green of spring and early summer, to the yellow of summer. The yellow flowers of the many common *Heilichrysum* species act to transform the landscape into yellow and this is added to by the pale yellowish brown of the *Merxmuellera* inflorescences (Figure 8.4 and 8.5). Despite their increased visibility, very little is known about the reproduction of the study species and tussock species in general. The causes of mast seeding and roles played by influential factors in the amount of seed produced remain mostly unknown (Rees *et al*., 2002; Hay *et al.*, 2008).



Figure 8.4: The study valley with yellow patches of *Helichrysum* sp. in the background and with *Merxmuellera* tussocks abundant with inflorescences in the foreground.

A Kruskal-Wallis test demonstrated that the difference in inflorescence numbers between each of the sites was significant for all three years in which data were collected ($p = 0.001$ in 2009, $p = 0.018$ in 2010 and $p = 0.048$ in 2011). The results agree with those of Mark (1964), who found that inflorescences of *Chionochloa rigida* in New Zealand vary greatly between sampled sites. This indicates that there are different factors or varying levels of factors acting on each site to produce significant differences in the average number of inflorescences between each site. This assists the hypothesis that the burning frequency and the time since last burnt have an influence on the production of seed.



Figure 8.5: Single inflorescence in late December 2010.

The average number of inflorescences increases from 2009 (mean = 9.85; Table 8.9) to 2010 (mean = 11.13). In 2011, the average number of inflorescences rose again to a mean of 13.1 inflorescences. Many factors have been proposed to influence the number of inflorescences. An early study suggested that the environmental conditions at the start of a growing season were the most influential factors on inflorescence production later in the growing season, as observed in Illinois prairie (Old, 1969). Lord (1998) suggests that a trade-off exists between the number of inflorescences produced and growth, especially in areas of reduced resource availability, a theory supported by the earlier findings of Mark (1965). The initiation of a *C. rigida* inflorescence begins in autumn and the partially developed inflorescence overwinter until conditions are suitable for it to complete its growth (Connor, 1966; Mark, 1964, 1968, 1970). This is a process common in many alpine species as an adaptation to the short growing season (Billings, 1974; Heide, 1997). Snow melt is often directly linked to the onset of warmer spring temperatures, which may have a role in both the initiation and number of inflorescences produced.

An increase in temperature from 9^{o}C to 21^{o}C in the initial phases of inflorescence production in a laboratory study resulted in a tripling in the number of inflorescences of *Carex brunnescens* (Heide, 1997). However, an increase in the number of inflorescences and temperature may not be beneficial, as a study on Alaskan tussock tundra observed a decrease in the developed and robust seed produced under conditions of increasing temperature (Hobbie *et al*., 1999). Yet, under

warmer conditions, more inflorescences were produced, supporting the earlier findings of Heide (1997). More recently, warm air temperatures have been linked to increased production of the plant hormone, gibberellin (Penfold, 2008; Turnbull *et al.*, 2012). The increased gibberellin levels produce increased inflorescence numbers in both the season of warm temperatures and in the following season (Turnbull *et al.*, 2012). In the Sani region, the mean air temperatures were observed to increase from 7.7^{o}C in 2009 to 8.9^{o}C in 2010 and 2011 for September. Hence, the small observed rise in temperature may explain the increase in mean inflorescence numbers. However, the mean inflorescence data also correspond to an increasing percentage of hours below 0 ^{o}C (1.21% in 2009, 5.78% in 2010 and 11.02% in 2011) and below 2 ^{o}C (6.32% in 2009, 11.16% in 2010 and 20.97% in 2011) in October each year. The increasing percentages of hours below growth thresholds (see Chapter 4) suggest growth may be reduced over the study period, yet more inflorescences are observed. Thus, the increasing air temperatures may be more influencial on inflorescence numbers than the growth rate.

Lord (1998) argues that the number of inflorescences produced in one year may influence the number produced in the following year, as observed on South Island, New Zealand. The production of inflorescences requires the use of both stored and available resources, which are potential limiting factors to the number of inflorescences which can be produced (Mark and Chapin III, 1989; Rees *et al*., 2002; Kelly and Sork, 2002; Hay *et al*., 2008). Thus, a large number of inflorescences in one year may result in a decrease in the number of inflorescences produced the following year as a result of a decrease in stored resources (Lord, 1998). This may explain the changes measured at sites V.S.N. and V.S.E., and may explain why a low number of inflorescences observed at some sites was followed by an increase the next year (e.g. S.S., S.Str., V.F. and to a lesser extent H.W., M.F. and along the A.P. site). A year of low inflorescence production would allow for an increase in stored resources allowing for an increase in flowering the following growing season.

Inflorescences were observed on most tussocks with varying numbers on each. Again, this result agrees with those of Mark (1964) from the New Zealand tussock, *Chionochloa rigida*, which also had varying numbers of inflorescences between individuals. The standard deviation highlights the variation and permits comparison between sites. In 2009, site S.Str. had the lowest standard deviation, implying that most of the individuals had similar numbers of inflorescences. In the same year, as well as in 2010, sites V.S.E. and V.F. had very high variations in inflorescence numbers on individual tussocks. This suggests that the mechanisms determining

the ability of an individual tussock to produce inflorescences differs between the locations of individual tussocks. Site H.E. was observed to be grazed more frequently than site H.W. (personal observations) and this may explain the difference in variation between the two sites. The lower M.F. site indicated more variation than the upper M.F. site for both 2010 and 2011 (Table 8.8), and is perhaps explained by differences in soil moisture. Soil moisture may limit the reproductive ability of plants (Craine *et al*., 2012). However, the soil moisture data indicate only marginal differences (Site M.F. upper: 17% and site M.F. lower: 18%). Yet, the moisture probe data indicate that the upper M.F. site was slightly wetter in May (site M.F. upper: 35% and site M.F. lower: 31%). It is unlikely that the marginal differences are responsible for the differences between the sites. Both sites were observed to be saturated during mid summer.

The GLMs indicated that both the burn frequency and the time since each site was last burnt were a poor fit to the inflorescence data ($R^2 = \leq 10\%$). Small increases in the ability of the predictor variables to account for the variation in the inflorescence data are reflected in the increasing significance of the relationship ($p = 0.005$ in 2009, 0.003 in 2010 and <0.0001 in 2011). In 2010 and 2011, the time since each site was burnt contributed more than the burn frequency, yet in 2009 the reverse occured. The coefficients indicate the relationship between the burn frequency and the number of inflorescences produced is direct (i.e. as the burn frequency increases, the number of inflorescences increases). However, an increase in burn frequency by a single burn in a decade may only increase the inflorescences produced between approximately 1 and 4 inflorescences per individual. The increase in the number of inflorescences may possibly be the result of increased crown temperatures due to increasing loss of shading biomass. Burning has also been observed to increase the available nutritional content of the soil (Whelan, 1995), providing more resources which may result in increased numbers of inflorescences. A decrease in inflorescence production was expected as a result of repeated damage to meristematic tissues within the crown as a result of the high temperatures experienced (Whelan, 1995; Peláez *et al*., 1997; Miller, 2000). Meristemtic tissues or growth points extensively damaged by burning may not recover, thus resulting in an increasing loss of tissue with the ability to produce inflorescences each time the tussocks are burnt (Richards and Cadwell, 1985). Similarly, the greater the burning frequency, the less time the tussocks have to recover and replenish stored nutrients, which are vital for inflorescence production.

The GLM indicates that as time increases between burn events, the number of inflorescences also increases. This allows for the time required for the recovery of the *M. drakensbergensis*

tussocks and production of new tissues. Flowering may have an initial burst of increased numbers of inflorescences in the growing season following burning but then the number of inflorescences may decline (Payton and Mark, 1979). The abundant production of inflorescences and resprouting after burning significantly lowers stored resources which take time to accumulate again (Payton and Mark, 1979). Burning and re-sprouting deplete the stored resources, which take time to re-accumulate (Payton and Mark, 1979), especially in harsh conditions such as those in the alpine areas of Lesotho. Thus, inflorescence production may increase as the amount or resources available for inflorescence production increase, as mentioned in the literature by Rees *et al.* (2002) and Kelly and Sork (2002).

The relationship between burning and inflorescence production was noted as early as 1953 (Barker, 1953, in Connor, 1966). The effects of fire on flowering may vary greatly with burning season and species (Bruhjell and Tegart, 2001), however, some generalizations may still be made. Tussocks burnt close to the onset of the growing season may show earlier inflorescence production than those not burnt, in the first growing season following burning (Mark, 1965; Daubenmire, 1968; Bruhjell and Tegart, 2001). The number of inflorescences is also noted to increase greatly in the growing season following burning in either winter or spring (e.g. Mark, 1965, 1994; Connor, 1966; Old, 1969; Payton and Mark, 1979; Lunt, 1994; Whelan, 1995; Bond and van Wilgen, 1996; Bond, 1997; Bruhjell and Tegart, 2001, Fill *et al*., 2012). However, increased numbers of inflorescences resulted in reduced size, height and seed weight of the inflorescences produced (Mark, 1965). Only a few exceptions indicate decreases in inflorescence numbers in the growing season following burning (e.g. *Sorghastrum nutans*) (Daubenmire, 1968). The increase is believed to be the result of increased temperatures experienced by the crown due to reduced shading by the tillers and increased absorption by the dark colour of the burnt crown (Mark, 1965; Rowley. 1969). Removal of leaf litter has also been linked to the increase in inflorescence numbers (Daubenmire, 1968), which corresponds with maximum soil temperatures (Old, 1969). High water availability may also be another reason for increased inflorescence numbers (Bond and van Wilgen, 1996). However, due to reduced vegetation cover, runoff on slopes may increase resulting in a decrease in infiltration and in soil moisture. Increased availability of nutrients through ash also increases the number of inflorescences, but not to the same extent as the removal of tillers and dead matter from around the crown (Mark, 1965; Old, 1969). Despite this, burning during or just before flowering may substantially reduce the number of inflorescences (Mark, 1965), and delay inflorescence emergence (Craine *et al*.,

2012). The resprouting of grasses directly following burning relies on the stored resources within the crown, thus limiting inflorescence production (Richards and Cadwell, 1985).

Defoliation through methods other than burning may also increase the number of flowering tillers, as observed on South Island, New Zealand (Mark, 1965). Hence, clipping is known to double the number of inflorescences in Alaskan tussocks (Shaver *et al.*, 1986). Thus, although both burning and grazing may increase the number of inflorescences, this is dependent on the season of burning, or the stage in the reproductive cycle the tussocks are in. Grazing after burning can have a strong negative effect on the number of inflorescences produced (Mark, 1965). Tussocks with new growth after burning are more palatable than usual, thus drastically reducing the number of seeds produced. Yet, burning and grazing were recently found to have little influence on grass inflorescence phenology in Kansas, U.S.A. (Craine *et al.*, 2012).

Mark (1968, 1970) argue that the number of inflorescences produced is dependent on the amount of reserves an individual tussock has, in addition to the persistence of warm temperatures. The production of inflorescences and their seed is the result of a large extraction of stored resources (Richards and Cadwell, 1985). *Chionochloa* tussocks were observed to produce inflorescences more frequently and in greater abundance when located on the edges of community patches, compared to those found in the middle (Hay *et al*., 2008). These researchers believe that this is the result of an increase in available resources. The relationship between flowering intensity and stored/available resources has been demonstrated by other researchers such as Rees *et al.* (2002) and Kelly and Sork (2002). A study on *Eriophorum vaginatum* in Alaska has observed that the allocation of resources to reproduction generally occurred in autumn and spring when resources needed for growth were low (Mark and Chapin III, 1989) and is supported by Trlica (1999). Resource accumulation and acquisition is a function of the water, light and nutrients available to each tussock (Hay *et al.*, 2008). These factors vary both across an area and within a community patch, and thus resource accumulation will vary between tussocks in an area and within a community (Hay *et al.*, 2008). Given that resources are accumulated and stored within the roots and crown (Trlica, 1999), thus the crown size may be an indication of the storage ability of tussocks and the number of inflorescences it is able to produce. This would explain the significant correlations between basal circumference and standing biomass in 2010 (Table 8.10).

In this study, the low mean number of inflorescences in 2009 may have been the result of an increase in inflorescence numbers drawing heavily on the stored reserves of each tussock. If the

climatic conditions had not allowed for adequate acquiescence of new stored resources, the result would be a decrease in the number of inflorescences the following year. Once the stored reserves had been replenished, inflorescence numbers could increase again. Similarly, the increase in the number of inflorescences in 2010 may be explained by the acquiescence of a greater number of resources than in 2009. This is perhaps the result of warmer mean air temperatures at the start of the growing season in October (7.7 ^{o}C in 2009 and 8.9 ^{o}C in 2010).

According to the theory, a trade-off should exist where increasing use of resources to reproductive efforts should result in a decrease in plant growth, especially in areas with reduced resource availability (Mark, 1965; Lord, 1998; Obeso, 2002). The decrease in growth may partially be as a result of a decrease in the number of tillers produced during a growing season, which are not for reproductive use, as well as the death of reproductive tillers once flowering has ended, as observed on South Island, New Zealand (Lord, 1998). However, very few studies have shown this (Lord, 1998), and the current study is no exception. Increased numbers of inflorescences at site V.F. were accompanied by decreases in basal circumference and average height, while the living basal area and the standing biomass increased. Yet, according to the literature, a reduced number of inflorescences under similar conditions should result in increased growth (Lord, 1998). At site V.S.E., the average height was the only measured growth component observed to increase, thus this theory may not be true within this environment. However, the effects of grazing cannot be discounted and may have resulted in the lack of positive growth, especially as this site is located near a grazing post (Figure 3.4). Lord (1998) concludes that this trade-off between growth and reproduction may not be a significant factor in long lived grassland communities. However, earlier results of this study may suggest that perhaps the trade-off exists between stored reserves and the number of inflorescences produced. This idea is supported by Obeso (2002), but further research is required to verify this. The trade-off may have both direct effects and long-term effects on growth rates and future inflorescence production (Obeso, 2002), which also requires further research in terms of *M. drakensbergensis*.

Seed dispersal in grasslands is mainly the action of wind (Mark, 1964; Hay *et al.*, 2008). Thus, new seedlings are likely to be observed in down-wind locations in the direction of the dominant wind. In the case of Sani top, this direction is towards the escarpment edge due to a dominant westerly to north-northwesterly wind direction. Few (<5 per site) seedlings were observed and most were observed downwind of mature individuals. The up-slope movement is reliant on up-slope wind and animal movements (Crawford, 2008). The increasing number of livestock within

the study region would suggest that more seed is being transported further and thus the likelihood of seed resting within a suitable place is higher. This suggests more seedlings and new populations should be observed, however, this was not the case. It is suggested that the same livestock are grazing the new more palatable shoots of emerging seedlings as they are roughly the same height as the other more palatable over-grazed species. The seedlings also emerge with a vivid green colour, perhaps making them more appetizing to grazers.

The unstable nature of slope soils makes germination and settling difficult (van Zinderen Bakker, 1981). This may explain why small *Merxmuellera* individuals (possibly seedlings) were observed mainly at the flatter sites. The observations correspond with those of Bullock *et al*. (1994), who observed less than 200 seedlings within two different tussock species during the same time period. Large amounts of seed are produced each growing season, but result in very few seedlings. This suggests that a large seed bank must exist within the study region. However, grass seeds generally have a soil life of 5 years, although a small proportion of the seed may last longer (Gibson, 2009). The seeds in the soil are lost due to pathogens, aging, decomposition, or are eaten by many organisms (Gibson, 2009). However, the effects of fire on the seedbank are not known, but personal observations suggest that burning does not promote germination in *M. drakensbergensis*. Similarly, *Danthonia* species (similar to *Merxmuellera*) were observed to regenerate from their crowns after burning, but no seedlings were observed in the burnt and unburnt sites by Lunt (1990).

The majority of the reproduction within all the study sites is vegetative, where segments from fragmented tussocks are believed to become independent individuals (Danin and Orshan, 1995; Lewis et al., 2001). Thus, many species at higher altitudes are more reliant on vegetative reproduction as a means to maintain and increase the altitudes at which they are found, as observed in Patagonia (Oliva *et al.*, 2005). This is perhaps a tradeoff between survival of a current population in one location and colonization of a new area, as suggested by Hedberg (1970). *Merxmuellera* species produce both male and female inflorescences, thus the seed produced may be only slightly genetically different to the parent plants (Zaitchik *et al.*, 2000). This suggests little genetic diversity within *Merxmuellera* communities, but could not be tested due to the costs involved. As genetic diversity of separated communities decreases with increasing time and grassland maturity (and is influenced by land-use and management), it is suggested that further research is required on this theme within the Lesotho highlands, especially in light of climate change.

8.5 Relationships between *M. drakensbergensis* characteristics

The number of inflorescences produced was significantly correlated to average height in 2009 and 2011 ($p<0.05$). In 2009, the relationship was not quite significant ($p = 0.05$). The correlation coefficientsindicate that as the average height increases, the number of inflorescences produced increases. This may be a reflection of the amount of stored resources enabling leaf extension and inflorescences produced. Thus, the greater the stored resources, the greater the leaf growth and the more inflorescences are able to be produced. This suggests that grazing may have a negative influence on the production of inflorescences especially if the inflorescences are grazed directly. Grazing during the summer months could potentially remove the inflorescences along with the leaf matter being consumed. This would result in a lower average height as well as low inflorescence numbers. The data gathered for this study were collected during the summer months when both grazing and inflorescences were present. Thus, the results are not surprising. Lee *et al.* (1993) believe that grazing through the removal of young reproductive tillers may influence the number of inflorescences. Reducing the non-reproductive tillers will cause the resources of tussocks to be allocated to replacing tillers rather than to inflorescence production. This suggests that had the data been collected towards the end of summer, when grazing intensity was slightly reduced, the relationship may have been indirect. Tiller growth is a function of the amount of photosynthesis taking place, the amounts of available minerals and the amount of stored resources within the crown. Thus, the more tiller growth there is, the more resources may potentially be available for reproduction. The longer the leaves, the bigger the surface area in which photosynthesis can occur and thus the more glucose produced.

Basal circumference was observed to be extremely significantly linked to average height in 2009 and 2011 ($p <0.0001$). However, in 2010 the relationship is not quite significant ($p = 0.076$). The lack of a significant relationship in 2009 may be the result of increased grazing of *M. drakensbergensis* occurring. Yet, this is not supported by the grazing study data (Figure 5.6). The relationship indicates that as the basal cover increases, the average height will also increase. Again, this may be related to the stored resources within the crown. Increasing crown size also suggests a greater capacity for stored resources, which may allow for greater numbers of inflorescences (e.g.Lord, 1998). However, this was not significantly observed in this study.

Average height was significantly correlated to the percentages of living basal area in 2009 and 2011 ($p<0.05$) and extremely significantly in 2010 ($p<0.0001$). This finding corresponds with the literature, as an increase in the amount of living tissues would mean an increase in resource uptake, photosynthesis and stored resources and thus the more resources available for growth.

Changes in the living basal area are able to explain 23% of the variation in average height in 2009 and 18% in 2011. In 2010, 32% of the variation may be explained by changes in the other variable. The increase may be the result of slightly more precipitation in October, November and December 2010 than experienced in 2011, and possibly 2009 (See Figure 4.7). This would have the potential to increase the availablility of soil nutrients, allowing for increases in growth.

Increases in the significance and strength of the relationship between standing biomass and average height are observed during the three years of study. The relationship is not significant in 2009 ($p = 0.768$), significant in 2010 ($p = 0.02$) and very significant in 2011 ($p = 0.002$). This result is not surprising as standing biomass is a function of tiller length and basal circumference. Thus, increases in tiller numbers and size will increase the standing biomass observed. A large proportion of standing biomass is the tiller length which extends the ground cover of tussocks beyond merely basal measurements. The greater the percentage of *M. drakensbergensis*, the more plant matter there is of this species within the ecosystem, and potentially more inflorescences. However, the data collected was only significantly correlated to the number of inflorescences produced in 2009 ($p = 0.033$). Similarly, basal circumference was also significantly related to the standing biomass in 2010 ($p=0.013$), but not in 2009 and 2011 ($p>0.2$). Due to the lack of correlation during two of the three years of study, these two significant relationships are potentially anomalies and thus do not indicate general trends.

8.6 Summary

The baseline study aims to compare the *M. drakensbergensis* communities of different sites to indicate which communities are surviving at their optimum, and compare the growth, and growth patterns of *M. drakensbergensis* between sites. Five main features were outlined for comparison: basal circumference, average height, basal area, standing biomass and inflorescences produced. The following are the key findings from this section:

1. Across all the sites, the average height was 41.6cm, well below 1m. Decreases in height were linked to grazing and wind pruning.
2. An overall increase in basal circumference was recorded for most sites between 2009 and 2011, and attributed to intensive grazing as well as an increase in the percentage number of hours below $0^{o}C$ and $2^{o}C$.
3. A general increase in living basal area was observed across all sites (mean increase = 10.7% across all sites) between 2009 and 2011.

4. Standing biomass measurements were greater than 30% (apart from site S.E.). A decrease across all sites of 0.71% was observed between 2009 and 2010, but increased by 2.4% from 2010 to 2011.
5. The combined relationship of the burn frequency and the time since each site was last burnt were significantly correlated to the basal circumference data for all three years. The time since burning contributed more to the relationship yet increases in both predictors resulted in increases in basal circumference. Movement through the age states increases with increasing burn frequency and time since last burnt.
6. The average height becomes an increasingly poorer fit with burn frequency and time since last burnt with increasing time. Average height potentially becomes more influenced by grazing and its associated variation with time since burning.
7. Burn frequency and the time since last burnt were a good fit to standing biomass in 2009 but the relationship and influence of the predictors decreased with time. As the burn frequency and time since last burnt increases so the standing biomass may decrease.
8. An increase in the number of inflorescences were observed each year between 2009 and 2011 (mean = 9.85 in 2009; mean = 11.13 in 2010; mean = 13.1 in 2011). A small rise in temperature each year in October and November may explain the increase in mean inflorescence numbers. The mean inflorescence data also correspond to an increasing percentage of hours below 0 ^{o}C and below 2 ^{o}C in October each year.
9. Together burn frequency and time since each site was last burnt are a poor fit to the number of inflorescences produced, but the overall relationship is very significant. As the time since burning increases so the number of inflorescences also increases.
10. The number of inflorescences observed may be linked to stored resources. Basal circumference and living basal area significantly correlated to indicate that as basal circumference and living basal area increase, so would the number of inflorescences.
11. Average height correlated in direct relationships to basal circumference, living basal area and inflorescences produced, and were attributed to stored resources and greater capacity of resource capture. The relationship between average height and standing biomass was direct and increased in significance during the years of study.
12. The main method of propagation of *M.drakensbergensis* is through vegetative propagation, possibly due to a low seedling survival rate associated with grazing and climatic factors.

Chapter 9:

Monitoring Study of *M. drakensbergensis* Recovery



CHAPTER 9:

Monitoring Study of *M. drakensbergensis* Recovery

The survival of a tussock is determined by the production, turnover, growth and longevity of tillers. This chapter aims to examine the extent of recovery which *Merxmuellera* tussocks are able to achieve between burning events through growth rates, both following recent burning and without burning during recent years. The literature suggests that it may take tussocks many years to recover from a single burning event, for example 14 years were required for *Chionochloa rigida* to recover to pre-burn levels in alpine New Zealand (Payton and Mark, 1979). Thus, growth was monitored across all sites as well as at sites recently burnt. The produced growth rates should also provide a good indication of the health of the *M. drakensbergensis* tussocks within this region, assuming a healthy tussock has a positive growth during the growing season. Each site is characterized by a slightly different microclimate, and thus growth may not be the same between each site, and so are compared. Growth rates may also not be evenly distributed throughout each individual and hence the growth rates are compared for three positions within the studied tussocks: along the edge, at the radial midpoint and in the center. This was also monitored in recently burnt *M. drakensbergensis* tussocks and the study compares whether growth is concentrated in the same place in the individuals, both before and after burning. Growth rates are also compared to the burn frequency and the time since last burnt, so as to determine any relationships. Similarly, growth rates are compared across various aspects within the studied tussocks and between sites.

9.1 Methodology

Monitoring the recovery and growth of *M. drakensbergensis* tussocks was undertaken over the course of the growing seasons during the study period. Monitoring the recovery of tussocks within recently burnt sites and sites with known post-burn ages was undertaken every second month, following the growing season for three years: 2009, 2010 and 2011 (e.g January, March, May, July, September and November). Sites were selected as for the baseline study, with some sites similar in topographical position but of differing post-burn age, and others of the same post-burn age but differing in topographical position. Altitude, aspect and slope gradient were noted at each sampling site as site-specific environmental conditions may have a possible impact on growth, hence the use of different sampling sites for similar post-burn ages. Typically, ten tufts are observed from each sampling site (Rowley, 1970; Payton and Mark, 1979; Ferwerda *et al.*,

2006; Payton and Pearce, 2009), but in this study an average of 30 were initially aimed at for each site. However, the number actually measured was dependant on the number occurring within the site, weather conditions during sampling, and occasionally herdsmen prevented the measuring of all 30 tussocks. Unlike the study by Monteiro *et al*. (2011), who eliminated the very young and very old individuals based on their shape, all *M. drakensbergensis* individuals were measured in the current study. The tussock communities in this region are small and all individuals interact and are affected by burning and grazing.

Leaf elongation has been used in the past to indicate grass growth or rate of growth (e.g. Cole and Wilkins, 1958). Although methods such as precision resistors have also previously been used (Busso and Richards, 1992), this method is time consuming and uses equipment which may be stolen or vandalized. An alternative approach is to estimate grass growth by simply measuring the highest blade for each of twenty plants (Everson and Everson, 1987). However, this method was decided against as it was tested and provided highly variable results. Leaf length has also been measured from the base at the crown to the tip of a straightened out tiller, and proven to be effective (Volesky, 1994). This study measured leaf length of the same tiller each time by markings made on the leaf blade using brightly coloured nail varnish. Nail varnish was ideal for this study due to its water resistance, durability and compact size. However, some brands are known to kill the leaf, thus several brands were tested beforehand. Markings were made 2mm from the tip of the leaf blade and where the blade joins the crown, as well as 3cm down from the tip. The blade length was then held straight, measured and recorded in millimeters between the base of the leaf at the crown and the tip. This was performed on three tillers within each tuft (as used by Payton and Mark, 1979; Snyman, 2004), with different positions (i.e. the perimeter, radial midpoint and centre of the tuft diameter, as in Figure 9.1) and orientation (e.g. northern/eastern/southern/western tuft sides) within each tuft for each selected tiller. In a previous study, only two tillers were observed: at the perimeter and center (Volesky, 1994), yet in the current study it was felt that adding a third measurement would be beneficial. Thus, a tiller measurement at the radial midpoint was added. The three measurements would provide more accurate data when hollow centers (or ring shapes) were encountered, as the central tillers are influenced differently to those in the middle of an un-hollow tuft. This was due to different microclimates and levels of exposure to insolation within different micro-spatial positions across the tuft (e.g. to temperature). Thus, the three measurements will assist in providing a detailed depiction of *M. drakensbergensis* and its associated processes. This procedure was also undertaken within the burnt *M. drakensbergensis* communities.

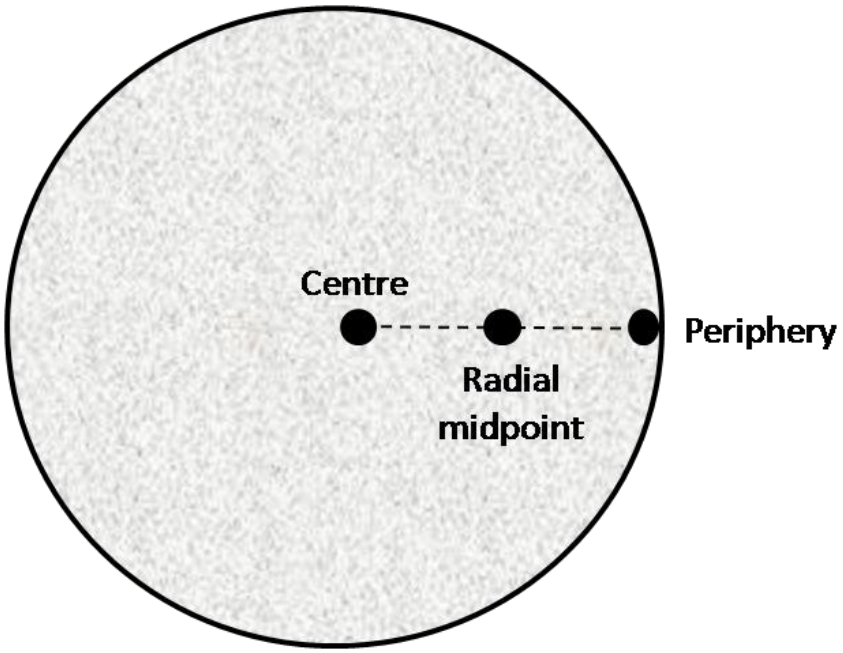


Figure 9.1: A tussock with the location of the measured positions marked.

Detailed drawings and measurements were made of each study site which depicted the positioning of each studied tussock. This enabled the marked tussocks to be located on repeat visits. Other international studies have used thin coloured wire to mark tussocks and individual tillers (e.g. Olson and Richards, 1988; Guàrdia *et al*., 2000), but such an approach could not be used within the study region due to the interference of livestock, vandalism and theft. On returning to the site, any grazing activity on the tuft (marked by the nail) was indicated by a loss of nail varnish markings and flat edges to the leaf blades. Grazing activity was noted, a different colour was then applied to the new tip and the leaf blade was measured. Any fire between sampling was noted and damage classes, as outlined by Conrad and Poulton (1966), were allocated to tufts marked with nails, with the study resuming using the same tufts. Another layer of different colored varnish was applied to the new tip following each measurement. Although only the length of green tissue was measured by Calder *et al.* (1992) to avoid measuring any dieback of the leaf, this was not followed in the current study in order to improve accuracy. Where a measured tiller died, its death was recorded and a new nearby tiller was selected. Repeat photography is a useful tool which was used in this study and aided sampling sites and tufts, in addition to providing visual evidence of changes occurring within the study areas over time, as well as the recovery/degradation of the studied tussocks. To assist the accuracy of the measurements, a field sheet was generated (see Figure A.15 in the appendices).

9.1.1 Data Analysis

Graphs were generated in Microsoft excel to show the height changes for each position (periphery, centre and radial midpoint) across the tussock over time. Leaf elongation was converted into a growth rate by subtracting the leaf length from the previous measurement of leaf length. The result is a growth rate for each individual leaf, where possible, per two month period. The standard deviation of each bimonthly average growth rate was added to the graphs to give an indication of the vaiation experienced within the communities at each of the sites. A two way repeated measures ANOVA was performed to indicate if differences were present between the different tiller positions (peripheral, radial midpoint and central), and between the different sites. The two-way ANOVA allows for the effect of two separate independent variables on a dependent variable to be determined, as well as the effect of the interaction between the independent variables on the dependent variable (Townend, 2002; Dytham, 2011). Differences between the different tiller positions may indicate that different recovery times may be required by different tillers in order to reach average pre-burn heights. Similarly, differences between sites may indicate different growth rates at sites with differing burn histories. The growth rates were entered into SPSS version 21 and met the assumptions for repeated measures ANOVA (e.g. normal distributions and homogeneity of variance). A 95% confidence level was used to indicate significant values.

The relationship between the growth rates and the factors possibly influencing the growth rates were investigated. A multiple regression analysis is appropriate when one dependent variable and many selected predictor variables are present (Townend, 2002; Dytham, 2011). In this case, the dependent variable was the growth rates for each site and tiller position. Using the raw data and not averaging the tiller position measurements allowed for the location of the tillers to be added to the analysis, thus providing more detailed information on the relationships present. The other predictor variables used were altitude, average temperature, average precipitation, average wind speed, site aspect, season and the burn history variables. Altitude was calculated at the center of each site, and thus *M. drakensbergensis* community. The climatic data were obtained from the weather station present in the study region and presented in Chapter 4. The site aspect was observed in the field using a compass and the values allocated a value from one to four. This allowed for the flat site, A.P., to be assigned a value of zero. However, once the multiple regression had been performed the aspect indicated a non-significant result ($p>0.1$). Thus, the values were reassigned into broader classes. Using knowledge of the position of the sun in summer and winter, aspect was divided into north and west facing aspects and south and east

facing aspects. A value of one was assigned to aspects from west-northwest to east-northeast, while the remaining aspects were assigned a value of two. Site A.P. retained a value of zero. The multiple regression was performed again, but this time the site aspect predictor indicated increased significance. Similarly, the multiple regression was initially performed using the original burn frequency and time since last burnt values, but these proved to be insignificant contributors. The regression was re-run using the square of burn frequency and the time since last burnt as this may have increased its significance. SPSS version 21 was used to perform the analysis at a 95% confidence level. Scatterplots were also produced to observe whether any of the relationships might have been non-linear, and the results presented.

In order to observe the role played by site aspect and the aspect of the individual tillers within individuals, the growth rates were averaged between the tiller positions. The average growth rates were then divided according to site aspect, and separately according to tiller aspect. The growth rates for each two month period were categorically assigned a colour on a grey scale according to their value. These were plotted on a chart to allow for visual comparison.

A multiple regression was undertaken to test whether there was a linear relationship between the growth rates and several potentially influencing variables. The raw growth rates were entered into SPSS version 21 and tested for normal distributions and homogeneity of variance, two of the assumptions required for a regression analysis (Townend, 2002). The data was significantly different from a normal distribution so a transformation was applied. A transformation of the data is a process in which each of the measured values is altered in the same way, producing a transformed data set (Townend, 2002; Dytham, 2011). However, negative values in the data set were problematic as many transformation techniques will either omit these values or alter their magnitude. A logarithmic transformation was applied to the data (see equation 5). One was added to each data value as $\log_{10}(0)$ is undefined (Townend, 2002). Once transformed, the data met the assumptions of normal distribution and homogeneity of variance.

$$\text{Growth rates transformed} = \log_{10}(\text{raw growth rates} + 1) \quad (5)$$

The influencing variables chosen were the different sites, tiller location, season, altitude, average temperature, average precipitation, average wind speed and site aspect. The variables were calculated in the same manner as for the multiple regression using the growth rates measured between burning events. The significance level used was 95%.

The predicted time to recover to pre-burn average heights was calculated as per equation 6.

$$\text{Time in years to pre-burn height} = \left(\frac{\text{Dec 2011 average height} - \text{Pre-burn height}}{\text{Growth rate after first 2 months}} \right) / 6 \quad (6)$$

9.2 Results

9.2.1 Growth rates between tiller position

The onset of spring growth occurs in October, and growth rates increase towards the middle of the growing season (i.e. late December to early January) (Figures 9.2, 9.3, 9.4 and 9.5). Summer growth is high and reached a maximum of 25.1cm between January and March 2009 at site S.E. The average growth rate dissipates in late May and June, with almost no growth (<1.0cm) between mid June to mid August. Negative growth, indicating a loss in plant matter and reduced average height, is indicated for each of the sites once or more during the study. Losses are not confined to a particular season, but are greater in the summer months (e.g. for the A.P. site: periphery = -5.74cm, radial midpoint = -5.78cm and centre = -2.69cm between December 2009 and February 2010) than during the winter months (e.g. for the A.P. site: periphery = -0.2cm, radial midpoint = -0.3cm and centre = -0.4cm between June and August 2010).

Across the duration of the study (January 2009 to January 2011), positive growth was observed for all sites except site H.W. (periphery = -11.3cm, radial midpoint = -9.3cm and centre = -5.3cm), while the greatest average growth was observed for site V.F. (periphory = 49.4cm, radial midpoint = 54.0cm and centre = 51.0cm). The change in average height between January 2009 and December 2009 was greatest at site V.F. (periphery = 46.7cm, radial midpoint = 47.6cm and centre = 52.9cm) and lowest at site H.W. (periphery = -9.7, radial midpoint = -11.4cm and centre = -3.5cm). Sites S.S., V.S.N., V.S.E., S.Str. and H.E. were observed to grow between 0.1 and 10.0cm during the course of 2009. Similarly, sites S.E., V.F., the upper and lower M.F. sites, the A.P. and the Kotisephola Pass sites were observed to have growth greater than 20.0cm during the course of the year. During 2010, the majority of the sites measured growth between 0.1cm and 10.0cm, except site V.F. and the lower M.F. site which grew more than 10.0cm.

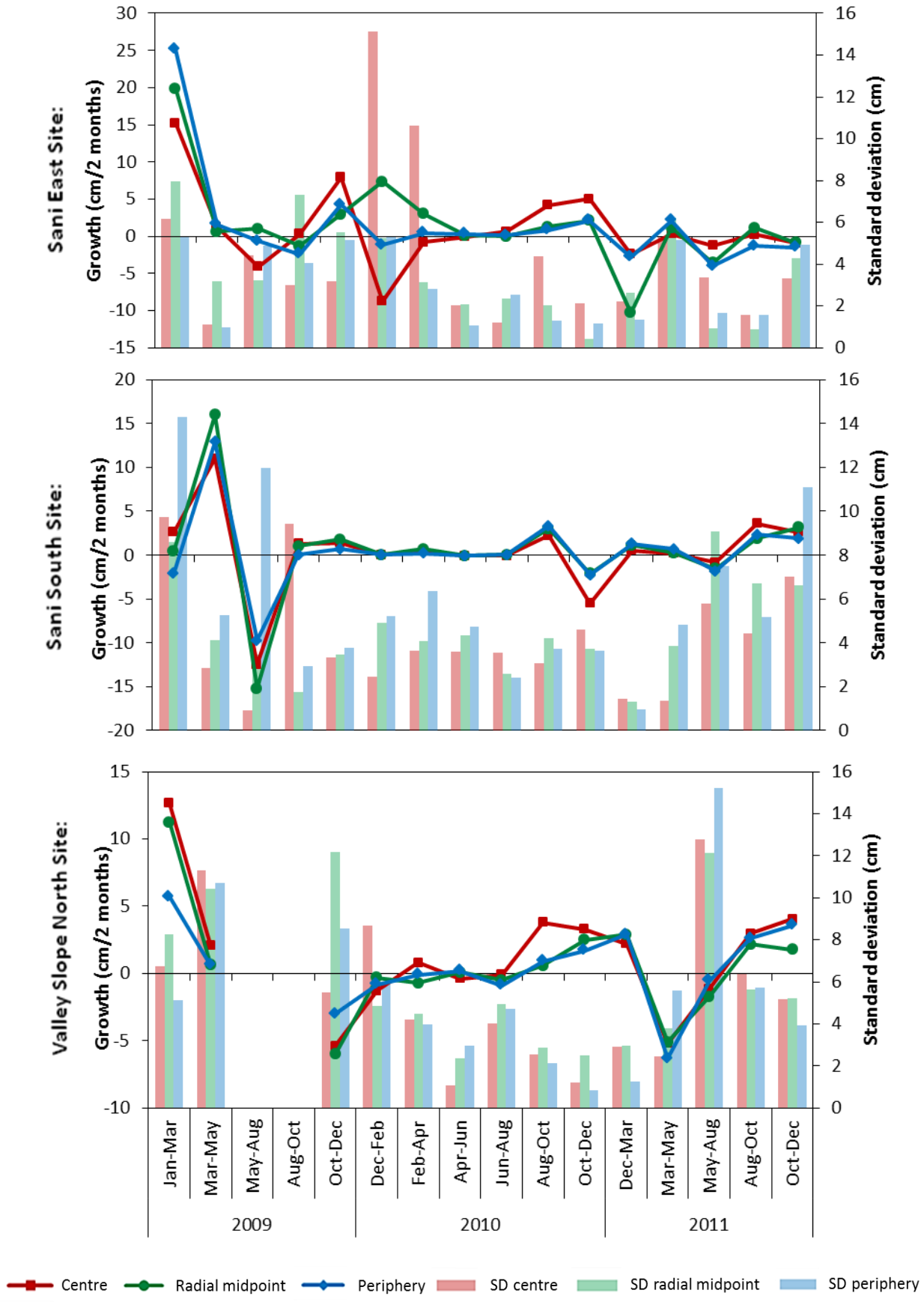


Figure 9.2: Average growth of *M. drakensbergensis* at sites S.E., S.S. and V.S.N. between January 2009 and May 2011.

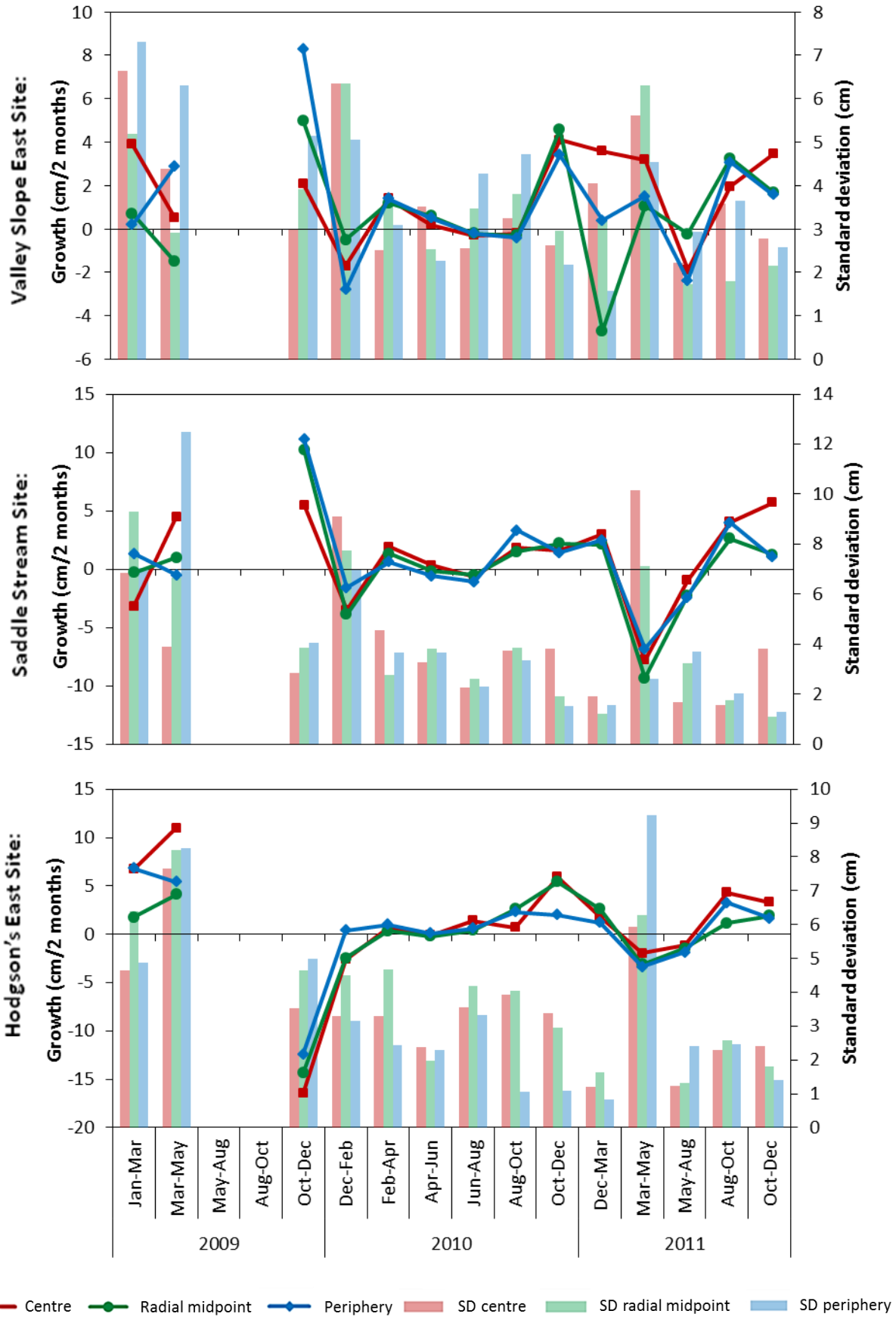


Figure 9.3: Average growth of *M. drakensbergensis* at sites V.S.E., S.Str. and H.E. between January 2009 and May 2011.

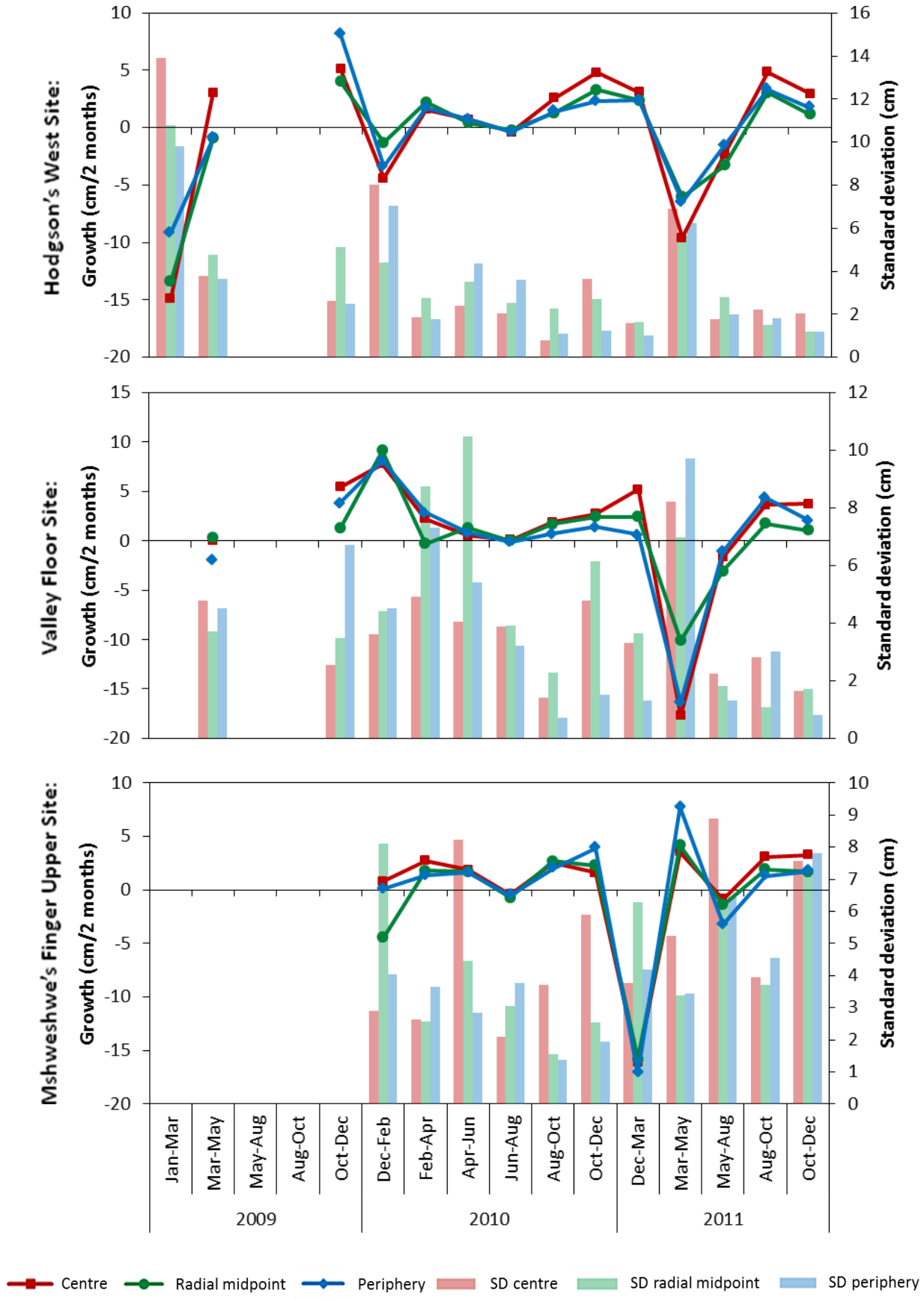


Figure 9.4: Average growth of *M. drakensbergensis* at sites H.W., V.F. and the upper M.F.site between January 2009 and May 2011.

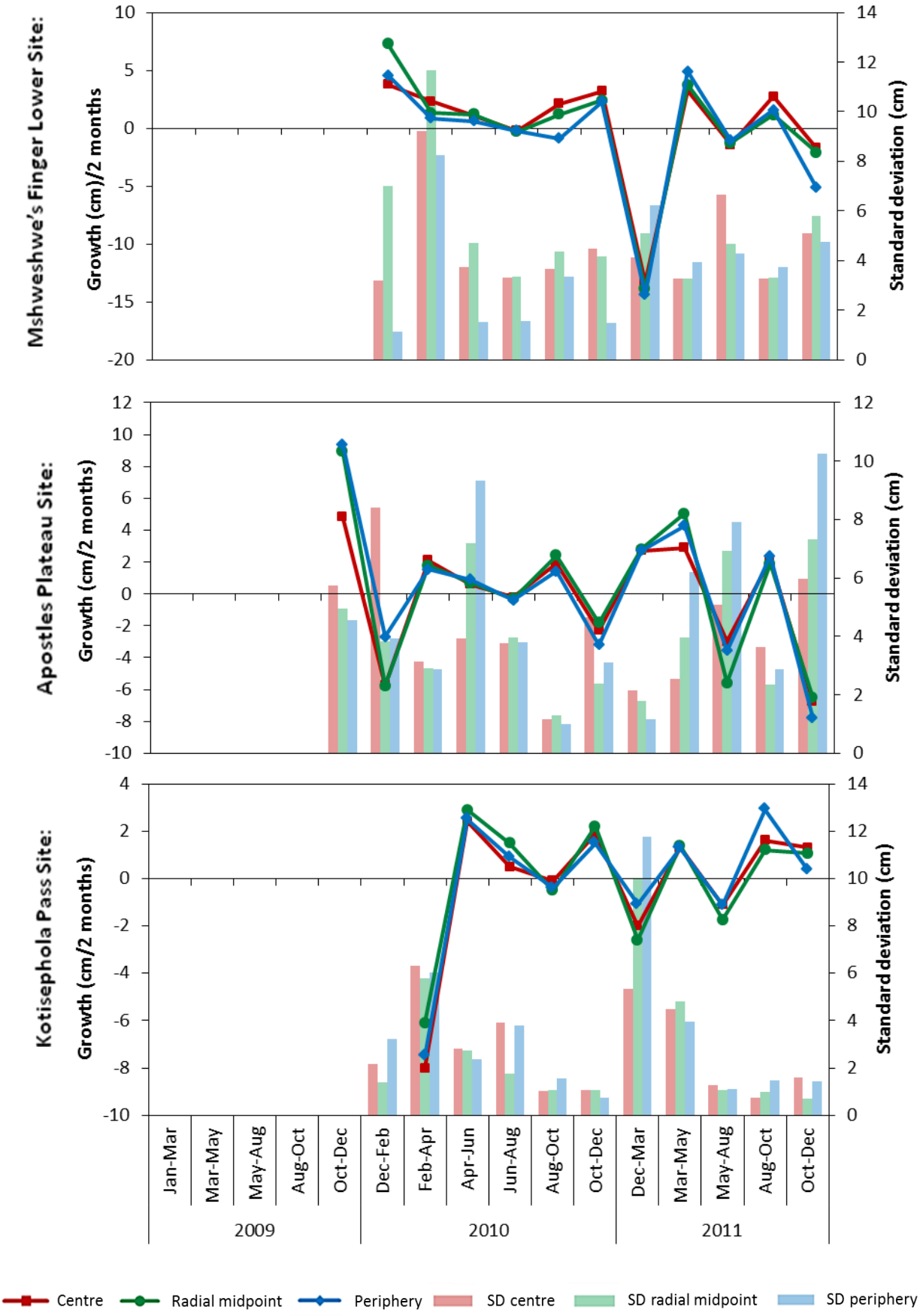


Figure 9.5: Average growth of *M. drakensbergensis* at the lower M.F.site, the A.P. site and the K.P. site between January 2009 and May 2011.

The standard deviations of the growth rates were calculated for each position within the studied individuals (periphery, radial midpoint and centre) at each site (Figure 9.2, 9.3, 9.4 and 9.5), and as an average across all sites (Figure 9.6). The standard deviation increases during the summer months (average value = 4.69cm/2 months) and decreases (3.56cm/2 months) during the winter months. The highest average standard deviation (7.85cm/2 months) is measured for the growth period from January to March 2009. This value decreases in 2010 (5.32cm/2 months) and 2011 (5.23cm/2 months). The lowest average standard deviations were recorded during the winter periods with the lowest value observed between August and October 2010 (2.54cm/2 months). The majority of the variation in the growth rates was linked to losses in plant matter. The standard deviation increased as the number of individuals experiencing losses increased. Yet, individuals with all tillers experiencing losses indicated low standard deviations (e.g. site S.E. between December and March 2011). Site S.E. was observed to display the highest variation across the study period. Sites H.W. and A.P. indicate a seasonal trend in the position of the highest and lowest standard deviations. During the summer months, the highest standard deviations are along the periphery, while in winter they are at or near the centers. Yet, the K.P. site suggests the opposite to this trend. No observable trends were noted at sites V.S.N., V.S.E. and for the upper M.F. site.

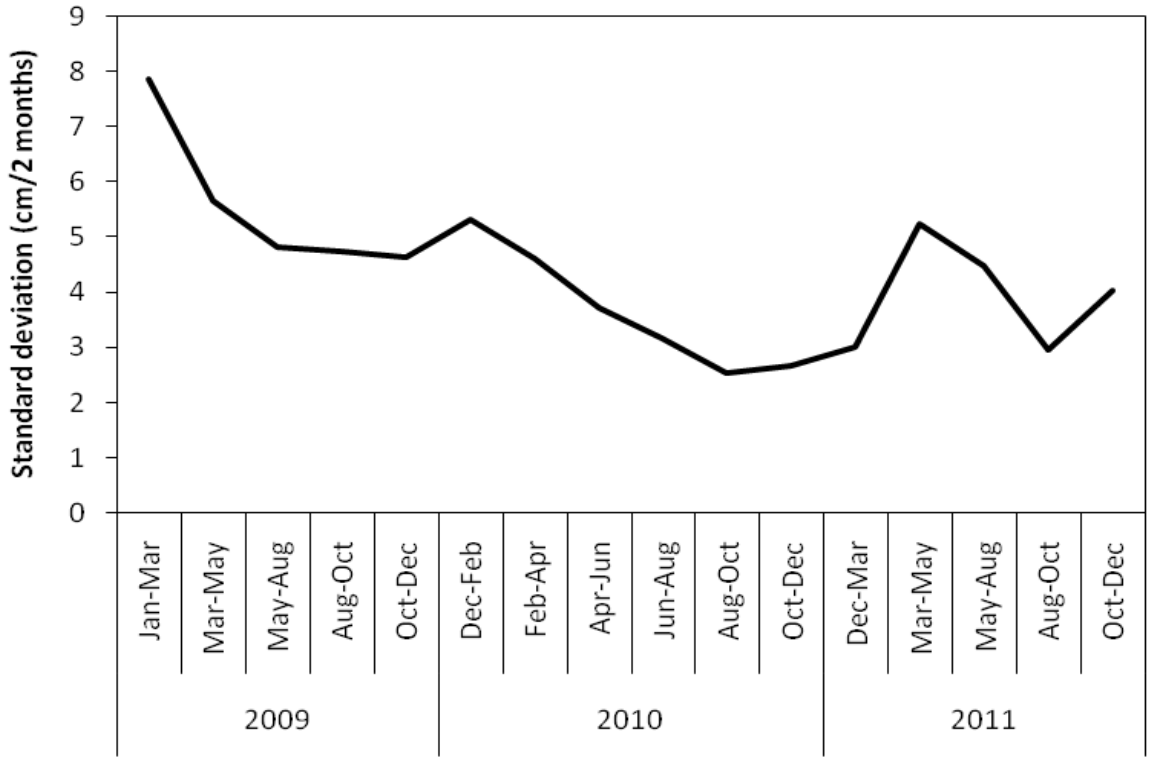


Figure 9.6: Average standard deviations in growth rate for all sites across the study period.

9.2.2 Growth rates between sites

The growth rates averaged for all positions (centre, radial midpoint, and periphery) for all sites indicates an overall decreasing growth rate across the tree years of study and increasing numbers of sites with negative growth rates (Figure 9.7 and 9.8). The greatest losses are recorded in late summer each year (March to May) and are associated with high variation within the studied communities. Conversely, winter months experience low standard deviations but also low average growth rates. The two way repeated measures ANOVA indicated that the location of the tillers (peripheral, radial midpoint and central) had no significant effect on the growth rate mean (F_2 =0.51, p = 0.6; Table 9.1). Similarly, the site also had no significant effect (F_{11} =0.32, p = 0.983). There was also no significant interaction between the factors (F_{22} = 0.19, p = 1.0).

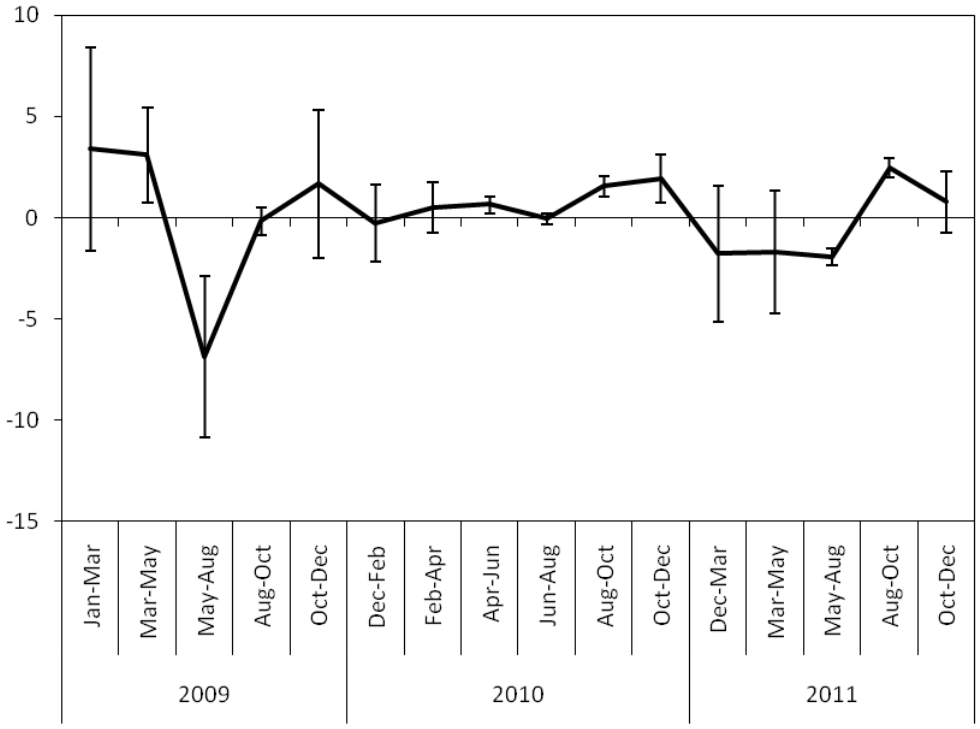


Figure 9.7: Average growth across all areas of *M. drakensbergensis* for all sites.

Table 9.1: Two-way repeated measures ANOVAs used to indicate the differences between tiller location and sites.

Source of Variance	Df	SS	MS	F	P-value
Location	2	13.298	6.649	0.512	0.600
Site	11	45.027	4.093	0.315	0.983
Location x site	22	53.728	2.442	0.188	1.000

SS: sum of squares; Df: degrees of freedom; MS: mean square; F: the ratio of MS_{group} to MS_{error}; P: significance value.
* P<0.10; **P<0.05; ***P<0.01

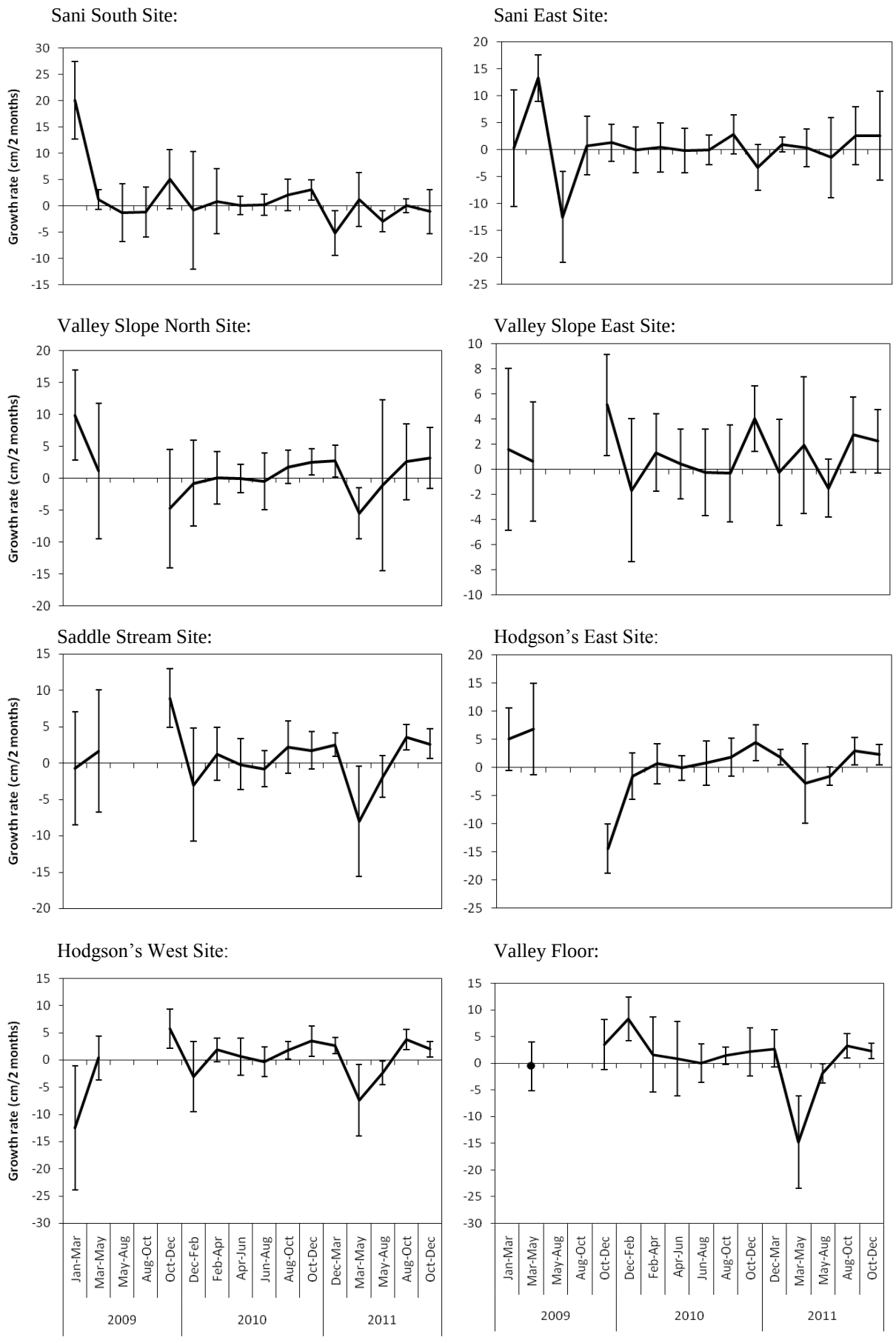


Figure 9.8: Average growth rates for each site, with the standard deviation.

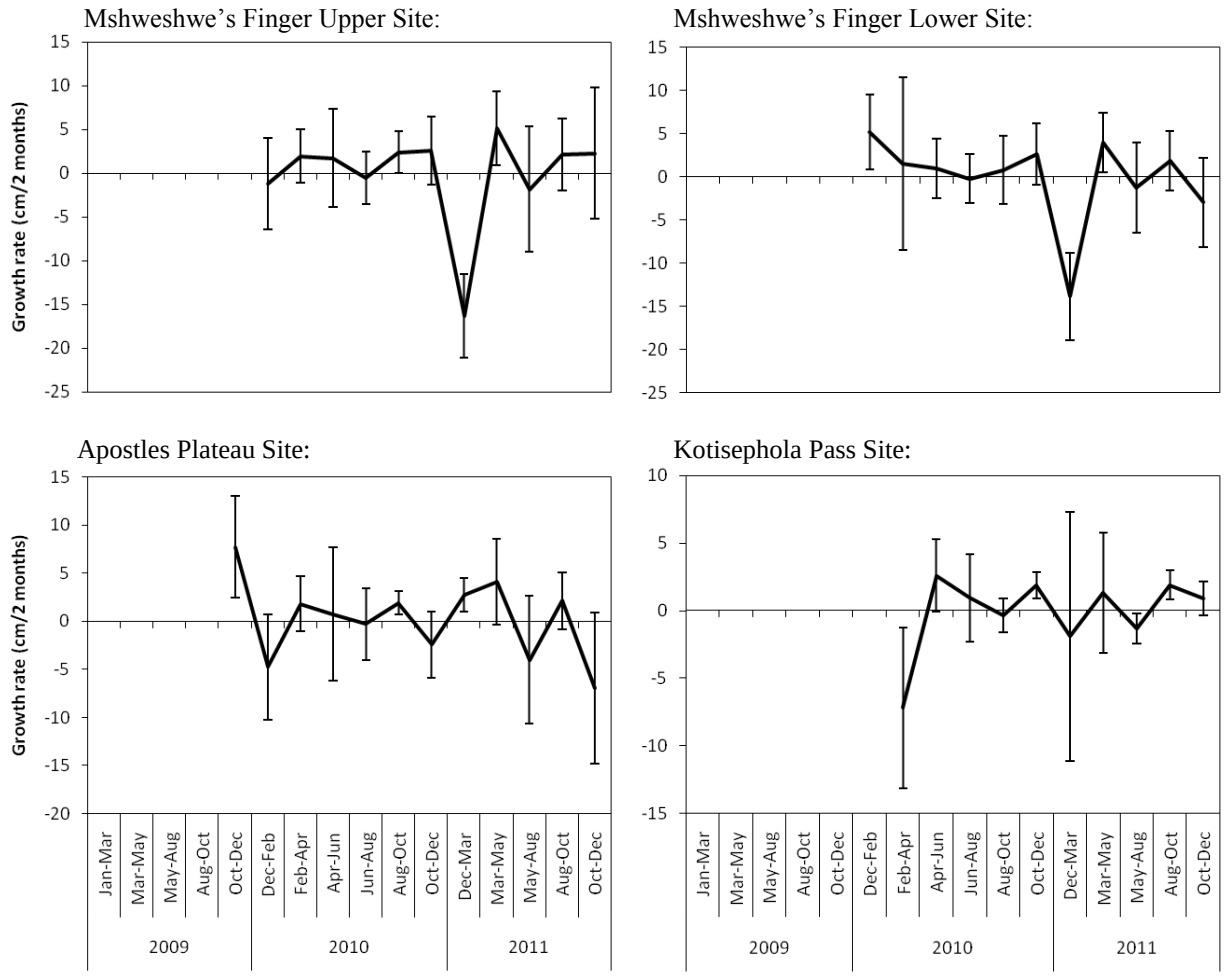


Figure 9.8 continued: Average growth rates for each site, with the standard deviation.

9.2.3 Relationship between growth rates, burning history and other variables

The multiple regression was extremely significant ($p<0.0001$), and provided a good description of the average relationship between the variables. The adjusted R^2 value is a better indication of the fit than the unadjusted coefficient of determination. Yet, the adjusted R^2 value indicated that the best-fit line only explained 10% of the variation. The slope (b-values) indicated indirect relationships were present between growth rates and tiller location, average rainfall, average wind speed, burn frequency and the time since last burnt (Table 9.2). However, insufficient evidence was present to indicate that average wind speed, site aspect, burn frequency and the time since last burnt had real relationships with measured growth rates. Temperature, precipitation and season indicated the most significant relationships to the growth rates.

Growth rates observed at each of the sites are presented in Figure 9.9 in relation to the aspect of each site. The results confirm the seasonal trend observed earlier in this chapter and in the multiple regression (Table 9.2). The north facing sites are observed to have the most consistently

high growth rates, while the south facing sites appear to vary the most. Increased spring growth rates are indicated to start earlier on the south facing sites in 2010, but not 2011. Similarly, the east facing site may also increase earlier than the north and west facing sites, but this is hindered from detection by grazing.

Table 9.2: Multiple regression analysis results.

Factor	Standardized coefficient (b)	SE	t	P-value
Site	-0.025	0.063	-0.324	0.746
Tiller location	-0.099	0.187	-1.855	0.064
Burn frequency2	-0.131	0.031	-1.417	0.157
TSLB2	-0.145	0.005	-1.566	0.118
Altitude	0.145	0.002	2.294	0.022*
Average temperature	0.448	0.115	3.992	0.000†
Average precipitation	-0.544	0.010	-3.091	0.002***
Average wind speed	0.021	0.156	0.241	0.810
Site aspect	0.111	0.341	1.744	0.082
Season	-0.290	0.612	-2.891	0.004***
(constant)	-10.088	6.170	-1.635	0.103

SE:Standard error; P-value: significance value.
*P<0.05; **P<0.01; ***P<0.005; †P<0.001

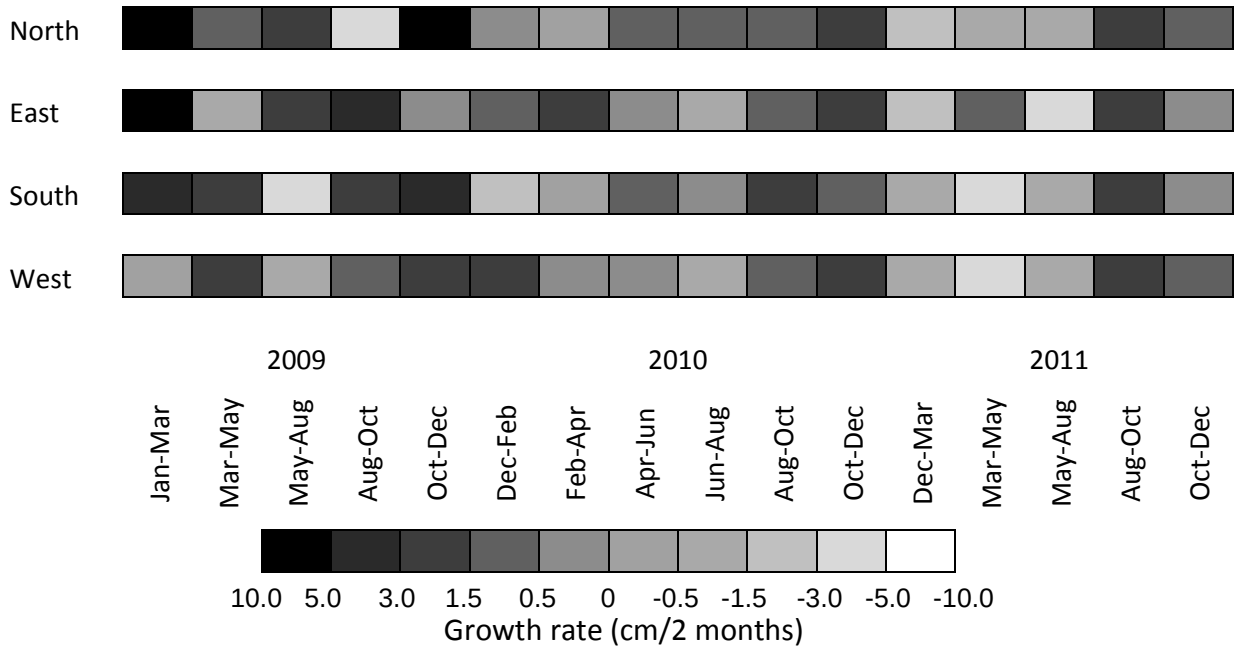


Figure 9.9: Average growth rates for the aspect of each site.

9.2.3.1 Relationship between growth rates and aspect within individuals

The growth rates for each aspect within each site are indicated in Figure 9.10. Losses in tiller height were generally observed in mid to late summer each year (December to May). The growth rates appear to be high during the summer months and low during the winter months. No clear trends are observed between the different tiller positions (north, south, east or west) each season and each year. Towards the end of 2010 and during 2011, the different tiller positions appear to have similar growth rates at each site.

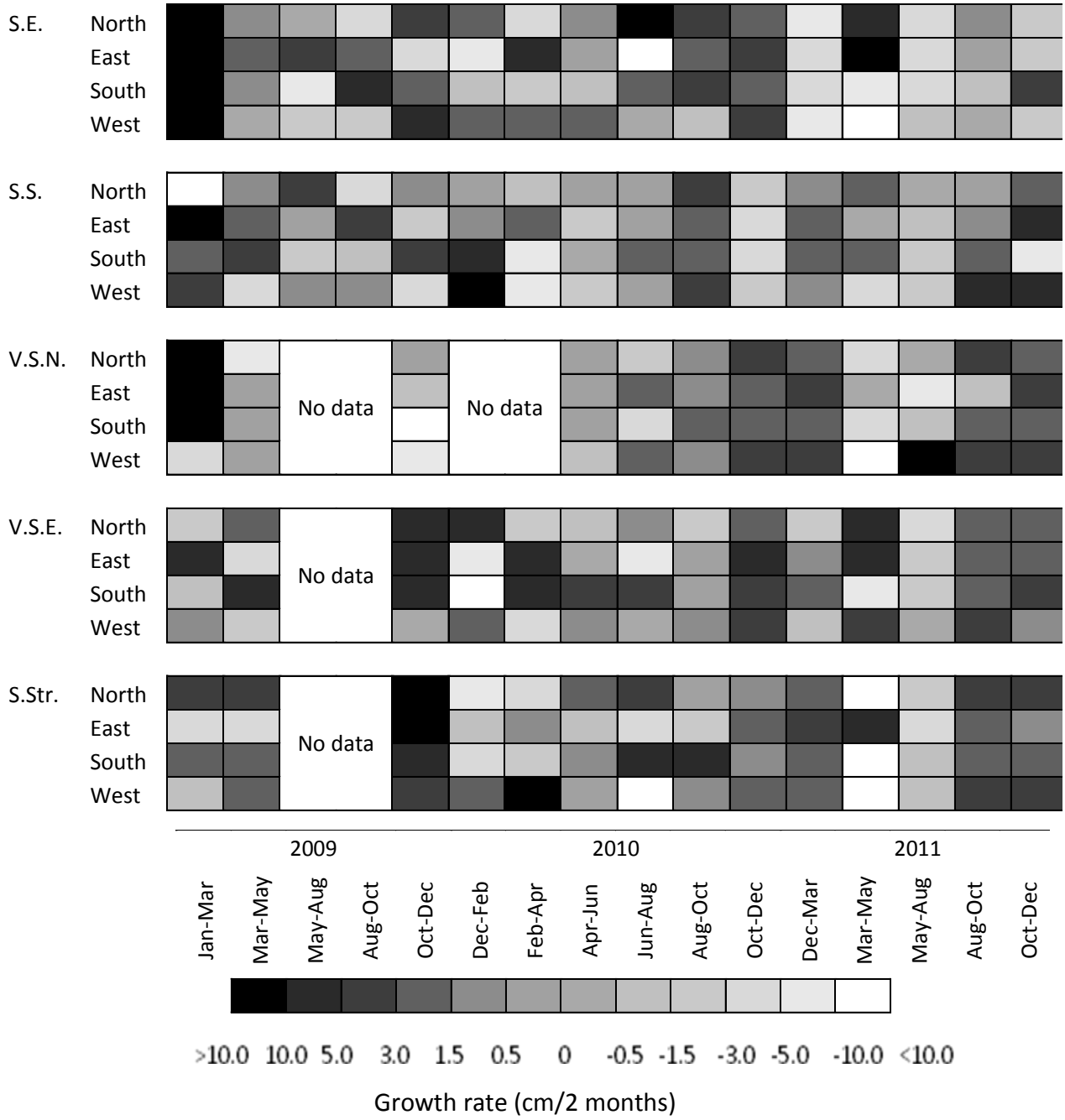


Figure 9.10: Average growth rates for each of the measured tiller aspects at each site.

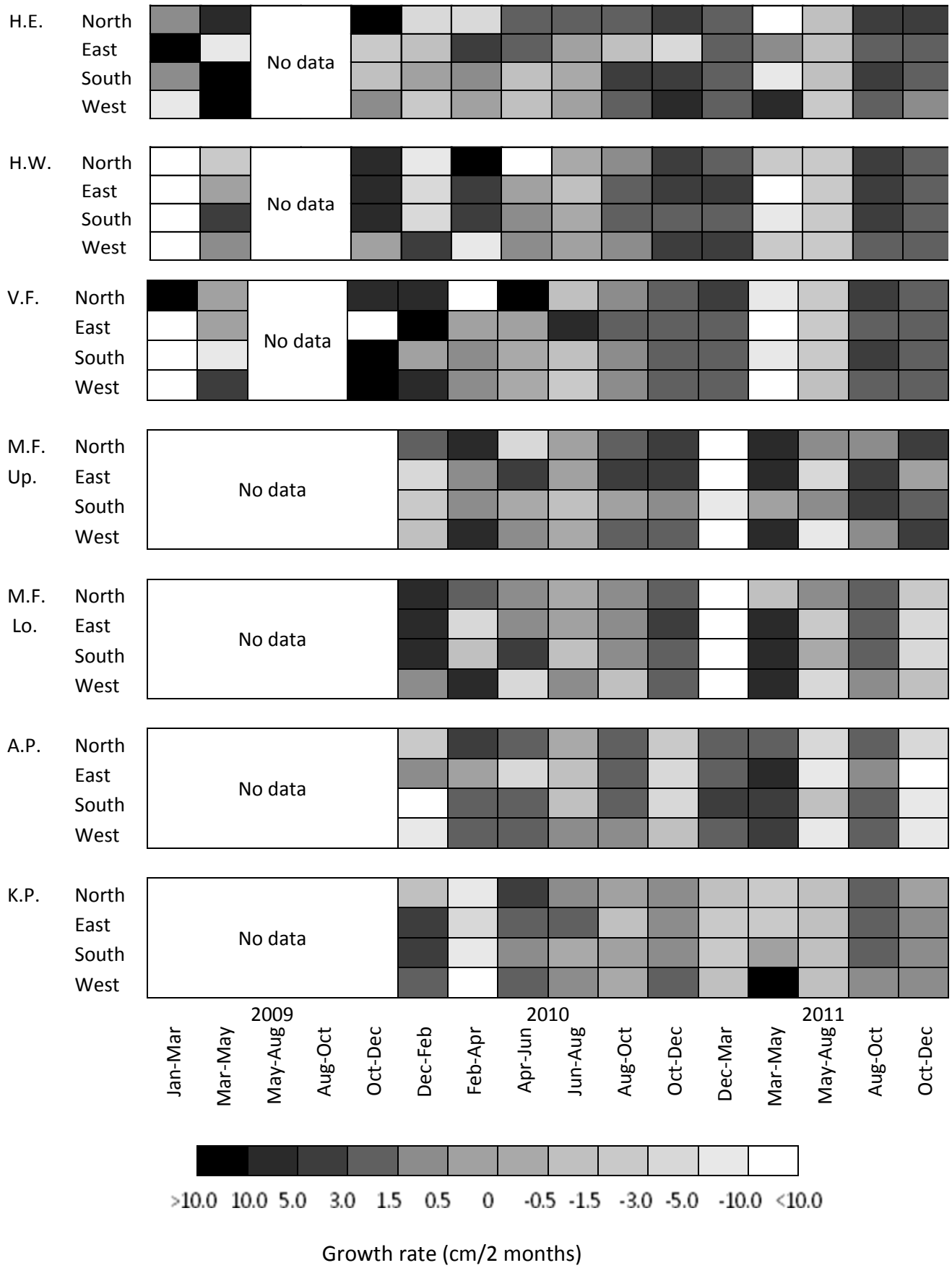


Figure 9.10 continued: Average growth rates for each of the measured tiller aspects at each site.

9.2.3 Growth rates following recent burning

9.2.3.1 Burnt sites and resprouting

During the first year of the study, burnt tussocks within or near three of the sites were burnt, and burnt only once during the study period. These three sites were marked and monitored to demonstrate the patterns, ability and rate of *M. drakensbergensis* tussock recovery from single burning events. A group, 8m to the north-west of site S.S., were burnt in late April 2009 (Figure 9.11). The burnt site is in close proximity to two grazing posts (in the village and one on the southern slopes of the study valley). The inter-tussock vegetation was not burnt, suggesting that each tussock was lit separately, and perhaps burnt through the night. Fifteen months after burning, leaf elongation acted to shade areas of bare soil and leaf litter visible after burning (a), and shrubs (*Helichrysum flanaganii*) had begun encroaching (Figure 9.12, c). A pathway created by an ice rat is observed at the bottom of the image (blue dashed lines), and is clearly visible soon after burning but is difficult to detect due to shading of new leaves (Figure 9.12). New shoots were observed to emerge out of the remaining protective leaf sheathes (Figures 9.17 and 9.18). As time progressed after burning, smaller flowering plants were observed to colonize the exposed soil in inter-tussock spaces as well as within the hollows in the interior (Figure 9.15).

A portion of site V.S.E. was burnt in mid December 2009 and was the result of a lightning strike (pers. comms. Herdsmen; Figure 9.16, top). The fire was quickly beaten out by herdsmen but not before it had consumed approximately $10m^2$. The burnt site is in close proximity to the grazing post in the study valley, hence the rapid extinguish time. Two months after burning, the inter-tussock vegetation such as *Berkheya multijuga* and *Cotula socialis* were again present (Figure 9.16, bottom). *Trifolium burchellianum* was also abundant and could often be found growing inside hollow centers such as in photo F in Figure 9.17. Rocks, as at the burnt site near S.S., were removed to create new and mend existing grazing markers. Being in close proximity to the grazing station, the burnt tussocks were observed to be frequently grazed as sheep were moved to and from the grazing station (Figure 9.18). Two years after burning, the burnt individuals were still identifiable from those not burnt, even though many of the flowering plants are no longer present (Figure 9.19).

The A.P. site was burnt in late August 2009 as the result of a runaway fire started in the Drakensberg at the base of the escarpment. The fire moved up the sides of the Sani Pass valley, aided by strong winds, and onto the Apostles plateau. The fire did not spread inland very far,

merely burnt an approximately 4m wide area along the length of the escarpment edge. The majority of tussocks resprouted, but a few did not resprout and were still leafless 27 months after burning. Many individuals displayed only partial regrowth (Figures 9.24, 9.25 and 9.26).

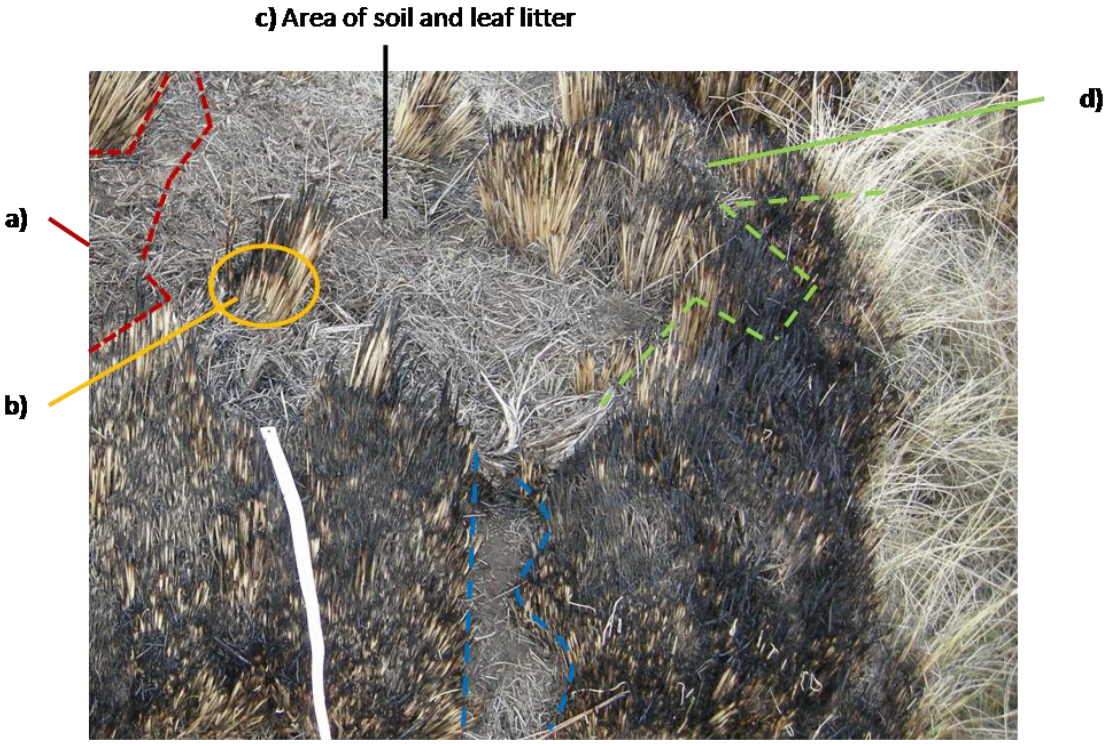


Figure 9.11: The burnt *M. drakensbergensis* tussocks near site S.S. (tape measure is 1cm in diameter).

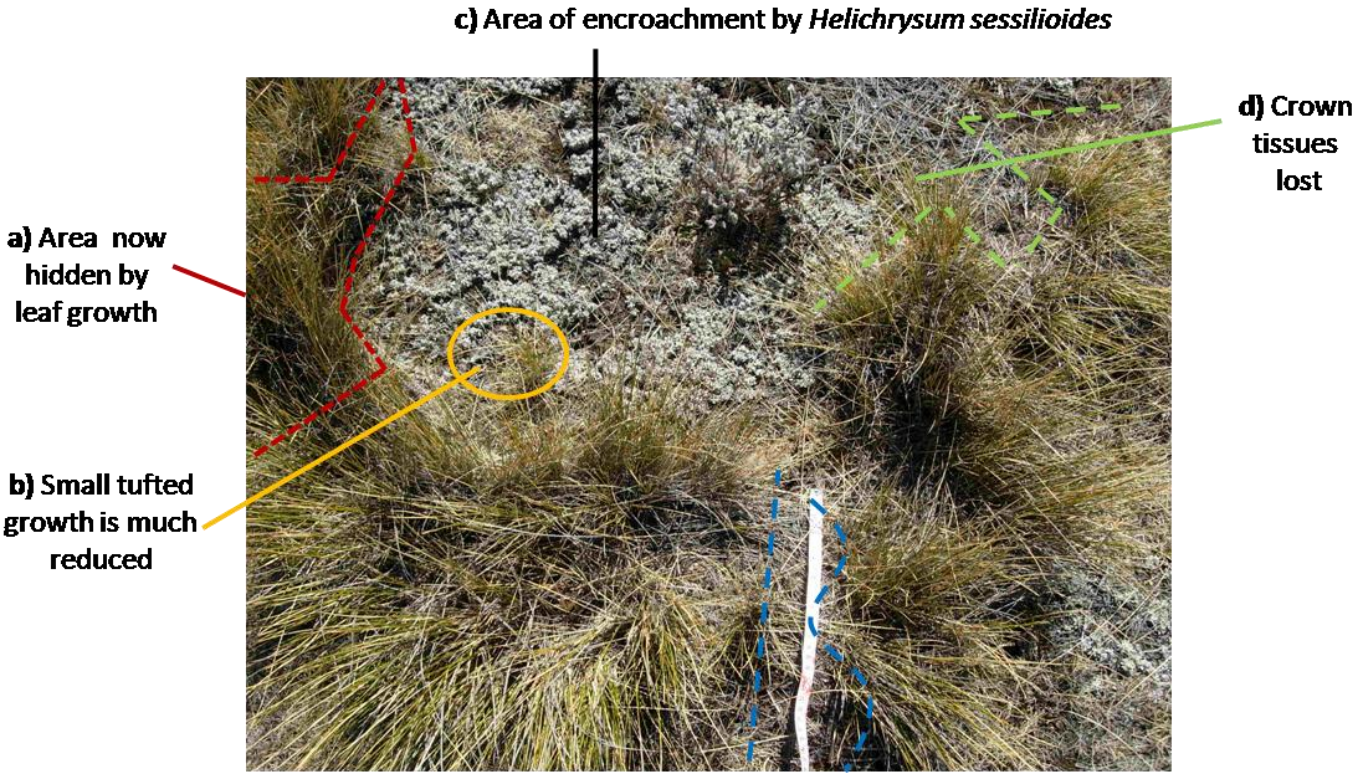


Figure 9.12: The burnt patch near site S.S. 15 months after burning.

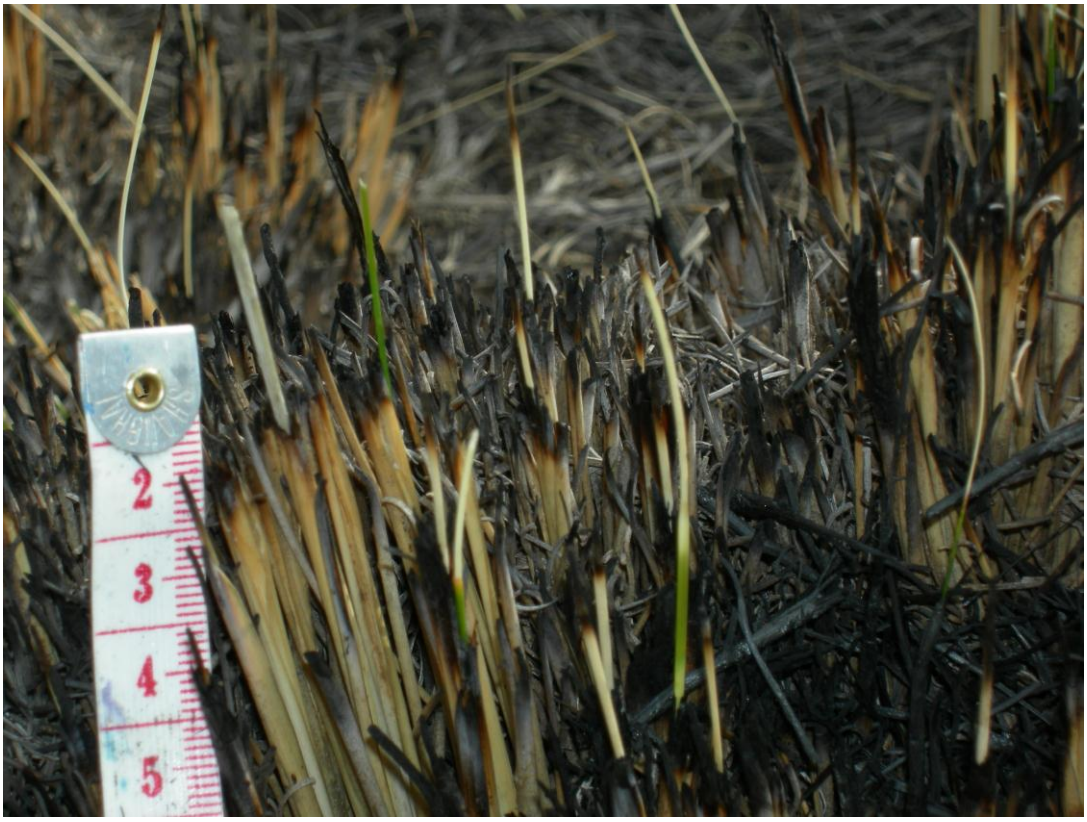


Figure 9.13: New shoots pushing up through the dead matter at the crown.



Figure 9.14: New growth along the edge of a tussock.



Figure 9.15: A burnt tussock two years after burning, highlighting the invasion of small plants into the center.



Figure 9.16: The burnt area of site V.S.E. in late December 2009 (top) and two months after burning in February 2010 (bottom).



Figure 9.17: *M. drakensbergensis* individuals just after burning in late December 2009 (left) and two months after burning in February 2010 (right).

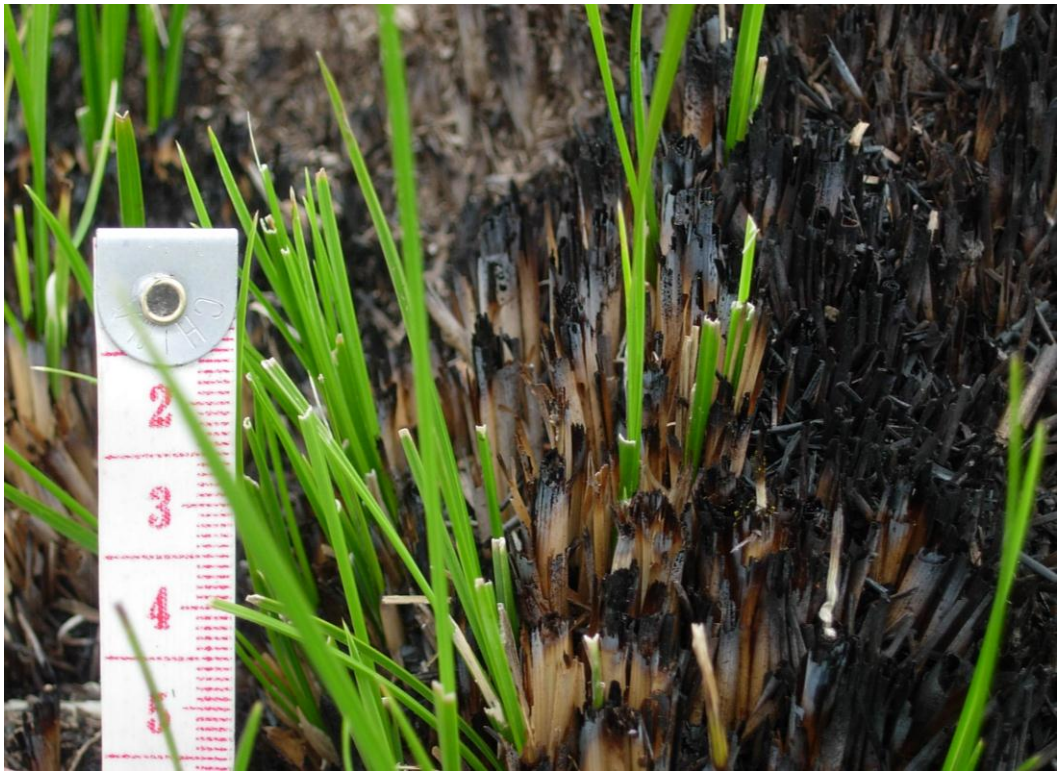


Figure 9.18: New shoots along the edge of a *M. drakensbergensis* tussock which have been grazed.



Figure 9.19: Two years after burning (December 2011), the difference between the burnt and non-burnt tussocks is separated by a dashed line.

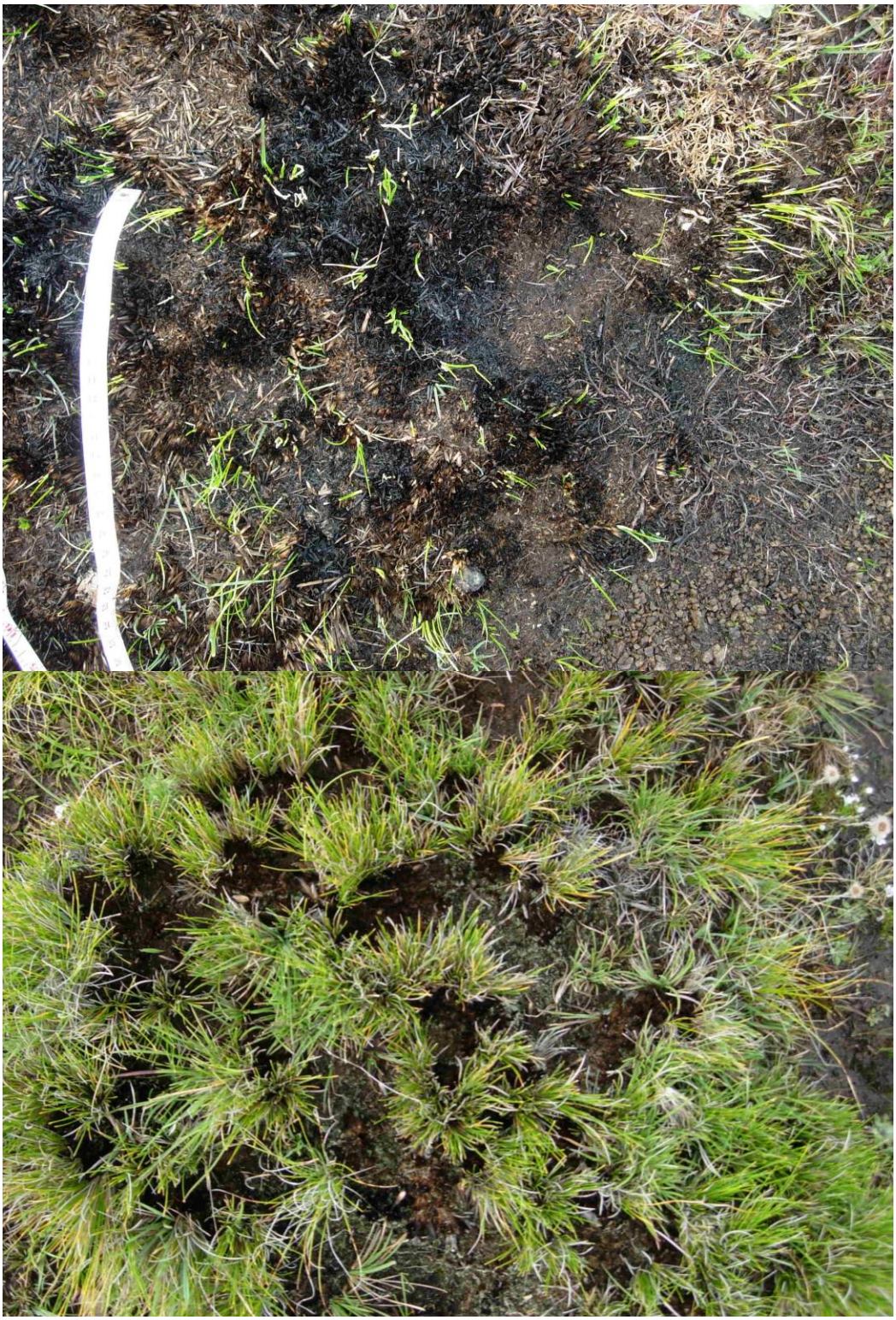


Figure 9.20: A tussock from the burnt A.P. site one month after burning (2 October 2009) and 15 months after burning (December 2010).



Figure 9.21: A *M. drakensbergensis* individual from the burnt A.P. site a month after burning (October 2009) and 27 months after burning (December 2011).



Figure 9.22: A burnt tussock in late December 2010 (left) and again in late December 2011 (right).

Rhodohypoxis baurii was first noticed a month after burning and became abundant on the burnt site and were still abundant more than two years after burning (Figure 9.23). A distinction may be seen in the *Rhodohypoxis baurii* distribution between the burnt and non-burnt areas (Figure 9.23). Months after burning some individuals had still not shown any indication of resprouting and others had only resprouted across a portion of the crown (Figure 9.24). The changes in tussocks was also evident with the majority of changes occurring within the central tissues as central tissues deteriorated and became colonized (Figure 9.25).

A patch of the *M. drakensbergensis* tussocks at Kotisephola Pass were burnt on the 2 March 2009, most likely as a result of herdsmen seeking warmth and protection for their herds (Figure 9.26). Not all the tussocks in the patch were burnt, suggesting that individual *Merxmuellera* tussocks were perhaps chosen for burning. This suggestion is supported by the lack of burning of the inter-tussock vegetation. Many new shoots were observed two months after burning (Figure 9.27), with the majority of the new shoots appearing on the warmer north-facing sides of tussocks (Figure 9.28).

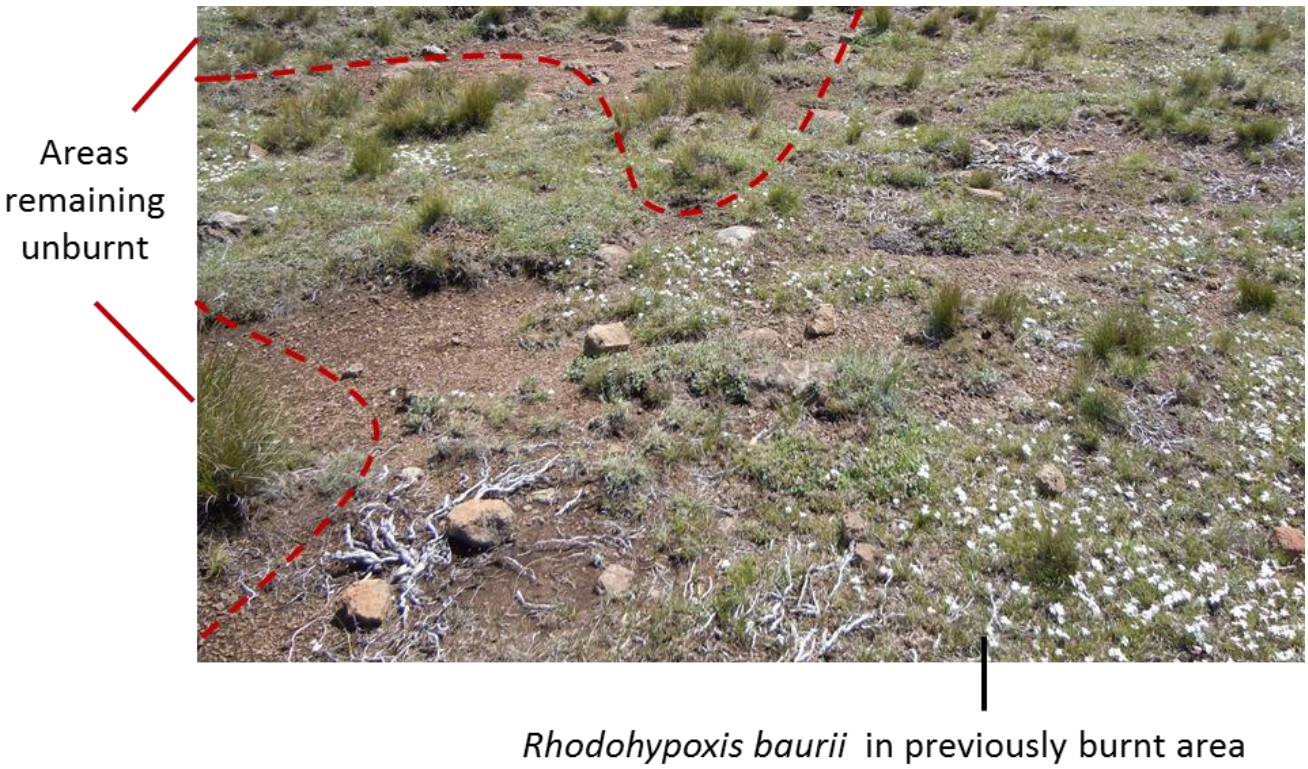


Figure 9.23: The presence of *Rhodohypoxis baurii* in previously burnt areas of site A.P. (December 2010).



Figure 9.24: The burnt tussocks of the A.P. site in February 2010, June 2010, May 2011 and December 2011.



Figure 9.25: Six months after burning some tussocks had no visible signs of resprouting (left) and some had only resprouted on one side (right).

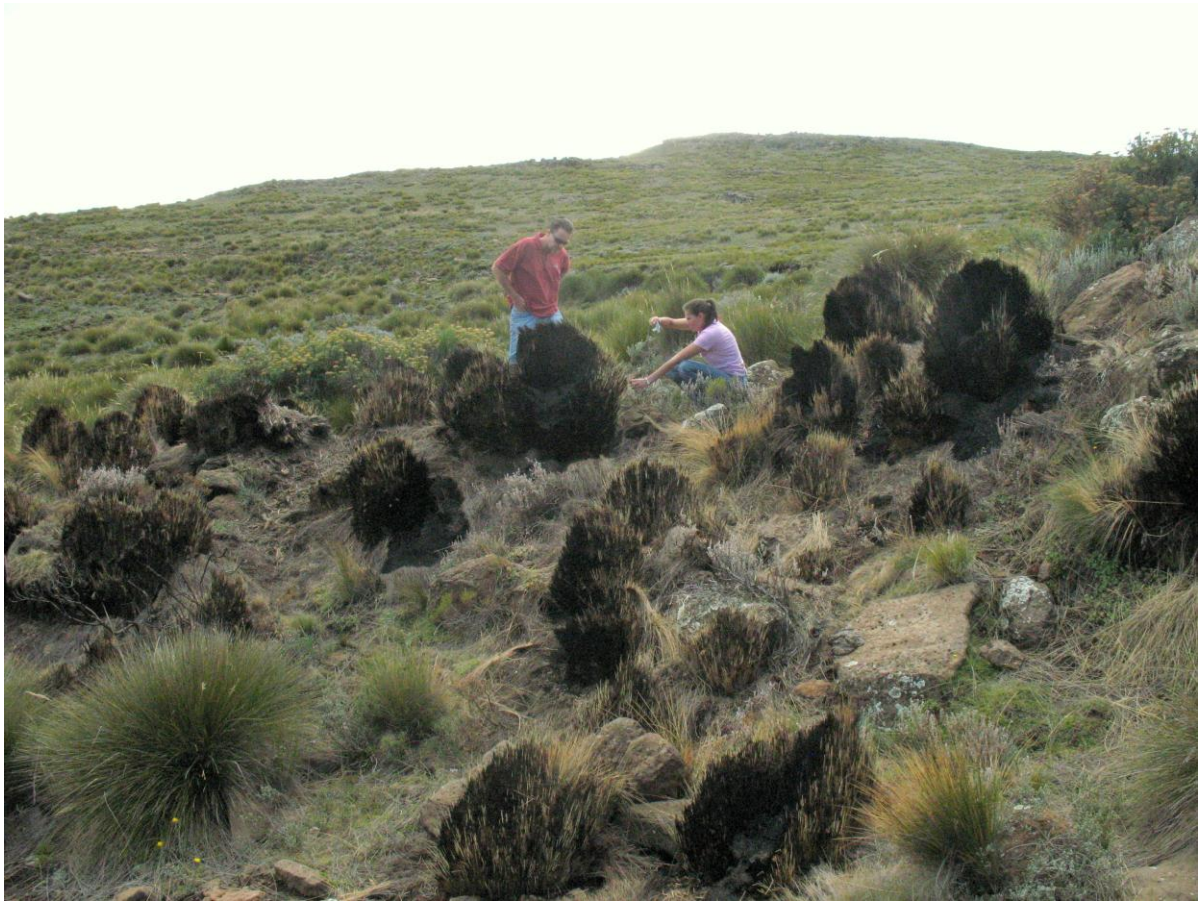


Figure: 9.26: Photographing the burnt patch just below the summit of Kotisephola Pass 2 months after burning.



Figure 9.27: Tussock two days after burning (left) and 2 months after burning at Kotisephola Pass (right).



Figure 9.28: The same tussock as in Figure 9.27 showing the regrowth on the north-facing side on the left of the image and the regorwth on the south-facing side on the right of the image.

9.2.3.2 Monitoring of burnt sites

Once identified, the tillers of the burnt *M.drakensbergensis* communities were measured every second month until December 2011. Initialy, the burnt patch near site S.S. exhibited very rapid re-sprouting in the outer tillers after burning (average height of 15.5cm), yet the tillers nearer the centre had a much slower growth rate (average height of 5.8cm and 4.1cm) (Figure 9.29 and Table 9.3). During the winter periods of the study, the growth rate decreases and during the winter of 2010 a growth rate of -2.14cm/2 months was experienced. A period of accelerated growth was experienced each year between August and October for all tiller positions, especially the central tillers in 2010 (max 4.2cm/2 months) and the periphery in 2011 (4.7cm/2 months). The growth rates remained positive but below 1.5cm/2 months during the summer months with all the tillers of a similar average height. The greatest losses were experienced at the periphery and tillers at the radial midpoint. Losses in plant tissues were experienced by all tiller positions between December 2010 and March 2011 (min -2.3cm/2 months at the periphery). Throughout the study period at this burnt patch, the central tillers had the lowest average height and lowest growth rates during the summer months.

Table 9.3: Initial growth rates of the tillers of all burnt sites in the 2 months following burning.

Site:	Periphery	Radial midpoint	Centre	Month:
S.S.	-2.0	8.1	6.4	May - August
V.S.E.	7.3	4.4	4.0	December - February
A.P.	7.6	6.4	5.0	August - October
K. P.	7.6	8.1	7.2	March – May

The summer burnt *M. drakensbergensis* tussocks of site V.S.E. have little significant difference in average height between differing positions during resprouting (periphery = 4.2cm, radial midpoint = 3.9cm and centre = 3.8cm). Yet, over the first two months, the peripheral tillers have a faster growth rate than that of the central tillers (7.3cm/2 months as opposed to 4.0cm/2 months) (Table 9.11). Spring growth rates reached a maximum between August and October 2010 and 2011 with the central tillers growing fastest in 2010 (1.5cm/2 months) and the peripheral tillers in 2011 (4.0cm/2 months). All the positions experience no growth or losses between December 2010 and March 2011 (periphery = 0cm/2 months, radial midpoint = -0.6cm/2 months and centre = -1.2cm/2 months), and continue until spring 2011. Between October and December 2011 the growth rate at the periphery drops to 1.0cm/2 months for both the central and radial midpoint tillers.

Burnt area near site S.S.:

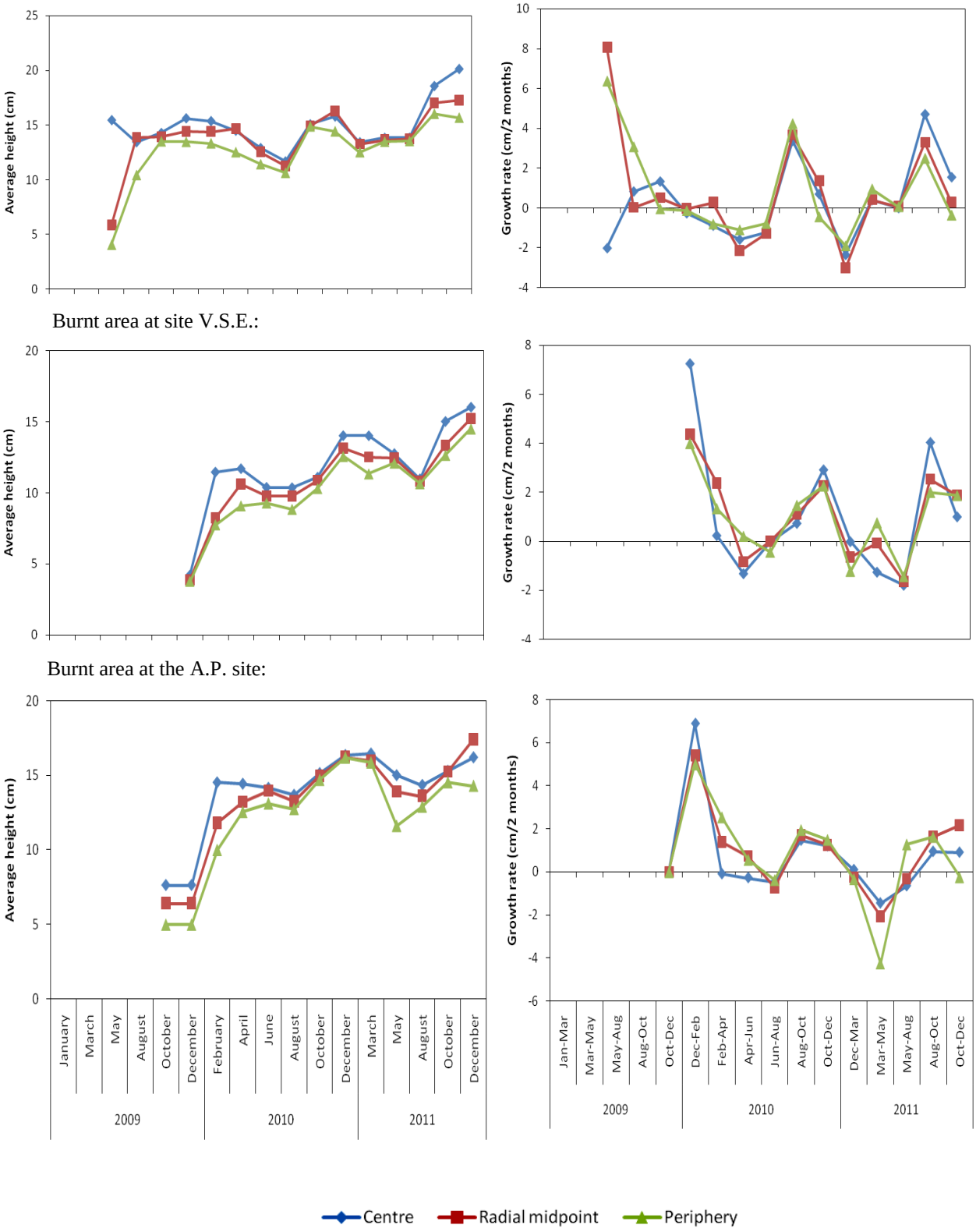


Figure 9.29: Average height of the burnt tussock tillers and growth rates of the outer, radial mid points and middle tillers.

The burnt tussocks at the A.P. site were experienced almost no growth during the first two months after an initial growth spurt after burning (Figure 9.29). The lack of growth was observed to be the result of grazing as leaf blades had flattened edges. This is followed by a larger growth period between December 2009 and February 2010 (max 6.9cm/2 months). The growth was led by the peripheral tillers. The average height declined in June and August 2010 and 2011, and was reflected by negative growth rates and similar growth rates among all tillers. From October to December, the growth rates increased, especially by the central tillers in 2010 (max 2.0cm/2 months) and the radial midpoints in 2011(max 2.2cm/2 months). A large decline in the growth rates was observed in May 2011, with central tillers experiencing the greatest losses (-4.2cm/2 months as opposed to a drop of -1.5cm/2 months at the periphery).

The burnt *M. drakensbergensis* tussocks of the Kotisephola Pass site similarly indicate a rapid growth after burning, with little difference between tiller positions (periphery = 7.6cm, radial midpoint = 8.1cm and centre = 8.2cm) (Figure 9.30 and Table 9.3). The average growth rate decreases over the winter of 2009 and the average height in December remains similar to that before winter. The highest growth rate in October 2010 was measured at the tillers positioned at the radial midpoint (2.8cm/2 months). The highest growth rate between December 2009 and April 2010 was recorded at the periphery (maximum of 2.3cm/2 months) and this is reflected in the average height (maximum of 11.6cm) and again in 2011 (average height of 14.9cm). The average heights decreased and become similar between June and August 2010 and 2011.

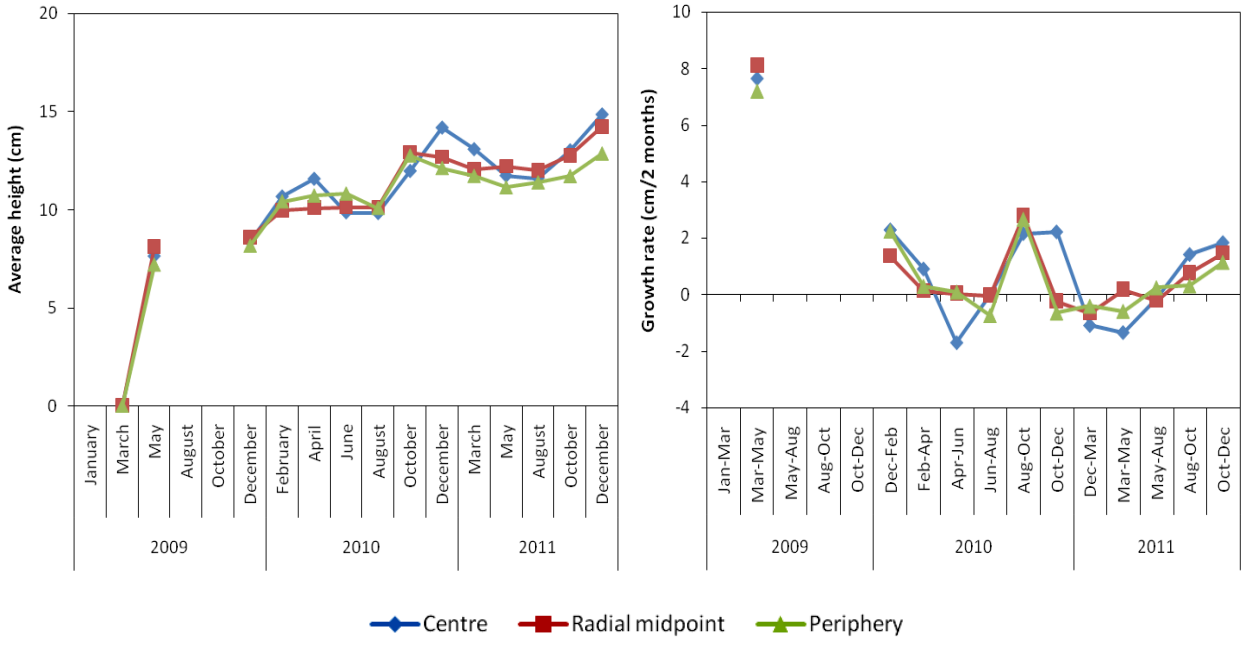


Figure 9.30: Average height of the burnt tussock tillers and growth rates of the outer, radial mid points and middle tillers for the burnt tussock grasses of Kotisephola Pass.

The multiple regression of the recently burnt site growth rates indicated that the influencing factors were an extremely good fit to the data ($p<0.0001$). The adjusted R^2 value indicated that the influencing factors explained 26% of the variation in the growth rates. Average temperature and precipitation were extremely significant factors influencing the growth rates ($p<0.0001$). Additionally, average wind speed was also a significant contributor ($p = 0.032$). The slope values (b-values) indicated a direct relationship between the growth rates and average temperature, and indirect relationships between average precipitation and wind speed.

Table 9.4: Multiple regression analysis results for recently burnt sites.

Factor	Standardized coefficient (b)	SE	t	P-value
Site	-0.205	0.029	-0.824	0.412
Tiller location	0.016	0.014	0.184	0.855
Season	0.111	0.035	0.813	0.418
Altitude	0.177	0.000	0.931	0.354
Average temperature	1.123	0.008	6.462	0.000†
Average precipitation	-1.188	0.001	-5.004	0.000†
Average wind speed	-0.308	0.011	-2.180	0.032*
Site aspect	0.080	0.034	0.475	0.636
(constant)	0.208	0.498	0.418	0.677

SE:Standard error; P-value: significance value.
*P<0.05; **P<0.01; ***P<0.005; †P<0.001

Individual *M. drakensbergensis* tussocks were divided into quarters according to their aspect and the average height of each quarter calculated and plotted over the course of the study period (Figure 9.31). The burnt patch near site S.S. indicated that initially after burning the south side of the tussocks grow faster (average height = 23.5cm) than the other aspects, especially the western side (average height = 11.8cm). The south-facing sides were usually tallest during the winter months, and the east or west-facing sites during the summer months. The north-facing aspects usually had the lowest average heights. The burnt individuals of site V.S.E. began to resprout with almost no difference between aspects (Figure 9.32). However, the south-facing side had the greatest average height two months later (10.7cm), while the other aspects all remained similar. The south-facing slopes were generally tallest until late winter each year, when all aspects were similar. The north and west-facing aspects were usually the lowest. The growth rates of the south-facing sides were either the highest when there was a general increase in growth or the lowest when there was a decrease. The north and west-facing growth rates remained the most consistent

throughout the study period. The A.P. site indicates similarity between the aspects until April 2010, when north-facing sides were the shortest (9.3cm) and west-facing were the tallest sides (16.1cm). During the summer months of the study period the average heights became similar. March 2011 brought a division of the average heights, with east-facing (18.4cm) and south-facing sides (18.0cm) becoming taller than west (15.2cm) and north-facing sides (12.6cm). In December 2011, the west facing sides became the tallest (19.6cm) while the other sides remained close together. The growth rates of the north and west-facing sides were generally highest in summer except for the decline between December 2010 and March 2011. Similarly, the growth rates in winter were generally highest on the east-facing sides of the tussocks.

The burnt *M. drakensbergensis* tussocks of the Kotisephola Pass site differ from the other burnt patches as the highest growth rates were experienced on the north- and west-facing tillers (8.8cm/2 months and 8.0cm/2 months respectively) (Figure 9.32). The following summer (December 2009) the north- and west-facing tillers had the highest average height (9.1cm and 9.4cm respectively), yet the highest growth rates were experienced on west- and east-facing tillers (2.5cm/2 months and2.2cm/2 months). However, the high growth rate does not last and decreased to 0.3cm/2 months on the north-facing sides and 0.1cm/2 months on the west-facing sides, with the summer of 2011 following a similar pattern. A decrease and grouping of the average heights was observed in June and August 2010, as well as between May and August 2011, as a response to low growth rates. The north-facing tillers were the last aspect to reduce their average height and experienced a negative growth rate between June and August 2010 (-1.2cm/2 months). Between August and October 2010, the north and west-facing sites experienced the highest growth rates (4.0cm/2 months and 3.8cm/2 months respectively).

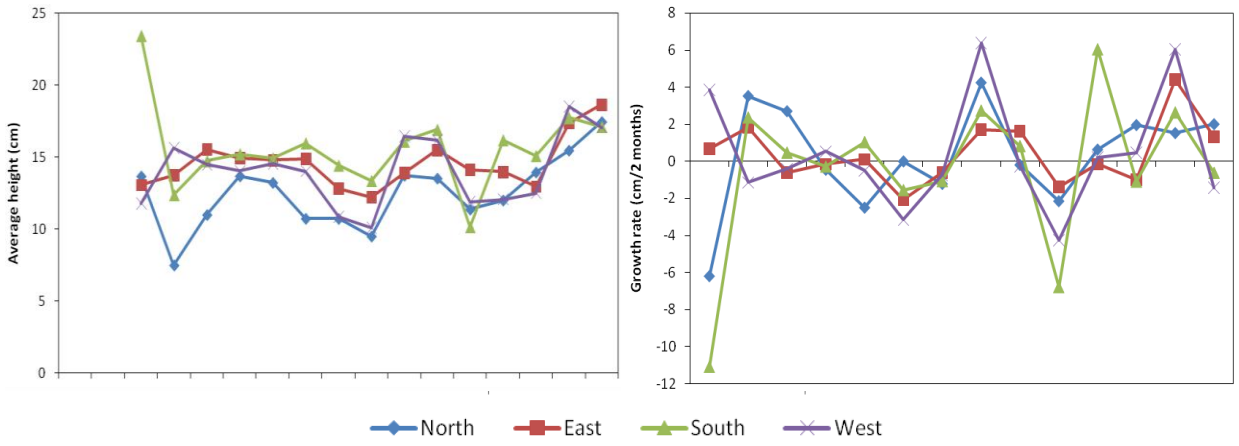
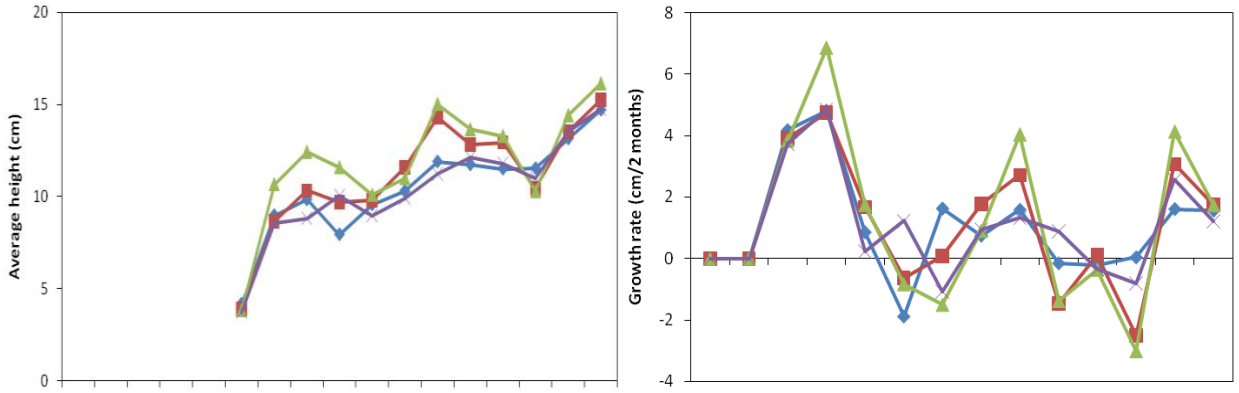
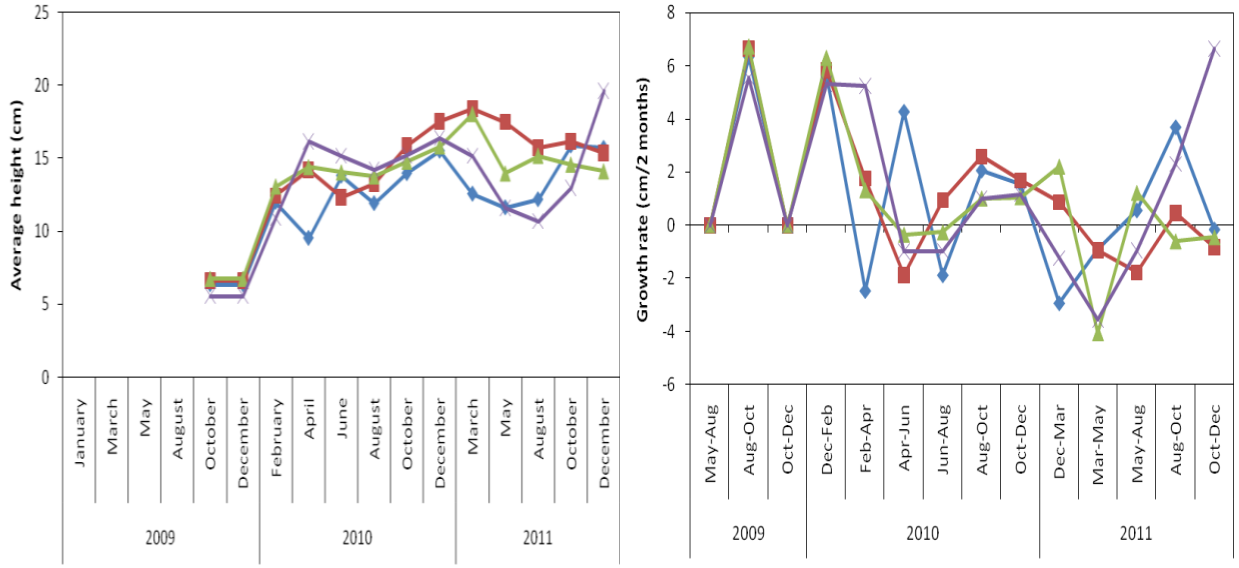


Figure 9.31: Average tiller height for each aspect of the burnt tussocks at site S.S. over the study period.

Burnt area of site V.S.E.:



Burnt area of the A.P. site:



Burnt areas of the K.P. site:

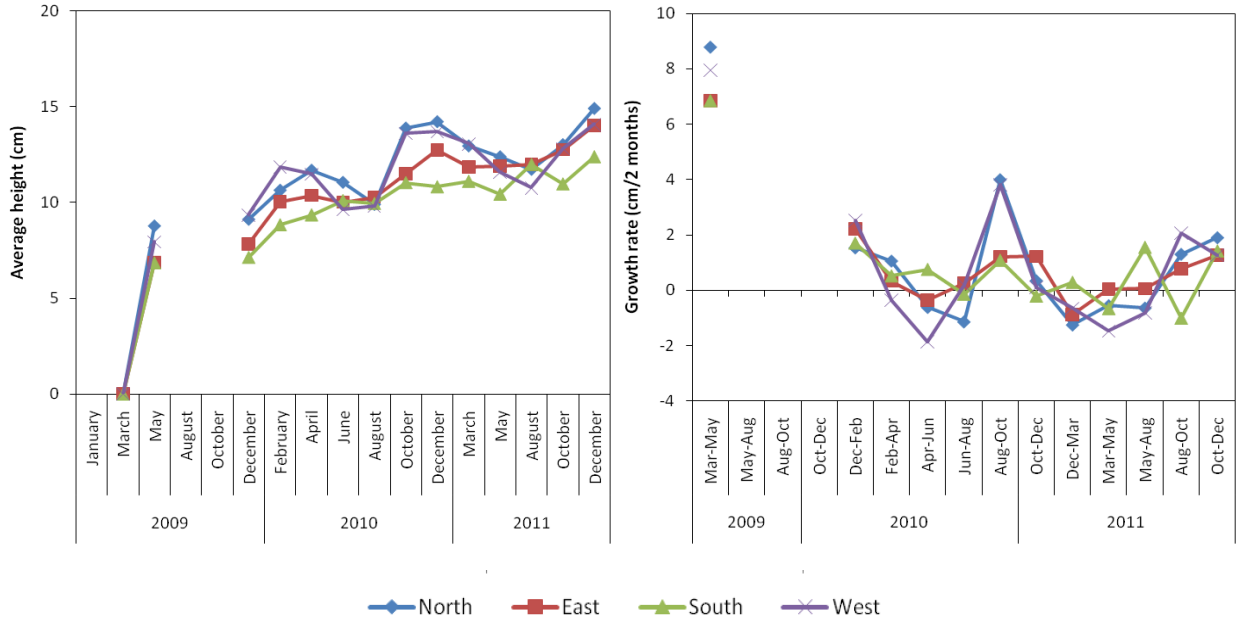


Figure 9.32: Average tiller height for each aspect of burnt tussocks over the study period for the burnt tussocks at sites V.S.E., A.P. and K.P.

9.2.4 Predicted time for recovery

The time required for the burnt *M. drakensbergensis* tussocks to reach pre-burn average heights was calculated in years (Table 9.5). The peripheral tillers of the A.P. site require the most time to recover (12 years) while those at the periphery of the burnt site near S.S. require only 2½ years to recover to pre-burn heights. Overall, the burnt patch near site S.S. requires the least time to recover, whilst site V.S.E. and K.P. burnt tussocks require the most time to recover (average of 7.2 years).

Table 9.5: Predicted recovery time required to reach pre-burn average heights at the completion of the study for tillers at the periphery (P.), radial midpoint (R.M.) and centre (C.).

	Final Height (December 2011) (cm)			Growth rate after Initial Spurt (cm/2 months)			Pre-burn Average Height (cm)			Time to Reach Height (years)		
Site:	**P.**	**R. M.**	**C.**	**P.**	**R. M.**	**C.**	**P.**	**R. M.**	**C.**	**P.**	**R. M.**	**C.**
S.S.	20.1	17.3	15.7	0.5	0.3	0.4	22.2	24.4	27.1	2.6	7.9	5.3
V.S.E.	16.0	15.2	14.5	0.5	0.7	0.7	37.8	41.3	44.8	7.9	6.2	7.5
A.P.	16.2	17.4	14.3	0.2	0.6	0.4	28.1	29.7	28.5	11.8	3.7	5.5
K. P.	14.9	14.3	12.9	0.6	0.5	0.4	29.8	36.4	38.6	4.2	7.2	10.1

9.4 Discussion

9.4.1 Growth rates

There are no observations in the literature of recorded growth rates for *M. drakensbergensis* or for other alpine tussock grasses within the Lesotho highlands. However, similar tussock grasses in New Zealand (*Chionochloa rigida*) have an average seasonal growth (September to April) of between 34.1cm (Rowley, 1969) and 28.0cm (Payton and Mark, 1979). During a similar seasonal time period, the tussock grasses in the current study were only able to produce a highest growth of 12.0cm at site S.E. and 6.7cm at site S.S. The findings by Payton and Mark (1979) roughly equates to 7.0cm per 2 months, a figure only reached on average by site S.S. (March to May, 2009) and site V.F. (December 2009 to February 2010). A comparison of individual growth rates suggests that a growth rate of 7.0cm/2 months is possible by *M. drakensbergensis.* However, these tussocks were subjected to grazing while those in the study by Payton and Mark (1979) were not. Hence, the impact of grazing on the average seasonal growth rates may be quite severe. *Chionochloa macra*, an alpine New Zealand tussock, has an average growth rate of 1.65mm/day (Williams, 1977), a rate almost three times higher than the maximum observed by the current study. Similarly, a maximum undisturbed growth rate of approximately 13.8cm/2 months

(2.3mm/day) was measured by Hofstede *et al.* (1995) in the alpine grasslands of Colombia, while moderately disturbed and heavily grazed conditions resulted in approximate growth rates of between11.4cm/2 months (1.9mm/day) and 2.4cm/2 months (0.4mm/day). The results from the current study support the possibility that the average growth rate for *M. drakensbergensis* could potentially be higher than that measured when excluded from grazing.

Plant growth is the result of photosynthesis, allocation of resources to plant components and results in differing plant morphologies, while still allowing for some resource loss (Monteiro *et al.*, 2011). The rate of photosynthesis is influenced by temperature, moisture availability, sunlight duration and strength, factors which may be influenced by changes in the seasons. The graphs for growth at each site (Figures 9.2, 9.3, 9.4 and 9.5) indicate distinct seasonal trends. Growth increased during the month of October and continues to increase slowly until February or March, similar to the pattern observed in New Zealand by Williams (1977). Increased soil temperatures above 12^{o}C at depths of 10cm for most of the day were required to initiate accelerated growth rates in spring in the North American Great Basin (Eissenstat and Caldwell, 1987). The continued warm soil and air temperatures combined with a longer photoperiod results in rapid growth as plants take advantage of the favourable conditions. The cooler temperatures indicating the approach of winter causes a decrease in the growth rates at each site. Between June and mid August each year, a very low rate of growth was observed at all sites (<1.0cm). Harsh climatic conditions may result in slower growth rates and a greater allocation of resources to below-ground structures (Monteiro *et al.*, 2011). The slower growth rate is often the result of plants favouring survival rather than production (Monteiro *et al.*, 2011). However, growth continued all year round, including the winter months although growth was minimal. This result is not unique and has previously been noted in Austtralian grasslands by Williams and Biddiscombe (1965) and South American savannas by Sarmiento (1992).

The increase in the growth rate at the start of spring is typically between late August and early October each year. Some sites record later onsets of increased growth rate, such as site V.S.N. in 2009 and the K.P. site each year when the growth rate only increased between late October and December. The observation at the K.P. site was not unexpected as this site is at a higher altitude and associated with lower temperatures. There was more variation in the effects of the onset of winter on the growth rates, yet most sites were observed to have a slower growth rate between March and May, which decreased further between May and August. Some sites indicated earlier

decreases in growth, such as site S.S., which decreased between December and February 2010 and between February and March 2011 for site H.E.

Losses in plant matter are indicated as negative growth rates on the graphs (Figures 9.2, 9.3, 9.4 and 9.5) and may be the result of grazing, damage by wind or passing livestock. The losses are not confined to one particular time of the year, yet become less severe during the winter months (e.g. for the A.P. site: periphery = -0.2cm, radial midpoint = -0.3cm and centre = -0.4cm between June and August 2010). It is likely that the greater losses observed in the summer, autumn and spring months were the result of intensive grazing (e.g. for the A.P. site: periphery = -5.74cm, radial midpoint = -5.78cm and centre = -2.69cm between December 2009 and February 2010). During these months, the study valley is heavily grazed (see Chapter 5), and thus it is likely that losses were as a consequence of such activity. During the winter months, the climatic conditions become harsh and some of the livestock are moved to lower pastures. Thus, winter losses may be the result of damage and breakage caused by increased wind speeds and durations observed during May (4.73ms^{-1} in 2010 and 2.37ms^{-1} in 2011), June (6.29ms^{-1} in 2010 and 3.53ms^{-1} in 2011), July (5.14ms^{-1} in 2010 and 4.83ms^{-1} in 2011) and August (6.55ms^{-1} in 2010 and 4.70ms^{-1} in 2011; Table 4.4, Figure 4.6 and 4.7). This would be observed as small decreases in the growth rate. Some livestock does remain within the Sani study region, mostly sheep, hence grazing may be the cause of the large negative growth rates during the winter months as other more palatable forage may be scarce.

The greatest average growth rate across the years of study were indicated for site V.F. (periphery = 49.4cm, radial midpoint = 54.0cm and centre = 51.0cm) and was reflected in the average heights and positive growth rates for 2009, 2010 and 2011. Site V.F. is situated on the banks of a non-perennial stream and is relatively sheltered from the wind. The site is also situated on relatively flat land and not along a grazing route (sees Chapter 5). The lack of disturbance within this site, as well as potential access to water during the summer months, may account for the height these *M. drakensbergensis* tussocks are able to achieve. Tillers at sites S.E., V.F., the upper and lower M.F. sites, the A.P. and the K.P. sites experienced growth of over 20.0cm during the course of 2009. Yet, these growth rates were only measured once at each site as grazing often followed. Site V.F. and the lower M.F. site were the only two sites observed to grow more than 10.0cm in 2010. These two sites are at low altitudes and sheltered from the wind. An increase in average height across a year or growing season is most often the result of consistent positive

growth, which allows low levels of grazing to be overcome. However, two or more instances of severe grazing were observed to have a negative effect on overall growth.

9.4.2 Growth rates between individuals

Here, the standard deviation is used to indicate the amount of variability in growth rates observed for each time period, each site and each position within the individual tussocks (periphery, radial midpoint and centre). The results indicate a general decline in the standard deviation values, which indicates less variability occurring within the data across the years of study (Figure 9.2, 9.3, 9.4 and 9.5). Variability may be the result of influential factors on the growth rates, such as aspect, soil moisture, radiation strength, photoperiod and many others. Yet, ultimately losses and growth determine the variability within the data and the greater the mixture of positive and negative values the more variable the data are likely to be. The decrease in the average standard deviation values suggests that either the growth rates are becoming similar or that grazing is becoming more widespread and more frequent, with greater intensity within each site.

The average standard deviation confirmed the seasonal trend observed within the larger standard deviation data set (Figure 9.6). The variation is observed to be at a maximum each year during the growing season, more particularly between the mid and late growing season. Consequently, the standard deviation values decreased during the winter months, reaching a low towards spring. The increased variation may be a consequence of competitive tussock growth taking advantage of the optimum conditions, and the effect of grazers on this growth. The number of livestock in the study valley increases during the summer months and thus the likely quantity of grazing resource intake may also increase. The combination of uneven grazing patterns and competitive growth will result in varied growth rates. However, in winter the livestock are mostly moved to lower altitudes and the grazing intensity consequently decreases. Growth rates are also lower during winter due to cold conditions and limited soil moisture. Together, the lower growth rates and lower grazing rates are likely reasons for lower levels of variation.

Site S.E. recorded both the highest (15.11cm/2 months between December 2009 and February 2010) and the lowest (0.42cm/2 months between October and December 2010) standard deviation values, hence, this site was recorded to have the greatest variability between individual *M. drakensbergensis* tussocks. Site S.E. is the most exposed of all the sites to grazing, wind, rain, hail or snow, as little shelter is offered by the landscape or even other tussock individuals. Yet the low number of individuals means less competition for resources and thus better growth during

favourable periods than at other sites where competition is high. Therefore, it is not surprising that this site has the highest variability. Data from site S.S. indicate that the peripheral tillers generally have lower standard deviations than the central tillers until June 2010. After this, tillers at the periphery have higher standard deviations than in the centres. Sites S.E., S.Str. and V.F. also indicate this trend, but to a lesser extent. The low standard deviations found on the periphery may be the result of similar growth rates and grazing intensities, while the high standard deviations may be the result of uneven grazing. The central tillers would be more protected from grazing by the tussock size and the prickly nature of the leaf tips, hence the central tillers at site H.E. have lower standard deviations from December 2009 onwards.

Sites H.W. and the A.P. site indicate a seasonal trend in the position of the highest and lowest standard deviations. During the summer months, the highest standard deviations are found along the edges of the tussocks, while in winter they may be found at or near the centers. This may again be the result of increased grazing and increased competitive growth during the summer months, and a decrease in both in the winter months. Yet, the K.P. site suggests an opposite trend, perhaps as a consequence of altitude. Grazing was not commonly observed at this site and may be responsible for the different result. No observable trends were noted at sites V.S.N., V.S.E. and for the upper M.F. site. The lack of a trend in the variation may simply be due to a lack of predictable grazing. All three of these sites are situated on slopes and this may have an influence on the growth rates. Grazers prefer flatter slopes and thus grazing along the valley sides may be reduced and perhaps restricted to livestock moving between pastures.

9.4.3 Growth rates across individual tussocks

The two way repeated measures ANOVA indicated that different tiller positions within the tussocks were not significantly different from the mean (Table 9.2). Thus, there is no significant difference in the average growth rate between the periphery, radial midpoint and central tillers. This result is surprising after as trends appear to be present within the data. However, this result is in agreement with a New Zealand study comparing leaf elongation rates within tussock grasses (Mark, 1965). The *M. drakensbergensis* tussocks measured belong to communities and have different positions within these communities. Some measured tussocks are located in the central regions of the community, while others are located along the community periphery. The central tussocks are more likely to be more sheltered than the tussocks at the periphery of the community and thus the growth advantages and disadvantages of the peripheral tillers become reduced.

Solar radiation is an important component of photosynthesis and thus is required for growth (Robinson, 2007), yet light is not distributed evenly within a tussock. Shading by tillers means that central regions near the crown may receive only 10% of the potential light (Monteiro *et al.*, 2011), reducing photosynthetic activity in the central regions and hence reducing the growth rate (Ryel *et al.*, 1994). The access to light and thus photosynthetic activity increases towards the periphery of the tussock (Ryel *et al*., 1994; Monteiro *et al.*, 2011). Thus, it is expected that growth will be focused on the outer edges of the tussock, supporting the findings in this study. However, the peripheral tillers are the most exposed to harsh conditions and the impacts of herbivores or passing livestock, as observed in western Bolivia (Monteiro *et al.*, 2011). Yet, growth rates are often higher in younger tillers, and the majority are found along the periphery (Williams and Biddiscombe, 1965; Gatsuk *et al*., 1980), thus peripheral growth may be high and was supported by the results. After defoliation, growth rates initially increase and gradually slow until the optimum length for that species is reached (Williams and Biddiscombe, 1965). This study supports the literature and indicates increases in the growth rates after grazing.

Resource competition and sharing between tillers may also result in different growth rates and emerging tiller numbers within the tussock structure *Schizachyrium scoparium* tussocks in Texas (Derner *et al*., 2012). Initial observations indicate that at site S.E., the central tillers of individual *M. drakensbergensis* tussocks were taller than those at the periphery late in the growing season and during the winter months, while the peripheral tillers were taller during the spring and early summer months (Figures 9.2, 9.3, 9.4 and 9.5). The centres of the tussocks are more protected than the periphery from damage by wind, hail and passing livestock. The thermal properties of the tussock grasses also suggest slightly warmer conditions within the core, a factor which would assist growth during winter. The outer edges are more exposed to an increased frequency and velocity of winds during the winter months and this may result in losses of plant matter. Despite this, the periphery is the least shaded by other leaf blades of the same plant, particularly at site S.E., due to their upright shape, and thus are able to take full advantage of snow melt and warmer temperatures, as well as increased duration of sunlight in early spring. Hence, the peripheral tillers may begin their spring growth earlier in the growing season than the centres. However, the peripheral tillers are also the first to experience cooler temperatures and the strong winds characteristic of the winter months. This trend was also observed at the lower and upper M.F. sites.

At many sites (sites V.S.N., H.E., H.W. and V.F.) the peripheral tillers were taller than the centres, unless grazed, as noted by Volesky (1994). This may be due to less competition for moisture, light, nutrients and minerals, necessary for growth along the edges when compared to the central regions. Thus, it is not surprising that the periphery is taller than the central tillers, especially where the studied tussocks are sheltered from wind, hail and passing livestock by topographical features or other individuals. The peripheral tillers of the individual tussocks at K.P. encompass both of the ideas put forward as the leaves were lowest in winter while the centres were lowest in summer. The low height of the periphery as well as negative growth during the winter months suggest breakage is high, while the centres are protected. However, during the summer months the peripheral tillers are less shaded than the central regions and face less competition for resources. Each tiller position has advatages and disadvantages to growth, which according to the lack of significant difference between them, may even out resulting in similar mean values.

The lack of significant differences between the means for each site is not surprising due to the closeness of the values of all the sites, especially between December 2009 and August 2010. The results also suggest that factors such as site aspect, slope angle, altitude and soil texture and fertility may not have any influence on the growth rates experienced at each site. However, the average height may be more influenced by aspect than the growth rates as observed in Chapter 7. The sites used in the current study may perhaps not be far enough apart for differences in precipitation to be significant. The soil temperature was not monitored in this study, but may also be a potential influencing factor within the region. The factors influencing the growth rate will be discussed in greater detail in the following section.

9.4.5 Relationship between growth rates, burning history and other variables

A multiple regression analysis was undertaken to indicate the contribution made by potential influencing variables on the measured growth rates. The low value indicated by the adjusted coefficient of determination (R^2 = 0.095) may indicate that additional factor, such as grazing, may play a greater role in explaining the variation of growth rate. The lack of significant relationships between burn frequency and the time since each site was last burnt are surprising. Burning may result in reduced productivity, potentially from reduced growth rates (Vogl, 1974, Trollope, 1999). Yet, growth is likely to occur if the parameters for photosynthesis, metabolic functioning and resource uptake have been met. The *M. drakensbergensis* communities at each

site within this study have not been burnt recently (>2 years), suggesting that growth rates potentially altered by burning may become more significantly influenced by other variables within 2 years after resprouting.

Temperature extremely significantly ($p<0.0001$) contributes to the variation in the growth rates and is not surprising. Additionally, average precipitation and season contribute highly significantly to the relationship ($p<0.005$). Both temperature and rainfall are linked to season, and thus it is not surprising that all three variables are significant contributors. Above ground tussock cover increases during wet months and decreases during dry months in semi-arid Australia (Bennett, 2002). In Lesotho, the pattern of wet and dry periods follows the summer and winter seasons respectively, and was reflected in the growth rate patterns observed at all the sites. The growing season occurs when the availability of growth requirements are at an optimum. The growing season length is influenced by the solar angle and thus, the amount of solar radiation received by an area depends on the aspect, slope angle, albedo, angle of the sun, altitude, latitude and vegetative cover (Nagy and Grabhorr, 2009). The strong radiation received by high altitude plants may curb their growth (van Zinderen Bakker, 1981), which suggests that as radiation increases from winter to summer, the growth is first accelerated then declines slowly, as was observed in this study.

Tussock growth has been demonstrated to correlate with temperature fluctuations in both the air and the soil, with positive growth occurring under warm temperatures (Williams and Biddiscombe, 1965; Everson and Everson, 1987). Longer periods of warm temperatures are also influential in promoting consistent positive growth, as observed in Canberra, Australia (Williams and Biddiscombe, 1965). The seasonality of the growth rates observed supports these theories. As the warmer summer months approach, growth rates increase rapidly towards the middle of the growing season and then decline slowly over the course of the growing season, supporting the alpine findings of Williams (1977), in New Zealand. The growth rates are higher in spring than autumn in both this study and that of Williams (1977), suggesting that general trends in temperature may have a more important impact than daily temperatures. However, these trends are not always observable as grazing increased during these months and losses to the grasses may have altered the growth rates as well as reduced the quantity of plant tissue at each site. Seasonality also significantly affected the amount of variation between individual tussock grasses, as well as individual leaves. The variation decreased during winter when growth was at a minimum and increased during the summer months, and is supported with similar results by

Mark (1965) in New Zealand and Eissenstat and Caldwell (1987) in Utah, U.S.A. These authors related their findings to snowmelt and increasing soil temperatures. The increased variation in growth rates, in this study and that of Eissenstat and Caldwell (1987), may also have been due to selective and random grazing, differences in micro-climate, soil moisture differences within a site, soil nutrient distributions and competitiveness.

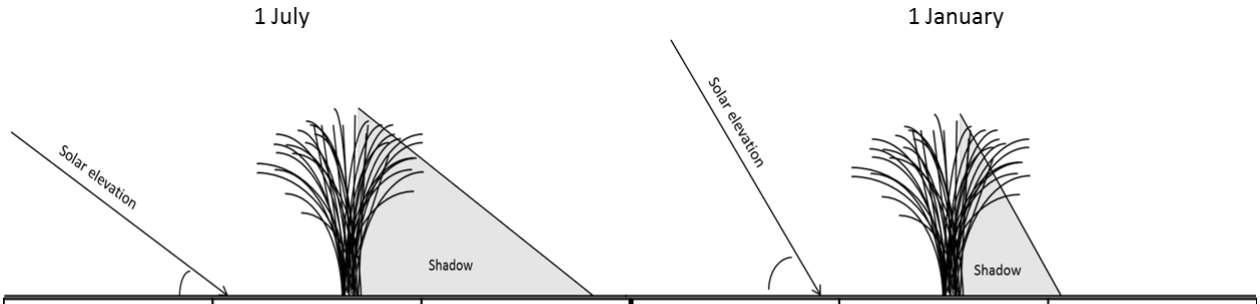
Precipitation may be more important for growth than temperature, and hasbeen strongly linked to grass growth worldwide, as it has in this study (Radcliffe and Baars, 1987; Gibson, 2009). Differences in precipitation and soil temperatures are responsible for the differences in growth rates in both North American and British tussock grasses (Radcliffe and Baars, 1987). Yet those studies were not at high altitudes or in alpine areas. However, no relationship was found between precipitation and soil moisture and growth in the Underberg region of KwaZulu-Natal (Everson and Everson, 1987) or in the Kansas prairies (Briggs and Knapp, 1995). Similar results have been observed by others such as Mark (1993), who believed strongly that air temperature and soil temperatures have the greatest control on alpine tussock growth rates. Soil richness is also believed to be an influential factor on grass growth (Guàrdia *et al*., 2000; see Chapter 4).

Altitude contributes significantly in explaining the variation of growth rates. This is not surprising as altitude is able to influence ambient temperature. The relationship is direct indicating that as altitude increases so does temperature, yet the b-values indicate that this change is marginal (b = 0.005). Thus the increased growth rates may be influenced by an additional factor linked to altitude. Site K.P. is at the greatest altitude (±3230m a.s.l.) and grazers were seldom observed at this site due to cooler temperatures and increased exposure to wind. Thus, grazing patterns may be linked to altitude, with the lower altitudes grazed more frequently, and is supported by the data gathered in Chapter 5. The literature suggests that the cooler conditions of high altitudes are able to slow the growth rates (Körner, 2006). However, the lack of a relationship indicated that this is not true for this species of tussock. *M. drakensbergensis* is a specialized plant which is highly adapted to live at high altitudes, so this result is not surprising, despite not being expected.

The contribution of the aspect of each site towards explaining the variation in the measured growth rates of each site were investigated using a multiple regression analysis (Table 9.9). The results indicate that site aspect does not significantly influence the observed growth rates. Thus, aspect exerts a direct influence on plant growth through differences in temperature generated by

the amount of solar radiation received (Morris *et al.*, 1993). The vegetation may be both positively and negatively affected by the amount of solar radiation it receives, yet the multiple regression analysis implies that neither the potentially moister south facing aspects nor the warmer north facing aspects influence the growth rates significantly.

No clear trends are observed between the different tiller positions (north, south, east or west) at each site, each season and each year. This is not unexpected as the tussocks will be influenced by inter-tussock spaces through shading from their nearest neighbours as well as themselves. Shading of tussocks by the tussock itself (especially when the sun is at a low angle such as in the early mornings and late afternoons or during the winter months) may not be effective enough to alter the growth rates (Figure 9.33). Additionally their position within each community may play a role. The *M. drakensbergensis* individuals observed on the edges of communities may be influenced less by shading from others. Thus, the role played by aspect within each individual may be decreased, resulting in the lack of relationships observed. The intercalary meristematic tissues, located at the base of the leaf blade, are responsible for leaf elongation (Trlica, 1999). It is these tissues which are responsible for the growth rates observed in this study through cell division and growth. Thus, it is possible that the conditions at the base of the leaf are more influential on leaf growth than conditions outside the tussock, except at the periphery. Additionally, the youngest leaves usually depend on stored resources and parent tillers for growth materials (Wolfson and Tainton, 1999). Towards the end of 2010 and during 2011, the different tiller positions appear to have similar growth rates at each site. The reasons for this are unclear.



Year	Solar Elevation 10:00	15:00	Shadow Length 10:00	15:00	Year	Solar Elevation 10:00	15:00	Shadow Length 10:00	15:00
2009	61.01°	50.78°	23.05cm	33.95cm	2009	29.19°	22.90°	74.47cm	98.48cm
2010	61.05°	50.77°	21.19cm	31.27cm	2010	29.18°	22.88°	68.59cm	90.76cm
2011	61.08°	50.75°	20.33cm	30.07cm	2011	29.18°	22.87°	65.90cm	87.25cm

Figure 9.33: Shading produced by each individual tussock (average height of 41.6cm) during mid summer (1 January) and mid winter (1 July) during the study period, at mid morning (10:00) and mid afternoon (15:00). Solar elevation is measured in degrees rising from the horizon.

Despite unfavourable conditions, such as those experienced during the winter months, minimal growth was observed to continue in order to maintain the studied tussock grasses, as was observed. Grasses found in other alpine areas have adapted in order to maintain physiological processes at low temperatures, thus extending the growing season (Billings, 1974). However, the growth may not be all above ground as many perennial plants direct growth to below-ground organs when conditions are unfavourable above ground (Monteiro *et al.*, 2011). Three arctic graminoid species have been observed to have a growth rate of between 1 and 2mm per day at 0^{o}C (Billings, 1974). Growth rates observed in this study for the winter months were less than or equal to 1.5cm/2 months. The highest average growth rates were observed at sites S.E. with 1.4cm/2 months for the peripheral leaves and 1.5cm/2 months for the tillers at the radial midpoint at K.P. The growth rates for the remainder of the sites were below 1cm/2 months, or were negative. The records of positive growth support the theory that *M. drakensbergensis* tussocks do continue to grow slowly during the winter months. However, a limit to growth processes seems to occur between 0^{o}C and 2^{o}C in all plant types (Körner, 2006), with growth unlikely below 0^{o}C (Woodward *et al.*, 1986; Körner, 2003; Hayashi, 2007). The temperatures experienced in the current study region during winter are low with a mean air temperature of 1.81 ^{o}C during the winter months (May to August). Additionally, more than 30% of the winter months are below 0^{o}C in the Sani region and more than 70% in the Kotisephola region (Table 4.2 and 4.3). Despite this, some studies observed little or no relationships between growth and temperature (Patton *et al.*, 2007), yet few of these were at high altitudes or in alpine areas. The positive growth occurring at some sites (e.g. site S.E.) suggest that *M. drakensbergensis* has adapted to take advantage of temperatures suitable for growth during winter

Tussock growth cannot be attributed to a single factor such as precipitation; rather, it is the response to the collective role of light, moisture, nutrients, temperature, topography and fire (Briggs and Knapp, 1995). Essentially, growth is linked to carbon uptake, carbon loss and the movement of photosynthetic products to various organs (Monteiro *et al.*, 2011). Carbohydrates are necessary molecules for cell wall synthesis and expansion, and if they are lacking, may reduce growth despite optimum conditions, hence the decrease in grass growth when light is limited (Busso and Richards, 1992). Thus, many carbohydrates are stored in the crown for the onset of growth, yet as the season progresses these stores will become depleted and the plant will focus on reimbursing these stores rather than growth. The results in this study indicate carbon gains and losses, and are a reflection of the environment including the grazers. Growth is generally believed to be slow in high altitude environments due to the harsh conditions. Thus,

alpine tussock grasses favour protection and survival growth strategies rather than production strategies (Monteiro *et al*., 2011). Production and leaf elongation will only occur when survival is not paramount and conditions are suitable.

9.5 The burnt tussock study

The first leaves to appear after burning in all four of the burnt patches were the outer tillers (e.g. Figures 9.21 and 9.29), and is not unique to the current study (Miller, 2000). New leaf tissues are produced by the intercalary meristematic tissue at the base of leaf blades, within a protective sheath (Bond, 1997; Trlica, 1999). Some of the new leaves appeared pale green to white as they emerged and began to coil as they lengthened (Figure 9.14). The leaves were soft and pliable where they were white and less pliable where green near the crown. During burning, the aerial parts of the leaves are burnt and reduced to ashes, while the sheathed leaf tissues at the base remains protected from the direct effects of the flames (Trlica, 1999; Bruhjell and Tegart, 2000). However, they are not entirely protected from the heat, and the water content of the leaves may rise, essentially cooking the leaf tissue inside the sheathes. The meristematic tissues remain safe in the crown tissues and are thus able to produce new growth pushing upwards underneath the old leaf. This would push the 'cooked' tissue upwards on top of the new leaf tissue, resulting in the pale coiled tissues observed.

New leaf tissues are produced using resources stored in the crown until the resprouted plant is capable of providing enough fuel to support the new growth and replenish used stores (Trlica, 1999; Miller 2000; Monteiro *et al*., 2011). Damage to the central tissues may occur through longer burning durations and increased temperatures (Peláez *et al*., 1997), resulting in stunted growth or a lack of growth. Hollowing of the tussock centres may also be the result of the natural progression of tussock grasses (Feldman *et al*., 2007). Hence, many tussock individuals appear to change shape after burning, and tend to become 'hollow' in the centers (type 'B') or result in patchy growth around the edges (type 'C') (see Chapter 7). This was suspected by the researcher and by Gagnon *et al.* (2012), to result in an increase in basal circumference, but was not the case in either study.

A high initial growth rate was common to all the burnt sites but drops after a short period. Observed initial increase in growth rates, and thus productivity after burning, is believed to be the plants response to increased nutrients, increased light availability, increased soil temperature and the removal of competitive vegetation (Whelan, 1995). A high growth rate lasts approximately 5

months in the burnt patch near site S.S., but at site V.S.E. it declines after the first month. However, grazing was regularly observed at this site and may be the cause of the growth rate decline (see also Gitay *et al*., 1992; Hardy, 1999). At the A.P. site, a high initial peripheral growth rate lasted an average of two months with grazing on only a few individuals.

The initial appearance of the leaves at the tussock edges resulted in the peripheral tillers being, on average, taller than those closer to the centre of the studied tussocks during the first measurements. At site V.S.E., the peripheral tillers were on average only slightly taller (4.2cm) than the tillers at the radial midpoint (3.9cm) and in the centre (3.8cm). However, the periphery experienced a much faster growth rate (7.3cm/2 months) than those of the central regions (4.4cm/2 months at the radial midpoint and 4.0cm/2 months in the centre) (Figure 9.29). The stored resources used to fuel initial re-growth are found within the crown (Fynn *et al*., 2005; Feldman *et al*., 2007). Thus, when growth resumes, the meristematic tissues of the center of the crown are likely to be the most damaged and less likely to be able to resprout. The outer tillers have room for expansion and the meristematic tissues are more likely to be intact. Hence growth, when it resumes, is likely to appear and grow at a faster rate in the peripheral tissues than in the centers. Yet, the outer leaves are more exposed to defoliation, while the leaves at the radial midpoints remain more protected resulting in a high growth rate.

After the initial growth phase following burning, the growth rates, where positive, are similar to that measured after heavy burning of tussocks (2.4cm/60 days) in the Columbian páramo belt (4 000-4 100m a.s.l.; Hofstede *et al*., 1995). Yet, less than that of tussocks only moderately burnt at 11.4cm/2 months (Hofstede *et al*., 1995). Raised growth rates have been observed to continue for at least two years following burning in the Murchison Mountains, New Zealand (Lee *et al.*, 1988). Yet, in the current study the growth rates decrease after an initial spurt perhaps as a result of grazing. The alpine tussock grasses of New Zealand were observed to have a higher growth rate than that of unburnt plants during the first two growing seasons following burning, but the growth rate declined for the next few growing seasons (Mark, 1965). Similarly to this study, reduced growth rates occurr in semi-arid and arid regions, such as found in western Australia, as burning may exacerbate water and nutrient stresses (Bennett, 2002).

All four of the burnt sites indicated that in the spring following burning, the central tillers experienced faster growth rates than that of the periphery (Figures 9.33 and 9.34). Towards the middle of the growing season, the peripheral tillers became the fastest growing tillers. All four

burnt sites indicated less variation of average heights and growth rates in the winter months, while summer months depict varying growth rates and average heights between tiller positions. However, unlike the other burnt sites, at K.P. the highest growth rate was that of the tillers at the radial midpoint. The measurements leading to these results were made in autumn, when the dominant winds were increasing in force (mean wind speed was 4.73ms^{-1} in May 2010 increased to a mean wind speed of 6.29ms^{-1} in June 2010; Table 4.4) and the temperature was dropping (Table 4.2). The cooler conditions may have the effect of slowing the growth rate and the outer tillers are the most exposed to these conditions. The high altitude and exposed nature of the north-facing K.P. burnt site may explain why this phenomenon was not experienced at any of the other sites. Similar to the other sites, the peripheral tillers experienced the highest average height as a result of high average growth rates during the summer months. Similar to other sites, the winter months are represented by low growth rates and similar average heights.

It was observed that above ground biomass in the second growing season following burning was significantly less than that of the growing season directly following burning in Highveld grassland near Bloemfontein (Snyman, 2005). Yet, all four of the burnt sites in the current study did not support this. The average heights of the tussocks were higher in the second growing season than that at the end of the growing season directly following burning, even when taking grazing into account. This suggests that the productivity of the tussocks was maintained in the current study allowing for a greater standing crop. Grazing has been observed to cause a further decrease in the second growing season growth (Snyman, 2005). Tussock grasses were used in both studies but grazing intensity may have been different in the current study, and in addition, the difference may be the result of differing growth rates and thus differeing productivities.

The importance of the seasonality of burning is well established in the literature (e.g. Daubenmire, 1968; Wright and Bailey, 1982; Trabaud, 1987; Malanson, 1987; Bond and van Wilgen, 1996; Brown, 2000; Uys *et al.*, 2004) and the effects impact regrowth, especially in regions with limited resources. Each of the sites was burnt at different times of the year; the patch near site S.S. in autumn, V.S.E. in midsummer, the A.P. in late winter and K. P. in autumn. The time each site had to recover before the end of the study period in 2011 was reflected in the final average heights. The burnt site near S.S. had the longest period to recover and had a maximum average height of 20.1cm, while the site with the least time to recover, site V.S.E., had a maximum average height of 16.0cm. Burning in midsummer, such as at site V.S.E., may be very destructive as much of the stored resources would have been used in spring growth, leaving little

to fuel resprouting (Daubenmire, 1968; Everson and Everson, 1985a; Everson and Everson, 1987; Hardy, 1999). Regrowth may also be initiated by slower auxillary buds rather than the faster growing intercalary meristematic tissues (Bond and van Wilgen, 1996). Yet, the growth rate is not the lowest and the low average height is more a reflection of losses during the winter months and as a result of grazing. K.P. had two weeks longer to recover than site S.S., yet has a lower maximum average height (14.9cm) than the site V.S.E. This may be attributed to lower temperatures, lower metabolic rates and a slightly shorter growing season. Ultimately, site S.S., burnt in autumn, and the A.P. site, burnt in late winter, had achieved the most growth by the end of the study period, while site V.S.E., burnt in midsummer and the high altitude K. P. site indicated the least average growth.

Both burnt patches near site S.S. and K. P. were burnt in autumn and entered winter with new shoots. Burning prior to winter may leave the new shoots exposed to frost, especially as some believe new shoots contain higher water contents than older leaves (Daubenmire, 1968). The reduced ability to withstand freezing temperatures may explain the very low growth rate experienced in the peripheral tillers between May and August 2009 (-2.0cm/ 2 months) at site S.S., whilst the central tillers were thermally protected through their position. Alternatively or jointly, the low average height is perhaps due to the effects of depleted stored reserves within the crown, which enabled regrowth following burning (Wolfson and Tainton, 1999). Defoliation before winter has also been linked with large resource use during dormancy, to maintain minimal functioning (Lee *et al.*, 1988). The tussocks will have had to draw on stored resources until tillers had grown sufficiently to allow photosynthesis to fuel post-fire re-growth. The strategy allows the tussock to have photosynthetic leaves ready at the start of the next growing season, yet the stored reserves would not have been replenished. Thus, the low amount of average growth taking place between December 2009 and December 2010 was possibly the result of a shift in focus to replenishing reserves rather than only growth. The burnt areas of site V.S.E., burnt in midsummer (December 2010), followed a similar trend.

The A.P. site was burnt in late winter, when stored reserves would be reduced as they provide fuel for respiration to maintain the crown and roots during winter (Trlica, 1999). Thus, initial growth would have drawn on remaining reserves and been less than observed when stored resources were higher. The first growing season after burning allows the tussock to grow and begin to replenish stored resources. Burning in late winter increases the length of the first growing season through an earlier start of increased growth (Vogl, 1974). The closer the burn is

to the onset of the growing season, the faster the resprouting (Tainton *et al.*, 1977). Yet, the initial rates were not greater than observed at the other burnt sites. As with the other burnt sites, the maximum average height gained between the second and third growing seasons following burning is low, perhaps due to the focus being shifted entirely to replenishing stored resources. However, all of the above conclusions have not taken into account the effects of grazing, which may be the cause in the low growth rates after the first growing season after burning through additional strain on stored resources (Hardy, 1999).

Average temperature and precipitation contribute to the variation in the growth rates extremely significantly ($p<0.0001$), as observed through a multiple regression analysis. Water availability may be important in the resprouting of tussocks (Defossé and Robberecht, 1996; O'Connor and Bredenkamp, 1997; Bennett, 2002), particularly in Lesotho where it can be a limiting factor to growth. As a result of precipitation, the A.P. burnt site was saturated in October 2009, a month after burning and remained saturated until at least late December 2009. If water is an important factor for regrowth, it stands to reason that if it is in abundance that the growth rate would be high, however, the growth rate remained almost constant between October and December 2009. Supporting this, a previous study reported that neither early spring rainfall, nor soil moisture, was positively correlated to tussock growth in the Drakensberg (Everson and Everson, 1987). However, the slope coefficient indicates that the opposite is true and too much water can slow the growth rate, possibly through solution of other necessary nutrients, as well as through reducing gaseous exchange. Water is believed to be vital to resprouting and tussock recovery by Gittins *et al.* (2011). However, these sites all show rapid initial growth, suggesting water may not be required for initial resprouting, rather the growth is a result of stored resources. However, declines in the growth rate after the initial measurements may indicate a delayed requirement for water. Rather, temperature (as a result of aspect and shading) has been demonstrated to be more influential on the rate of re-growth in the Drakensberg (Everson and Everson, 1987). Yet, initially the colder K. P. site has a higher initial rate of growth than sites A.P. and V.S.E. The average wind speed was also indicated to significantly ($p<0.05$) explain the variation in growth rates observed. This may be explained through the observed wind chill factor resulting in decreased ambient temperature and thus growth rate.

Site V.S.E. and the A.P. site were heavily grazed soon after resprouting, while almost all sites showed signs of grazing two months after burning. After burning, the palatability of the new shoots increases as well as their accessibility and are often subsequently grazed (Daubenmire,

1968; Vogl, 1974; Wright, 1974; Mark, 1994; Williams, 2003; Cremer, 2004; O'Connor *et al*., 2004). However, the impact of grazing is greater when grazing occurs during resprouting (Trlica, 1999). Grazing negatively impacts the growth rates and the average height of tussock grasses (Lee *et al*., 1988; Mark, 1994; Hofstede *et al*., 1995; Moore, 1995; Snyman, 2004b). Intensive grazing was observed for similar species, *M. disticha,* in the Karoo mountains after burning (Novellie, 1990). When growth is rapid, tussocks can withstand defoliation better than during periods of low growth (Trlica, 1999), as stored resources are further depleted to replace the lost tissues. Thus, a reduction in the growth rate is often the response by plants, as they have lost tissues which provide the fuels necessary for growth to occur and maintain the general functioning of the plant (Moore, 1995; Trlica, 1999). Despite being evenly grazed, outer tillers were observed to initially grow faster than those of the central regions of a tussock in Oklahoma, USA (Volesky, 1994), and perhaps partially explains the increased growth in peripheral tillers throughout the growing seasons at many sites.

Projected times for recovery are an average of seven years after the completion of the study, meaning a total of nine years are required for recovery. However, the burnt patch near site S.S. only had an average minimum of 5.3 years after December 2011. Within the first two growing seasons following burning, this site had recovered more than 90% of the pre-burn average height of the edge tillers, more than 70% of the tillers at the radial midpoint and 58% in the central areas. Hence less time is needed to reach pre-burn average heights. The A.P. site had managed to recover more than 50% of its pre-burn height after two growing seasons, but at a maintained post-burn growth rate requires seven years after the first two growing seasons to recover. Both the burnt tussocks of site V.S.E. and K. P. require an average of 7.2 years to recover to pre-burn heights. The intensity of grazing may decrease further as the tussocks age due to decreasing palatability. Hence the predicted time to reach pre-burn average heights may be a slight over estimation. However, the losses experienced during the winter months and the level of grazing remain similar to those of unburnt individuals. The calculated recovery times correspond with observations in the literature. At least 10 years are required in order for vegetation to recover to the same levels as before burning, in the Bogong High Plains, Australia (Camac *et al*., 2012). In contrast, burnt individuals of the Greek tussock *Ampelodesmos mauritanica* were larger than that of unburnt individuals after two years (Vilà *et al.*, 2001). Yet, at least two full growing seasons are required for tussock grasses to recover their above ground tissues to that before burning (Snyman, 2001a), as was also found to be the case in alpine grasslands of south-eastern Australia (Wahren *et al*., 2001b). Similarly, the tussock grasses of semi-arid, sub-tropical Australia are

expected to require between four and five years to recover (Bennett *et al*., 2002). After two years, the burnt *Chionochloa rigida* tussocks of New Zealand were only 25% of the size of those that remained unburnt (Yeates and Lee, 1997). The same species was observed to have stunted leaf growth rates for more than 13 years following burning and was attributed to the time taken to replenish stored resources (Payton and Mark, 1979; Mark, 1994). More recently, the same reason was also attributed to the long duration of tussock recovery in semi-arid Australian grasslands (Bennett, 2002). Plants tend to devote more energy to resource storage in cold or harsh climates as these are relied upon for growth (Monteiro *et al*., 2011), such as those in the study region.

The burnt *M. drakensbergensis* tussocks near site S.S. follow an expected trend with the peripheral leaves requiring the least time (2.6years) to recover while the central tillers and those at the radial midpoint require more time (5.3 and 7.9 years respectively). The central tillers require the greatest amount of time to recover as they are likely to have received the most damage during burning. The tillers at the radial midpoints will compete with the peripheral and central tillers for resources and thus, should require a longer time to recover. The recovery times required for the burnt tussocks to reach pre-burn heights at site V.S.E. are similar for all the tillers (7.9 years, 6.2 years and 7.5 years). This may be due to the high level of grazing. The A.P. burnt individuals indicate a different relationship, perhaps due to more intensive grazing than that at site V.S.E. The peripheral tillers experience the slowest growth rate (0.2cm/2 months) as a result of tissue losses due to both grazing and other defoliant factors such as strong winds. As a result, the edge tillers will take an additional 12 years to grow, assuming the growth rate remains constant as do the losses. The peripheral tillers at the burnt K.P. site experience the highest growth rate for longer than the other sites and thus require a shorter time to recover to their pre-burn heights. This may be a function of the higher altitude or slightly shorter growing season. The central tissues require the longest time to recover (10.1 years), a time longer than that for any other burnt sites central tillers. Perhaps before burning, the central regions are more tightly packed as this is where the most growth is able to occur in the harsh conditions. Hence, when burnt, the central tissues burn for longer and result in greater tissue damage. Consequently, the central tissues require a greater amount of time to reach pre-burn average heights.

Burnt tussock grasses in an alpine region of south-eastern Australia were observed to flower abundantly a year after burning (Wahren *et al.*, 2001a). This study is not unique, leading to the general belief that burning results in proliferation of inflorescences in most monocotyledons (Whelan, 1995; Bond and van Wilgen, 1996; Bond, 1997; Bruhjell and Tegart, 2001). Yet, in the

current study, *M. drakensbergensis* individuals burnt during the first year of the study still had not flowered after two growing seasons following burning. The environmental conditions of the Lesotho highlands result in a generally slow level of growth and thus a slow rate of resource accumulation for storage. This suggests that once a tussock has been burnt, it will take a long time to re-accumulate the stored resources to levels required for inflorescence production. This finding is supported by observations of cool-season grasses burnt in late spring, which were unable to produce many inflorescences due to depleted stored resources (Miller, 2000). Yet, in the alpine grasslands of New Zealand, *Chionochloa rigida* flowered abundantly in the growing seasons following burning (Mark, 1965, 1994; Lee *et al*., 1988; Yeates and Lee, 1997). This suggests that perhaps the response of flowering to burning is species specific, or that a limiting factor occurs in the alpine regions of Lesotho.

9.6 Summary

This chapter examined the ability of *Merxmuellera drakensbergensis* to recover through its growth rates, both following recent burning and without burning during recent years. The following are the main findings:

1. Losses in leaf tissue are not confined to one particular time of year, yet become less severe during the winter months. It is likely that thelarge losses observed during summer, autumn and spring months were the result of intensive grazing.
2. The results indicate a general decline in the variability of individual leaf growth rates, and are potentially an indication of increased grazing pressure. The variation is observed to be at a maximum each year during the growing season, more particularly between the mid and late growing season.
3. At many sites (sites V.S.N., H.E., H.W. and V.F.), the peripheral tillers were taller than the centres of the tussocks, unless grazed, due to less competition for resources and increased shelter when compared to the central regions. Tiller position and associated height may be seasonal due to grazing intensity and wind pruning effects.
4. Average temperature, precipitation and wind speed were significant contibutors towards the variation in the measured growth rates between burning events and after recent burning.
5. Altitude contributes significantly to the growth rates between burning events but not after rescent burning.

6. Positive growth occurring at some sites (e.g. site S.E.) suggests that *M. drakensbergensis* has adapted to take advantage of temperatures suitable for growth during winter.
7. The first leaves to appear after burning in all four of the burnt patches were the outer tillers, and is perhaps the result of increased damage to central regions. This was linked to higher growth rates. Yet, the leaves at the radial midpoints remain more protected, resulting in a high growth rate later.
8. The average heights of the tussocks were higher in the second growing season than that at the end of the growing season directly following burning, even when taking grazing into account.
9. Site S.S., burnt in autumn, and the A.P. site, burnt in late winter, had achieved the most growth by the end of the study period, while site V.S.E., burnt in midsummer and the high altitude K. P. site indicated the least average growth.
10. Projected time for the recovery of *M. drakensbergensis* to pre-burn heights is a total of nine years within the study region.

Chapter 10:

Synthesis



CHAPTER 10:

Synthesis

To demonstrate the health and survival of *Merxmuellera drakensbergensis* communities in the Sani region of Lesotho various research questions were investigated. Defoliation as a result of burning and grazing were thought to be the main contributors towards landscape degradation within this region. This study aimed to determine the spatial and temporal patterns of tussock burning over an 10 year period (1997-2007) in the Sani study region ($25km^2$) and assess relative recovery rates after burns in various spatial contexts (aspect, altitude). The long-term conditions of tussock grasses under various burning frequencies were determined for the last 18 years. This was aided by the construction of a burn frequency map using satellite images.

10.1 First hypothesis

It was hypothesized that as a result of frequent burns, biomass is becoming reduced over time and inter-tussock plant communities are being degraded through loss of protective cover. If mortality increases, and sprouting and the reproduction of new plants is slowed or suppressed (Mark, 1994), the ground cover is likely to decrease. Changes in growth form and standing biomass, inflorescence numbers and variables influencing growth were observed in order to assess the hypothesis.

10.1.1 First hypothesis: conclusions

The first hypothesis states that a decrease in biomass is expected as a result of frequent burning and grazing. Four measurements were observed: basal circumference, average height, living basal area and standing biomass. General increases were observed for basal circumference, living basal area and a small increase in standing biomass across all sites during the three years of study. However, a decrease was observed in average height and was attributed mainly to grazing. Burn frequency and, to a greater extent, the time since each site was last burnt were extremely significant contributors to basal circumference variation. The basal circumference increased with increasing burn frequency and time since last burnt, suggesting both an accelerated and natural progressions through the age states presented by Gatsuk *et al.* (1980) respectively. Average height indicated a quadratic relationship with burn frequency and the time since each site was last burnt for all three years of study. The results suggest that a burn frequency of three burns per decade is the optimum for retaining the greatest average heights, while a lack of burning for more

than ten years may result in a decrease in average height. Similarly to average height, living basal area was indicated to decrease at burn frequencies greater than four burns/decade, while a lack of burning for more than ten years may result in a decline in living basal area. Biomass decreased on east- and south-facing aspects. Warm temperatures of the north-facing slopes in the southern hemisphere are more likely to have a positive effect on metabolic functioning and growth rates and thus may be preferable.

Biomass may also be a function of birth and death rates of individual plants and tillers, hence, an indication of the potential for reproduction through inflorescence production was observed. The average number of inflorescences across all sites increased across the years of study (mean = 9.9 in 2009; mean = 11.1 in 2010; mean = 13.1 in 2011). However, the viability of the seed was not investigated and has the potential to decrease with an increasing number of inflorescences. Significantly different numbers of inflorescences were observed between the sites, supporting the findings of Mark (1964). The increases were linked to increases in basal circumference, average height, living basal area and standing biomass through direct relationships. This supports the idea that the larger and more productive the tussock, the more inflorescences may be produced. Inflorescence numbers have been largely attributed to the amount of stored resources available for inflorescence production during the growing season when temperature cues have been reached (Mark and Chapin III, 1989; Lord, 1998; Rees *et al*., 2002; Kelly and Sork, 2002; Hay *et al*., 2008). However, a decrease in inflorescence numbers was observed when the standing biomass was greater than 55%, perhaps as a result of shading and competition between individuals. Similarly, the number of inflorescences is expected to decrease under higher grazing pressure. Almost no relationship was indicated between burning and inflorescence production, however, the ability of burning history variables to account for variation in inflorescences numbers increases each year. This may be due to the influence of increasing time since last burning increasing inflorescence numbers. Yet, despite all this, very few seedlings were observed and suggests that grazing may have a negative impact on seedling survival, hence, reproduction is mainly through vegetative means. Vegetative reproduction produces individuals with small crowns but with a good average height, allowing them to quickly produce and maintain stored resources.

10.2 Second hypothesis

It was hypothesized that the alpine tussock grasses of Lesotho are being burnt and grazed at a rate that does not enable them to adequately recover between burns. Under the harsh alpine climatic

conditions and the short growing season of alpine grasses, the herdsmen are required to turn unpalatable tussock grasses into valuable forage for growing livestock numbers and longer durations of grazers through burning, at a rate that is not sustainable over the long-term (Quinlan, 1995b).

10.2.1 Second hypothesis: conclusions

The second hypothesis suggests that there is not enough time available between burning of the tussock communities for their recovery. However, only 4 of the 12 sites were burnt during the three year study period, and only a small proportion of each site was burnt (less than 25%). The observations and the burn scar mapping suggest that the frequency may not be as high as previously thought. Rather, the burn frequency is patchy with some *Merxmuellera* tussock individuals receiving a high burn frequency while others are burnt only every 3 years or more. Yet no *Merxmuellera* communities remained unburnt between 1993 and 2008 (Figure 6.12). By the end of the study period, tussocks from site S.Str. and the upper and lower M.F. sites had not been burnt for 13 years. Tussock grasses have been observed to 'collapse' if not burnt after 11 years in the Derrimut Grassland reserve, Australia (Morgan and Lunt, 1999), but this was not observed in the current study. The absence of 'collapse' is likely to be due to grazing which reduces aerial tissues and thus prevents them from becoming top heavy.

The leaves observed at the radial midpoints were calculated to require the least time (6.3 years) to recover to pre-burn heights, while leaves observed at the periphery require 6.6 years to recover to pre-burn heights. The results do counteract the observation that the outer leaves had the highest initial growth rates directly following burning. However, the peripheral leaves are exposed to wind and hail, and were observed to be more readily grazed. The leaves at the radial midpoints were more sheltered from environmental impacts and grazing, but the competition would be higher for stored reserves. The central tillers generally required the most time to recover to pre-burn heights (7.1 years) but this was not surprising as the greatest competition for resources may be found in the centers which may also suffer the most damage from burning (Paysen *et al.*, 2000), and thus recover slowly (Bruhjell and Tegart, 2001). The longest time to recover to pre-burn rates was 11.8 years, while the lowest time to recover was 2.6 years. Thus, a burning frequency of burning every 7 years should be adequate for this species, taking into account current grazing levels.

The second hypothesis also questions the sustainability of the current use of the Sani region as a forage resource. Many researchers refer to the process of burning to increase forage availability and its nutritious state (e.g. Vogl, 1974; Daubenmire, 1968; Van de Vijver *et al.*, 1999; Körner, 2004; Vermeire *et al*., 2005). Yet, despite literature confirming this practice in the Lesotho highlands (Guillarmod, 1969; Nüsser, 2002b), it was not observed within the study region. Rather, burning was observed on a small scale (<100m^2), mainly for warmth or livestock protection. A lightening strike observed at site V.S.E. started tussocks burning, yet was quickly extinguished. Yet, grazing of new shoots following the burning of *Merxmuellera* tussocks were observed at all sites within two months following burning. This observation supported the idea that the regrowth of these tussocks is more attractive, accessible and palatable to livestock.

Sufficient reserves exist within alpine plants to allow them to withstand grazing despite harsh climatic, edaphic and topographical conditions (Crawford, 2008). Yet, the low height of forage grasses and evidence of overgrazing, particularly the presence of invasive species, suggests that the current levels of grazing are unsustainable. The number of grazing posts within the study region has decreased since Grab and Nüsser (2002) undertook their study, yet many are still being used during the winter months. The result is continuous grazing of the study region, mostly by sheep and goats during the winter months. The palatable forage (grasses) will therefore have very little time between grazing events to recover stored resources and aerial tissues. It is predicted that as the forage species become overgrazed, livestock will turn more to the less palatable *Merxmuellera* species. The evidence suggests that the amount of grazing taking place on *M. drakensbergensis* has increased between late winter and early summer (May to February). Adding to this, the high percentage of *M. drakensbergensis* tussocks grazed at each site every second month during the monitoring study (Figure 5.6) suggests that grazing of the tussocks may become more intense if more palatable species are overgrazed.

10.2.2 Soil erosion

The potential for and recorded instances of soil erosion in the Lesotho highlands has been well documented (e.g. Guillarmod, 1971; O'Connor *et al.*, 1997; NES, 2000; Mbata, 2001; Grab and Deschamps, 2004). The maintenance of soils for their ability to supply water and nutrients for vegetative growth is the heart of sustainability of an ecosystem (Yeates and Lee, 1997). Plants are able to anchor soil on slopes and prevent erosion through their root systems as well as through improving infiltration, delaying runoff and breaking the force of rain drops and wind (Binnie and Partners, 1972; Körner, 2004; Breebaart *et al.*, 2006). The removal or loss of tussocks reduces

soil stability, especially on slopes. Grasses also act to filter out the soil being washed away by trapping soil grains between the leaf blades (Showers, 1996). The multi-stemmed structure of tussock grasses also increases their ability to prevent erosion by water or wind (Breebaart *et al.*, 2006).

The burning of *M. distacha and M. drakensbergensis*, as well as other tussock species growing in moist soils, appears to have a negative effect on water supply and erosion (Guillarmod, 1971). The vegetation and litter covering alpine soils is removed during a fire exposing the soil to erosion through increased runoff immediately after a fire (Hester *et al.*, 1997; O'Dea and Guertin, 2003; Williams *et al.*, 2006). Runoff removes ash and leaches the soil in alpine environments, resulting in poorer soils and the vegetation becoming shorter with less diversity over time (Smith, 1985). Both unburnt and burnt tussocks have showed they were able to prevent more soil loss than short grasses, making them somewhat more valuable in reducing soil erosion (Hester *et al.*, 1997). Erosion and leaching in areas of reduced tussock cover are a precursor to nutrient loss through early desertification, and not the consequence of desertification (McIntosh, 1996).

Exposed soil is also at the mercy of strong winds, leaving behind typical wind erosion features (van Zinderen Bakker, 1981). Burning well before the start of the growing season can leave soils exposed for long periods of time and render these more prone to erosion (Everson *et al.*, 2004). Increased levels of erosion have been observed in the first and second wet summers after a spring burn in southern Arizona grassland (O'Dea and Guertin, 2003). Individual tussocks dying through frequent burning and grazing leave longer-term gaps in the ground cover. As little as 1% bare ground is known to induce soil erosion in mountainous environments under the right conditions (Bridle *et al.*, 2001). As bare ground increases, so will the potential for erosion. Thus, defense against erosion is dependent on post-fire regeneration (Snyman, 2006; Williams *et al.*, 2006). Frequent burning may also lead to soil erosion which negatively impacts long term biodiversity and conservation goals (MDTP, 2007).

10.2.3 Fire exclusion

Burning is known to increase the crude protein content of *Merxmuellera* tussocks under conditions of clipping (Donaldson, 1991). Similarly, a higher quality forage of burnt as opposed to that of unburnt grass has also been indicated by Tainton *et al.* (1977) and Mbatha and Ward (2010). This suggests that if grazing is to continue within the region, then regular burning may be beneficial. However, the exclusion of fire must also be discussed as a possible pathway, perhaps

already occurring in many areas. Burning of fire adapted communities may increase the diversity through burning (Masocha *et al*., 2011). Under the exclusion of fire, re-vegetation and succession can be patchy and slow due to small seed reserves and low vegetation cover (Hunter and Boswell, 2003). This may also be due to reduced growth rates and low tillering rates (Everson, 1999). Additionally, woody vegetation may increase along with invasive species (Titshall *et al*., 2000; Masocha *et al*., 2011). Under the exclusion of fire, it is suggested that only a matter of decades is required to complete the vegetation cover (Bridle *et al.*, 2001). In severe environments, such as those in Lesotho and the eastern central plateau, Tasmania, regeneration of vegetation exposed to grazing and erosion will be extremely slow (Bridle *et al.*, 2001), but the rate of recovery may be increased through regular burning. If left for longer periods, the aerial organs of the tussocks 'collapse,' covering the ground in a thick layer of plant litter (as seen in temperate Australian grassland after 11 years without fire and in South African grasslands after seven years) (Morgan and Lunt, 1999; Zimmermann *et al*., 2010). Yet, drought and reduced growth may prevent or slow the rate of collapse (Prober *et al.*, 2007). Similarly, tussocks may suffocate if excess plant tissues are not removed through reduced gaseous exchange. When an area which has been excluded from burning does eventually burn, it will burn with significantly higher temperatures than an area burned regularly (Mistry, 1998). The higher temperatures and burning of a high fuel load are likely to result in high damage to plant tissues, reducing resprouting. Additionally, suppressed burning and increased biomass from tussocks can pose a fire risk to communities, farmland and wildlands. Thus, the exclusion of burning may not be the best option for the Lesotho highland region.

10.3 Conclusion

Ultimately, this study aims to determine whether recent (last 19 years) burning and grazing practices in the Sani region of eastern Lesotho are sustainable for the long-term survival of tussock grasses in the alpine belt. The observed erosion occurring within the Lesotho highlands and the increasing extent of invasive species are predicted using the synthesis model proposed by Cingolani *et al.* (2005). Additionally, the model suggests that the study region grasslands are not resilient enough to cope with long term continuous grazing (Cingolani *et al.*, 2005). Alpine grasslands, in particular the upper subalpine belt, have been refered to as 'fragile' under conditions of increased grazing (Quinlan and Morris, 1994). A decrease in the number of grazing posts is observed, yet the number of perennially utilized grazing posts increased. Thus, not only has the frequency of grazing changed since the studies by Morris *et al.* (1989) and Quinlan and Morris (1994), but also the number of livestock has increased. It is predicted that overgrazing of

the forage grasses will result in an increase in the grazing of *Merxmuellera* tussocks, particularly the newer outer leaves as these are the softest and most accessible. The increasing grazing practices will reduce the stored resources available for reproduction through inflorescences produced (already being observed) and resprouting. Grazing as an alternative form of defoliation to burning may not be suitable for the study region as overgrazing has led to invasion by Karroid shrubs. Currently, the *M. drakensbergensis* tussocks are not being outcompeted by the invasive shrubs, but under increasing grazing pressure the faster growing shrubs may dominate the landscape in the future. Grazing has comparatively less impact on species composition than burning (Prober *et al*., 2007). However, burning may lessen the impact of invasive species.

According to Ellis (1981), Schmitz (1984), Killick (1990) and Pooley (2003), *M. drakensbergensis* may reach a maximum height of 100cm, a height reached by only a few individuals at site V.F. in 2009 and at Kotisephola Pass in 2010. Across all the sites, the average height was 41.6cm, well below 100cm. The shortening of tall-grasslands to short-grasslands is an indicator of reduced biomass as a result of burning and grazing (O'Connor *et al.*, 1997). Yet, biomass has not been indicated to decrease, but rather to increase slowly across all the sites. However, some sites have decreased in terms of biomass and this is largely linked to grazing rather than burning. Hence, current levels of grazing and burning may be sustainable but are not optimal.

The reproductive study observed a general increase in the average number of inflorescences across all sites. Yet, seedlings were rare, suggesting that seedling survival was low. This finding implies that the main pathway of reproduction is through vegetative means, as a function of increasing age state. The majority of the *M. drakensbergensis* tussock grasses within the study region are considered 'young' due to their shape, and this may have implications for future numbers of the *Merxmuellera* communities as the rate of progression through the age stages to the segregated stage may take many years. The research suggests more than 10 years may occur between each age stage under a low burning frequency. With increasing grazing pressure, the age state progression may be shortened, but this could not be determined in this three year study, and thus the extent remains unknown. Similarly, no relationship was observed between tussock shape and burning.

Burning may prove advantageous to *M. drakensbergensis* through its ability to increase the growth rate directly following burning, remove dead tissues which may suffocate productive

leaves, increase light penetration, and reduce competition from less fire tolerant plants. However, a burning regime which is too frequent has been indicated to potentially reduce biomass and tussock resilience. Yet, this was not observed to be a risk factor within the study region. Rather, burning events are becoming increasingly patchy and on smaller scales (less than $100m^2$ burnt each time, usually only a few *Merxmuellera* individuals). Thus, the risk may be that individuals may remain unburnt but grazed for long periods. The result may be a vertical change in shape as grazing may act to increase tillering, but consistent grazing would reduce the average height.

Bullock *et al.* (1994) demonstrated that two different species of tussock differed in their rate of response to grazing, yet responded in very similar ways. Similarly, the tussock grasses showed similar seasonal growth patterns and population dynamics. This finding, along with others, suggests that despite differences in tussock species, *Merxmuellera* species may follow similar seasonal patterns and responses. This PhD research may thus be relevant and beneficial to studies concerning *Merxmuellera* species across the alpine region of Lesotho and assist in their conservation.

10.3.1 Future of *M. drakensbergensis* communities

M.drakensbergensis communities are listed as 'unprotected' despite many communities falling within the Maluti-Drakensburg Transfrontier Project and have thus been determined critically endangered (Bureau of Statistics, 2012). *M.drakensbergensis* communities also fall within Lesotho Highlands Basalt Grassland, another poorly protected vegetation classification which has decreased by 55.3% between 1950 and 2007 (Bureau of Statistics, 2012).

This study has noted the increased use of grazing posts in the study region and the movement towards continuous grazing. The effects will be increased pressure placed on the grasslands (Quinlan and Morris, 1994). Whether *Merxmuellera* communities can withstand the increased pressure remains to be seen, but the results of this study suggest their growth will be stunted. Not only does the future of Merxmuellera tussocks remain uncertain, but they perform many roles within their niches and the future of these roles and those they affect remain just as uncertain. The interspace areas between tussocks often contain the greatest diversity (Körner, 2004). This is often attributed to the sheltering effect of the taller denser tussock grasses. Similarly, increased pressure leading to a decrease in tussock numbers will decrease the shelter and food available for many invertebrate species. The extent of the reliance of the native Sloggets' ice rat (*Otomys slogetti*) on *Merxmuellera* tussocks remains unknown, thus their futures may be uncertain too.

As Sloggets' ice rats are only found within the Lesotho highlands this species remains fragile due to climate change, land use change and erosion. Tussock grasses act to stabilize slopes through their dense root structures and runoff slowing abilities. If they disappear or their extent is reduced, the open spaces may either be filled with smaller flowering plants, invasive species or the top soil may become eroded. The result may be the degradation of slopes and wetlands, putting many more species and livelihoods at risk.

The future of many alpine species is uncertain due to climate change and its variability. Minor changes in soil fertility and moisture are suggested to affect the abundance of *M. drakensbergensis* (Morris *et al.*, 1989). As the tussocks are mostly found in areas of seepage or at stream heads, this would directly relate to the amount of precipitation occurring. Yet, the the slow rate of percolation and the high soil moisture content may act as a buffer to short term changes in precipitation. Additionally, alpine species may be more likely to persist given the topographic variability of alpine landscapes (Scherrer and Körner, 2011). However, competition for space on cooler slopes will be at a maximum (Scherrer and Körner, 2011) and if burning and grazing continue to increase or even be excluded from some areas in the case of burning, the future of *M. drakensbergensis* remains bleak.

10.3.3 Recommendations

Only 1% of the Lesotho highlands grasslands are conserved within the Sehlabathebe and Bokong National Parks (Du Preez and Brown, 2011). Yet many consider the wetlands and grasslands under threat due to their land use (e.g. Du Preez and Brown, 2011). However, attempts to conserve the grasslands and wetlands will face many obstacles. This has caused doubt in the effectiveness of the Transfrontier Park to which the study area belongs (e.g. Du Preez and Brown, 2011). Mofokeng and Ge (2009) strongly believe that the unique nature of alpine environments demands appropriate conservation as well as sustainable land use above all else. Competition for resources (such as ecotourism, grazing, water collection) is becoming more evident and needs to be included more in rangeland management programs (Motsamai, 1995). The aim of many management strategies is to increase inter-tussock species as this promotes an increase in species richness (Lunt, 1994). Additionally, monitoring of indicator species is particularly valuable as changes in their abundance usually correlate to a change in an abiotic element (Morris *et al.*, 1989). Yet, management and monitoring will be difficult due to the low accessibility (only 5-10% is considered accessable) of the region (Schwabe, 1995; Grab and Nüsser, 2001). Thus

management and monitoring of the region will require the inclusion of local people as well as a greater reliability on aerial, satellite and photographic monitoring measures.

Excluding fire from grasslands leads to degradation and thus is not an option (Gibson, 2009). Controlled burning through building on the communal knowledge and capacity as well as implementation of by-laws has been suggested as the solution to this problem in the Mokhotlong-Sanqebethu Managed Resource Area Management Plan, while management grazing plans have been suggested as the solution to overgrazing (MDTP, 2007). However, the issue of controlled burning has been allocated to the Ministry of Forest and Land Reclamation and the Ministry of Local Government (MDTP, 2007). The Ministry of Forest and Land Reclamation does not have the knowledge base nor the capacity to regulate and monitor controlled burning, while the local government deals with many issues and has little capacity for regulation of burning. Patchy burns which create a mosaic landscape are described as being the best for biodiversity conservation (De Ronde *et al.*, 2004). The patchy nature of the Sani region burns is observed both in the satellite images and personal ground level observations as the burning of a few tussocks by herdsmen is common. However, this means that not all tussocks are burnt and many years may pass before they are burnt and this may be detrimental to them. Yet, burning may be a means of rehabilitation in regions where shrubs and invasive species have reduced the forage.

The exclusion of all livestock is one possible solution (Schwabe, 1995). However, this would not benefit local people and may not potentially benefit the grasses. Thus, other options need to be considered. The seasonal movement of livestock can be a useful management tool. Selective grazing cannot be prevented but resting grazing land can allow preferred vegetation to recover (Hormay, 1956). By altering the burning regimes to accommodate tussock grasses and prevent the overgrazing and over burning of the flora in general, the biodiversity will be better conserved in a healthier ecosystem. It has been suggested that protection of auxillary buds thus the regrowth of grasses through a period of no grazing following burning is the best solution (Peláez *et al.*, 2009). This will allow for the re-growth of tillers and new growth to take place. It is suggested that more than six months is a sufficient period of no grazing, as grazing within this period can be damaging, especially within the first month (Peláez *et al.*, 2009). The moving of livestock should be controlled by the amount of leaf material that remains rather than a predetermined date (Trlica, 1999). Yet this is already in practice, however, observation has shown that the livestock is allowed to remain for too long.

In 1946 a system of including grass rows in terrace systems to decrease erosion was attempted. However, the reports stated that the grasses were flooded in the summer and died of cold in winter (Showers, 2005). Where the grasses did survive they were grazed by livestock (Showers, 2005). Perhaps tussock grasses such as *Merxmuellera* species which grow well under those conditions as well as having a lower palatability than other grasses would be ideal to make this scheme work.

Alpine and mountainous areas are being shown by many studies to be sensitive to changes as a result of climate change (Löffler and Pape, 2008). High altitude grassland regions are usually dominated by C_3 grasses as well as regions with high moisture availability, while C_4 grasses tend to decrease at altitudes above 2 400m a.s.l. (Morris *et al.*, 1993; Sieben *et al.*, 2009). However, the mountainous regions of eastern Lesotho are dominated by C_4 grasses that do not fit with our current understanding of temperature regimes. They are thought to be the result of low CO_2 levels during glacial times (McDonald *et al.*, 2002). Previous disturbances, invasive species and climate change may significantly alter the response of communities to a disturbance (Davies *et al.*, 2009). The literature suggests that the more alpine or the more tundra-like the vegetation and climate the greater the destruction by fire, grazing and climate change (Racine *et al.*, 2004). In the light of climate change, multilevel adaptive management practices have emerged as a potential way of mitigating the effects (Bisaro *et al.*, 2010). Yet, more research and willingness of stakeholders need to be in place before this may be successful.

The improvement of ranges is dependent on management practices. Management involves the manipulation of four factors: grazing season, frequency of grazing, distribution of livestock and stocking rate according to Hormay (1956). However, as the literature review has shown the effects of fire cannot be ignored. Altering the management of the Lesotho grasslands is not an easy task. In order to formulate effective policies an understanding of the current communal grazing system is vital (Swallow and Bromley, 1995). This includes detailed knowledge of institutions, government and the relationships between government, institutions and individuals (Swallow and Bromley, 1995). Individuals are particularly important as the success of new policies is dependent on them. The expectations and incentives of individuals need to be taken into account (Swallow and Bromley, 1995). Livelihoods of many Lesotho people are centered around grasslands (e.g. for livestock products or thatching grass). Yet these grasslands are sensitive to changes within the climate (Bisaro, 2007). Collective local adaptive knowledge may be crucial to governing these landscapes in an uncertain future.

Local knowledge may be valuable but as the current study has suggested, livestock may be remaining within the highlands for greater periods of time. The local knowledge may not be adequate to deal with these alterations to the transhumance system, suggesting that new forms of livestock management may be needed in order to prevent degradation in the years to come. The conservation of the biodiversity is vital but the focus also needs to be on the communities and their livelihoods. If the people are not supportive and their needs understood and met in a sustainable way, conservation methods would be futile (Pooley, 2003). People of all ages as well as local institutions need to become involved if biodiversity conservation is to become a success (Pomela *et al.*, 2000).

In the recent past a few different attempts have been made to conserve the Lesotho highlands. One such program was the Range Management Improvement Program (assisted through USAID) relied on the implementation of sustainable management practices (Mbata, 2001). However, this program failed due to the unpopular idea of paying to use more sustainable pastures. Many other projects have failed due to their inability to alter underlying causes of the deterioration (Mbata, 2001). Currently the Lesotho National Parks department gives presentations in schools and public gatherings (Leipzig, 1996). However, these generally focus on the prevention of erosion and not the broader issues behind it. The problem of grassland degradation in Lesotho grasslands may be the result of socio-economic issues rather than ecological resource based (Thomas, 1995). This suggests that grassland researchers should broaden their focus to include the social aspects related to grasslands in order to generate better solutions (Thomas, 1995).

Perhaps the way forward lies in combining the historical practices of *Leboella* and *Maboella.* Rotational grazing and zoning of the highlands for the use of villages and livestock owners may be a step forward in allowing vegetation recovery between grazing and burning events. However, traditionally these systems relied heavily on the participation of the local Chiefs (Mdee, 2004). The strength and leadership abilities of both Principal and village chiefs is vital to any conservation measures or policy implementation (Swallow and Bromley, 1995). However, in many areas chiefs may be ineffective due to a breakdown of the chieftain system or a lack of leadership qualities and might (Swallow and Bromley, 1995; Tshabalala, 1995). This places more emphasis on government institutions and on the individual. Thus, the success of conservation within current political and social frameworks relies on the capacity and leadership of Principal Chiefs, village chiefs, the Ministry of Agriculture, Range Management Associations, individuals and various agencies and their projects.

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Appendices



APPENDICES

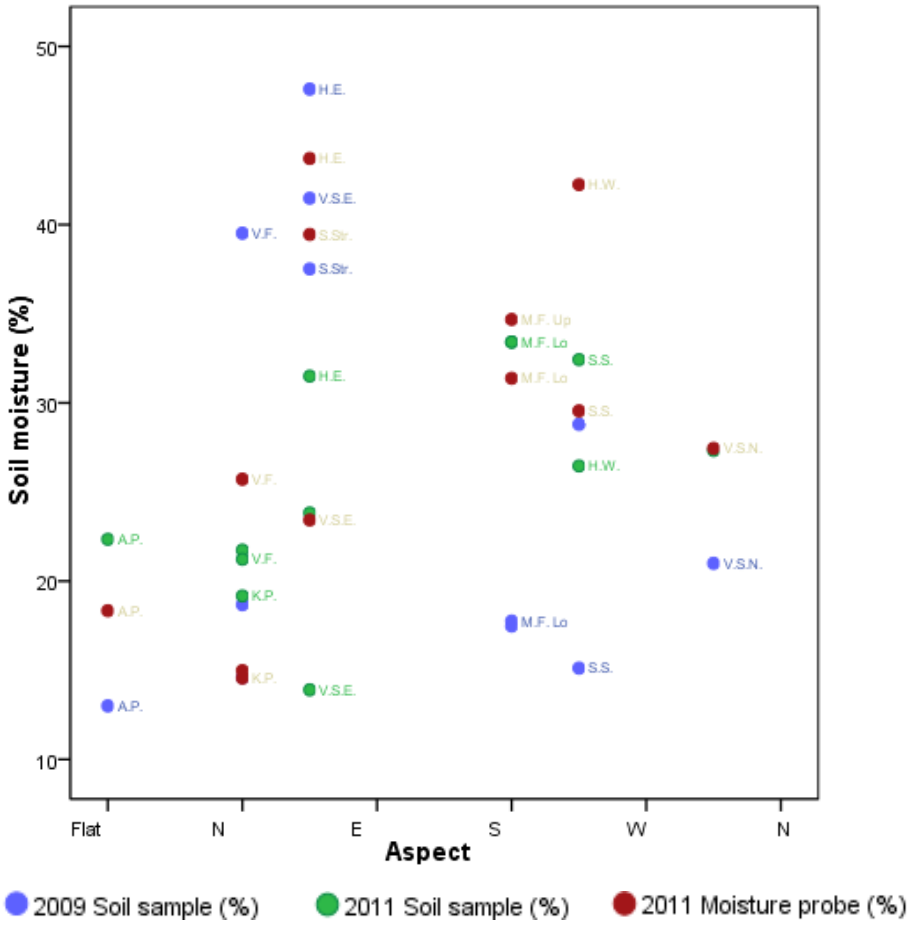


Figure A.1: Soil moisture (%) plotted against aspect.

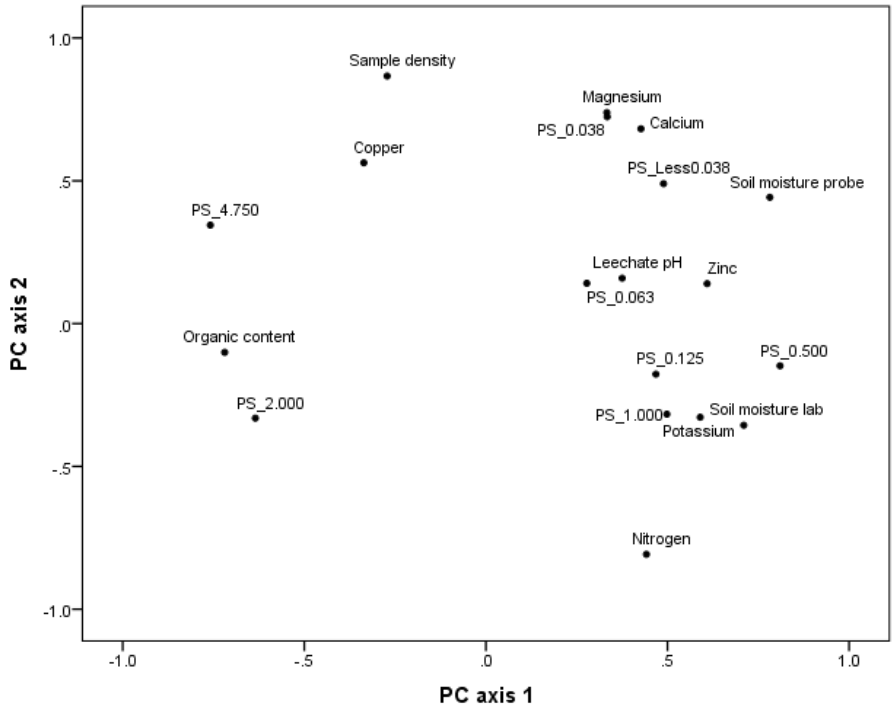


Figure A.1: PCA ordination diagram.

Table A.1: PCA structure matrix.

Soil Characteristics	Component						
	1	2	3	4	5	6	7
Soil moisture lab	0.434	-0.174			-0.841	-0.400	-0.474
Soil moisture probe	0.637	0.644	0.116	0.166	-0.484	-0.442	-0.163
Organic content	-0.314	-0.445		-0.404	0.454	0.407	0.521
Leechate pH	-0.100	0.299	0.161	0.395	-0.711	-0.197	-0.144
Sample density	-0.220	0.754	0.248		0.472		0.483
Potassium	0.355	-0.181		0.806	-0.188	-0.361	-0.514
Phosphorus				0.923			
Calcium	0.224	0.925	-0.187		-0.334	-0.263	
Magnesium	0.123	0.977	-0.169		-0.115	-0.263	
Zinc	0.861	0.293		0.112		-0.266	-0.201
Manganese	0.888		0.254		-0.254	-0.397	
Copper			0.218	-0.179	0.316		0.912
Nitrogen	0.322	-0.524			-0.257		-0.862
PS_4.750	-0.719	0.145	-0.269		0.182	0.389	0.710
PS_2.000	-0.186	-0.181	-0.208	-0.106	0.291	0.942	
PS_1.000	0.564	0.151	-0.673	-0.147	-0.192	-0.105	-0.570
PS_0.500	0.630	0.126	-0.413	-0.120	-0.472	-0.625	-0.456
PS_0.250	0.529	0.113			-0.349	-0.882	-0.435
PS_0.125	0.243	-0.322	0.536	0.351	0.175	-0.597	-0.379
PS_0.063		-0.103	0.959		-0.160	-0.283	
PS_0.045	0.128		0.960		0.110		
PS_0.038	0.104	0.545	0.198	0.221	0.214	-0.695	0.278
PS_Less0.038	0.229	0.224	0.756	0.239	-0.153	-0.635	0.125

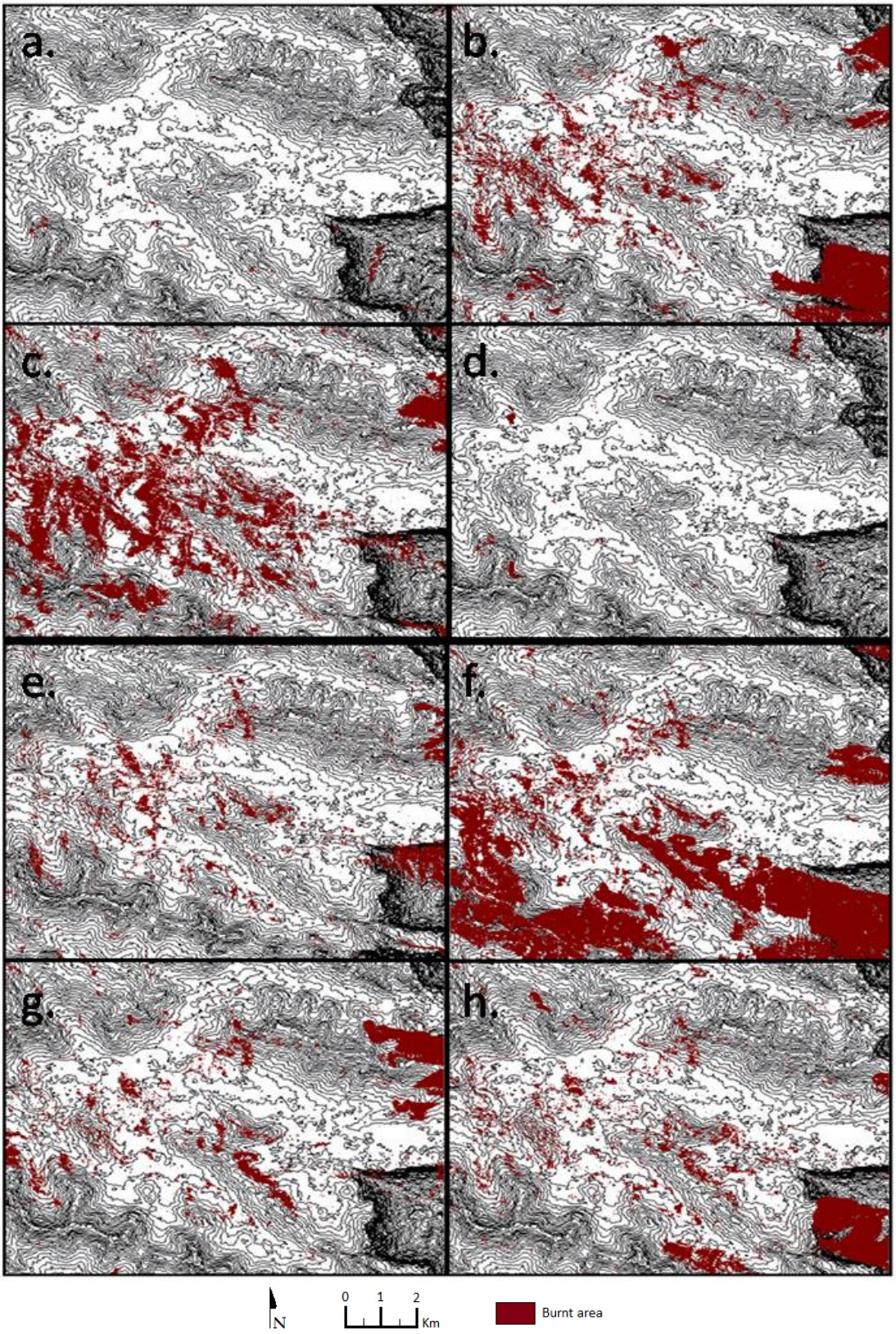


Figure A.3: Burnt areas of the Sani region at the end of August 1993 (a.), and 1994 (b.), as well as early September 1995 (c.), 1996 (d.), at the end of August 1997 (e.), 1998 (f.), 1999 (g.) and 2000 (h.).

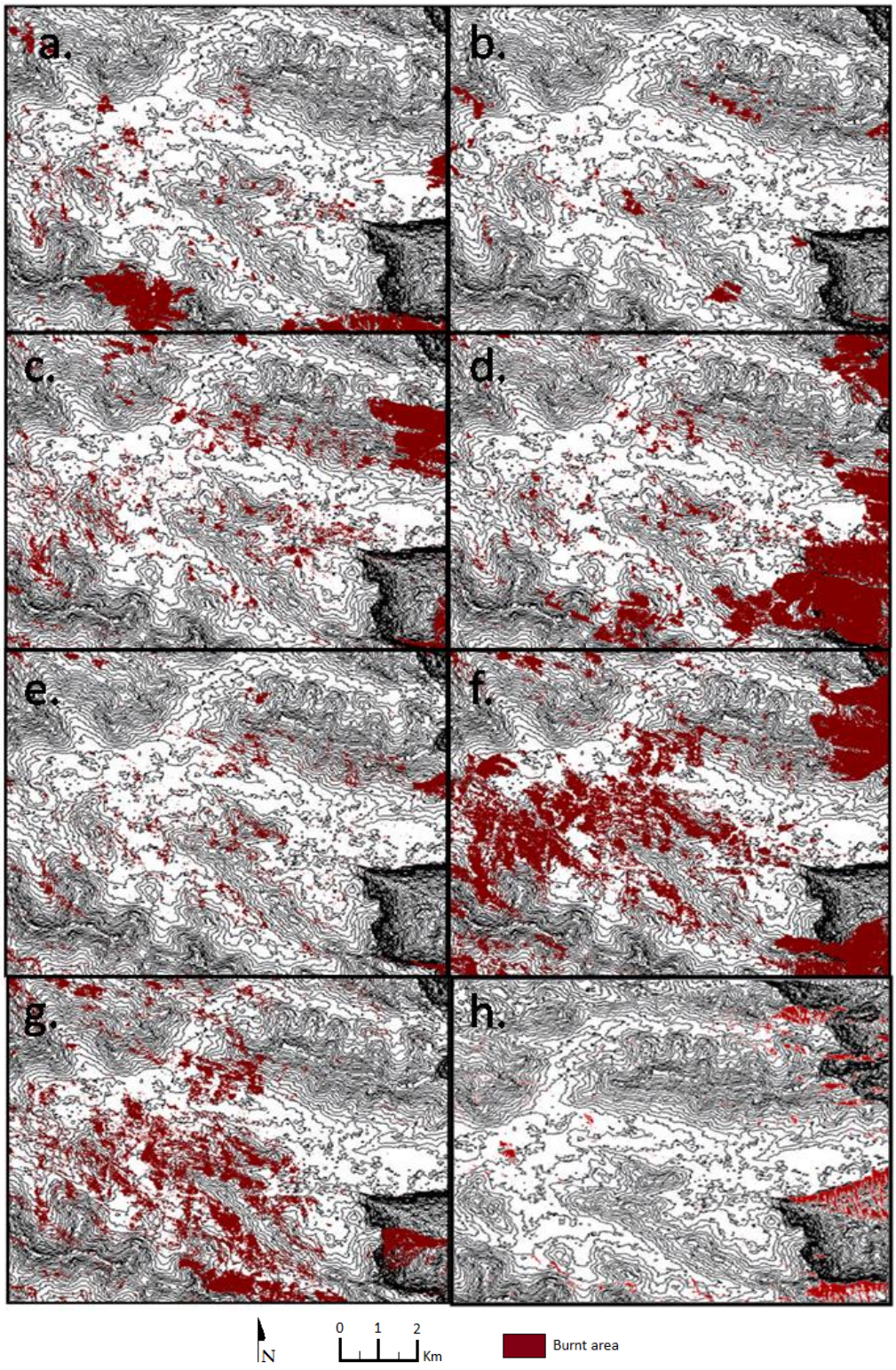


Figure A.4: Burnt areas of the Sani region at the end of August 2001 (a.), early September in 2002 (b.), 2004 (c.) and 2005 (d.), at the end of August in 2006 (e.) as well as early September 2007 (f.) and 2008 (g.), as well as early November 2009 (h).

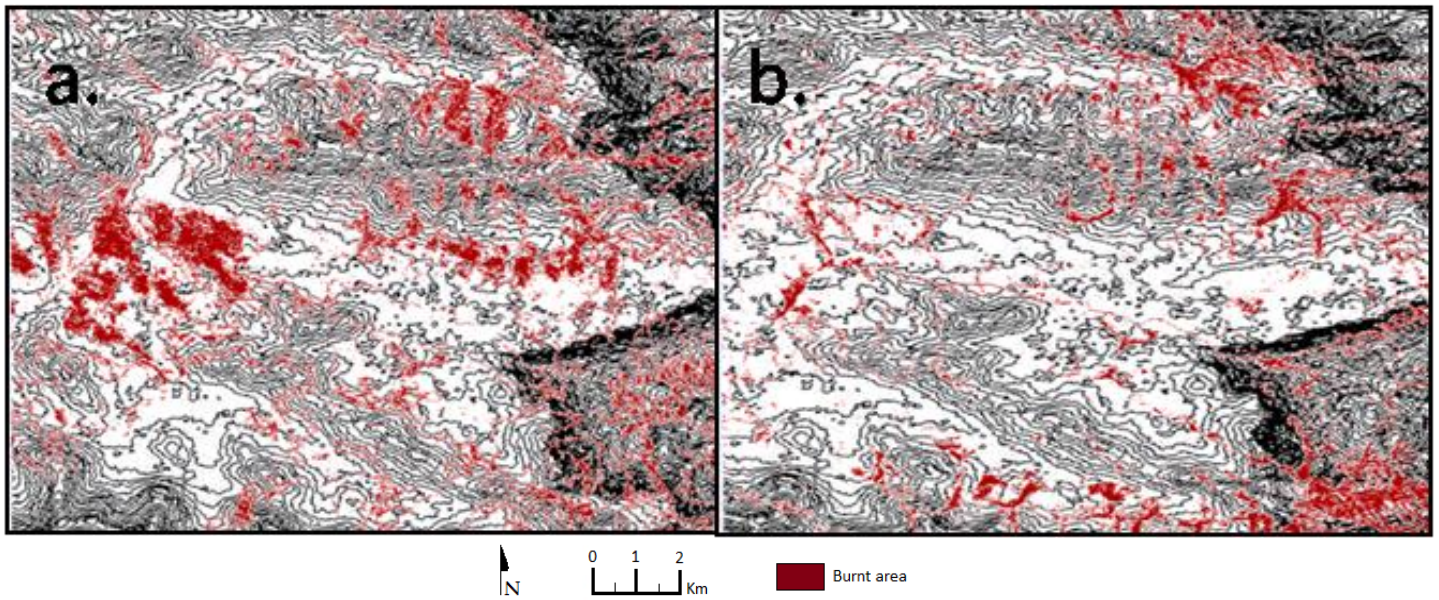


Figure A.5: Burnt areas of the Sani region in early September 2010 (a.) and early August 2011 (b).

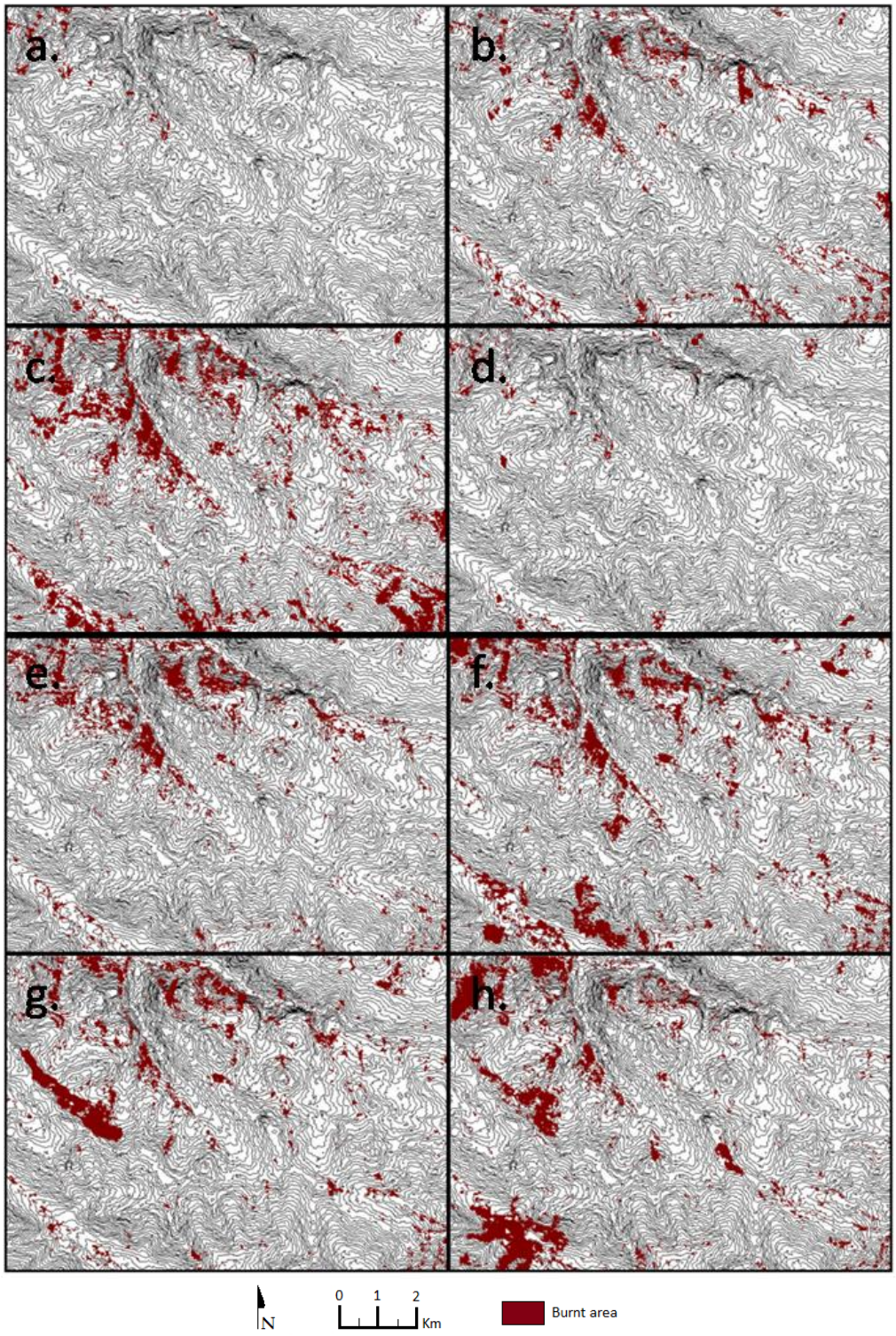


Figure A.6: Burnt areas of the Kotisephola Pass region at the end of August in 1993 (a.) and 1994 (b.) as well as early September in 1995 (c.) and 1996 (d.), the end of August 1997 (e.), 1998 (f.), 1999 (g.) and 2000 (h.).

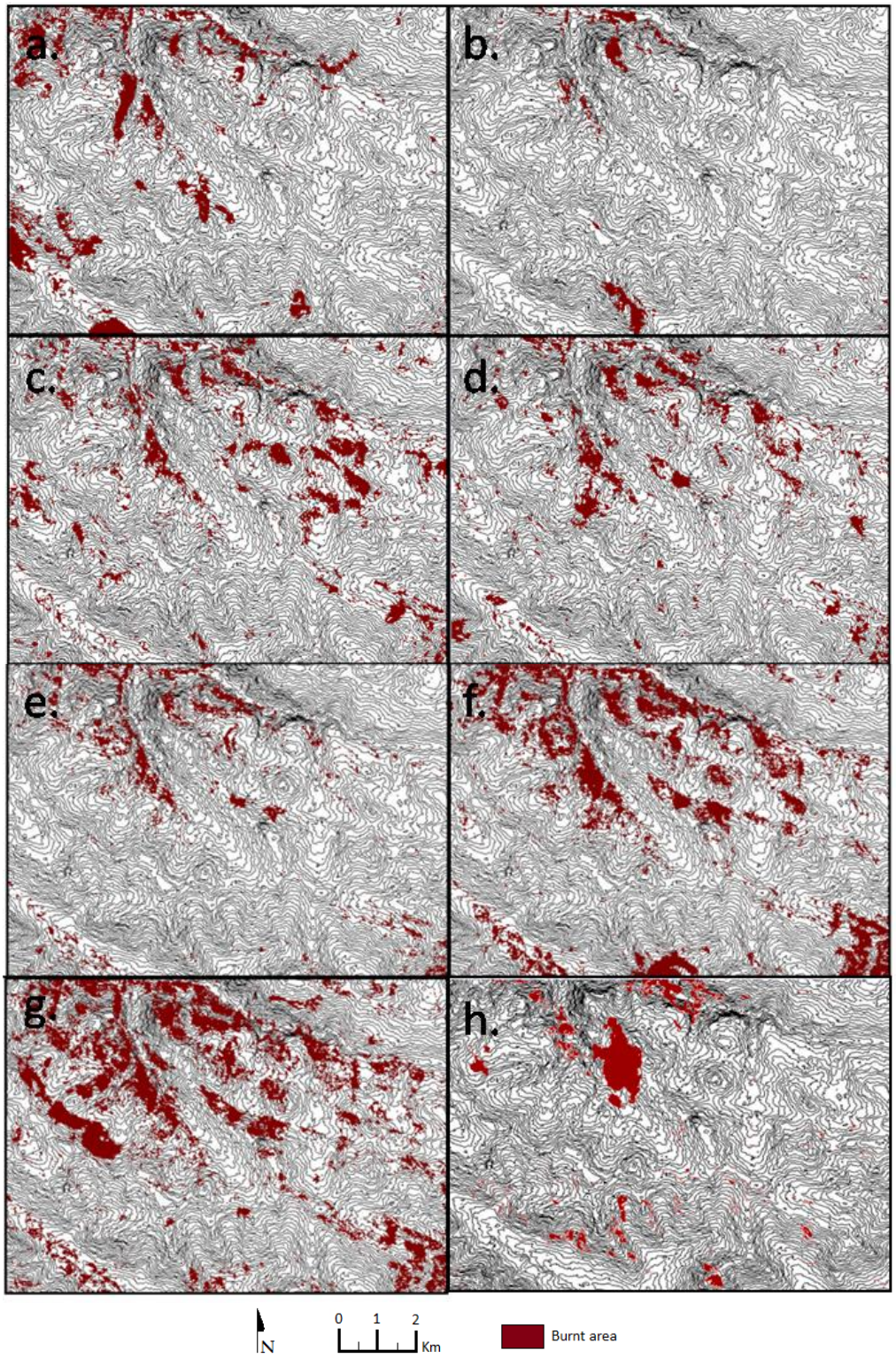


Figure A.7: Burnt areas of the Kotisephola Pass region at the end of August 2001 (a.) as well as early September in 2002 (b.), 2004 (c.) and 2005 (d.), at the end of August 2006 (e.) and early September 2007 (f.) and 2008 (g.), as well as early November 2009 (h.).

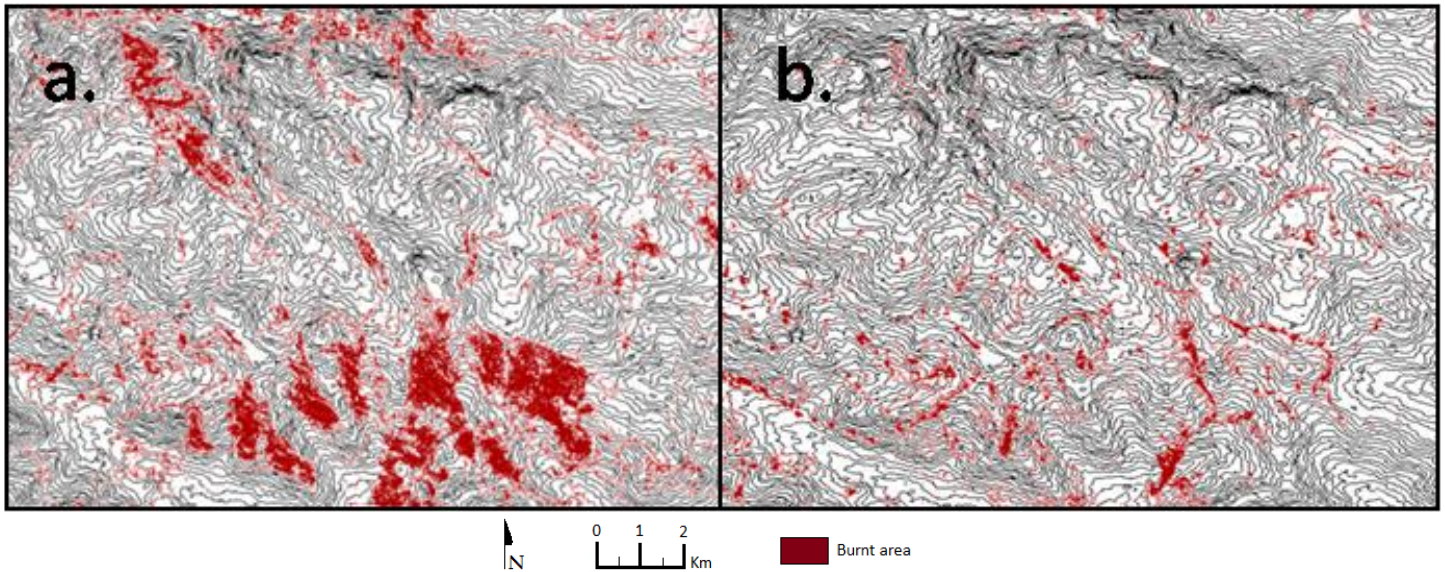


Figure A.8: Burnt areas of the Kotisephola Pass region in early September 2010 (a.) and early August 2011 (b.).

Score Sheet – Baseline Study

Site no.:
Location:

Basal Circumference:

	Diameter				
Quadrat	1	2	3	4	5
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					

Average Height:

	Average Height				
Quadrat	1	2	3	4	5
1					
2					
3					
4					

5					
6					
7					
8					
9					
10					

Basal Area:

	Basal Area				
Quadrat	1	2	3	4	5
1 Living					
1 Dead					
2 Living					
2 Dead					
3 Living					
3 Dead					
4 Living					
4 Dead					
5 Living					
5 Dead					
6 Living					
6 Dead					
7 Living					

7 Dead					
8 Living					
8 Dead					
9 Living					
9 Dead					
10 Living					
10 Dead					

Standing Biomass:

Species 1:

Species 2:

Species 3:

Species 4:

Species 5:

Quadrat/ Rank	1	2	3
1			
2			
3			
4			
5			
6			
7			
8			
9			
10			

11			
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Figure A.9: Field sheets used for the baseline study

Table A.2: GLM full equations (TSLB: time since each site was last burnt)

Variable	**Equation**
Basal circumference	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}14.669\\13.006\\12.167\end{pmatrix}$ Burn frequency $+ \begin{pmatrix}9.114\\8.401\\7.264\end{pmatrix}$ TSLB $+ \begin{pmatrix}10.542\\12.641\\14.935\end{pmatrix}$
Average height	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}3.941\\1.275\\1.007\end{pmatrix}$ Burn frequency $+ \begin{pmatrix}2.697\\1.231\\2.411\end{pmatrix}$ TSLB $+ \begin{pmatrix}15.747\\24.449\\21.902\end{pmatrix}$
Living basal area	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}5.209\\2.151\\2.188\end{pmatrix}$ Burn frequency $+ \begin{pmatrix}1.197\\0.864\\0.890\end{pmatrix}$ TSLB $+ \begin{pmatrix}37.212\\59.209\\57.740\end{pmatrix}$
Standing biomass	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}-6.491\\-2.676\\-4.279\end{pmatrix}$ Burn frequency $+ \begin{pmatrix}0.172\\0.859\\0.111\end{pmatrix}$ TSLB $+ \begin{pmatrix}59.140\\42.584\\57.047\end{pmatrix}$

Table A.3: GLM full equations (TSLB: time since each site was last burnt)

Variable	**Equation**
Inflorescences	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}2.765\\1.211\\3.316\end{pmatrix}$ Burn frequency $+ \begin{pmatrix}0.671\\1.241\\1.926\end{pmatrix}$ TSLB $+ \begin{pmatrix}-0.773\\2.089\\-10.573\end{pmatrix}$

Table A.4: Quadratic polynomial regression full equations for average height (BF: burn frequency, TSLB: time since last burnt).

Variable	**Equation**
Average height	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}27.723\\6.282\\15.188\end{pmatrix}$ BF $+ \begin{pmatrix}-5.037\\-1.246\\-2.581\end{pmatrix}$ BF^2 $+ \begin{pmatrix}15.639\\30.643\\19.886\end{pmatrix}$ $y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}15.244\\3.790\\9.421\end{pmatrix}$ TSLB $+ \begin{pmatrix}-0.990\\-0.197\\-0.521\end{pmatrix}$ $TSLB^2$ $+ \begin{pmatrix}-2.679\\22.071\\1.252\end{pmatrix}$

Table A.5: Quadratic polynomial regression full equations for Living basal area (BF: burn frequency, TSLB: time since last burnt).

Variable	**Equation**
Living basal area	$y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}20.797\\5.326\\6.391\end{pmatrix} BF + \begin{pmatrix}-3.043\\-0.856\\-1.053\end{pmatrix} BF^2 + \begin{pmatrix}31.495\\64.977\\63.487\end{pmatrix}$ $y\begin{pmatrix}2009\\2010\\2011\end{pmatrix} = \begin{pmatrix}8.331\\3.426\\5.011\end{pmatrix} TSLB + \begin{pmatrix}-6.649\\-0.211\\-0.273\end{pmatrix} TSLB^2 + \begin{pmatrix}40.145\\60.298\\51.560\end{pmatrix}$

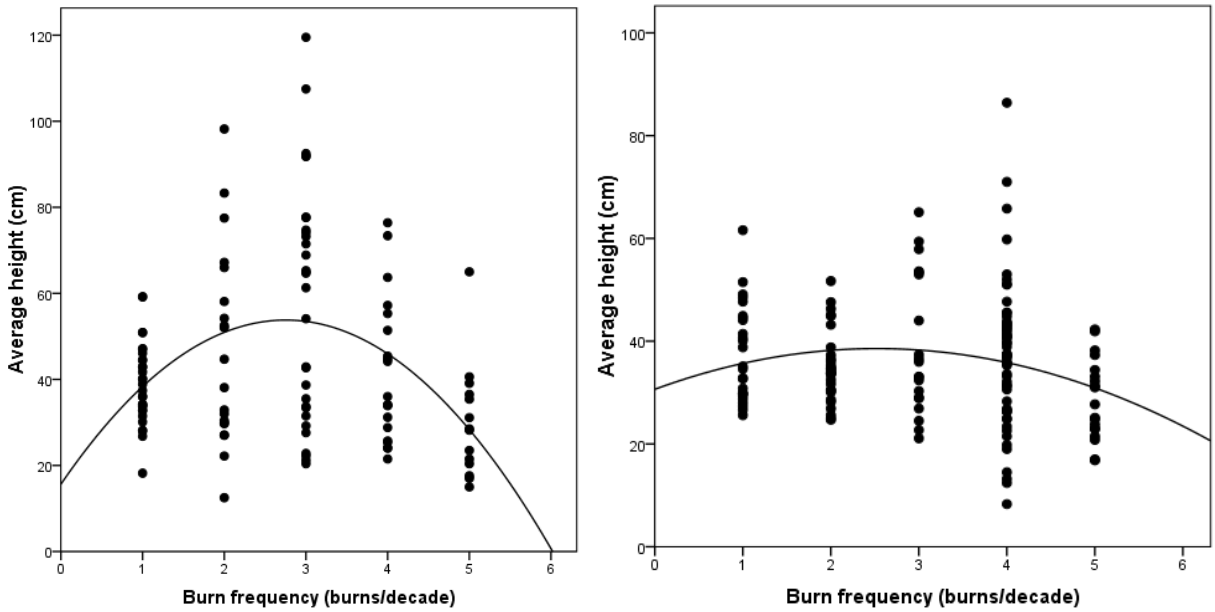


Figure A.10: Scatterplot for average height and burn frequency in 2009 (left) and 2010 (right), indicating the quadratic trendlines.

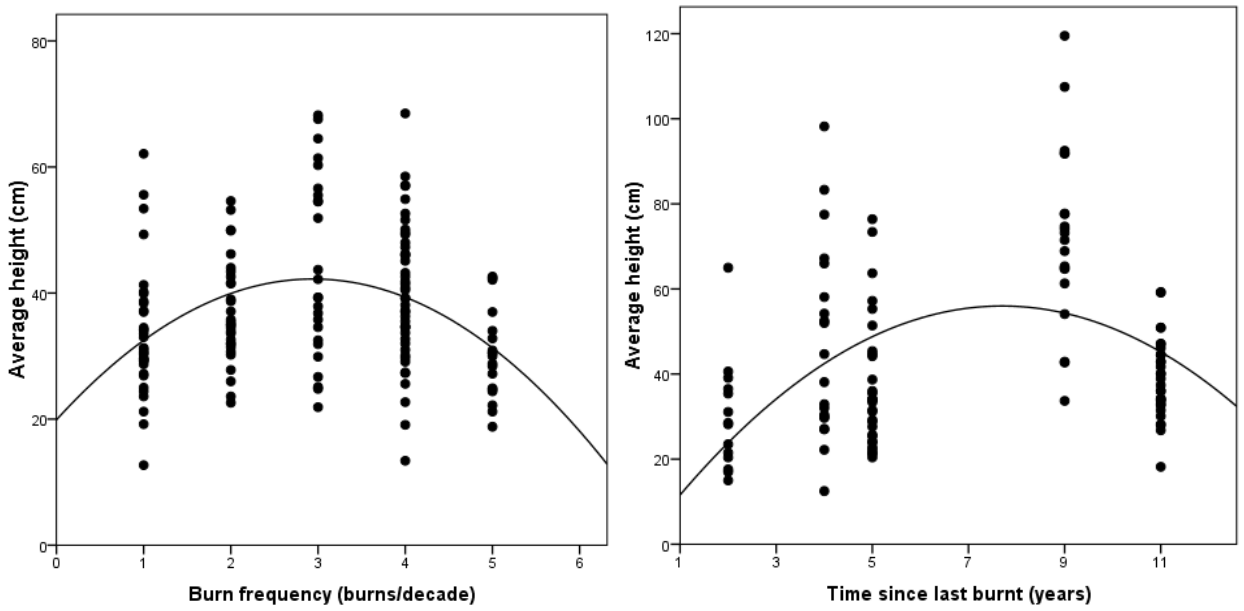


Figure A.11: Scatterplot for average height and burn frequency in 2011 (left) and for average height and time since last burnt in 2009 (right), indicating the quadratic trendlines.

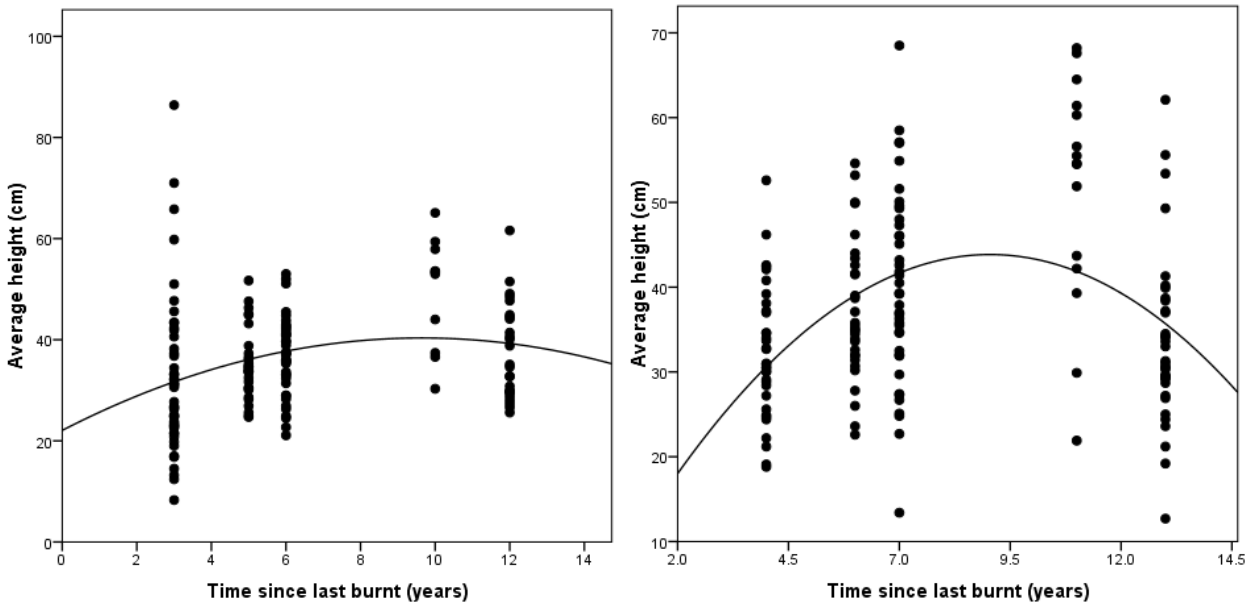


Figure A.12: Scatterplot for average height and time since last burnt in 2010 (left) and 2011 (right), indicating the quadratic trendlines.

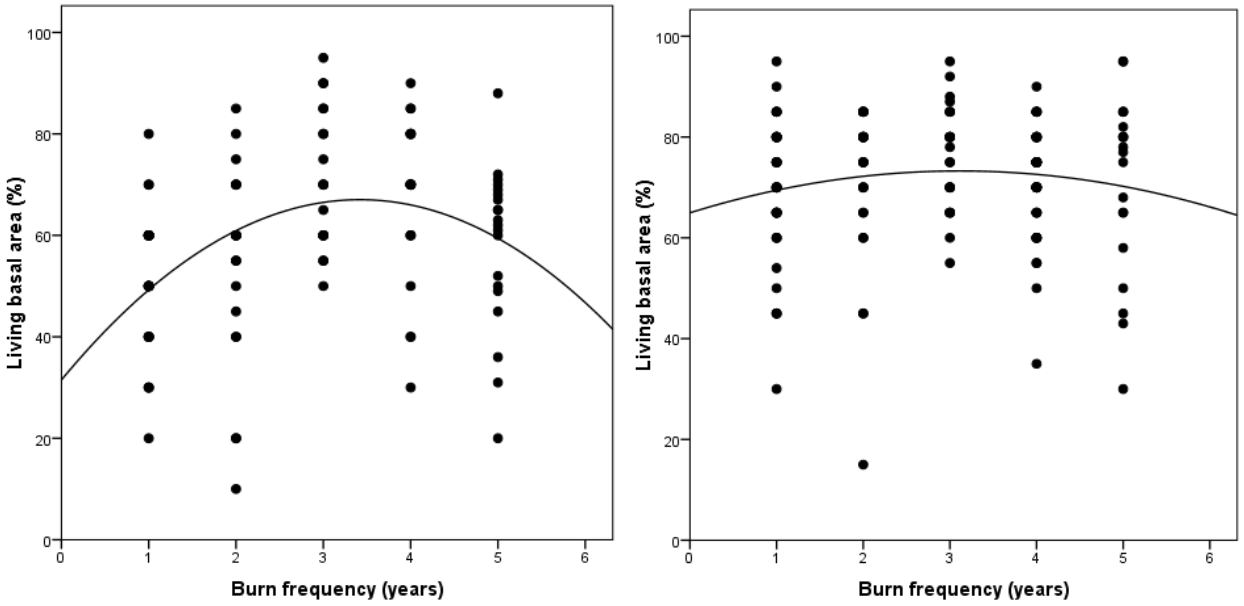


Figure A.13: Scatterplot for living basal area and burn frequency in 2009 (left) and 2010 (right), indicating the quadratic trendlines.

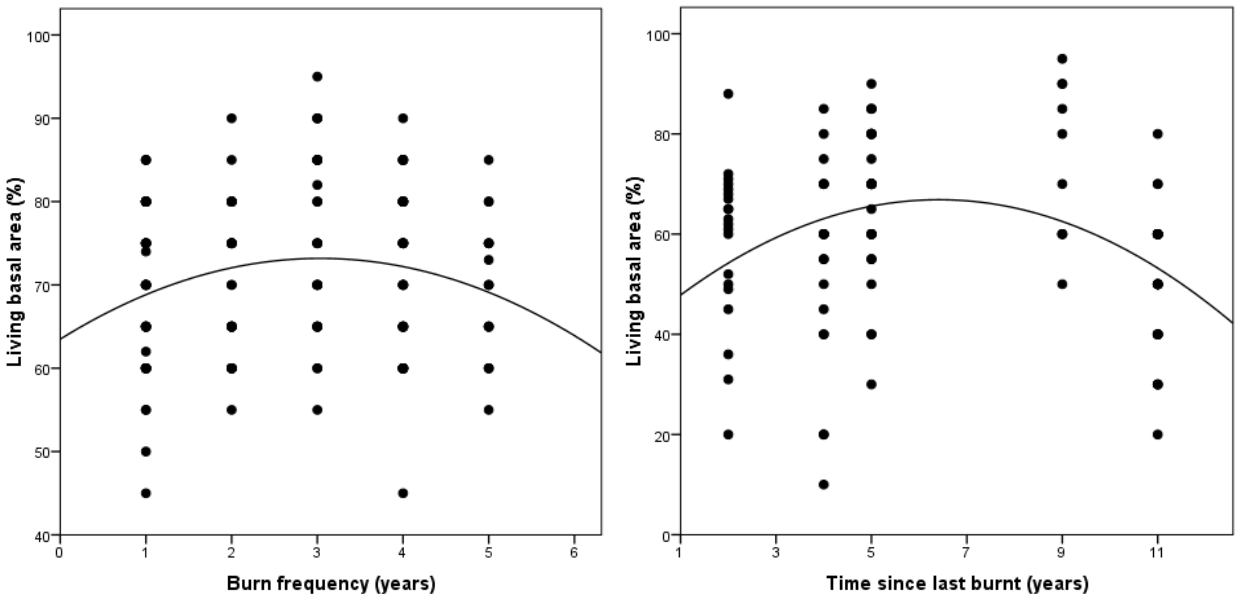


Figure A.14: Scatterplot for living basal area and burn frequency in 2011 (left) and time since last burnt in 2009 (right), indicating the quadratic trendlines.

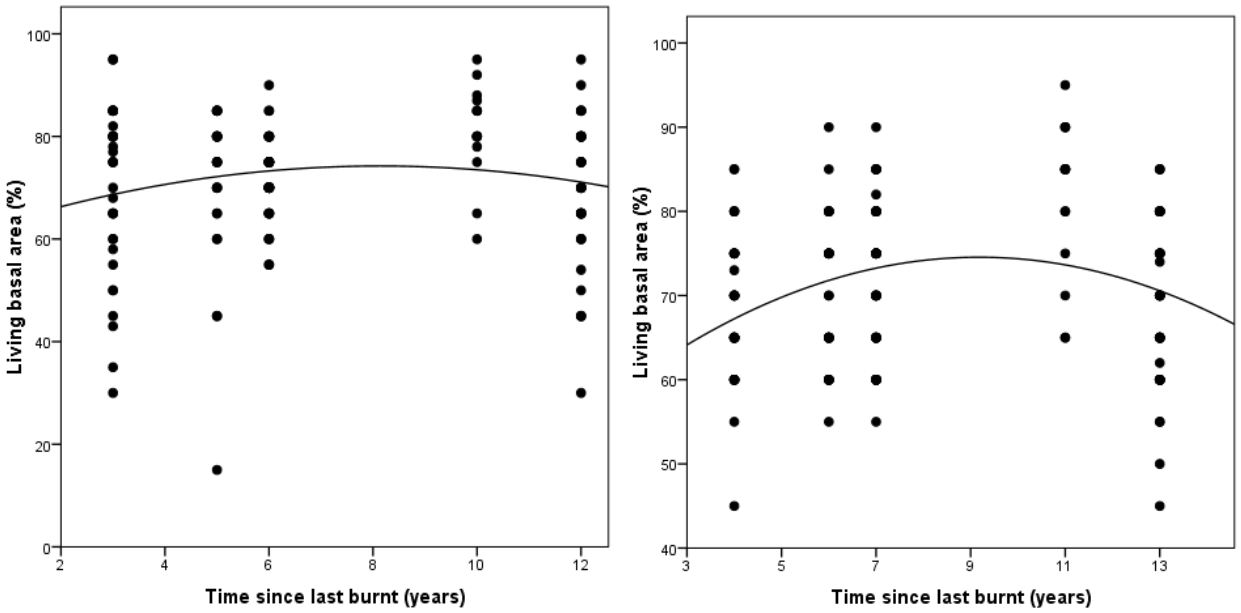


Figure A.15: Scatterplot for living basal area and time since last burnt in 2010 (left) and in 2011 (right), indicating the quadratic trendlines.

Site:
Number/Location:.....
Vegetation type:.....

Soil Characteristics: Moisture: % stones/pebbles:
(out of 5) Compaction: Colour:

Slope:

Altitude:

Aspect:

Erosion:

Feature	Number
Wind-blown soil mounds	
Pedestals	
Exposed roots	
Terraces	
Rills	

Vegetation cover:
% vegetation cover:..... % bare ground:.....

Conrad and Poulton Class:

Leaf length:

Number	Tiller 1 (edge)	Tiller 2 (middle)	Tiller 3 (1/3)	Grazed (y/n)
1				
2				
3				
4				
5				
6				

Leaf length:

Number	Tiller 1 (edge)	Tiller 2 (1/3)	Tiller 3 (middle)	Aspect	Grazed (y/n)
11					
12					
13					
14					
15					
16					
17					
18					
19					
20					
21					
22					
23					
24					
25					
26					
27					
28					
29					
30					

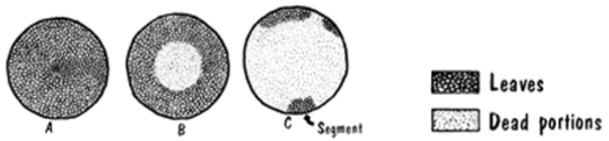


Figure A.16: Field sheet used for the monitoring study.