An analysis of the fire regime and its effects on post-fire recovery in Namaqualand Granite Renosterveld vegetation

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A thesis submitted in fulfilment of the requirements for the degree *Magister Scientiae*, in the Department of Biodiversity and Conservation Biology, Faculty of Natural Science, University of the Western Cape.

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Keywords:

- Namaqualand Granite Renosterveld
- Fire regime
- Livestock grazing
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- Plant diversity
- Forage productivity
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- Heterogeneity

Abstract

Namaqualand Granite Renosterveld (NGR) is a fire-prone shrubland largely confined to the Kamiesberg Uplands. More than 20% has been transformed by agricultural activities such as grazing and dryland cropping. The history of fires of Namaqualand Granite Renosterveld was assessed over a period from 2000 – 2015. Furthermore, this study aimed to assess the effects of fire and grazing on the dynamics of *Elytropappus rhinocerotis* and the post-fire vegetation recovery. Satellite imagery, field observations and combination of field sampling techniques were used to compile a comprehensive fire database and to collect data on vegetation parameters. The Kamiesberg Uplands has a human-induced fire regime with a fire frequency of 4.2 years, which appears to be shorter for NGR compared to other Renosterveld – types. The majority of the fires were in summer and early autumn which is the ecologically accepted season with high fire intensities. Fire Danger Index ratings for summer burns were in the dangerous and very dangerous categories. In terms of cover, the distances between *E. rhinocerotis* individuals were significantly more on the burnt than on the unburnt sites. The density of unpalatable species on the burnt sites is also strongly associated with *E. rhinocerotis*. *Elytropappus rhinocerotis* appears to facilitate a role as a nurse-plant for various species in renosterveld. The local biotic and abiotic conditions, allows *E. rhinocerotis* to maintain its abundance, thus displaying niche construction by a single species. Plant diversity and vegetation composition was not significantly different across the two treatments. Grazing only had a significant effect on species richness in the burnt sites. As the ratio of increaser species became higher, decreaser species declined significantly seven years after the burn. Since, renosterveld is situated in an agricultural landscape, the area should not be burnt more frequently than seven years as to improve forage productivity in an area with high stocking densities. Burning the area every seven years will not have a negative effect on species richness, as all the species recorded for this study are least concerned. Lastly, the vegetation has not

become homogenous in response to land use pressures such as livestock grazing, fire, climatic changes and microhabitats. The different burnt patched combined with grazing created heterogeneity across renosterveld rangeland.

Declaration

I declare that '*An analysis of the fire regime and its effects on post-fire recovery in Namaqualand Granite Renosterveld vegetation*' is my own work, that it has not been submitted for any degree or examination at any university, and that all the sources which I have used or quoted have been indicated and acknowledged by complete references.

…… ……

Megan Brigette Simons Date

Dedicated to my parents Bernard and Celeste Simons

> "*Happiness can be found, even in the darkest of times, if one only remembers to turn on the light*." ~ Professor Albus Dumbledore

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Acronyms and abbreviations list

ARC: Agricultural Research Council

BI: Burning Index

CFR: Cape Floristic Region

CSA: Conservation South Africa

CV: Coefficient of variation of annual precipitation

DAFF: Department of Agriculture, Forestry and Fisheries

ECR: Extra Cape Subregion

FDI: Fire Danger Index

FRI: Fire return interval

FRP: Fire Radiative Power

GCFR: Greater Cape Floristic Region

GPS: Global Positioning System

MAP: Mean annual precipitation

MODIS: Moderate Spatial Resolution Spectroradiometer

MTE: Mediterranean-type ecosystem

NASA: National Aeronautics and Space Administration

NGR: Namaqualand Granite Renosterveld

PCA: Principal Component Analysis

SSU: Small Stock Unit

RP: Return period

Table of contents

List of Tables and Figures

List of Tables

List of Figures

1. Chapter one: Introduction and literature review

1.1. Background and Rationale

Renosterveld remains one of the least understood vegetation types of the Fynbos biome (Jacobs and Jangle 2008). It is characterised by the dominance of Asteraceae species, especially *Elytropappus rhinocerotis* (L.f) Less.[1] (renosterbos) (Boucher and Moll 1981), and an understory of shade tolerant herbs and geophytes (Mucina and Rutherford 2006; Milton 2007; Jacobs and Jangle 2008). Renosterveld typically occurs on clay – rich soils derived from shale of the Malmesbury and Bokkeveld Groups and the Karoo sequences. In more arid regions it is confined to granite derived soils (Mucina and Rutherford 2006; Keeley *et al*. 2012). Renosterveld is divided into two broad groups based on their geographical distributions - West Coast Lowland Renosterveld (extending from the Cape Peninsula to the Cape Fold Mountains) and Mountain Upland Renosterveld (occurring around the Cape South Coast to the western fringes of the Northern Cape). These two groups have several different types of Renosterveld communities within themselves (Winters 2007; Parker and Lomba 2009).

Mountain Upland Renosterveld occur in small pockets and has semi-arid Mediterranean climates (Rebelo 1995; López-Bermúdes *et al*. 1998; Bergh 2006; Bergh *et al*. 2007; Parker and Lomba 2009). Mountain Upland Renosterveld exhibits strong karroid affiliations (Van der Merwe and Van Rooyen 2011a) and occupies an intermediate zone between the Fynbos biome and Succulent Karoo biome (Mucina and Rutherford 2006). This boundary is defined by the interplay between fire and succulence of the vegetation. Though there have been a few studies which focused on the endemic flora and vegetation, post-fire vegetation trends, soil microbial ecology and insect diversity (Helme and Desmet 2006; Colville 2006; Van der Merwe and Van

[1] The name *Elytropappus rhinocerotis* is given preference throughout this thesis as the new name *Dicerothamnus rhinocerotis* (by M. Koekemoer) has not been published.

Rooyen 2008a; Van der Merwe and Van Rooyen 2008b; Van der Merwe and Van Rooyen 2010; Van der Merwe and Van Rooyen 2011; Solomon 2015). A few of these studies have been conducted under private farming management and in conservation areas. However, no study has looked at these ecosystem components in a communal farming system.

1.2. Motivation for study

The natural fire frequency for Mountain Renosterveld, like its Lowland counterpart, is undefined and thus a critical component needed for management and conservation of this vegetation type is unknown. Furthermore, climate change models (Hoffman *et al*. 2009) predict temperature increases for the Namaqualand region, where Upland Renosterveld also occur, as well as an increase in summer rainfall incidences which could result in more erratic events such as lighting and thunderstorms (Helme and Desmet 2006). The Kamiesberg areas in Namaqualand is dry during summer and more unpredictable fire events could occur, which could be harmful to the biota. In addition, extensive burning from human activities has also contributed to fires in the region.

Fires are of ecological and social importance, thus altering (due to climatic changes and human interferences) the timing and frequency of fires will have negative consequences for affect biodiversity and agriculture (Buhk *et al*. 2007). This might result in damaging consequences for the conservation of the area, vegetation and faunal species (Buhk *et al*. 2007). More importantly this could have detrimental effects for communal livestock farmers in the region since their livestock primarily depends on the rangeland as a source of feed.

In addition, very little is known about the post-fire regeneration (resprouting from surviving parental tissue and seedling recruitment) and reproductive responses (fire-stimulated and fire intolerant recruitment) of Renosterveld species after fire (Curtis 2013; Pausas and Keeley

2014). Assessing whether regeneration mechanisms can survive with on-going land uses and changes in climate are crucial in understanding the ecology of Renosterveld (Buhk *et al*. 2007). Furthermore, investigating the dynamics of *E. rhinocerotis*, specifically focussing on how this shrub utilises fire and grazing to its advantage in order to outcompete the surrounding plant species are also important. *E. rhinocerotis* encroaches the vegetation of Namaqualand therefore determining the extent to which it affects conservation (biodiversity) and rangeland productivity (for grazing) of renosterveld could inform the management of the area.

1.3. Significance of the study

The study site is in the Leliefontein Communal Area (particularly around Leliefontein village), which is situated in the Kamiesberg Upland of Namaqualand. The village is home to large remnants of renosterveld vegetation. Namaqualand Granite Renosterveld is found mainly in the upper plateaus of the Kamiesberg but is also found elsewhere in Namaqualand with over 20 % transformed (due to agriculture, primarily by ploughing for cereal and planting of grazing) (Helme and Desmet 2006). However, further heavy grazing can have a significant impact on Namaqualand Granite Renosterveld threatening its conservation status. Namaqualand Granite Renosterveld has the second highest number of endemic species in the Kamiesberg as 21 endemic and near endemic species are found or restricted to this vegetation. Furthermore, fire could pose a threat to the agricultural system if it is not managed properly since livestock farmers will lose their valuable forage resource.

1.4. Literature review

1.4.1. The spread of fire in Africa

Fire is a natural disturbance process with spatial and temporal characteristics (Brown *et al*. 2007). Fires have been an essential ecological process burning terrestrial vegetation since the Mesozoic times (Bond and Van Wilgen 1996). Hominids mastered fires about 1.8 million years ago (Brain and Sillen 1988; Anderson 2012) putting in motion an environmental practise which, till this day, continues to transform various landscapes (Pausas and Keeley 2009). For example, the vegetation of Australia was first transformed by Aboriginal fire stick farming and then by burning practices of European settlers (Singh *et al*. 1981; Kershaw 1986; Pyne 1991). In North America, fires set by the first human settlers expanded the prairies and ate into woodlands and forests (Pyne 1982; Axelrod 1985). In Africa, the evergreen forests shrunk and grasslands and savannas expanded due to fires. Thus, this reinforces the fundamental conclusion that fire is a general phenomenon throughout the world (Bond and Van Wilgen 1996; Bond 2005) and cannot be ignored when considering the management of rangeland ecosystems whether for livestock or biodiversity.

The African continent is highly prone to lightning storms which allows the continent to support fire. In addition, it has a fire climate comprising of wet and dry periods which makes it ideal for fires (Bond and Van Wilgen 1996). Africa has the most extensive area of tropical Savanna in the world, which is characterized by the grassy understory that becomes extremely flammable during the dry season (Tainton 1999). The role of fire in Africa, south of the Sahara, ranges from “a rare phenomenon” in both the driest regions (including hyper arid and semi-arid deserts) and in the wettest region (including wet montane forests), to annual or biennial occurrences (Bond and Van Wilgen 1996). Fires are caused mostly by lightning and anthropogenic factors. In the Savannas of West Africa, human-caused fires are the most

important factor affecting plant communities. Pastoralists mostly use fire to stimulate grass growth for livestock whereas subsistence agriculturalists use fire to remove unwanted biomass when clearing agricultural lands. On the other hand, fires caused due to lightning are more common in the humid equatorial areas (Pyne *et al*. 2004).

Due to the widespread occurrence of biomass burning, specifically in the savanna and grassland biomes, Africa is referred to as the ‘’Fire continent” (Komarek 1971). As a result, prescribed burning has become an important ecological factor in Grassland and Savanna biomes of Africa and research investigating the effects of the fire regime on the biotic and abiotic components of the ecosystems has been conducted in these regions since the early periods of the twentieth century (Trollop 2004). The impacts of fires on the environment is not only complex but varies to a great extent depending on the fire regime (i.e. season of burning, intensity and fire frequency) and its interaction with other environmental parameters such as grazing pressure and patterns, weather (seasonality of climate and rainfall variability) and landscape heterogeneity (Hudak *et al*. 2004; Archibald *et al.* 2005; Govender *et al*. 2006).

The occurrence of fire is found in every biome in the world, however since 1990, 86 % of fires world-wide occurred in tropical Savannas and Grasslands. The increase of fires in regions such as western United States (Westerling *et al*. 2006), Canada (Gillett *et al*. 2004), and the Neotropics (Barlow and Peres 2004) have been associated with recent changes in climate (Reside *et al*. 2012). Furthermore, fire occurrences in Brazil, central Asia, south-eastern Europe, southern Africa and Australia (the world’s most fire-prone continent) are predicted to have a rapid increase in fire events (Reside *et al*. 2012). In South Africa, fire has an integral role in the vegetation dynamics in several biomes (Fig. 1.1).

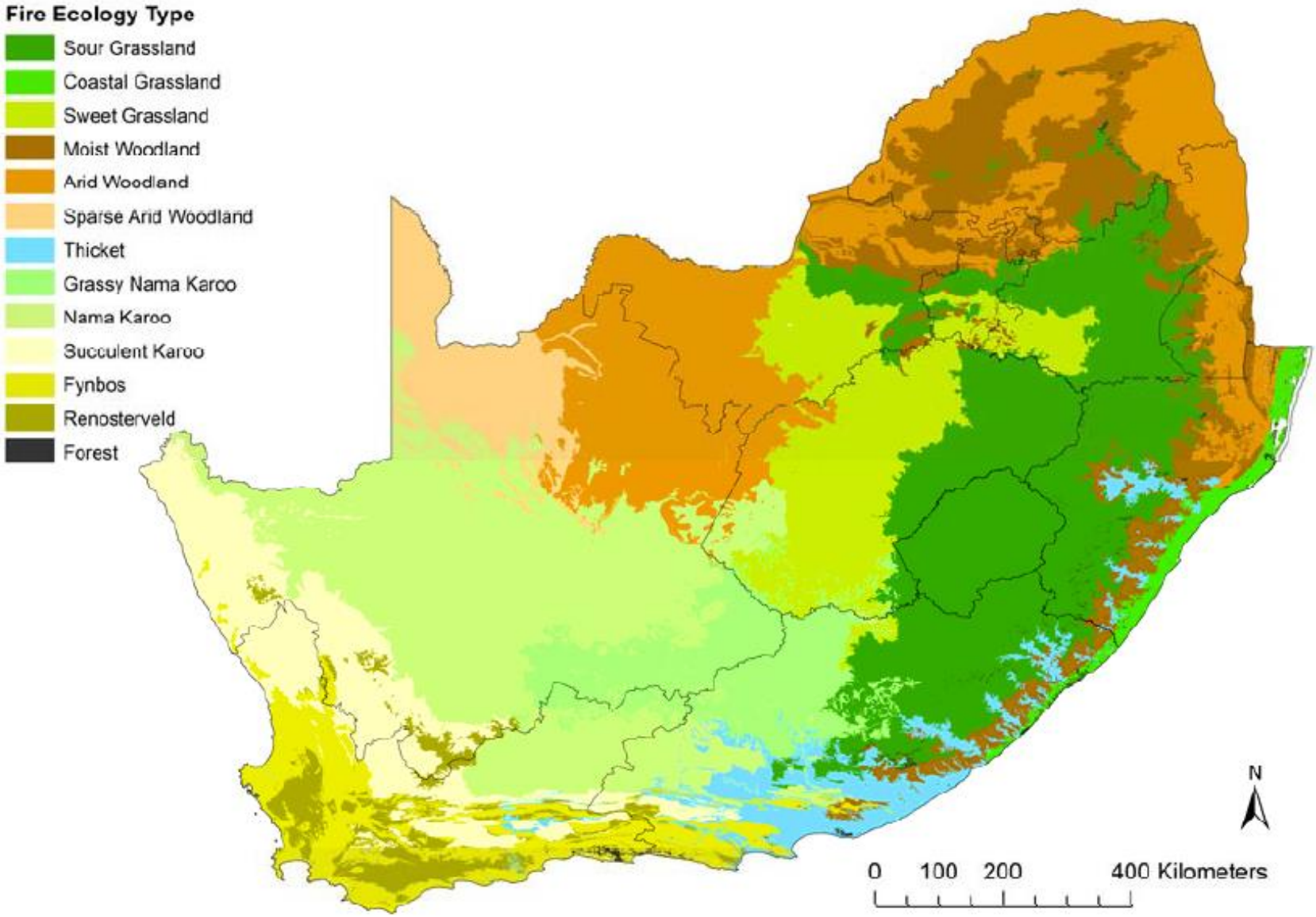


Figure 1.1: The distribution of the 13 fire based vegetation types in South Africa (from Forsyth *et al*. 2010).

1.4.2. Understanding fire regimes

Fire regimes are ecological drivers because it can alter the functioning, structure and composition of the ecosystem (Forsyth *et al*. 2010). Fire is a strong competitor because it is not limited by either toxins or protein deficiency and readily consumes dead woody biomass, but by contrast it is often limited by ignition sources and continuity of fuels (Van Wilgen *et al*. 2010). The environmental (weather) conditions must be such that the fuel is dry enough to burn, some source of ignition must be present and there must be sufficient fuel (also known as the fire triangle). All three elements are necessary for fire, and removing one, will make it possible to extinguish a fire (Trollope 2004). Understanding the characteristics which control fire regimes are of growing importance as the size and severity of wildfires increases in many regions (Westerling *et al*. 2006).

The primary driver of variable fire regimes has been regional and international climate fluctuations. The recent increase in temperature in certain ecosystems and regions has been partly responsible for the increase in the number and size of wild fires (Westerling *et al*. 2006). The fire regime thus has an important role in fire-maintained ecosystems (Trollope 1992).

Fire regimes are also driven by local conditions, particularly the spatial distribution of flammable fuels and ignitions as determined by complex interactions between physical and ecological processes (Turner 2010). Moreover, the fire regime may also be driven by either ecological conditions or anthropogenic conditions (such as human interferences), or even a combination of these factors. Therefore, to understand the dynamics of various fire regimes, it is key to have an understanding of the physical, ecological and human effects across multiple scales of time and space (Turner 2010). Historical fire events are essential for providing necessary data of past and present fire regimes and the changes in fire regimes over space and time (Falk *et al*. 2011).

1.4.3. The components of the fire regime

1.4.3.1.Type of fire

Fires vary in their types: a) head fires usually burns with the wind or upslope and b) back fires burns against the wind or down slope. The variability also includes ground fires which burn underground in the organic layer; surface fires which burn just above the ground surface (e.g. in Savanna and Grassland areas this is the most common types) (Bond and Van Wilgen 1996; Trollope 2007). Crown fires which burn in the canopies of trees and shrubs, for example South African fynbos has a crown fire regime (Forsyth and Bridgett 2004; Keeley *et al*. 2012). Crown fires are usually sustained by surface fires below. However, under extreme conditions "independent crown fires" occur and they can race ahead of the surface fires (Trollope 2007; Forsyth *et al*. 2010).

1.4.3.2.Fire ignition sources

Fire will only occur if some sort of ignition is present. The ignition sources do not have any ecological significance but a change in the primary ignition source can alter the entire fire regime (Forsyth *et al.* 2010). Lightning events has been the prime cause of fires in natural vegetation (Pyne *et al*. 2004). In the Cape Mountains, more than half of the fires recorded in Fynbos shrublands were ignited by lightning. Although sparks from hardened quartzite rocks have also been recorded as a cause of fires in the Cape. Fynbos is the dominant vegetation in Table Mountain National park and the majority of fires are either wildfires, fires of unknown origin, natural (as mentioned above) and to a lesser extent prescribed burning (Forsyth and Van Wilgen 2008). In Bontebok National park the cause of fire for renosterveld vegetation was similar, as the cause of fires were either accidental or prescribed (Kraaij 2010).

Humans remain the main cause of contemporary fires (Bond and Van Wilgen 1996), as they can alter fire by actively suppressing fires and altering vegetation by introducing new plants and land-use activity (e.g. increase/decrease grazing) (McWethy *et al*. 2013) Furthermore, human activities (such as arson, accidental or prescribed burns) can result in wildfires from a variety of heat sources (e.g. cigarette butts, matches and sparks from power lines). Prescribed burning for the removal of moribund material, unacceptable plant material, control and/or prevent the encroachment of undesirable plants in appropriate biomes are recommended (Trollope 2004). Climate affects the fuel characteristics (vegetation type, fuel moisture, continuity and abundance). The increase in large fires has been attributed to drier summers and severe drought and increased plant mortality have exacerbated fire hazard (McWethy *et al*. 2013). Today it is very rare to find fire regimes which are unaffected by humans (Fig. 1.2) whether intentionally (for land management or arson) or by accident (Bond and Van Wilgen 1996; Archibald *et al*. 2011).

1.4.3.3.Fire season

Fire seasonality is a function of the coincidences of ignition with conditions of the fuels (Archibald *et al*. 2010). The season of burning should also overlap with the reasons for burning and the type of vegetation (Forsyth *et al*. 2010). Most ecosystems have an ideal fire season when fires are more likely to occur (Keeley *et al*. 2012), which are usually centred on the hottest and driest time of the year (Kraaij and Van Wilgen 2014). In the South-Western Cape hot, dry summers (November to April) would be the recommended season for burning. In other areas dry winters (May to October) provide suitable conditions (Forsyth *et al*. 2010). Thus, for fynbos and renosterveld communities the fire season is in summer and early autumn which are the ecologically accepted season (Forsyth and Van Wilgen 2008). However, human

interferences have substantially influenced, and may also have altered the seasonality of fires as some fires occur outside recommended periods (Archibald *et al*. 2010).

1.4.3.4.Fire frequency

The fire frequency or fire return interval (FRI) refers to the interval between fires on the same area (Kraaij and Van Wilgen 2014). Interval dependent effects and event-dependent effects influence the fire frequency. As interval-dependent effects are quite predictable and is influenced by the growing conditions which occur during the interval between fires (Bond and Van Wilgen 1996). Event-dependent effects are less predictable and is influenced by weather conditions and fire behaviours and it regulates plant numbers (Bond and Van Wilgen 1996; Trollope 2007). The change in FRI is critical to ecosystem resilience and species can differ in their tolerance to high and low FRIs. The difficulty in using the term fire frequency is based on the fact that various ecosystems have a mixed fire regime which either compromised of surface and/ or crown fires as well as unburned patched (Keeley *et al*. 2012). When burning to remove moribund material the frequency of burning will depend on the accumulation of excess grass litter (Trollope 2004). Furthermore, renosterveld burns more frequently (FRI= 3-5 years / 6-7 years) than fynbos (FRI= 10-25 years) because fynbos species requires some years to mature and flower and are sensitive to short recurrent intervals (Forsyth *et al*. 2010; Kraaij and Van Wilgen 2014). The frequency of the fires may also change over time due to climatic changes and increased anthropogenic ignition (Keeley *et al*. 2012).

1.4.3.5.Size of fire

This refers to the area burnt by a given fire or in defined fire size classes (Forsyth and Van Wilgen 2008). Small fires tend to be numerous, with a few big burns. The bigger fires are more important as they can burn up to 90 % of the total areas of all fires in a given period (Forsyth *et al*. 2010). In fynbos vegetation large fires can burn a range of vegetation ages (Southey 2009). Fire size distribution for the Table Mountain National Park was stable as the largest fires on record were 3 363 ha and 3 204 ha, with numerous smaller burns. In the Cederberg, Swartberg and Hottentots-Holland nature reserves large fires burnt between 15 000 ha to 20 000 ha (Southey 2009). The size of fires is also influenced by wind, slope and aspect (Sharples *et al*. 2012). The wind and slope can tilt the flame towards the unburnt fuel (Fig. 1.3). In this way the flame height is reduced and the flame length remains unaffected.

Figure 1.2: An active human-induced burn site near the Leliefontein village in 2015.

Figure 1.3: Illustration of the difference between flame length and flame height (adapted from Goldammer and de Ronde 2004).

1.4.3.6.Fire intensity

The fire intensity indicates how fiercely the area burns (Bond and Van Wilgen 1996) and can be used to describe the general behaviour of fires and their effects on the vegetation (Trollope 2007). The fire intensity and flame height has a significant effect on the vegetation. The fire intensity is often defined as the rate at which energy is released from organic matter. (Forsyth *et al*. 2010). This is defined as the release of heat energy per unit time per unit length of fire front (kJ s^{-2} m^{-1}) (Byram 1959; Trollope 2004). Byram's (1959) defines fireline intensity as the rate of heat transfer per unit length of fireline:

$$I = HWR$$

Where H = heat of combustion (kJ^{-1} of fuel), W = mass of fuel consumed (kg m^{-2}) and R = rate of fire spread (m s^{-1}), this gives a fireline intensity (I) in kWm^{-1}. Byram's fireline is the radiant energy released in the flaming front and is a good measure of fire propagation, which is critical for fire suppression activities and has been incorporated into the fire danger calculations. It can be quite difficult to measure fireline intensity, particularly for unplanned wildfires therefore fire severity or burn severity has been used as a post-fire indicator of fire intensity. Studies which have used fire severity have had a common basis which centres on the loss or destruction of aboveground or belowground organic matter (Keeley 2009; Keeley *et al*. 2012). Satellite-derived information can be another measure to determine variables of the fire regime. Radiometer sensors which provide satellite data are more sensitive and resolved and can be used to provide an indication of the rate of energy released of fire, this is known as Fire Radiative Power (Giglio *et al*. 2009; Wooster *et al*. 2003). Fire Radiative Power gives an indication of both the biomass consumption rate and the fireline intensity (Archibald *et al*. 2010).

If one of elements of the fire regime is altered then an ecosystem may be stressed by significantly changing it structure, functioning and composition (Forsyth *et al*. 2010). Analysing the fire regime is of importance especially interpreting biotic, physical and soil responses to fire and to which extent management (e.g. grazing), anthropogenic changes and climate has influenced the fire regime (Fig. 1.4) (Van Wilgen *et al*. 2010).

1.4.4. The Fynbos biome as part of the Greater Cape Floristic Region

Mediterranean-type ecosystems (MTE) have a restricted distribution and occur in five parts of the world (Keeley *et al*. 2012). These areas include the Mediterranean Basin, west coast of California, central Chili, the Cape Region of South Africa, and western and south Australia (Dell *et al*. 1986; Rundel 1998). The annual rainfall may range from 250 to 1 200 mm per year, of which most fall during the winter months (Dell *et al*. 1986). The climate of these ecosystems are predicted to increase in temperature therefore becoming effectively drier, which would rapidly change the fuel load and, consequently, changes in the fire regimes (Pausas *et al*. 1999). In addition to climate, these regions are characterised by their relatively infertile soils, high frequencies of fire, species rich flora and vegetation generally dominated by evergreen, sclerophyllous shrubland or heathland vegetation (Datson *et al*. 2008; Underwood *et al*. 2009; Keeley *et al*. 2012).

Mediterranean-type ecosystems are one of the world's major fire-prone biomes (Bond *et al*. 2005) and are often characterised by the occurrences of wildfires and summer droughts (Pausas 2004). In these ecosystems fire is a dominant ecological factor controlling vegetation dynamics, structure, and patters and processes since the early Holocene at least (Pausas 2004; Keeley *et al*. 2012).

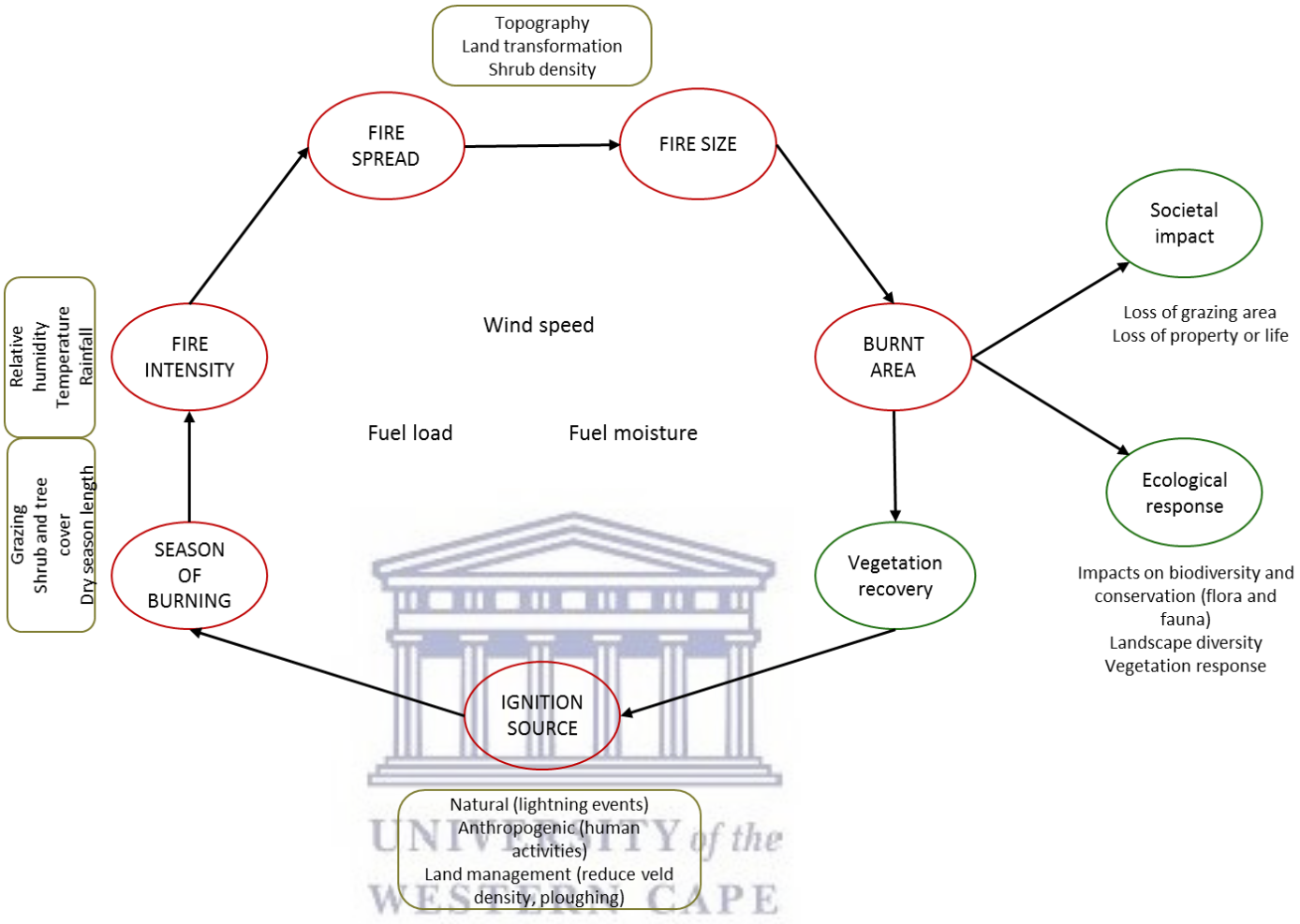


Figure 1.4: Conceptual model of factors contributing and affecting the fire regime. Items in the uppercase are components of the fire regimes, the items in the middle of the circle are direct drivers, while items on the outer side of the circle are all indirect drivers. The items circled in green is the way in which the vegetation responds to fire followed by a resting period. In combination, fires are controlled by all of these factors. Adapted from Archibald *et al*. (2009) and Keeley *et al*. (2012).

The occurrence of fires depends on the dry conditions which drastically increase flammability during summer (Mouillot *et al.* 2002). Any changes in the fire regime can have important consequences for the stability of these landscapes (Pausas 2004). Understanding changes in the fire regime and their relationship to climate is a key factor for predicting future changes in vegetation of MTE (Pausas 2004).

The Fynbos Biome is confined to the Cape Floral Kingdom, which is one of the world's six floristic kingdoms (Good 1947). The Fynbos Biome also constitutes the largest portion of the Cape Floristic Region (CFR), a region which is internationally renowned for its extraordinary species diversity and which is recognised as one of 34 global biodiversity hotspots due to the high concentration of endemic taxa (Myers *et al*. 2000; Reyers *et al*. 2001; Van der Merwe and Van Rooyen 2011). The CFR comprises an area of 90 000 km^2 and is estimated to have 9 030 vascular plant species, with 68 % endemism (Goldblatt and Manning 2002). This is an endemic biome to South Africa and its name is derived from the fynbos-the dominant vegetation of this region (Mucina and Rutherford 2006).

The Fynbos biome is well defined geographically and comprises three distinctive, naturally fragmented vegetation types (i.e. fynbos, renosterveld and strandveld) which occur in the winter-and summer-rainfall areas and are dominated by small-leaved, evergreen shrubs (Rebelo *et al*. 2006; Mucina and Rutherford 2006; Van der Merwe and Van Rooyen 2011). In contrast in the arid and semi-arid regions known as the Greater Cape Floristic Region (Greater CFR) less literature is available (Snijman 2013). Available literature has been centred mostly on plant and vegetation ecology in these arid regions (Jürgens 1991, 2004; Hilton-Taylor 1996; Cowling and Hilton-Taylor 1999; Cowling *et al*. 1999; Esler and Rundel 1999; Schmiedel 2004; Mucina *et al.* 2006; Van der Merwe *et al*. 2008a, 2008b; Helme 2009).

The Greater CFR mainly covers the winter rainfall areas in the west of southern Arica and also includes the non-seasonal rainfall areas in the extreme south and southeast (Snijman 2013). Bayer (1984) first included the 'winter rainfall region' flora, but Jürgens (1991, 1997) fully developed this concept by rearranging the Succulent Karoo biome with the CFR. This led to the recognition of the temperate Greater CFR (Snijman 2013). The winter rainfall areas are dominated by two biomes namely: Fynbos in the mountains on soils derived from sandstones and granites, and Succulent Karoo on the plains on soils derived from shale or rarely granites. The Greater CFR includes the Cape Floristic, Hatam-Tanqua-Roggeveld and Namaqualand Regions (Born *et al*. 2007). The Succulent Karoo and Fynbos biomes comprises 90 850 km^2 and 4 443 km^2 of the Extra Cape Subregion (ECR respectively (Snijman 2013). The area inhabits about 98 869 km^2 and has a flora of 3 715 vascular plant species, with 40.4% species endemism (Snijman 2013).

1.4.5. Renosterveld: The term Renosterveld

The Dutch Governor, Simon van der Stel was the first person to make reference to *Elytropappus rhinocerotis* in his journal (Newton and Knight 2004). In 1685 during his expedition, to what is now Springbok, he described the Olifants valley overgrown with rhinoceros wood (*Rhenosters bosch*) and made reference to rhinoceros (Parker and Lomba 2009). It was unclear whether the Dutch Governor was writing about the specific shrub or the vegetation in general because the term renosterbos has been used to refer to both (Stander 2016). Various names have been given to *E. rhinocerotis* by colonists particularly in their journals and these include: *rhinoceros bosjes* (Schrijver 1689); *rhinocer-bosch* (Valentyn 1726); rhinosceros-bush (Brink 1761-1762); rhinosterbosjes (Lichtenstein 1812); *rhinoster bosch* (Burchell 1822); *rhenosters bosch* (Waterhouse 1932); and *rhenosterbush* (Adamson

1938) (from Parker and Lomba 2009). *Elytropappus rhinocerotis*[2] is endemic to South Africa (Levyns 1926; Vermeulen 2010) and is the dominant shrub in renosterveld vegetation (Cowling *et al*. 1986; Walton 2006; Mucina and Rutherford 2006; Curtis 2013; Kraaij and Van Wilgen 2014). The distribution of *E. rhinocerotis* (Fig. 1.5) suggests that it requires more moisture than Succulent Karoo plant species but less than most fynbos taxa (Levyns 1935; Cowling and Holmes 1992).

The term 'Renosterveld' implies an association with *E. rhinocerotis*, even though *E. rhinocerotis* can be absent or scarce in different types of renosterveld. It is not restricted to renosterveld, as a result it can sometimes occur on ecotones between Fynbos, Succulent Karoo and Grassland biomes (Bergh *et al*. 2007; Allsopp *et al*. 2014).

Due to overgrazing by livestock in the late 19th and early 20th century, *E. rhinocerotis* became dominant, and started replacing more palatable grass and shrub species (Midoko-Iponga 2004). Farmers not only viewed *E. rhinocerotis* as an invader plant species but as an aggressive coloniser and a punishment for their sins during the 20th century (Beinart 2003). Furthermore, farmers used *E. rhinocerotis* as an indicator for soil suitable for wheat planting (Midoko-Iponga 2004).

Until today the origin of the name renosterveld has been unclear. Levyns (1972) suggested the name derived from the shrub, renosterbos. Due to the fact that from a distance the structure and the grey green appearance of renosterbos resembled that of a rhino hide. Another explanation was that rhinoceros occurred in the vegetation, hence the name 'Renosterveld' (literally meaning rhinoceros field) is derived from Afrikaans words 'renoster' (meaning rhino) and

[2] There are about eight species of *Elytropappus* in southern Africa of which one occurs in Namaqualand (Proksch *et al*. 1982).

'veld' (meaning vegetation). Boucher (1980) suggested that the term renosterveld should be given preference over *rhenosterveld*, *renosterbosveld* and *rhenosterbosveld*.

1.4.5.1. The components of Renosterveld

Renosterveld is found in the Western and Southern Cape (Parker and Lomba 2009) and is dominated by Asteraceae shrubs (especially *E. rhinocerotis*), with grass components and high diversity of geophytes (especially in the Iridaceae, Orchidaceae and Liliaceae families) (Milton 2007; Curtis 2013). Even though *E. rhinocerotis* does not appear to be very attractive it is an important component of renosterveld in terms of biodiversity and ecology of the vegetation (Vermeulen 2010).

Renosterveld is a severely fragmented fire-prone shrubland (Reyers *et al*. 2001) which occurs in diverse landscapes (Jacobs and Jangle 2008). Renosterveld is found on clay-rich, fertile shale-derived soils in low-lying areas (50-500 m above sea level), but it can also occur on granite, dolerite and silcrete soils (Cowling *et al*. 1986; Midoko-Iponga 2004). Renosterveld shrublands occurs in winter rainfall areas where the rainfall ranges between 250 mm – 700 mm per year (Krug 2004; Midoko-Iponga 2004; Vermeulen 2010; Stander 2016).

Renosterveld is one of the least-clearly defined vegetation types in South Africa, thus defining this vegetation can become quite complex due to the different types of renosterveld which occur on diverse landscapes (Vermeulen 2010). Furthermore, different species inhabit renosterveld vegetation and about one-third (1 000 species) of the endemic flora in the CFR has been identified in Renosterveld. (Vermeulen 2010; Solomon 2015).

Renosterveld is a complex vegetation in terms of its classification history. Various authors have grouped it with different vegetation for example in Fynbos as 'evergreen sclerophyllous bush'

(Pole-Evans 1936); in Strandveld as ‘temperate and transitional forest and scrub types’ (Acocks 1953); as ‘Mediterranean shrubland’ (Boucher and Moll 1981) or karoo groupings as ‘karroid and renoster shrubland’ (Campbell 1985; Cowling and Holmes 1992). However, renosterveld has been treated as a distinct vegetation as ‘Mediterranean shrubland’ in Specht and Moll’s (1983a) category, or as a ‘transitional small-leaved shrubland’ category as described by Cowling (1983a), Moll *et al* (1984), Vlok and Schutte-Vlok (2010) (Bergh *et al*. 2014).

1.4.5.2. The four distinct Renosterveld groups

Several types of renosterveld are recognised in terms of lithology (Fig. 1.6) of which Shale Renosterveld is the widest spread, comprising of 19% of renosterveld. This is followed by Granite and Dolerite Renosterveld 5%, Alluvium Renosterveld 2%, and Silcrete and Limestone Renosterveld 3% (Mucina and Rutherford 2006; Bergh *et al*. 2014). Granite Renosterveld units include the Kamiesberg in Namaqualand, Swartland and Robertson in the Western Cape. These three units are very distinct and unrelated from each other (Mucina and Rutherford 2006).

Moll *et al*. (1984) distinguished four groups of Renosterveld in South Africa. The four groups are: West Coast Renosterveld, South West Coast Renosterveld, South Coast Renosterveld and Mountain Renosterveld. These groups are distinctive in their biogeography, floral composition and ecological characteristics (Cowling *et al*. 1986; Moll *et al*. 1984; Milton 2007; Vermeulen 2010; Stander 2016).

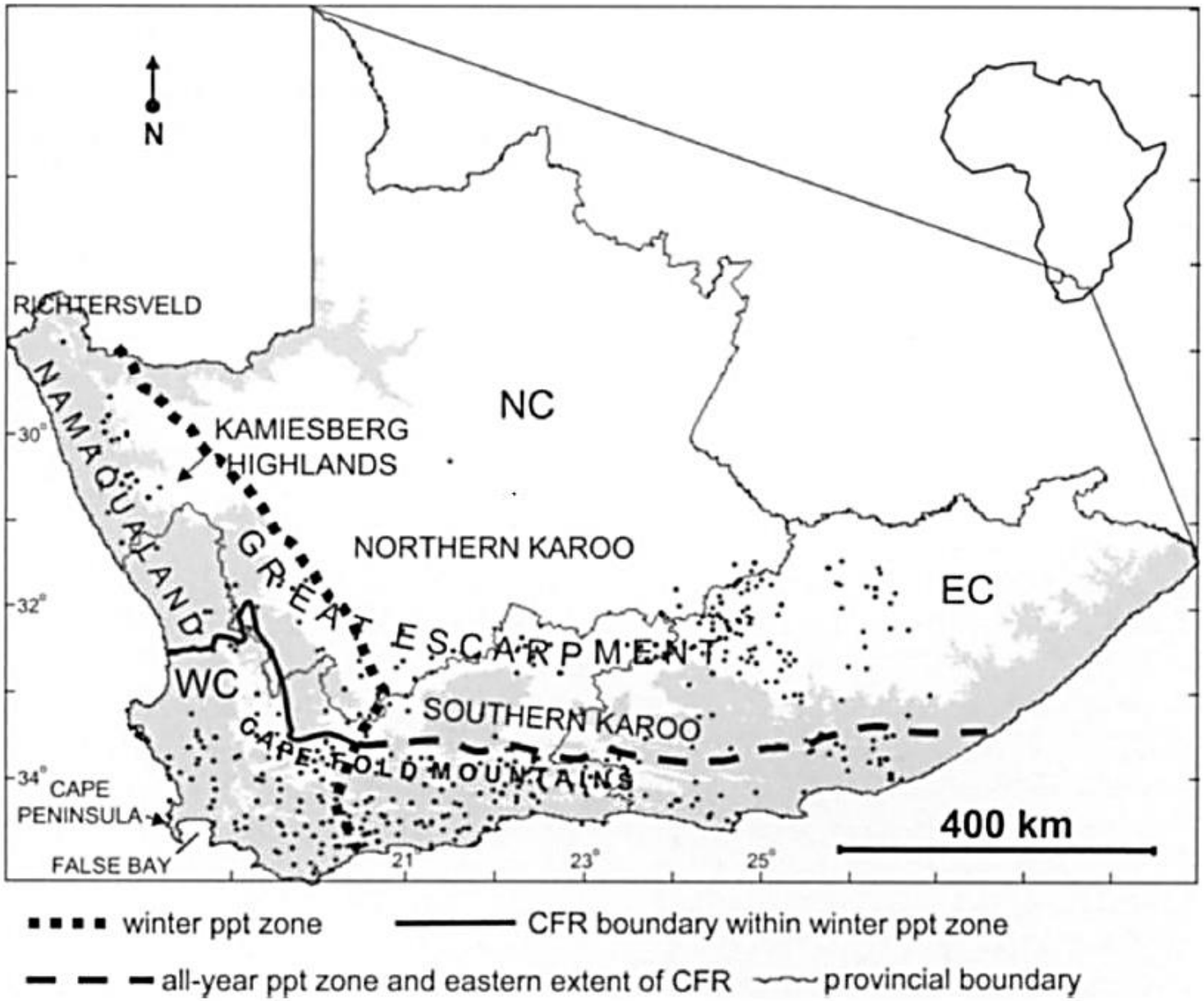


Figure 1.5: Distribution of *Elytropappus rhinocerotis* in the Cape Provinces of South Africa (Bergh *et al*. 2007).

1.4.5.2.1. West Coast Renosterveld

This group is found mostly on Malmesbury shale and Cape granites and comprises of mid-dense to closed (50 – 90 % canopy cover) cupressoid and small-leafed and evergreen shrubs. Annual and geophyte diversity is high while the perennial graminoid cover is spare (Moll *et al*. 1984). Dominant species include *Elytropappus rhinocerotis* (renosterbos), *Eriocephalus africanus* (kapokbos) and *Leysera gnaphalodes* (geelblommetjiestee). Only 3-4 % of West Coast Renosterveld remains and the remnants are scattered between agricultural fields (Vermeulen 2010; Stander 2016).

1.4.5.2.2. South West Coast Renosterveld

Found mostly on Bokkeveld shale this group's plant community essentially includes mid-dense (50 – 75 % canopy cover) cupressoid and small-leafed, mid-high evergreen shrubs and rare broad-leafed shrubs. The underlying layer of vegetation is mostly herbaceous with occasional perennial graminoids (Moll *et al*. 1984). Distinctive species such as *Helichrysum* spp. (sewejaartjie), *Relhania* spp. (gombos) and *Hermannia* spp. (poprosie) occur in this group of Renosterveld (Moll *et al*. 1984; Walton 2006). Approximately 5 -8 % of South West Coast Renosterveld remains (Stander 2016).

1.4.5.2.3. South Coast Renosterveld

Mainly found on Bokkeveld and Cango shales the vegetation is open to mid-dense (25 – 60 % canopy cover) cupressoid and small – leafed, low to mid-high shrubs with no emergent (Moll *et al*. 1984). The understory has scattered herbaceous shrubs. This group has a high proportion or perennial grasses (Midoko-Iponga 2004). South Coast Renosterveld has about 4-6 % remaining (Stander 2016).

1.4.5.2.4. Mountain Renosterveld

This group is found on Bokkeveld and Witteberg shale and occurs on less fertile soils (Moll *et al*. 1984; Mucina and Rutherford 2006; Parker and Lomba 2009; Curtis 2013). The plant community is more open to mid-dense (25 – 60 % canopy cover) cupressoid and small leafed with low to high shrubs (Moll *et al*. 1984). Succulents are scattered across this Renosterveld type and is dominated by *Relhania genisfolia* (Boucher 1995), *Pteronia incana* and *E. rhinocerotis* (Low and Rebelo 1996; Midoko-Iponga 2004; Walton 2006). Perennial grasses are often absent due to veld deterioration. Trees and large shrubs such as *Acacia karroo*, *Aloe ferox*, *Euclea undulata* and *Rhus* spp. are scattered (Walton 2006).

Mountain Renosterveld types include: North-Western Mountain, Escarpment Mountain and Central Mountain and are based on locality (Rebelo 1995; Walton 2006). North-Western Mountain Renosterveld is confined to the Kamiesberg Uplands nearby Leliefontein and on an ecotone between Fynbos and Succulent Karoo biomes (Midoko-Iponga 2004; Walton 2006). The Escarpment Mountain Renosterveld is found on the escarpment slopes and foothills from Calvina to Sutherland and towards Beaufort West, on the Roggeveld, Koms and Nuwerus Mountains (Midoko-Iponga 2004; Walton 2006). The Central Mountain Renosterveld is found on the fringes of the Great and Little Karoo basin and westward toward Worcester (Rebelo 1995; Midoko-Iponga 2004; Walton 2006). These renosterveld types are not well studied and are mostly restricted to the Kamiesberg Uplands (Rebelo 1995; Midoko-Iponga 2004; Vermeulen 2010). This is the most intact of the four Renosterveld groups (Kemper *et al*. 1999; Reyers *et al*. 2001) as only 20 % in the Kamiesberg has been transformed (Helme and Desmet 2006). Roughly 73 % remains (Stander 2016) and the remnant Mountain Renosterveld patches has irreplaceable conservation value (Ferrier *et al*. 2000; Vermeulen 2010).

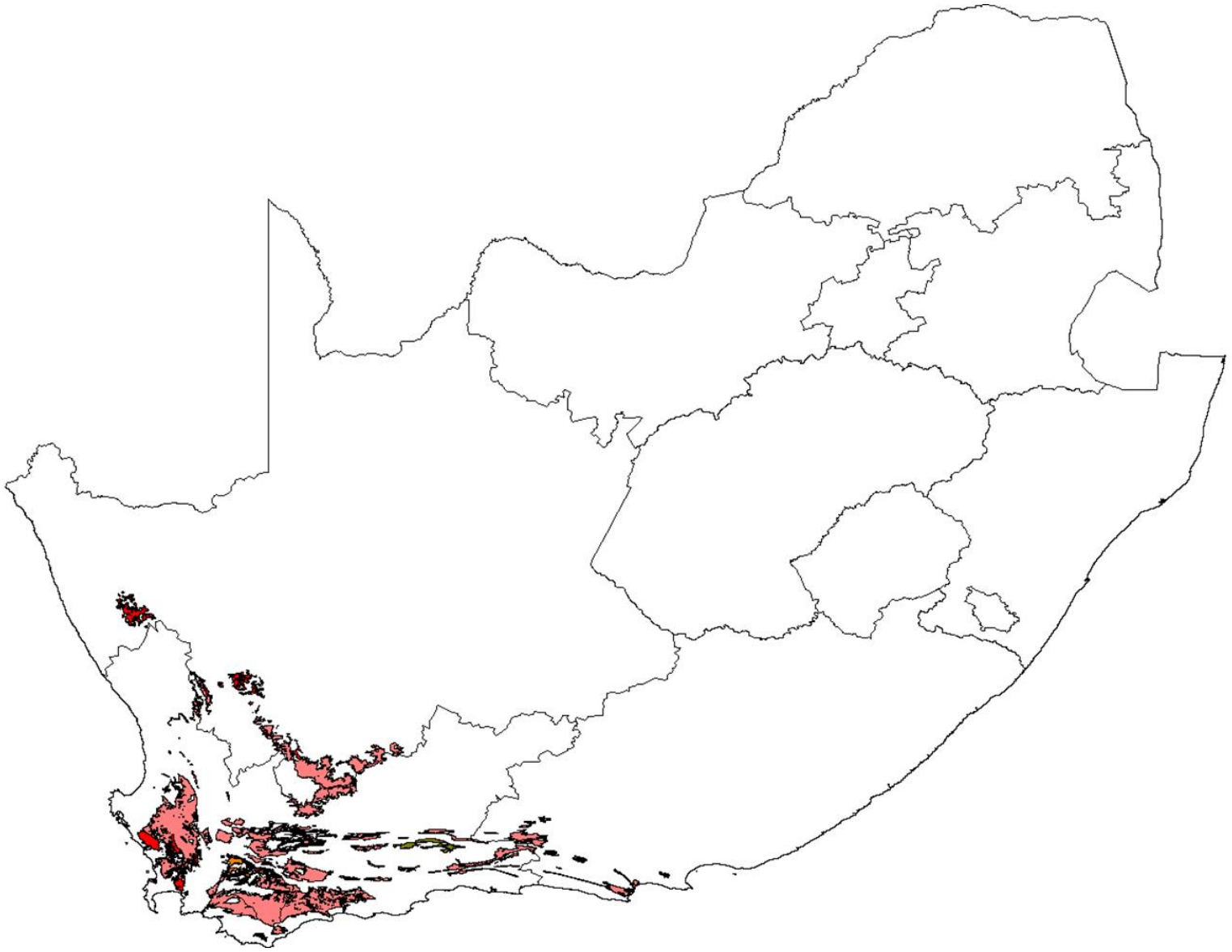


Figure 1.6: Distribution of the different types of Renosterveld along the Cape Floristic Region and Greater Cape Floristic Region.

1.4.6. The transformation and conservation of Renosterveld

Renosterveld has been heavily transformed for the development of field crops, vineyards and orchards (Kemper *et al*. 1999; Von Hase *et al*. 2003). Over 90 % of Renosterveld has been ploughed for agriculture (primarily for grains and artificial pastures) (Newton and Knight 2004). The remaining fragmented patches of Renosterveld occurs on agricultural landscapes (Newton 2008). In addition to its accessibility and arable quality of Renosterveld it was one of the first vegetation types to be changed by grazing and transformed for crop production in the 17th and 18th centuries (Milton 2007).

Large herds of game such as roan antelope, blue antelope, bontebok, red hartebeest and quagga use to roam in renosterveld in the past (Krug 2004; Parker and Lomba 2009). The vegetation was likely a mixture of grasses and shrubs with a substantial geophyte component. The grass component was probably much high in the past (Vermeulen 2010). About 2 000 years ago a group of indigenous herders called the Khoekhoen adopted a herding lifestyle. Their herds consisted of 10 000 – 20 000 cattle and sheep. They used Renosterveld for centuries as natural grazing for their livestock would often burnt patches of renosterveld to create more palatable grasses (Midoko-Iponga 2004) and to stimulate the growth of geophytes such as Watsonias, which is believed to have been part of their diets (Parker and Lomba 2009). The herders migrated seasonally over the region and returned one to four years later to the burnt patches. When the European settlers arrived in 1652, herds of game were hunted to extinction (Raitt 2005) and they transformed the vegetation for crop farming (Krug 2004). Overstocking, overgrazing and too frequent fires following European settlement has changed Renosterveld completely (Raitt 2005) from a ‘shrubby grassland’ to a ‘grassy shrubland’ (Newton and Knight 2004). In addition, the expansion of urban settlements further encroached into renosterveld (Krug 2004).

An estimated 160 000 ha of natural Renosterveld has been transformed (Fig. 1.7) by agriculture for wheat production and artificial pastures, since the 1920s (Cowling *et al*. 1986). The vegetation has less than 9% of the original extent remaining in the lowlands (Reyers *et al*. 2001). Less than 1% is formally conserved (Von Hase *et al*. 2003; Midoko-Iponga 2004) and remnant patched occur mostly on privately owned land (Krug 2004; Curtis 2013). Renosterveld which has no formal conservation is mostly subjected to grazing, trampling, crop spraying and burning (Kemper *et al.* 1999). Land use impacts on Renosterveld have resulted in the area becoming a conservation concern (Helme and Desmet 2006; Parker and Lomba 2009). At drier extremes (usually below 250-300 mm) it is replaced succulent karoo shrublands, and in wetter areas (usually over 500-800 mm) by fynbos (Cowling and Holmes 1992).

Little is known about the fuel dynamics and fire ecology of renosterveld (Van Wilgen 1987; Kraaij and Van Wilgen 2014). Consequently, evidence as to what constitutes the ideal fire frequency is lacking (Mucina and Rutherford 2006; Jacobs and Jangle 2008). Various authors have estimated the fire frequency of renosterveld. It is assumed that the natural fire frequency is shorter than that of Fynbos (Walton 2006), due to its inherit resilience and dominant species with short maturation times (Boucher 1983) and because of faster growth rates and dominance by finer grasses (Mucina and Rutherford 2006). Presumably heavy grazing areas seldom burn, due to the lack of burning material but by contrast un-grazed areas can accumulate sufficient fuel to maintain fire.

Fire is dominant during hot dry summers and is facilitated by secondary compounds in sclerophyllous leaves (e.g. volatile oils) (Walton 2006). Fire affect sclerophyllous vegetation by influencing the landscape patterns and monitoring community composition (Walton 2006). Small-and landscape fires contribute to the overall heterogeneity of renosterveld. Renosterveld diversity patterns seem to be maintained by fire which create gaps between plants (Walton

2006). Fire suppression or fires during spring and summer are rare features in renosterveld but they often favour the establishment of unpalatable shrubs, such as *E. rhinocerotis* (Krug 2004).

In addition to fire, herbivores also influence the structure and composition of renosterveld as fire attracts grazers to new plant growths (Bond and Van Wilgen 1996; Jacobs and Jangle 2008). Frequent burning of renosterveld, combined with overgrazing enables alien grass species to invade the vegetation, whereas infrequent burns with long fire intervals facilitate the invasion of alien trees and shrubs. Geophytes and grass species requires space between shrubs to establish, thus small scale fires are necessary to create these open patches within the vegetation (Krug 2004). If renosterveld is burnt too frequently it will be transformed into grassland and if fire is suppressed for too long it will transform into a thicket (Krug 2004). Consequently, fires should not be excluded from renosterveld environments in order to maintain this balance between grass and shrub species.

1.4.7. The impacts of fires on Renosterveld diversity

Diversity patterns in Fynbos have been studied extensively (Bond 1983; Cowling 1983; Cowling 1990; Goldblatt and Manning 2002; Kruger and Taylor 1980). The high species richness has been attributed due to high beta and gamma diversity with moderate alpha diversity (Curtis 2013). Fire is not the only factor maintaining the diversity of renosterveld, but grazing[3] is also important disturbances which maintain the floristic diversity (Walton 2006). Grazing can have both positive and negative impacts on the vegetation (Hoffman and Rohde 2007). As plant species richness can increase under light grazing but vegetation cover reduces with heavy grazing (Anderson and Hoffman 2007). Renosterveld is a very dynamic vegetation

[3] Grazing in this thesis refers both to grazing and browsing by herbivores.

type, as it can fluctuate between a grass-and shrub dominated state depending on the use of fire and other disturbances (such as grazing and bush cutting) (Walton 2006).

Renosterveld recovery initially has low floristic diversity but annual and pioneer species are dominant. Renosterveld is the richest plant community per unit land area with high conservation value (Cowling 1990; Kemper *et al*. 1999; Newton and Knight 2010). Renosterveld is well-known for its exceptionally high richness in geophytes (Cowling 1990) which are influenced by soil fertility, aridity and fire frequency (Kruger 1979). Fynbos and renosterveld are extremely diverse, in terms of rare and endemic species. These species are often associated with high extinction risk as many of them occur in fragmented patches (Helme and Desmet 2006, Curtis 2013).

Various studies (Shiponeni 2003; Krug 2004; Mdiko-Iponga 2004; Walton 2006; Newton and Knight 2010; Curtis 2013) have focussed on Lowland Renosterveld. Thus the ecological and conservation aspects of this critically endangered vegetation is well understood but there is a gap in knowledge regarding Mountain Renosterveld. As Mountain Renosterveld is mostly pristine and occurs on different landscapes across the CFR and Greater CFR. Heavy grazing and inappropriate fire management are seen as threats facing Mountain Renosterveld today and it is important to determine and understand which the main drivers of vegetation change are (Helme and Desmet 2006; Parker and Lomba 2009). The functioning and ecological requirements of Namaqualand Granite Renosterveld is poorly understood and needs to be determined.

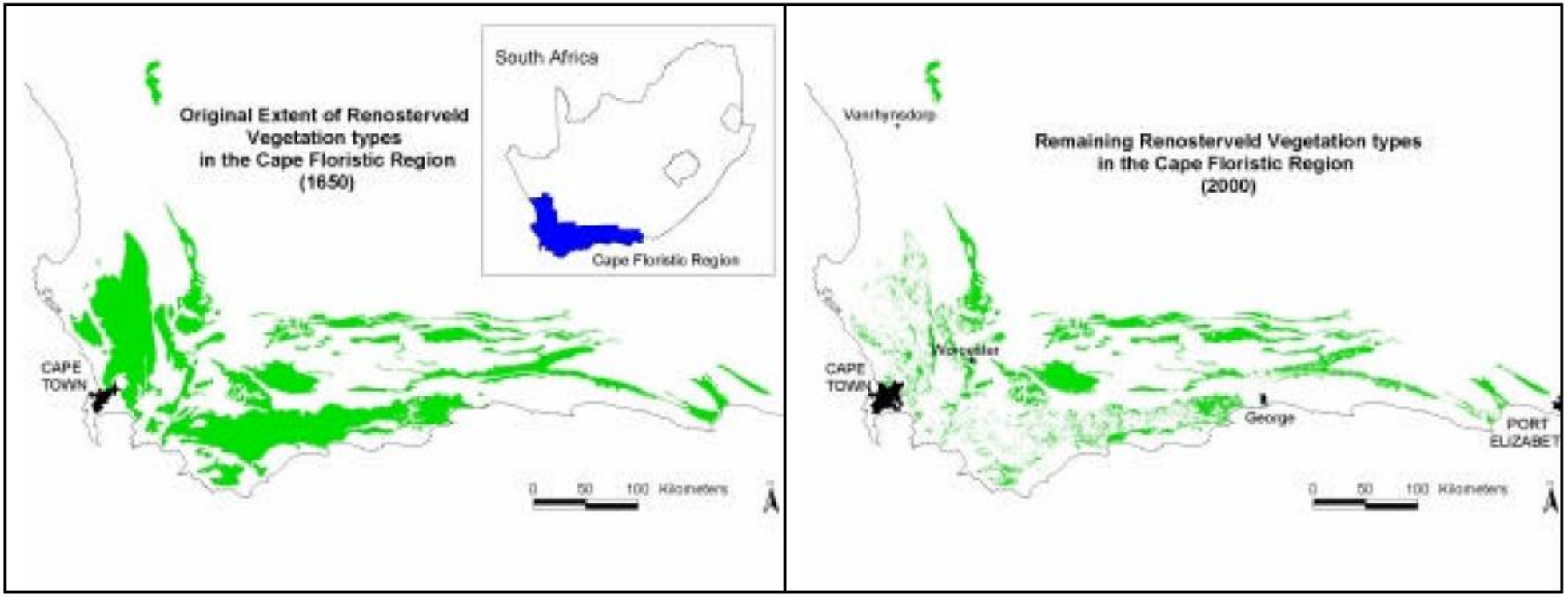


Figure 1.7: Illustration of a) Pre-colonial (1850) and b) remaining extent (2000) of Renosterveld in the Cape Floristic Region (Von Hase *et al*. 2003).

1.4.8. Effects of grazing on Renosterveld

In Renosterveld, livestock grazing from Khoi-Khoi pastoralists have been proceeding for centuries but the effects of their livestock on plant communities is debatable (Allsopp 1999; Kemper *et al*. 1999; Anderson and Hoffman 2007). The impact of grazing can vary across the landscape. Light grazing may have a positive impact as it increases plant diversity or the negative impacts in which heavy grazing can result in loss of palatable species and soil erosion (Hoffman and Rohde 2007). The nature of the impact varies, in accordance from the loss in cover, variation in plant diversity and even compositional shifts (Anderson *et al*. 2010). Reported compositional shifts are mainly from systems in which perennial species dominate and to systems dominated by less palatable species in response to the selective pressure of grazing (Hoffman and Rohde 2007). These types of selective pressures can result in a loss of heterogeneity, because different vegetation types may become increasingly homogenous in composition and cover because of dominance by a few unpalatable species (Lombard *et al*. 1999).

The role of variable habitats which either limits or influences grazing impacts. (Anderson and Hoffman 2007). Due to the inaccessible nature of rocky upland habitats they are targeted less by grazers. Therefore, plant communities in these areas are therefore less likely to show grazing responses (Lombard *et al*. 1999; Rohde *et al*. 2003; Anderson and Hoffman 2007; Samuels 2013). Grazing is an important factor in determining vegetation patterns as it inhibits vegetation succession, thus preventing the vegetation to move between states (Carmel and Kadmon 1999). Vegetation composition can therefore be altered by misguided grazing management (Tietjen and Jeltsch 2007; Du Toit *et al*. 2011).

1.4.9. Post-fire vegetation changes in Renosterveld

Mediterranean ecosystems are one of the world's most fire prone biomes (Capitanio and Carcaillet 2008). The post-fire regeneration of Mediterranean-type vegetation is highly dependent on the pre-fire vegetation (Capitanio and Carcaillet 2008). Studies have shown (Hanes 1971; Lloret and Vilá 2003) that vegetation composition and structure in MTE recover rapidly after a fire within the first eight months. As a result, many species have evolved strategies which enable them to survive and recover after a fire event (Pausas *et al*. 1999). The process of fire activates the germination of dormant seeds in the soil, breaking seed coats and permitting the entrance of water and subsequent imbibition recovery (Buhk *et al*. 2007). Plant regeneration mechanisms protect plant tissues from fire damage and improve survival, enhancing the fitness after the fire event.

The post-fire recovery determines whether species has an obligate or facultative requirement and this is coupled to the habitat in which they occur and the abiotic factors (Curtis 2013). Resprouters usually have buds in underground organs or in stems and this allow them to resprout after fire. Seeders are initially killed off by fire but the released seeds are able to germinate. The recovery rate of species varies due to different climate conditions and grazing pressures. These ecological conditions are specific to each site (Pausas *et al*. 1999; Buhk *et al*. 2007). Therefore, available resources are essential for the abundance of post fire seeders and resprouters. In certain Mediterranean ecosystems (such as Greece, Buhk *et al*. 2007) it has been hypothesized that the post-fire regeneration may be rich in nitrogen fixing species, such as legumes due to their high competitive ability in environments where parts of the nutrients has been lost through fires (Pausas *et al*. 1999). Knowledge relating to the success of regeneration mechanisms and their interrelations with ecological factors are essential (Buhk *et al*. 2007). It is important to assess whether theses regeneration mechanisms can survive with on-going land uses and climatic changes.

In order for plant communities to recover to their natural state, in a disturbed environment, new plant species has to establish themselves first and survive (Krug 2004). In order for species to recovery successfully their environmental conditions, such as available water and sufficient nutrients, has to be favourable (Krug 2004). Lowland Renosterveld shows a significant increase in plant flowering and germination in post-fire vegetation, while older fields are less productive (Curtis 2013). Levyns (1929) conducted experiments on the germination of *E. rhinocerotis* and found no seedlings in unburned sites and very high numbers of seedlings in burnt sites. Thus suggesting that fire plays an important role in the recruitment of *E. rhinocerotis* (Levyns 1929).

Vegetation recovery in Mountain Renosterveld began within the first few months following a fire. Vegetation recovery remained higher in older sites compared to younger sites (one to two years post-fire). Recovery is influenced and to an extent depended on rainfall following fire (Van der Merwe and Van Rooyen 2011). It is important to establish to which extent Renosterveld species are dependent on fires, as this may be helpful in understanding the natural fire regime under which this system functions best.

1.5. Research aim and objectives

Analysing fires and characterising their effects on vegetation change overtime are important in understanding their role in ecosystem function and for management of renosterveld. The aims of this study were to determine the fire regime for Namaqualand Granite Renosterveld and assess its impacts in association with grazing on the conditions of renosterveld rangelands in the Kamiesberg Uplands. Furthermore, this study investigated the dynamics of post-fire plant recovery adopted by the plants in Namaqualand Granite Renosterveld.

To achieve the study aims, the objectives of the research are as follow:

- To provide an overview of fires in southern Africa and an understanding of the effects of fire in renosterveld vegetation
- To assess the fire regime for Namaqualand Granite Renosterveld defined by frequency (periodicity), seasonality and scale (spatial)
- To enhance the understanding of *Elytropappus rhinocerotis* (renosterbos) dynamics as a facilitator of vegetation change following a disturbance
- To understand to what extent fire affect and change plant diversity and structure of Namaqualand Granite Renosterveld over a temporal scale

1.6. Research questions

These research questions underpin the aims and objectives and form the basis for this study:

1. Analysing the fire regime of Namaqualand Granite Renosterveld (Objectives 1 and 2)
    i. What are the ignition sources of fire in Namaqualand Granite Renosterveld landscapes?
    ii. What is the Fire Danger Index for Namaqualand Granite Renosterveld landscapes?
    iii. Which factors affect the spatial extent of fires in Namaqualand Granite Renosterveld landscapes?

2. The post-fire effects on the dynamics of *Elytropappus rhinocerotis* in Namaqualand Granite Renosterveld (Objective 3)

i. How does the fire intensity (as Fire Radiative Power) affect the density and cover of *Elytropappus rhinocerotis* overtime?

ii. How does stocking density affect the density and cover of *Elytropappus rhinocerotis* in the burnt and unburnt vegetation?

iii. Does the density and cover of *Elytropappus rhinocerotis* change overtime? And how will it compare to the unburnt vegetation?

iv. Which species are associated with *Elytropappus rhinocerotis* and does it change the dynamics of Namaqualand Granite Renosterveld.

3. Post-fire plant regeneration and diversity of Namaqualand Granite Renosterveld (Objective 4)

i. How will plant species diversity, vegetation cover and structure change in burnt and unburnt areas over time?

ii. Is growth form diversity affected in Namaqualand Granite Renosterveld?

iii. What is the effect of stocking density on species richness over time?

iv. How does fire and grazing affect biodiversity and forage productivity over a temporal scale?

1.7. Thesis outline

In order to understand the effects of unplanned veld fires on the ecology and management of Namaqualand Granite Renosterveld; this thesis has been divided into six chapters.

Chapter 1: Introduction and Literature review

Chapter one contextualizes the study and includes a background and rationale of the study. The study objectives and questions are appended to this chapter. Furthermore, this chapter provided a review of the literature pertaining fire ecology in Southern Africa. Specific focus was placed on fire in the Fynbos biome. I discuss the distribution and ecology of *Elytropappus rhinocerotis*, characteristics of Renosterveld, and current distribution of this endangered vegetation type.

Chapter 2: Study area

Chapter two introduces the study area, Leliefontein, by describing its biophysical environment and the people of the area.

Chapter 3: Analysing the fire regime of Namaqualand Granite Renosterveld

Chapter three determines the fire regime of Namaqualand Granite Renosterveld in terms of frequency (return interval), season, and intensity and ignition sources. This chapter discusses the impacts of unplanned veld fires on the fire regime.

Chapter 4: The effects of fire on the dynamics of Elytropappus rhinocerotis

Chapter four investigates how *E. rhinocerotis* use fire to facilitate and promote its growth. Furthermore, it will investigate how *E. rhinocerotis* encroaches renosterveld and to which extent it possibly facilitates the role of a nurse-plant.

Chapter 5: Post-fire plant regeneration and diversity of Namaqualand Granite Renosterveld

Chapter five assesses the recovery of species in Namaqualand Granite Renosterveld and assess the post-fire effects on plant diversity, cover and structure. This chapter also assesses whether fire is beneficial for conservation or agriculture in the Kamiesberg.

Chapter 6: Conclusion

Chapter six provides the study synthesis by highlighting the key findings and gives recommendations for further future studies in Namaqualand Granite Renosterveld.

2. Chapter two: Biophysical description of the study area

2.1. Introduction

In this chapter the study area will be described in terms of vegetation, biophysical environment and the history of the people will be given. Specific focus was given to the Kamiesberg Mountain region of Namaqualand, but more specifically the village of Leliefontein.

2.2. Landscape of Namaqualand

2.2.1. Topography

Namaqualand is situated in the north-west of the South Africa. The area is approximately 55 000 km^2 in extent (MacKeller *et al*. 2007) and comprises almost half of the size of Namakwa District Municipality (Nortjie 2011; MacKeller *et al*. 2007). Namaqualand has six communal areas which were formally known as 'coloured reserves' until democracy in 1994 South Africa. The Leliefontein Communal Area, one of the former coloured reserves (Fig. 2.1), which is focus of this study comprises of 192 719 hectares of rangeland, lying in an east-west band of about 50 km straddling the Kamiesberg mountain range.

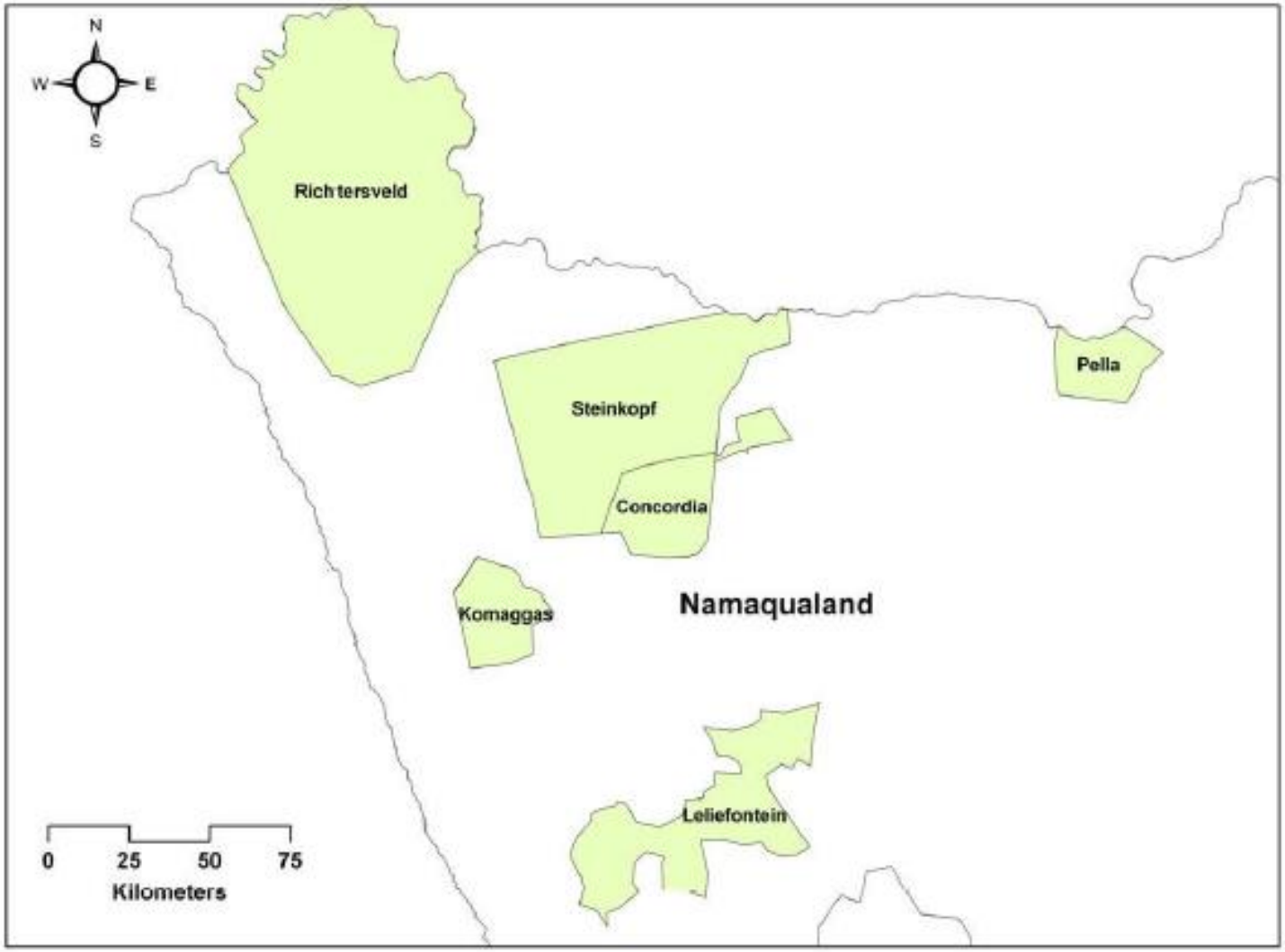


Figure 2.1: The six communal areas in Namaqualand including the location of Leliefontein Communal Area in the context of Namaqualand, Northern Cape Province of South Africa (Samuels 2013).

2.2.2. Geology

Namaqualand is characterized by a relatively wide (50-70 km), gently undulating coastal plain (Desmet 2007). The area is marked by distinctive round, rocky, granite hills separated by sandy plains (Todd and Hoffman 1999). Namaqualand comprises of circular intrusive igneous rocks with the centre comprising the high granite massifs of the Kamiesberg themselves (highest point is Rooiberg, 1700 m above sea level). Surrounding these massifs are variously metamorphosed granite rocks creating a landscape of rolling dome-shaped hills separated by sandy alluvial valleys (Desmet 2007). Valleys, particularly in the western side are dotted with "*heuweltjies*", which are large termite-derived mounds (Cowling *et al.* 1999; Desmet 2007). The Leliefontein village falls within the Namaqualand complex of the Great Escarpment, whose combined geomorphologic diversity and changes in the soils and climate, has profound effect on plant diversity (Simons and Allsopp 2007).

2.2.3. Soils

Namaqualand has a great diversity of soil types (Ellis and Lambrechts 1986; Watkeys 1998). The soils of Namaqualand are weakly structured grey, yellow or red medium grained sands of the sandveld and Bushmanland derived from aeolian reworking of marine or fluvial deposits (Watkey 1998). These soils have a relatively low clay content (< 3 %) and in most instances these sands are underlain by silica (dorbank) or calcium (calcrete) rich hardpans (Desmet 2007). The changes in soil properties brought about by the termite activity results in the areas being pockmarked by distinctive patterning regardless of termite activity on the heuweltjies (Desmet 2007). Thus, the soil properties (both physical and chemical) are extremely high and diverse and this could be fundamental in maintaining biodiversity and ecosystem functioning Namaqualand (Francis *et al*. 2007). The soils on the slope of the Kamiesberg is shallow,

homogenous and free-draining red and yellow. The soils are variably grained, sandy to loamy and are derived from in situ weathering of the underlying parent material (Desmet 2007).

2.3. Climate

Namaqualand is characterized as a semi-arid winter-rainfall region between the mesic Mediterranean climatic region south-western South Africa, with the summer-rainfall regions to the east and north (Desmet and Cowling 1999; MacKellar *et al*, 2007; Davis 2013). This area has a unique climate, characterised by dry and warm summers and wet and cold winter (Manning 2008). The climate is determined primarily by the southern subtropical high pressure system and the circumpolar westerly airstream (Tyson and Preston-Whyte 2000). Geographical features, such as the mountains of the escarpment and Benguela current, of Namaqualand drives landscape heterogeneity thus ensuring that the climatic conditions are variable (Mackellar *et al*. 2007; Davis 2013).

2.3.1. Temperature

Temperatures in Namaqualand are hot, during summer the mean maximum temperatures (Fig. 2.2) about 30 ^{o}C (Desmet 2007). The Namaqualand coast has one of the strongest wind systems in the world. Hot winds that originates from the high-altitude highlands occurs largely during winter and spring. The easterly winds known as 'berg winds' results from high temperatures (Desmet and Cowling 1999; Anderson 2008). In the Kamiesberg, the mean daily maximum and minimum temperatures for January are 25 ^{o}C and 10 ^{o}C (Snijman 2013). During the winter months (May to August) the temperatures in the uplands are often below freezing point (Hoffman *et al*. 2007). The mean temperatures range from 15 ^{o}C and 2 ^{o}C for July in the Kamiesberg (Snijman 2013). The high peaks, particularly Rooiberg (1 706 m), Eselkop (1 664

m), Sneeukop (1 588m), Kamiesberg (1 527 m), and Weeskind (1 440m) experience regular winter frosts and snowfalls are frequently found on these peaks (Snijman 2013; Samuels 2013).

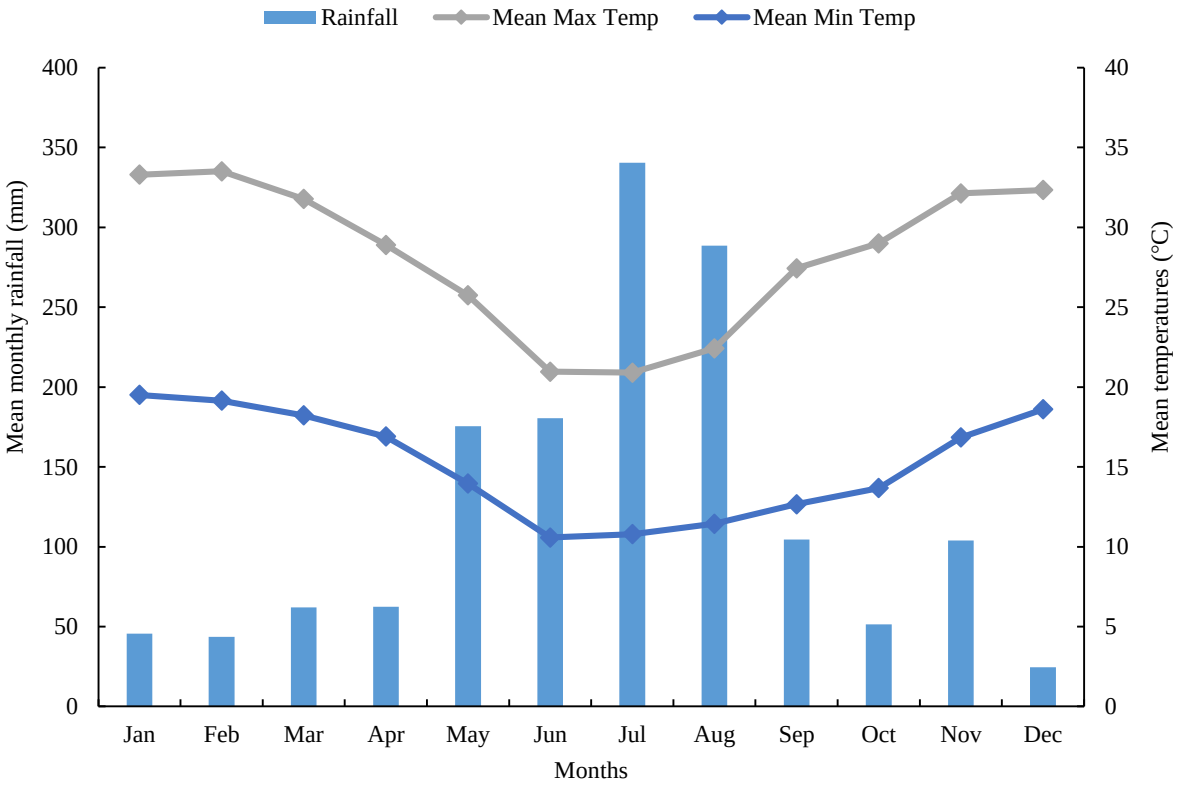


Figure 2.2: Mean monthly rainfall, maximum and minimum temperatures for the Leliefontein village from 2000 to 2015.

2.3.2. Rainfall

The climate of Namaqualand is characterized by two distinct geographic rainfall gradients (Desmet and Cowling 1998). The first is a gradual decrease in rainfall along an aridity gradient to the north into the Southern Namib Desert. The coastal region receives complete winter rainfall while the interior which extend about 300 km to the east receives entire summer rainfall (Desmet 2007).

The Kamiesberg Mountain Uplands receive nearly 400 mm rainfall per year (Snijman 2013) Peak rainfall is driven mostly by orographic effects (Kelso and Vogel 2007; David 2013). The

Leliefontein Communal Area experiences predictable rainfall with the western areas receiving mostly winter rainfall (May to August) and the eastern areas summer rainfall (Hoffman *et al*. 2007; Kotze *et al*. 2010; Samuels 2013). The rainfall received in the Leliefontein village is relatively higher than the surrounding villages due to its topographical location (Marinus 1998; Swarts 2014). The mean annual precipitation (map) for Leliefontein village (Fig. 2.2) in the uplands is 392 mm (CV = 0.44) and 145 mm (CV = 0.35) for Garies in the lowlands (Samuels 2013).

2.4.Biodiversity

2.4.1. Fauna

The Succulent Karoo has high levels of faunal endemism and is specifically dominated by diverse insects which are considered endemic to the Kamiesberg Uplands (Colville 2006). Moreover, Namaqualand Granite Renosterveld has over 46% endemic insects (Colville 2006). The study area currently has a wide variety of indigenous invertebrate species, reptile, bird and mammal species. The mammal species which occur in the study area include herbivores such as *Lepus capensis* (cape hare), *Procavia capensis* (cape hyrax), *Raphicerus campestris* (steenbok) and *Oreotragus oreotragus* (klipspringer) and carnivores such as *Canis mesomelas* (black-backed jackal), *Felis caracal* (rooikat) and *Panthera pardus pardus* (leopard). Both of these carnivores' prey on livestock and are thus viewed as pests to livestock keepers (Samuels 2013).

2.4.2. Flora

The flora of Namaqualand is extremely rich in diversity and is uniquely characterized among arid land floras by high numbers of species in the Mesembryanthemaceae (now forming part of Aizoaceae), Asteraceae and Iridaceae families (Cowling *et al.* 1999; Desmet 2007). Over

4750 plant species has been identified for the Succulent Karoo of which 75 % of these plant species occur in Namaqualand (Cowling and Hilton-Taylor 1994; Cowling and Pierce 1999). Despite the aridic conditions of Namaqualand, the region has about 3500 plant species in 135 families and 724 genera with approximately 25 % of these species being endemic to Namaqualand (Le Roux and van Rooyen 2006; Desmet 2007). Geophytes provide 16 % of this flora, which is much higher compared to other desert areas with winter rainfall (Desmet 2007).

The Kamiesberg has become known for its high levels of plant endemism, thus making it a biodiversity hotspot and an area of conservation concern (Helme and Desmet 2006; Anderson and Hoffman 2007; Snijman 2013). The Kamiesberg Uplands has 57 endemic species, along with a further 57 near endemic species. Kamiesberg Granite Fynbos and Namaqualand Granite Renosterveld supports the highest and second highest endemic species, in the Kamiesberg, respectively (Helme and Desmet 2006). The endemic species of Renosterveld may have lost significant portions of their population due to transformation by agricultural activities in the Kamiesberg respectively (Helme and Desmet 2006).

2.4.3. Vegetation of Namaqualand

Namaqualand forms part of the Succulent Karoo, a world renowned biodiversity hotspot of global importance as it supports the richest succulent vegetation in the world (Cowling *et al*. 1999). This biodiversity hotspot is well-known for its magnificent flower display during spring (occurring generally from mid - August to the end of September) depending on rainfall (Desmet 2007).

The Leliefontein Communal Area is home to nine vegetation types (Table 2.1) and is an area of noted biodiversity and conservation concern (Lombard *et al*. 1999). Six out of the nine vegetation types form 98.3 % of the vegetation cover in the entire Leliefontein Communal Area. The six vegetation types are: Namaqualand Klipkoppe Shrubland, Namaqualand

Blomveld, Namaqualand Heuweltjieveld, Namaqualand Granite Renosterveld, Kamiesberg Mountain Shrubland, and Kamiesberg Granite Fynbos (restricted to the area) (Mucina and Rutherford 2006).

Fynbos biome mainly borders the Succulent Karoo biome in the north and northeast. Furthermore, the two dominant plant species in each of these biomes can influence the boundary dynamics (Mucina and Rutherford 2006). Renosterveld occurs along the transitional area on shale alluvium between fynbos and succulent karoo shrublands (Mucina and Rutherford 2006). Mountain Renosterveld is an island of the Fynbos in the Kamiesberg surrounded by Succulent Karoo thus showing strong karroid affinities (Van der Merwe and Van Rooyen 2011).

Table 2.1: The nine vegetation types of the Leliefontein Communal Area and the biome in which they occur as well as their proportion (%) relative to the total size of the study area (Samuels 2013).

Vegetation Type	Biome	Proportion in relation to study area (%)
Namaqualand Klipkoppe Shrubland	Succulent Karoo	52.3
Namaqualand Blomveld	Succulent Karoo	20.3
Namaqualand Heuweltjieveld	Succulent Karoo	11.2
Namaqualand Granite Renosterveld	Fynbos	6.8
Kamiesberg Mountains Shrubland	Succulent Karoo	6
Kamiesberg Granite Fynbos	Fynbos	1.8
Namaqualand Riviere	Succulent Karoo	0.9
Namaqualand Inland Duneveld	Succulent Karoo	0.6
Namaquland Sand Fynbos	Fynbos	0.1

Renosterveld has been primarily transformed for field crops and vineyards (Milton 2007). Today renosterveld occurs as fragmented patched in transformed agricultural landscapes with a poor conservation status (Reyers *et al*. 2001; Milton 2007). In the Kamiesberg, Namaqualand Granite Renosterveld occurs on the deeper soils of the upland and lies between 1000 m and 1200 m (Helme and Desmet 2006; Allsopp *et al*. 2007). Over 20 % of this vegetation has been transformed through agriculture which includes cropping and grazing (Reyers *et al*. 2001; Helme and Desmet 2006). Grazing and natural and man-ignited fires are important conservation concerns to the area.

2.5. History of Namaqualand

2.5.1. Brief overview of Land Tenure and Land use history and the establishment of Leliefontein Communal area Namaqualand

Namaqualand owes its name to the pre-colonial Nama-speaking Khoikhoi pastoralists' whose ancestors took part in the pastoral expansion in southern Africa approximately 2 000 years ago (Rohde and Hoffman 2008). Namaqualand was historically known as the district of Little Namaqualand in the Cape colony, whereas the areas across the Orange River to the north (today known as Namibia) was referred to as Great Namaqualand. The pre-colonial Nama speaking Khoikhoi, are regarded as both a nomadic and pastoral society mainly because they always sought 'to maximize production by a seasonal exploitation of natural resources and movements that were largely related to seasonal cycles' (Penn 1986). Full control or access to the land in question was therefore of extreme importance to the Nama pastoralist in order to ensure prosperity and a lack of access at any time to a resource of the land could disrupt their entire cycle of transhumance and thus threatened their survival.

The Kamiesberg is situated in the very heart of Little Namaqualand (i.e. north of the Groen Rivier and south of the Buffels Rivier) and is often also referred to as to the Leliefontein Communal Area. The Leliefontein village[4], which is situated at the top of the Kamiesberg mountain range, originated in 1816 after the establishment of the Wesleyan Methodist mission station in the area (Kelso 2010; Samuels 2013). The Kamiesberg mountain range however has long been referred to as an oasis in the midst of the inhospitable semi-desert region of Namaqualand. Prior to colonial expansion into the region, it was an important summer rainfall season gathering point for the Nama speaking Khoikhoi pastoralists', which is illustrated in the name the earliest inhabitants referred to the area, namely, 'kamies' which means 'to gather' (Price 1976). But colonial expansion into the Namaqualand region both under Dutch and British rule and later Apartheid policy of 'separate development', restricted the transhumance movement of the Nama speaking Khoikhoi pastoralists and their descendants into confined communal areas (Samuels 2013). This consequently resulted in insecure land tenure, high unemployment and limited economic opportunities for these inhabitants (Rhode *et al*. 2003).

The arrival of the missionaries also impacted on aspect of governance and ownership in the current communal areas within Namaqualand and these aspects are also important to take into consideration when discussing who has access to the communal area and how enforcement of rules and regulations (e.g. grazing regulations, or rules regarding the burning of rangeland patches or croplands) have changed over the years. According to historical records chief Wildschutt of the Nama and four elders undertook a journey to Cape Town in 1816, in search of a missionary who could help them protect their land being grabbed by trekboers (Catling, 2008). Whilst on his way to Greater Namaqualand, the Wesleyan Missionary, Rev. Barnabas

[4] Both the Leliefontein Communal Area as well as the Leliefontein village are often referred to as 'Leliefontein'. This thesis will however differentiate between the two as done above.

Shaw, after an encounter with the Nama chief and the four elders agreed to settle in the Leliefontein village as the missionary of the Nama people who resided in the vicinity of the Kamiesberg under captain Wildschutt (Shaw, 1970). By 1824 the governor at the Cape approved that the resident missionary should be given full authority so as "to preserve order" in area (Swarts 2014). With the expansion of the British colony, on 17 December 1847 the entire Little Namaqualand region was declared as crown land and the Cape Colonial government awarded several "tickets of occupation" (not ownership rights) to the Nama inhabitants who had a missionary stationed at their respective reserves or prescribed territory (Rohde *et al*. 2003; Hoffman Rohde 2007). In 1854, a ticket of occupation was issued to the Nama-speaking Khoikhoi pastoralists' or 'Little Nama' in the Leliefontein Reserve (Rohde *et al*. 2003; Carstens 1966). This Reserve or Missionary Station became a central point for the pastoralist people to continue their practice of transhumance, albeit in a confined space that was limited to certain congregational outstations determined by the missionary and the local church council. In addition, the missionaries also introduced cropping (i.e. dryland cultivation of crops such as wheat, barley, oats, etc.) into the area which was adapted to the transhumant lifestyle of the Little Nama on the station (Rawson 2017). Price (1976) argued that the inevitable results of being a 'closed in' pastoral people began to become evident with signs of 'overpopulation, over-grazing and the soil being worked to the point of exhaustion.'

After a long and inconsistent history of governance and ownership (from the missionary to a Resident Magistrate in the late 1800s), the enactment of the Mission Stations and Communal Reserves Act 29 of 1909, brought Leliefontein Mission Station completely under the control of the State and saw the complete separation of power between church and State (Rohde *et al*. 2003; Anderson 2008). Thus, in just less than a century the land that was once governed for centuries past by the Namaqua Chief had been expropriated by church and finally the State.

In 1984 the Apartheid led government made attempts at privatising parts of the Leliefontein Reserve through the introduction of 'economic farming units' (Samuels 2013). Marinus, (1998) have argued that this was arguably one of the most explicit manner through which the State undermined the informal rights of the inhabitants to land and resources, as it meant a change in tenure. However, the abolishment of the economic farming units three years after its introduction, meant the communal tenure system of the inhabitants was reinstated throughout the Reserve. Efforts in post- Apartheid South Africa to reform communal land tenure and transfer land ownership to the inhabitants of these former Communal Reserves and Missions Stations in Namaqualand have not been finalised yet and the land is still presently held in trust by the State/ government under the guardianship of the Minster of Rural Development and Land Reform (Swarts 2014). However, with the introduction of local municipalities (local government) from the year 2000 onwards in the Namaqualand area, the responsibility for managing the access and use of the trust land/ communal areas, now resides with the local government and a group of locally elected committee members, that is until the Minister decides when land is ready to transferred to an entity of the inhabitants' choice.

2.5.2. People in the Leliefontein Communal Area and their land use livelihood strategies

Presently, the Leliefontein Communal Area is the third largest and most southerly of six communal areas in Namaqualand (Van der Horst 1976; Lebert 2005). The other five communal areas (formerly known as 'Coloured Reserves' under the Apartheid government of South Africa) in the Namaqualand region are Richtersveld, Pella, Steinkopf, Concardia, and Komaggas (Swarts 2014). The six communal areas, comprises approximately 1 010 703 ha of land (about 27 % of the Namaqualand region), and are home to roughly 40 % of the regions inhabitants (Lebert 2005). The communal areas are very isolated and geographically distant

from one another but the history of these areas are quite similar as all of them formally originated as mission stations (Kelso 2010). These communal areas are spread over three different municipalities, with the Leliefontein Communal Area falling within the jurisdiction of the Kamiesberg Local Municipality which was established in 2000 (Lebert 2005).

The Leliefontein Communal Area is divided into 10 village commons (Fig. 2.4) (Lebert and Rohde 2007; Samuels 2013). These village commons are not fenced into grazing camps and vary in size, between approximately 12 000 ha and 25 000 ha (Swarts 2014). The Leliefontein Communal Area has over 1 500 farming households and thus each village commons are used by a number of farmers (Stats SA, 2011). These village commons are Klipfontein, Kharkams, Spoegrivier, Kheis, Tweerivier, Leliefontein, Paulshoek, Nourivier, Rooifontein and Kamassies (Fig. 2.4). Kamassies and Rooifontein jointly utilize a single common (Samuels 2013).

These people are mostly descendants of the indigenous Nama speaking Khoikhoi pastoralist and the San, and comprise a range of mixed-race communities, which developed with the expansion of the colonial frontier and slave trade. The turn of the 20th century and later during Apartheid saw the descendants of the Nama speaking Khoikhoi pastoralist and other Khoisan[5] groupings being classified as 'coloured'[6]. The population of Namaqualand is almost entirely Afrikaans speaking although Nama is still spoken in the northern communal regions (Anderson *et al.* 2010).

Those living within the Local Kamiesberg municipality, as well as Namaqualand as a whole, are some of the poorest communities in South Africa crippled by high levels of unemployment

[5] The Nama are one of the five major groupings of the Khoisan in South Africa; the other four being the Korana, Cape Khoi, San, and Griqua.

[6] According to Adhikari (2005) in South Africa, the term 'Coloured' does not refer to black people in general but it instead alludes to a phenotypically diverse group of people descended largely from Cape slaves, the indigenous Khoisan population and other people of African and Asian descent who had been assimilated into Cape colonial society by the late nineteenth century (2005:468).

and poverty (Swarts 2014). In keeping with the tradition of their forefathers the area is still extensively used for agricultural purposes more specifically livestock grazing (Landman *et al*. 2006). These livestock herds consist of small stock such as goats and sheep which are usually accompanied by a herder. Cattle and donkeys are also found in some herds but are usually free ranged. Livestock also serve as an investment for these impoverished communities as they can be easily sold for cash in times of economic turmoil. The communal land in the Kamiesberg is also used for other activities such as the harvesting of firewood, the collection of medicinal plants, food gardening and the harvesting of reeds.

Dryland cropping have drastically declined since the mid-19th century and today an estimated 12 % of the demarcated cropping area is utilized for dry land cropping, mostly on the deeper soils of the valley bottoms. Individuals can access these croplands by paying an annual rental fee. Crops such as straw, barley and rye are grown and Lucerne are mostly used for livestock fodder, especially during dry periods or drought years. However, economic opportunities remain limited within the communal areas (Rohde *et al*. 2003) and in the past decade government welfare grants and remittances have become a major source for many residing within the broader Namaqualand region (Swarts 2014).

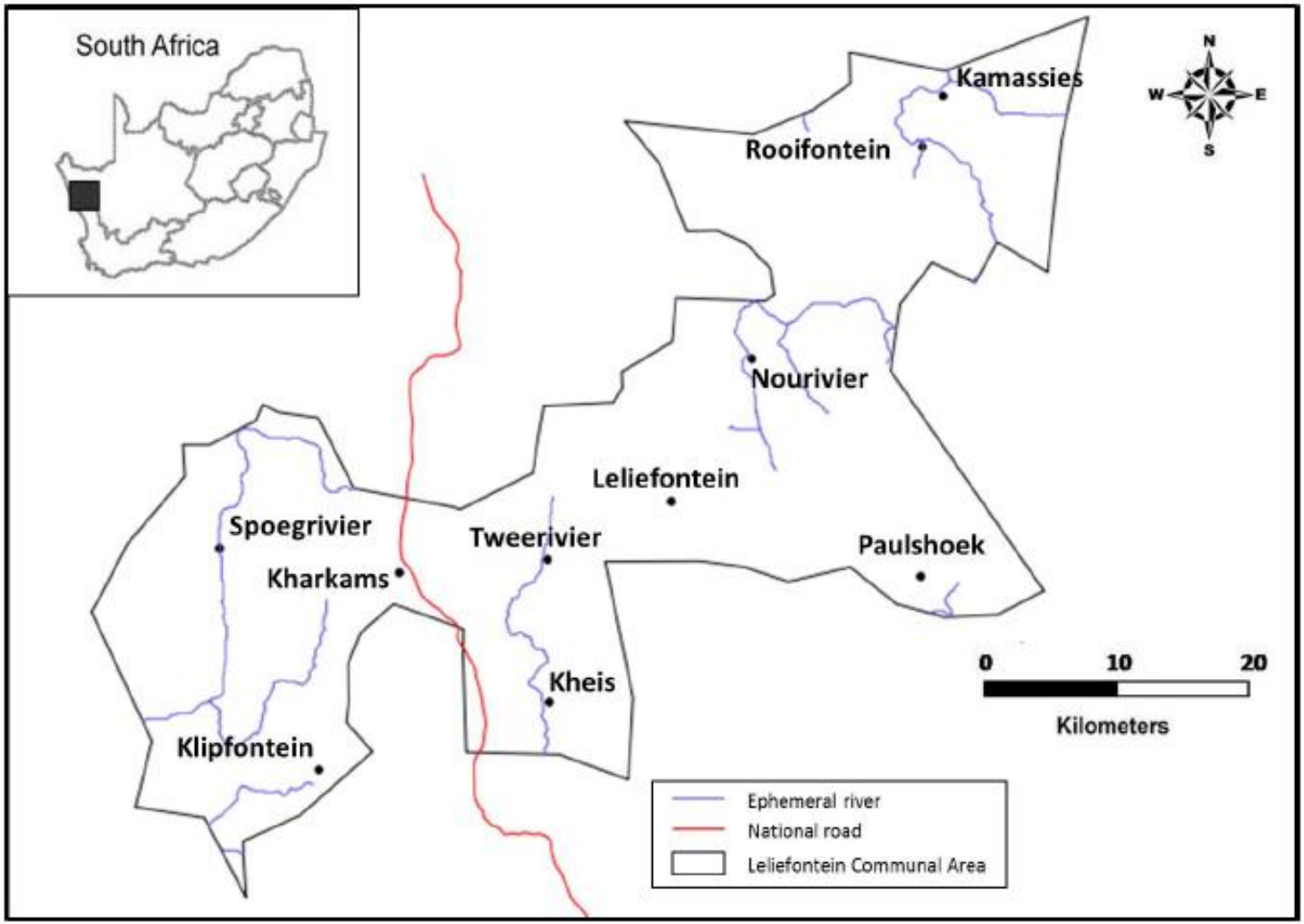


Figure 2.3: The location of the 10 village commons in the Leliefontein communal area in Namaqualand, Northern Cape of South Africa (Samuels 2013).

3. Chapter three: Assessing the fire regime for Namaqualand Granite Renosterveld in a semi-arid communal rangeland

3.1.Introduction

Fire is a universal terrestrial disturbance which plays a key role in ecosystem composition and distribution, as it consumes vast quantities of plant biomass annually (Bond and Van Wilgen 1996; Bond *et al*. 2005). Fires are key processes in the distribution, functioning and maintenance of fire-prone ecosystems (Bond and Keeley 2005). For this reason, it is crucial to describe and understand the fire regime (van Langevelde *et al*. 2003). In fire-prone ecosystems, the range of fire season, fire frequency (return period), size and intensity of fires describe the fire regime, which characterizes the ecosystem (Gill 1975; Stellmes *et al*. 2013). Within these ecosystems, the biota might adapt to, or depend on, a particular fire regime (Bond and Van Wilgen 1996; Van Wilgen *et al*. 2010).

Humans use fires for hunting, honey collection, pest control and cultivation and thus manipulate the fire regime (Stellmes *et al*. 2013). Moreover, farmers also utilise fires to provide fresh plant fodder for their livestock and cattle (Stellmes *et al*. 2013). Therefore, altering the fire regime of an area can lead to loss of biodiversity, land degradation, changes in species composition and vegetation structure (Govender *et al*. 2006).

Renosterveld is a fire-prone system in which its fire ecology is poorly understood (Mucina and Rutherford 2006; Curtis 2013). The fire return interval of renosterveld have been debated extensively by ecologists and is assume to be 3 -5 years (Boucher 1995); 3 -4 years or even up to 40 years in more arid regions (Mucina and Rutherford 2006); or 2 -10 years (Kraaij and Van Wilgen 2014). Fires in renosterveld largely occur in late summer and early autumn which is towards the end of the dry season (Kraaij 2010). These fires are thought to occur naturally but

with the increase in human densities more man-made fires have increased (Mucina and Rutherford 2006).

Due to the fact that renosterveld is subjected to regular fires, the frequency, season, size and intensity of fires are important determinates of vegetation structure, species composition and successional patterns (Van Wilgen *et al.* 2010). The fuel dynamics of pre-colonial renosterveld might have been affected by fire and grazing, which in turn would have had an impact on the fire regime (Kraaij and Van Wilgen 2014). However, the fuel dynamics of renosterveld is not well known, but it is thought to burn quite frequently due to fast plant growth and finer fuels (Boucher 1995). There is a concern that the absence of fire and herbivory in isolated renosterveld fragments may lead to woody shrub encroachment (Kraaij and Van Wilgen 2014). Therefore, studying the fire regime contributes to an improved understanding of the ecological role of fires (Kraaij 2010).

The impact of fires on various landscapes are complex due to the difference in fire regimes and its interaction with additional environmental factors such as grazing pressure, climatic changes and landscape heterogeneity (Govender *et al*. 2006). Fires can seldom be isolated from its association with grazing, which is responsible for much of the controversy surrounding the use of fire in southern Africa (Snyman 2015). Information regarding the response of semi-arid rangelands to fire is limited (Snyman 2003; Breedte *et al*. 2013) although fire is an important management tool from a plant conservation perspective and to improve forage productivity for livestock (Breedte *et al*. 2013). Accidental unplanned fires on rangelands can have detrimental consequences not only on the farm infrastructure but economic losses due to a decline in functionality and productivity on rangelands (Snyman 2006). Of the few accounts of fires in renosterveld, fires ignite either due to natural cause such as lighting storms in mountain renosterveld (Helme and Desmet 2006) or prescribed burning is applied in the case of south coast renosterveld (Kraaij and Van Wilgen 2014). In addition, humans appear to be the main

source of ignition which can increase the frequency of fires in renosterveld (Helme and Desmet 2006).

Therefore, fire cannot be ignored when considering the management of rangeland ecosystems whether for livestock or biodiversity. Understanding ecosystem dynamics under different fire regimes is essential to ensure sustainable plant and animal production in these semi-arid rangelands (Snyman 2015). This study provides the first comprehensive assessment of the fire regime in the Kamiesberg Uplands in Namaqualand. The study aims to assess the fire regime for Namaqualand Granite Renosterveld which occurs on communal and private land in the Kamiesberg region.

The following questions are addressed in this chapter:

3.1. What are the ignition sources of fire in Namaqualand Granite Renosterveld landscapes?

3.2. What is the Fire Danger Index for Namaqualand Granite Renosterveld landscapes?

3.3. Which factors affect the spatial extent of fires in Namaqualand Granite Renosterveld landscapes?

3.2. Methods

3.2.1. Study area

The study was carried out in the Leliefontein village commons. Refer to Chapter two for a detailed description of the study area. The 11 burnt sites are all located on Namaqualand Granite Renosterveld (Fig. 3.1) and are subjected to variant physical factors and grazing pressures.

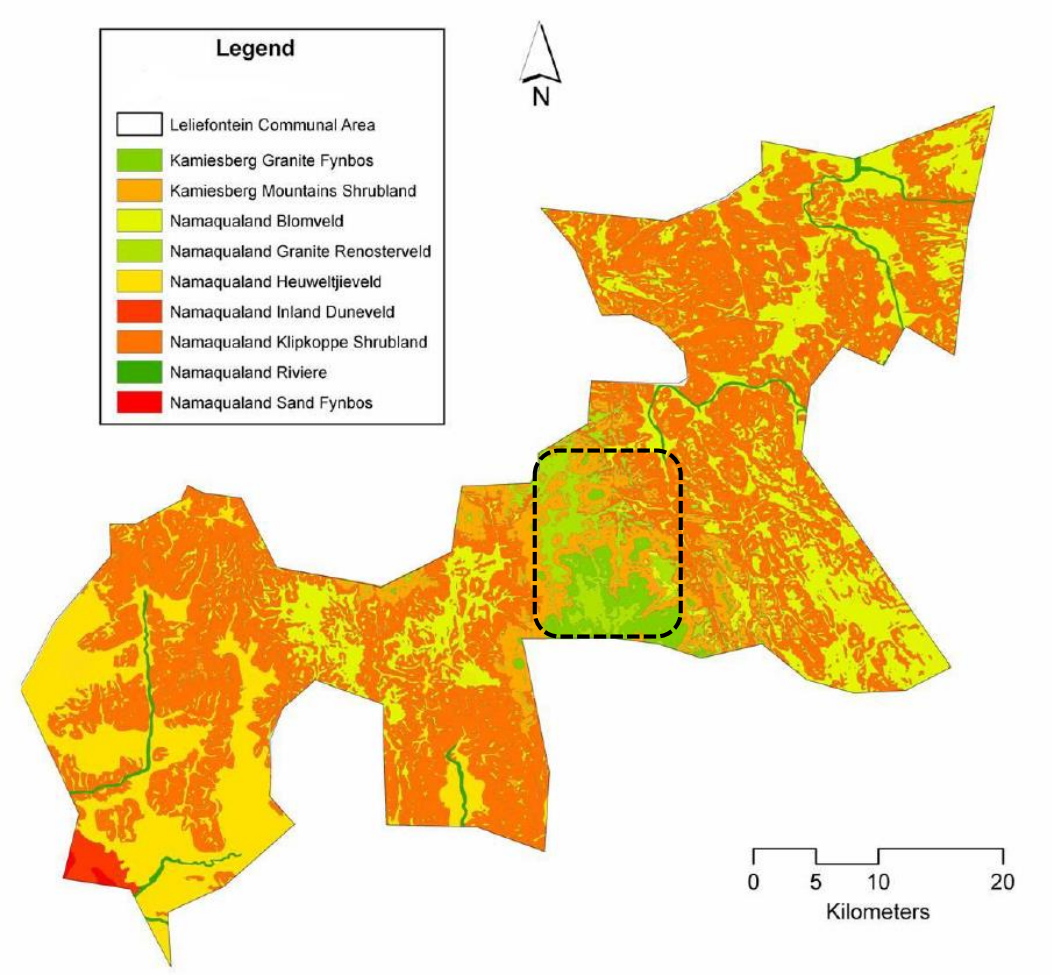


Figure 3.1: The distribution of vegetation types across the Leliefontein Communal Area. The area circled in black is Namaqualand Granite Renosterveld (adapted from Samuels 2013).

3.2.2. Data collection

3.2.2.1.Completion of a fire database

I compiled a database of all fires in the study area since the beginning of 2000 until 2015. Historical fire data were obtained from the National Aeronautics and Space Administration (NASA) Moderate Spatial Resolution Spectroradiometer (MODIS) fire archive, http://rapidfire.sci.gsfc.nasa.gov/. The study sites were overlaid onto the renosterveld vegetation map (Mucina and Rutherford 2009) t' determine if the location of fires fall within Namaqualand Granite Renosterveld vegetation (Fig. 3.2). Ground truthing and Google Earth

Pro (V 7.1.1.1580) were used to verify if these burns and to determine accessibility to these sites for ecological measurements.

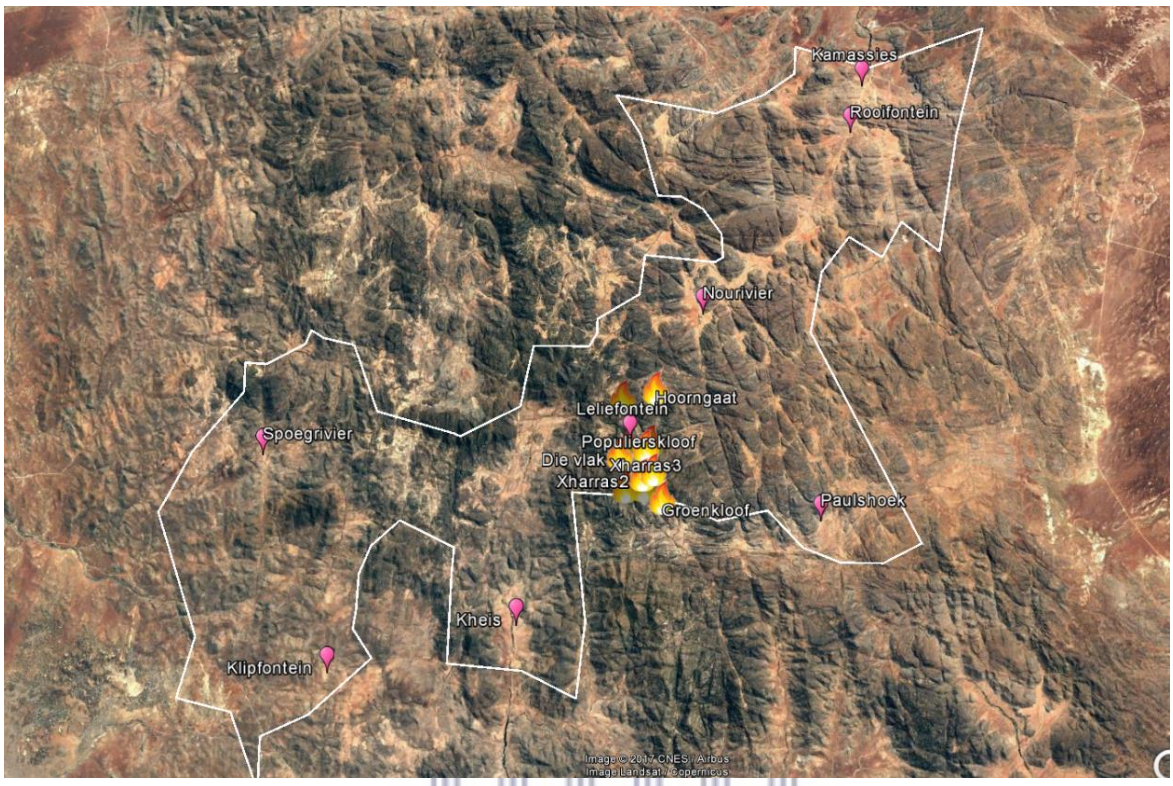


Figure 3.2: Distribution of burnt sites across Namaqualand Granite Renosterveld from 2000 to 2015.

3.2.3. Data analysis

3.2.3.1. The cause, size and season of fires

The cause of fires was obtained from informal interviews with farmers (Simons 2015) and once the main causes were established, these were classified as natural (ignited by lighting), anthropogenic activity (mostly unplanned burning) and fires of unknown origin. The size of the individual fires was examined using Google Earth Pro (Version 7.1.1.1580). For the size distribution I classified fires into small (<10 ha), medium (10-50 ha), large (>50 ha). Furthermore, these fires were categorized into different seasons such as: summer (December-February); autumn (March-May); winter (June-August); and spring (September-November).

3.2.3.2.Fire frequency

Fire occurrence data was used to calculate the mean fire return period as follow:

$$RP = y/(b/a)$$

Where *RP* is the return period in years, *y* is the number of years over which fires were recorded, *b* is the extent of all fires recorded over *y* years, and *a* is the area over which fires were recorded. I encountered difficulty in using this formula, as both the sizes of the total area and area burnt were less than 1000 ha. The above calculation has been used extensively in areas which has burnt sizes over 1000 ha (see Kraaij *et al*. 2013a; Forsyth and Van Wilgen 2008).

$$FRI = \frac{N}{a}$$

I developed my own formula, where *FRI* is the fire frequency in years, *N* is the total number of fires and *a* being the number of fires studied.

3.2.3.3.Fire Radiative Power (FRP) as a measure of fire intensity

The Fire Radiative Power (FRP) Satellite performing middle-infrared wavelength measurements over actively burning fires was used to calculate the rate of radiant energy release in units of megawatts per pixel (mW/ pixel) (Kaufman *et al*. 1996). The energy released by individual fires varies greatly over the duration of the fire (Smith and Wooster 2005).

3.2.3.4.Fire weather

I obtained daily weather data from the Agricultural Research Council (ARC) and Conservation South Africa (CSA) for the period 2002-2008 and 2008-2015 respectively. These environmental factors were selected based on the assumption that they may either be direct and indirect drivers of the fire regime. The most widely used fire danger indices (FDI) are the

McArthur Forest FDI (McArthur 1966) and the Lowland index (Meikle and Hein 1987). I used the Lowveld Model, as it has been approved as the National Fire Danger Rating for South Africa (DAFF 2013). The FDI for each burnt year was calculated, excluding 2000 and 2001 as no weather data was available for these years. The data used included maximum temperature, minimum relative humidity, wind speed and rain (correction factors). The Burning index (BI) was calculated using maximum temperature and minimum relative humidity.

$$BI = \{(T-35) – ((35-T)/30) + ((100-RH) *0.37) + 30\}$$

T: Temperature, Degrees Celsius

RH: Relative humidity (%)

Once the burning index has been calculated it is adjusted to wind speed thus adding a value according to the prevailing wind conditions at 14h00 (Table 3.1). The availability of excess moisture (above the plant saturation point) provided by recent rainfall and the number of days since that rainfall (Table 3.2), is taken into account by multiplying the burning index by rainfall correction factor.

$$FDI = (BI + \text{wind factor}) \times \text{rain correction factor}$$

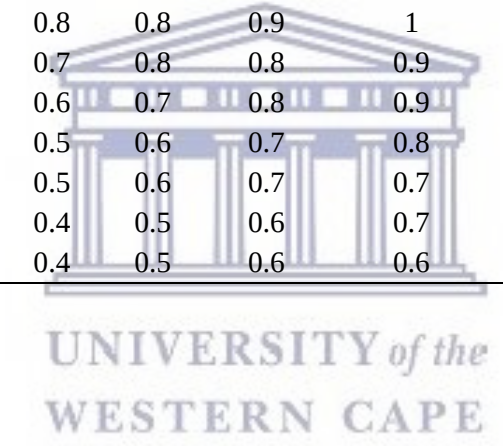
FDI values of 0 - 20 are considered low, 21 - 45 moderate, 46 - 60 dangerous, 61 - 75 very dangerous and 76 - 100 extremely dangerous. Each studied fire record was assigned a FDI value; where fire burnt more than one day, the highest FDI rating was used (DAFF 2013).

Table 3.1: The effect of wind speed (km/h) expressed as the wind correction factor which is added to the burning index (DAFF 2013).

Wind speed (km/h)	Add to BI value
0-2	+ nil
3-8	+05
9-16	+10
17-25	+15
26-32	+20
33-36	+25
37-41	+30
42-45	+35
46 +	+40

Table 3.2: The rainfall correction factors used to adjust the burning index for antecedent moisture conditions (DAFF 2013).

Rain (mm) following fire	Days since rain											
	1	**2**	**3**	**4**	**5**	**6**	**7 - 8**	**9 - 10**	**11 - 12**	**13 - 15**	**16 - 20**	**21+**
0.1 - 2.6	0.7	0.9	1	1	1	1	1	1	1	1	1	1
2.7 - 5.2	0.6	0.8	0.9	1	1	1	1	1	1	1	1	1
5.3 - 7.6	0.5	0.7	0.9	0.9	1	1	1	1	1	1	1	1
7.7 - 10.2	0.4	0.6	0.8	0.9	0.9	1	1	1	1	1	1	1
10.3 - 12.8	0.4	0.6	0.7	0.8	0.9	0.9	1	1	1	1	1	1
12.9 - 15.3	0.3	0.5	0.7	0.8	0.8	0.9	1	1	1	1	1	1
15.4-20.5	0.2	0.5	0.6	0.7	0.8	0.8	0.9	1	1	1	1	1
20.6-25.5	0.2	0.4	0.5	0.7	0.7	0.8	0.8	0.9	1	1	1	1
25.6-38.4	0.1	0.3	0.4	0.6	0.6	0.7	0.8	0.9	1	1	1	1
38.5-51.1	0.1	0.2	0.4	0.5	0.5	0.6	0.7	0.8	0.9	1	1	1
51.2-63.8	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.7	0.8	0.9	1	1
63.9-76.5	0.1	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.8	0.9	1
76.6+	0.1	0.1	0.1	0.2	0.4	0.5	0.6	0.6	0.7	0.8	0.9	1



3.2.4. Statistical analysis

Linear regressions were used to test whether a relationship exist between the Fire Radiative Power and number of fires. Pearson correlations were performed to test if a) Fire Danger Index and b) Fire Radiative Power has an effect on the size of fires. These statistical analyses were performed in STATISTICA (StatSoft, Inc., Tulsa, OK, version 8, 2007).

3.3. Results

3.3.1. The fire record

A total of 42 different fires recorded over 15 years (Fig. 3.3) of which only 11 burnt sites were studied extensively. This was due to the difficulty in determining the boundaries of the fire scars. Fires of natural origin were not selected as most occurred high up in the mountainous areas (e.g. Eselskop) and were difficult to access due to steep slopes and no access roads. Therefore, no results were generated for natural fires and this study has only reported results for accidental unplanned veld fires in Namaqualand Granite Renosterveld. The years 2003 and 2015 were classed as drought years, however surprisingly most of the fires were recorded in 2003 and only one fire in 2015.

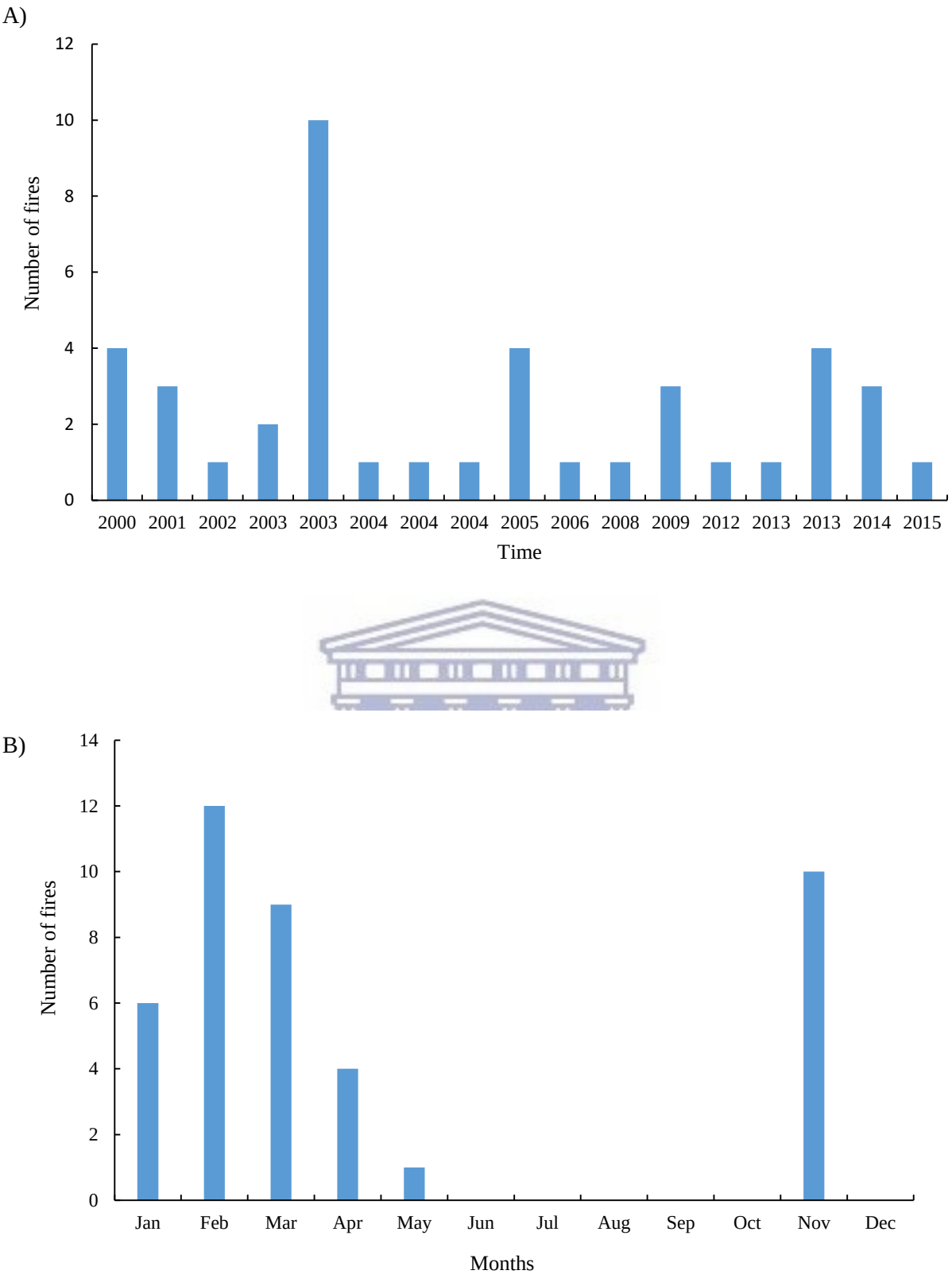


Figure 3.3: The total number of fires recorded over a) 15-year period and b) monthly fires over 15 years in Namaqualand Granite Renosterveld in the Kamiesberg.

3.3.2. Causes of fire

The majority of the fires in the Kamiesberg are accidental fires (Table 3.3). Six ignition sources were identified for all fires across the Kamiesberg. The two main causes of fires identified were burning for wild honey harvesting (31 % of all fires) and lightning (21 % of all fires). In 2010 most of the fires were attributed to natural events (mainly lightning strikes) but the extent of the areas was not determined as they occurred in the high peaks and was not included into the study sites. Lightning-induced fires mostly occurred on the mountain peaks such as Eselskop. However, fires ignited due to falling rocks have been recorded in the Kamiesberg (Helme and Desmet 2006).

Table 3.3: Perceptions from farmers on the causes for igniting fires in Renosterveld vegetation in the Kamiesberg (Simons 2015).

Ignition sources	Percentage (%) of fire
Wild honey harvesting	31
Lightning	21
Herders making tea in the veld	21
Cropland clearing before planting	11
Dense vegetation burnt to eradicate livestock predators or snakes	11
Kraal manure burnt to eradicate parasites	5

The causes of fire were incompletely recorded for the study sites, as five of the 11 studied fires were of unknown origin. Four fires were caused due to wild honey harvesting, whereas one fire was due to cropland clearing and the other due to burning kraal manure. The eleven fires studied were all human-induced fires (Table 3.4).

Table 3.4: Fire ignition sources for 11 studied fires and the extent of the area burnt in total (ha) for each study site in Namaqualand Granite Renosterveld from 2000 - 2015

Cause of ignition	Area burnt (ha)	Sites
Wild honey burning	940	Skaaprivier Toringberg Populierskloof Xharras 3
Unknown	51	Hoorngaat Xharras 2 Groenkloof Xharras 1 Kliphoek
Cropland clearing before planting	15	Bailylesvlakte
Kraal manure burning to eradicate parasites	6	Die Vlak

3.3.3. Sizes of fire

The 11 fires accounted for 1 012 ha, 7.7 % of the fire prone area, on burnt renosterveld. Burning to harvest wild honey accounted 940 ha of the total burnt area, whereas fires due to cropland clearing, kraal manure and unknown origin were less important in terms of areas burnt (Table 3.4). Fire sizes ranged from 1.47 to 922.2 ha (Fig 3.4). Numerous small fires of < 10 ha occurred throughout the study sites. Most of the reported fires in the study area were small (< 10 ha) and medium (10 – 50 ha) sizes, and collectively accounted for 9 % of the total burnt area. Large fires (> 50 ha) were infrequent and only one was recorded for the study period. This fire accounted for 91 % of the total burnt area. The FRI for the study sites in the Kamiesberg was 4.2 years, with at least two fire events in one year.

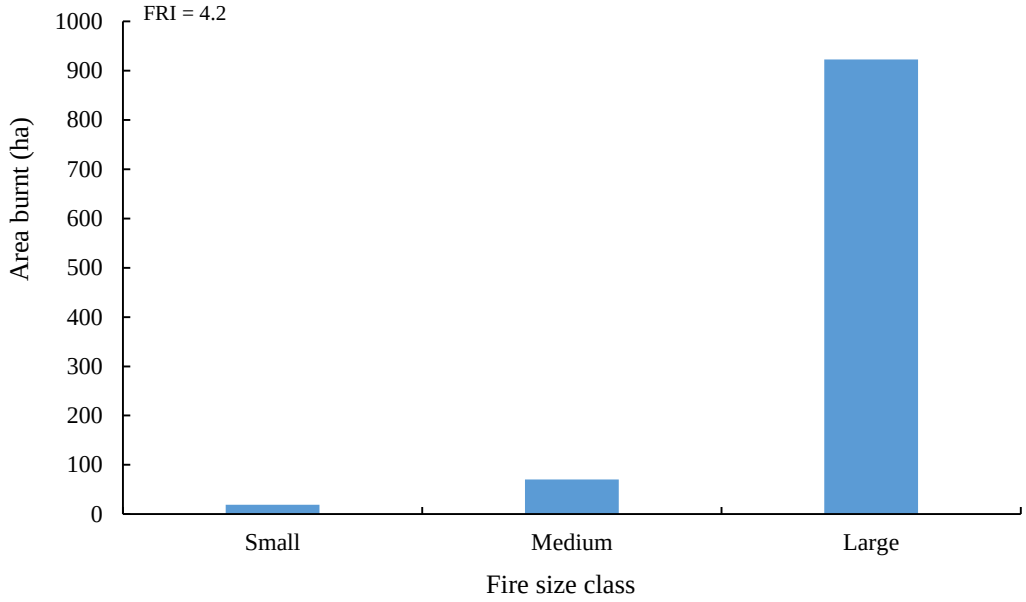


Figure 3.4: The distribution of fire size classes across the Kamiesberg Uplands.

3.3.4. Seasons of fire

Seasonal fires occur throughout the Kamiesberg with most of the fires concentrated predominantly in the dry summer (December - February) months (Fig. 3.5). Subsequent to this fuels are usually more dry and flammable during the summer season. Fires in autumn and spring burnt over a relatively small area (4 %) of the total burnt areas, while the main causes of these fires were either ploughing or unknown. Fires due to honey harvesting burnt approximately 96 % of the area during summer. No fire was recorded during the winter season over the study period.

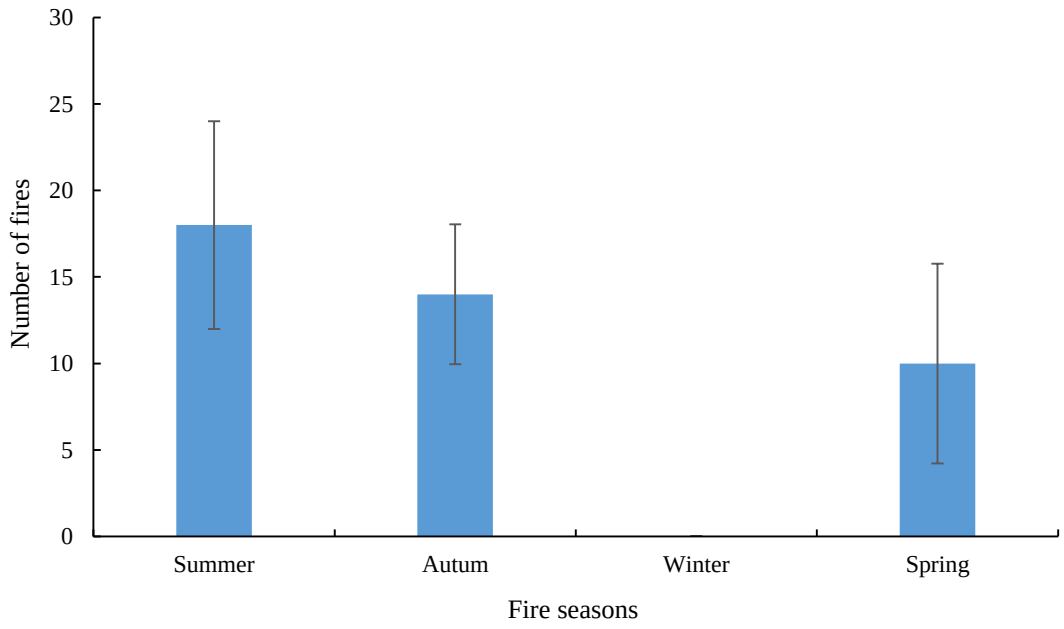


Figure 3.5: Seasonal distribution of fires across the Namaqualand Granite Renosterveld

3.3.5. Fire intensity as Fire Radiative Power (FRP)

The energy released by individual fires varies greatly over the duration of the study period. Fire Radiative Power increases during the summer season and reached a maximum of 2565.3 MW during the peak of the dry season (February) (Fig. 3.6). Whereas the lowest FRP of 8.3 MW was recorded in late autumn.

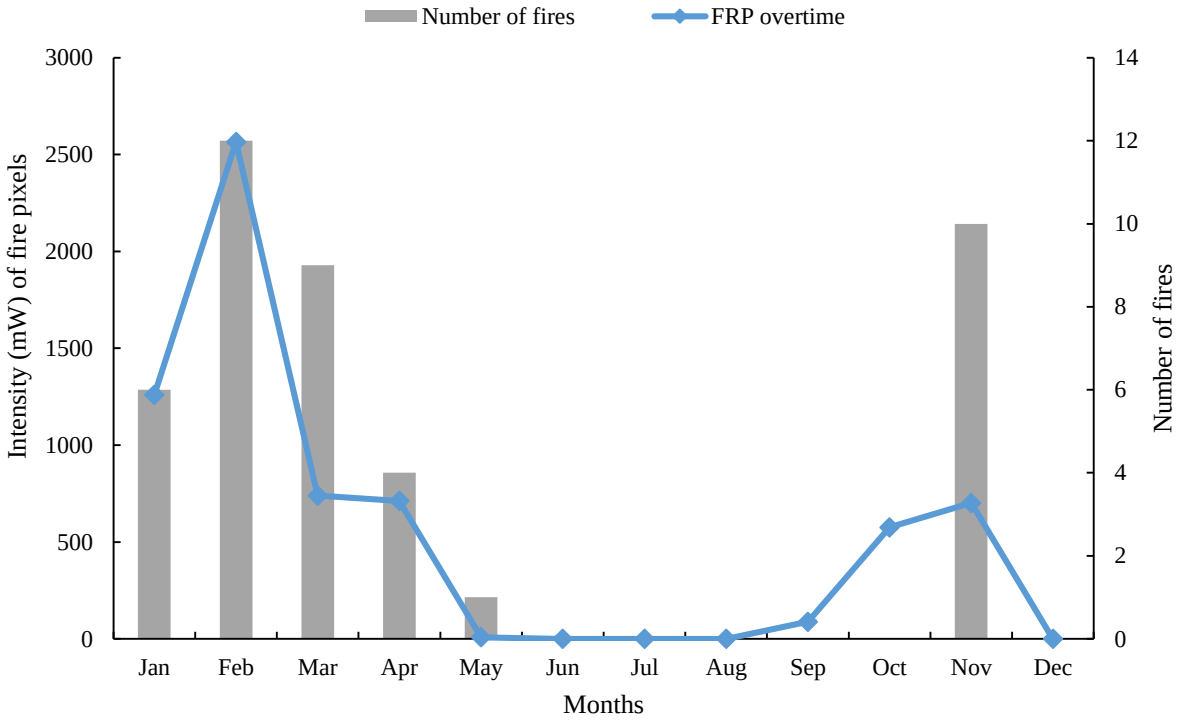


Figure 3.6: Seasonal pattern of fire intensity (FRP) across Namaqualand Granite Renosterveld in the Kamiesberg Uplands.

The number of fires over each month significantly increased the FRP, y= 136.77x + 75.45, β = 0.82, t (10) = 4.59, p = 0.001, n = 12. Number of fires also explained a significant portion of variance in the FRP, R^2 = 0.67, F (1,10) = 21.11, p = 0.001. The maximum FRP recorded in each MODIS pixel was isolated as an indication of the maximum rate of energy released. The number of fires continuously increased at the beginning of the summer season. In the dry summer season more than six fires were recorded during January to February. However, these fires often burn for several hours during the day and can continuously burn for > 1 week.

3.3.6. Fire weather

Analysing the fire weather using the Lowveld Fire Danger Index (FDI) model; the data showed that fires can take place under a wide range of conditions. The FDI ranged between dangerous to very dangerous across the burnt sites. None of the sites exceeded an FDI of 76, hence an extremely dangerous fire rating was not recorded for the burnt sites. Small (<10 ha) and medium (10-50 ha) fires were not always associated with low FDI's as medium to high (54-72) fire ratings were recorded. Lastly, the largest fire (> 900 ha) had a considerably high (66) FDI (Fig. 3.7). Overall summer burns had the highest FDI values[7].

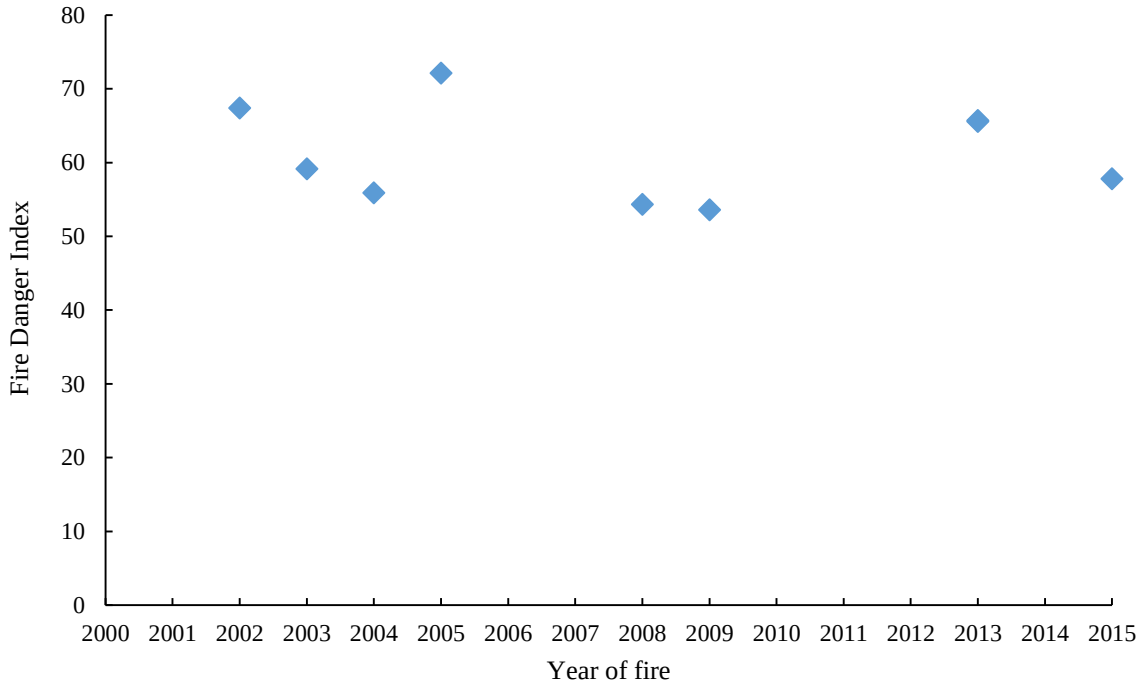


Figure 3.7: The fire danger index associated with fires of differing sizes in the Kamiesberg Uplands.

[7] Only eight values are seen in Fig. 3.7, due to the fact that there were insufficient weather data for the years 2000 and 2001 thus no FDI could be calculated. Furthermore, two fires were recorded in 2013 (one in early January and the other in late November) and both has an FDI of 66.

3.3.7. Factors influencing the size of fires

The Fire Danger Index and Fire Radiative Power does not significantly ($p > 0.05$) affect size of the burnt area. There is a weak, positive correlation between the FDI and area burnt ($r = 0.23$, $n=10$). These variables tend to increase together (i.e. larger burnt areas are associated with a larger FDI value). In contrast to the above results, FRP was negatively correlated ($r = -0.47$, $n=11$) with the area burnt. Even though the Pearson correlation coefficient was moderate.

3.4.Discussion

3.4.1. The cause of fire in Renosterveld

Fires ignited due to lightning, mainly resulted from thunderstorm activity (Bond *et al*. 1988). This is a distinct feature for most MTE such as California chaparral, Australia and Mediterranean basin (Montenegro 2004). Natural fires burnt mostly in the Kamiesberg mountainous areas, sometimes for more than a week (Simons 2015). The current known fire record for the Namaqualand Granite Renosterveld shows that the majority of fires are human-induced. Collectively human - induced fires accounted for 79 % of the ignition sources, of which wild honey harvesting was the main cause. Fires of human origin have gradually been increasing over time in the Kamiesberg. According to the fire record no prescribed burning has been done in the area but farmers have indicated that they often burn the veld once it becomes too dense, as wild animals not only pose a threat to the livestock but to the herders as well. Secondly, before farmers plough their land they burn it to remove *E. rhinocerotis*, which usually forms dense strands on fallow croplands as shown by Solomon (2011). It should be noted that even though farmers do not practise prescribed burning, they do plan their fires when burning croplands and kraal, and this study shows that these burns are not extensive and thus have a lower impact on the environment.

The primary cause is burning for wild honey. Burning for wild honey has becoming a popular activity in the Kamiesberg and this coincides with the peak of the burning season (i.e. summer). This activity is mostly practised by children whom ignite branches of *E. rhinocerotis* to smoke the honey hive, in most instances these branches are not extinguished properly which leads to most of the wild fires in the area. These fires are quite ferocious as the increase in wind can spread the fire over a substantial area. These fires can pose a great threat to the community and specifically the children who attempt to extinguish them (Fig. 1.2).

3.4.2. Fire frequency

In Namaqualand Granite Renosterveld the frequency of fires might vary but the seasonality remains stable. Due to the aridity of the area, the fire frequency for Renosterveld in the Kamiesberg is less than typical fynbos areas which occur further south (Helme and Desmet 2006). A FRP for Namaqualand Granite Renosterveld of 4.2 years, is not enough time for fuel to accumulate under the prevailing often heavy grazing regime to support an intense fire. For this reason, it is only one area (Skaaprivier) that has burnt twice during the study period (personal observation). In the past fires of human origin use to be less frequent in the Kamiesberg and natural fires (due to lightning) use to burn across the mountains (Simons 2015). The increase in short-interval fires in the study area is mostly attributed to the increased ignition sources generally resulting from human population expansion (Geldenhuys 1994; Forsyth and Van Wilgen 2008; Van Wilgen *et al*. 2010).

The recovery of each burn site is highly dependent on sufficient rain and rest from grazing. A mosaic of different post-fire ages not only promotes biodiversity (Weir *et al*. 2013), but also reduces fire hazards and a more tourism product in terms of annual flower display from mostly geophytes (Kraaij 2010). Also, different mosaics have diverse grazing patches with different fodder quality since forage is of different ages.

3.4.3. Fire season

The effects of fire season relate to the seasonality of growth, reproduction in species, and the floristic composition in fire-prone Mediterranean-climate shrublands (Booysen and Tainton 1984; Bond 1984; Kraaij *et al*. 2013b). In the Kamiesberg the fire danger conditions gradually increased from November and peaked during January until February. These dry periods increased the severity of the fire weather (Forsyth *et al*. 2010). This coincides with the summer drought climate of the Fynbos Biome (CapeNature 2013). Furthermore, this is consistent with climatic gradients, across the CFR, of increasing winter rainfall and summer drought from east to west and increasing summer evapotranspiration from the coast to the interior (Tyson and Preston-Whyte 2000; Kraaij *et al*. 2013a; 2013b). Consequently, burning during the acceptable season avoids negative impacts on the vegetation (Forsyth and Van Wilgen 2008). Burning for wild honey burnt the largest areas between November and February. Whilst natural fires burned in the peak of the dry season (February) (Helme and Desmet 2006). However, burning in the summer months, specifically November and February, is at great risk of runaway wildfires (CapeNature 2013). Additionally, fires in the ecologically undesirable seasons should be monitored, as they can burn extensive areas. Overall fires in the Kamiesberg remained within the accepted fire season. Therefore, inappropriate or out-of-season burns (e.g. spring) does not pose a threat to the management of Renosterveld, but it could have various effects on the composition and structure (see chapter 5) in the area.

3.4.4. Size of fires

The relationship between the fire size classes and area burnt showed that few large fires dominated the fire regime. This relationship between the number and size of fires is characteristic of numerous vegetation types worldwide (Keeley *et al*. 1999; Forsyth and Van Wilgen 2008; Van Wilgen *et al*. 2010; Kraaij *et al*. 2013a; Kraaij and Van Wilgen 2014). Small

fires do not remarkably contribute to the total burnt area and larger fires often burn the majority of the area as seen in the boreal forests (Strauess *et al*. 1989) and Australian savannas (Yates *et al*. 2008, Archibald 2010). In the Kamiesberg, large fires do burn larger areas in the mountainous areas and apparently natural fires are also often very large (B Links, pers.comm.). However, due to the inconsistency in the official record for earlier years (pre-2000) this has led to the absence of these fires from the fire record.

The size of the burnt area and severity of the fire behaviour on the day of burning can be due to strong winds driving these fires (Van Wilgen 2009; Sharples *et al*. 2012; Van Wilgen 2013; Kraaij and Van Wilgen 2014). The combustion rate of a fire is positively influenced by the rate of oxygen supply to the fire (Trollope 2007). Fires burning due to berg winds are likely to follow the direction of the wind and could be extremely intense. Similarly, sporadic fires under extreme bergwind conditions might destroy the vegetation beyond the margin of regular fires (Geldenhuys 1994). Nevertheless, this is highly unlikely to occur within the Kamiesberg as rocks often act as natural fire-breaks. In the Kamiesberg bergwinds are often experienced in the dry summer (Anderson *et al.* 2010) and the largest fires were recorded within this season. Increased wind speed can result in more intensive fires (Luke and McArthur 1978). However, in the Kamiesberg the flame height, which is approximately 3 m high (personal observation), does not necessarily increase with increased wind speed. In addition to wind, steep slopes and aspects are conditions which further influences the distribution of fires (Sharples *et al*. 2012). Wind and slope has the effect of tilting the flame towards the unburnt fuel (Fig. 1.3). In this way the flame height is reduced and the flame length remains unaffected.

The steepness of a slope can have a great influence on the fire behaviour. The degree of the steepness will determine the extent to which fuel is dried out before the fire front (Trollope *et al*. 2004). Furthermore, a slope can influence the rate of spread of fires. Only two fires occurred on a steep slope, while most fires had a relatively moderate slope. Wind is the most dynamic

variable influencing fire behaviour (Trollope *et al*. 2004). An increase in wind speed can cause greater rates of spread and as a results fires are more intense. However, crown fires do not always occur during high winds. Wind causes the angle of the flame to become more acute and with increased wind velocities the flames are often forced into the unburnt vegetation. That being so, the flame height does not necessarily increase with wind speed (Brown and Davis 1973; Luke and McArthur 1978; Booysen and Tainton 1984).

3.4.5. Fire intensity

Fires in renosterveld are not very intense although their intensities range widely. Williams *et al*. (1998) and Yates *et al*. (2008) have shown that in North Australia fires occurring in the early dry summers are much smaller and less intense than those burning later in the dry season. Namaqualand Granite Renosterveld does not follow the same pattern, as small fires occur during the peak of the fire season and has a higher intensity than the larger fires. In comparison to grass fuelled fire regime such as grasslands, savannas, thicket, forest and arid shrubland vegetation; renosterveld burn at a much higher intensity (Van Wilgen and van Hensbergen 1992). As they have more fires and a different burning season. In South African vegetation, the fire size distribution and fire intensity remains stable throughout the fire season, except forests which shows a declining trend in fire intensity (Archibald *et al*. 2010). In the Kamiesberg fires are mostly driven due to *E. rhinocerotis* and to a lesser extent *Euryops lateriflorus*. Fires can burn even if weather conditions are not conducive for burning because the above species are woody perennial shrubs and can form dense strands. Infrequent fires display a wide range of fire intensities (Archibald *et al*. 2013). These infrequent fires are associated with intense crown fire events. When fire is frequent, there is insufficient time to accumulate the fuel needed for very high-intensity fires, and these systems are climatically too flammable to produce creeping, low-intensity fires.

3.5. Conclusion

Fire is a fundamental driving force in the dynamics of renosterveld (Kraaij and Van Wilgen 2014). Analysing the fire regime of Namaqualand Granite Renosterveld provided an understanding of the fire behaviour and its ecological role. Human interferences have substantially contributed mostly to the total area burnt as they can change the seasonality, frequency, and timing of ignitions, changes in fuel load, fuel distribution and landscape patterns, which could become a conservation concern for fynbos (Kraaij and Van Wilgen 2014). Most of the human ignited fires occur in the fire season, which coincided with honey harvesting. Burning for wild honey is dangerous to the children running away from these fires. If too frequent fires, for honey harvesting, occur it could not only pose a threat to the bee population but small mammals, reptiles, grazing animals and farmers. Also keeping in mind that many of the perennial species are slow growing (Simons 2013) and frequent fires could be harmful to fire sensitive species and even supress the growth of others.

Renosterveld is thought to burn more frequently (FRI = 3-5 years) than fynbos (FRI = 10-25 years) due to faster plant growth and finer fuels (Rebelo *et al*. 2006). The FRI for Namaqualand Granite Renosterveld is 4.2 years, and a slight longer interval as expected due to heavy grazing that removes some fuel load. If fire is regularly required in renosterveld, then it can be expected that fire-stimulated recruitment would be a common feature of certain species, as it is in fynbos (Keeley *et al*. 2012; Curtis 2013). In Chapter 5, I will discuss the effects of short- interval fires on plant populations.

However, other studies on renosterveld in Bontebok National Park concluded a FRI of 6 – 7 years between 1972 and 2009, when prescribed burning and intensive grazing was applied (Kraaij 2010). In the Table Mountain National Park, no prescribed burning or intense grazing were applied, the mean FRI for the entire area between 1970 and 2007 was 22 years (Forsyth and Van Wilgen 2008). This increase in fire frequency in the Kamiesberg was primarily due to

an increase in human ignitions over time or the lack of fire as a management tool in those conservation areas (Forsyth and Van Wilgen 2008).

Climate change projections suggest that the CFR will become much hotter and drier, which could result in an increased fire frequency under these changes (Kraaij and Van Wilgen 2014). Similar predictions have been made for the Kamiesberg, with increased summer rainfall projections (Helme and Desmet 2006). I found that changes in climate has not had a major effect of the fire regime of Namaqualand Granite Renosterveld over the past 15 years. However, it is unclear whether future changes in climate will have an impact or be a cause of concern on the fire regime. Increased human populations, their land use practises and climate changes are all factors which will change the fire regime over time (Bowman *et al*. 2009; Flannigan *et al*. 2009), however the manner in which these interacting factors will change the fire regime in the future is unclear and thus would need regular re-evaluation as to inform the management of these rangelands.

4. Chapter four: The effects of fire and grazing on the dynamics of *Elytropappus rhinocerotis* (renosterbos) in Namaqualand Granite Renosterveld

4.1. Introduction

Plant-plant interactions play an important role in regulating vegetation dynamics as they influence the composition of plant communities, effects resource availability, and vegetation structure (Brooks 2006). External drivers such as climate, nutrient availability and even anthropogenic effects can alter the impacts of plant-plant interactions (Brooks 2006).

Climatic changes are expected to elicit considerable changes in vegetation structure and function and can affect the plant species composition, abundance and their function and distribution (Mouillot *et al*. 2002; Capitanio and Carcaillet 2008). These changes include carbon dioxide (CO_2) affecting both plant productivity and the water use efficiency (Tietjen and Jeltsch 2007). The rising atmospheric CO_2 levels could change species composition in favour of woody shrubs, with negative implications for nutritive values and may alter conditions for livestock productions (Svenning and Sandel 2013). In semi-arid regions pastoral systems such as livestock production and cereal cultivation are often painted as the main cause of land degradation. However, altered climate might have a greater influence on the vegetation composition as these plant communities have a higher capacity to recover from a grazing disturbance (Allsopp 1999). The dynamics of the relationship between climate, soil and vegetation is influenced by physiological characteristics of the ecosystem (Rodriquez-Iturbe 2000).

Fire events has positive effects on vegetation dynamics, as it can alter the structure and promote species richness (Jing *et al*. 2017). Burning removes the litter which then enhances the availability of light for a few competitive species (Jing *et al*. 2017). Livestock grazing can have a similar effect as fire for instance grazing animals can affect productivity and biodiversity by

removing certain species. Grazing can also change the availability soil resources and overall heterogeneity of plant communities (Jing *et al*. 2017). The time scale of vegetation change is difficult to estimate due to unpredictable climate changes (including low rainfall), event driven dynamic behaviour, and complex interactions between species (Wiegand and Milton 1996). Moreover, sudden changes in the socio-economic circumstances of people who live off the land make the rate, magnitude, and direction of vegetation change unpredictable (Woodward and Beerling 1997).

Elytropappus rhinocerotis commonly known as renosterbos, is the dominant plant species occurring in renosterveld (Kraaij and Van Wilgen 2014). *Elytropappus rhinocerotis* is not always present in renosterveld and therefore cannot be considered as an indicator of renosterveld (Krug 2004). The vegetation is named after this indigenous shrub which is abundant on road verges and reaches high densities on disturbed areas (Bergh 2006). *Elytropappus rhinocerotis* is widespread throughout the Cape Floristic Region (CFR) and occurs throughout Namaqualand as far north as the Richtersveld. And it is also found in the Eastern Cape Province (Bergh 2006). Therefore, the distribution of *E. rhinocerotis* would suggest that the plant requires more moisture than Succulent Karoo plant species but less than fynbos plant species (Cowling and Holmes 1992). As a result of this, *E. rhinocerotis* often occur on ecotones between Fynbos, Succulent Karoo and Grassland communities (Bergh *et al*. 2007; Allsopp *et al*. 2014).

Elytropappus rhinocerotis is about 2 m in height and has a smooth grey bark. The fine white hairs between the leaves gives the plant a woolly and distinct grey appearance (Le Roux 2015). Even though renosterveld has been severely impacted by agriculture and is critically endangered, *E. rhinocerotis* it is not a conservation concern yet due to the woody nature (Bergh 2006). In humid regions *E. rhinocerotis* is susceptible to fungal infestations, but overall it is tolerant to both snow and frost and can adapt to a wide range of conditions (Levyns 1926;

Levyns 1956). *Elytropappus rhinocerotis* is confined to fine-grained soils such as shale or granite in high rainfall regions but in arid regions soil type has less of an effect (Levyns 1956). Fire can remove *E. rhinocerotis* and acts as a stimulus for seed germination (Bergh 2006). The seeds are shed in winter and become dormant and will germinate the following year. During the early stages the environment has a great influence on seedling growth, as even a short drought period can kill seedlings (Bergh 2006). The seedlings of *E. rhinocerotis* seedlings are intolerant to shade therefore it will not reproduce and has to be disturbed in close communities (Walton 2006). However, in arid regions where the vegetation in more open, *E. rhinocerotis* seedlings are able to establish themselves as shade no longer acts as an inhibiting factor (Levyns 1956).

It is believed that renosterveld, especially in the east, was once a grassland with an invasion of *E. rhinocerotis* (Keeley *et al*. 2012). In the late 1700s a decline in grassland species and the increase in of *E. rhinocerotis* has been observed (Cowling *et al*. 1986). This interaction was primarily a result of changes in the disturbance regime, especially fire and grazing (Cowling e*t al*. 1986). Early settlers use to frequently burn renosterveld to increase palatable vegetation for their livestock (Parker and Lomba 2009). The immediate post-fire grazing by livestock was a harmful practice to the grass sward, which ultimately led to its ruin and shrubs becoming greater in abundance (Cowling *et al*. 1986). During the 19th and early 20th centuries, these practises led to the dominance of *E. rhinocerotis* in renosterveld (Krug 2004; Van der Merwe and Van Rooyen 2011).

Shrub encroachment has resulted in the loss of available fodder for livestock grazing, species richness decline and loss of biodiversity (Tedder *et al.* 2014). The invasions of indigenous species are usually episodic, resulting in even-aged stands of the species concerned, and are often associated with a local disturbance, increases in available nutrients (especially nitrogen and phosphorus), or a change in water supply or climatic conditions (Davis *et al*. 2000; Bond

2008). Encroachment can be successional process thus taking place over centuries (Archer 1989). Factors such as fire, rainfall, grazing and browsing has been considered as the main cause of encroachment. Whilst these factors have been studied individually (Trollope 1984; O`Conner 1995) their relative combined effects are poorly understood and the influence of the density dependence has not been integrated into studies of encroachment.

The encroachment of *E. rhinocerotis* has been identified as an emerging problem since the nineteenth century (Beinart 2003). *Elytropappus rhinocerotis* encroachment can be viewed as an indigenous encroacher in renosterveld vegetation. It stores seeds in the soil and germination is stimulated by fire (Levyns 1929). The fruit and feathery pappus dispersed by wind over long distances. The hard, woody tap root is able to penetrate down to 6 m and is drought resistant (Scott and van Breda 1937). These biological attributes are efficient for its successful encroachment in renosterveld (Cowling *et al*. 1986). It is unpalatable and livestock will only graze on it when little else is available.

The precise effects of *E. rhinocerotis* on the plant community is unknown. It is difficult to develop an effective plan to control the spread of the species as not much research has been done on the effects of fire in Mountain Renosterveld including the role of *E. rhinocerotis*. The objective of this chapter was to focus on the ecological dynamics of *Elytropappus rhinocerotis* following unplanned veld fires in Namaqualand Granite Renosterveld.

The following questions are addressed in this chapter:

4.1. How does the fire intensity (as Fire Radiative Power) affect the density and cover of *Elytropappus rhinocerotis* overtime?

4.2. How does stocking density affect the density and cover of *Elytropappus rhinocerotis* in the burnt and unburnt vegetation?

4.3. Does the density and cover of *Elytropappus rhinocerotis* change overtime? And how will it compare to the unburnt vegetation?

4.4. Which species are associated with *Elytropappus rhinocerotis* and does it change the dynamics of Namaqualand Granite Renosterveld.

4.2.Methods

4.2.1. Study area

The study area was in the vicinity of the Leliefontein village common in the Kamiesberg Uplands, Namaqualand. Refer to Chapter two for a detailed description of the study area. The study sites are all located on Namaqualand Granite Renosterveld. These study sites are subjected to variant physical, environmental and grazing regimes.

4.2.2. Data collection

The study describes the dynamics of *E. rhinocerotis* succession over 15 years within 10 burnt and 10 adjacent unburnt Namaqualand Granite Renosterveld sites with differing histories of fire and grazing. The unburnt sites act as the 'control' of the study.

4.2.2.1.Fire Radiative Power (mW) data collection

Fire Radiative Power (FRP) is an index of fire line intensity. Fire line intensity is a measure of the rate of energy released from a fire per unit length of the burning front (measured in kw/m/s) (Williams *et al*. 1999; Wooster *et al*. 2003). Energy released during wildfires are often lost through processes other than radiation (e.g. convection, conduction and vaporisation), but wildfire radiative power emissions are intense and suitable for assessment via satellite earth

observations (Wooster *et al*. 2003). Historical data on fire occurrences in the Kamiesberg Uplands were obtained from NASA's fire map archive, http://rapidfire.sci.gsfc.nasa.gov/ (Please refer to Chapter 3 for a detailed description).

4.2.2.2.Stocking densities (SSU / 10ha) data collection

Samuels (2013) calculated the current and historical stocking densities using monthly and annual livestock numbers and he mapped the grazing pressure across the Leliefontein rangeland. The average grazing densities from 1998 to 2006 were calculated by Samuels (2013) and used for this study.

4.2.2.3.Density, distance and radius data collection

Field sampling commenced from March to September in 2014 and September 2015. The density of *E. rhinocerotis* was collected by placing a random 100 m^2 quadrats (Fig. 4.1) at each site and the total number of *E. rhinocerotis* individuals were recorded. (Guo 2001). This method was repeated thrice at each burnt and unburnt site. Density was determined by quantifying the number of individuals of a species per unit area. The density *E. rhinocerotis* was calculated as:

$$Density\ (D_i) = \frac{A_i}{Total\ sampled\ area\ (m^2)}$$

Where Abundance (A_i) = total number of individuals of species *i*.

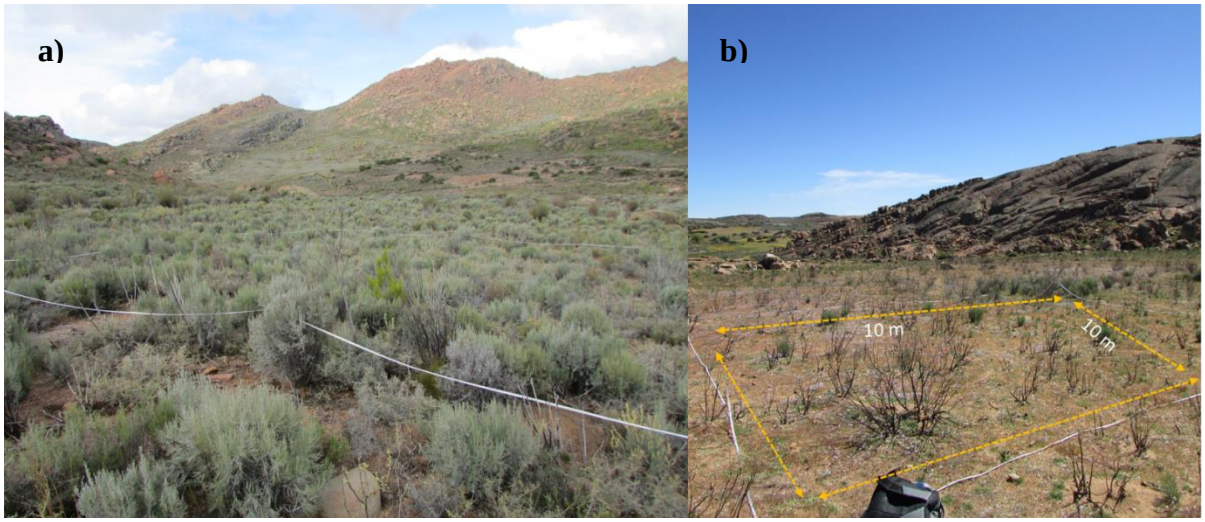


Figure 4.1: Density collection of E. rhinocerotis at an old (a) site burnt in 2001 and young (b) site burnt in 2013 in Namaqualand Granite Renosterveld.

To determine the distance between *E. rhinocerotis* individuals, the distance sampling method (Fig. 4.2) was used (Thomas *et al*. 2002). A 100 m line transect was randomly placed, at each burnt and adjacent unburnt site. At every one-meter interval the nearest *E. rhinocerotis* to the point was measured (Thomas *et al*. 2002). In addition to this, at each 10 m interval plants growing underneath or within a 1 m radius of E. *rhinocerotis* were identified, counted and recorded (Fig. 4.2).

Figure 4.2: Line-intercept used for collecting distance data and the sampling layout of each one-meter radius along this transect.

Vegetation cover was assessed using the descending point method (Roux 1963). According to the descending point method, a strike is recorded once any plant material intersected a one-meter interval of the line transect. In total four 50 m line transects (ca 200 m) were conducted at each site. These transects were not necessarily in a straight line but were in some cases adjacent to each other depending on the terrain and characteristics of the burn scar. For this study, the number of strikes per species was expressed as a percentage of the 200 points surveyed and these totals were added to determine the total percentage vegetation cover.

$$\%\ Cover\ (C_i)\ of\ Spp_i = \left(\frac{Total\ number\ of\ Spp_i}{Total\ points}\right) \times 100$$

Where cover (C_i) = Total % cover of species $_i$ and Spp_i = number of individuals of species $_i$

The physical attributes (slope, aspect, elevation, and rockiness) at each site and additional disturbances, which influences vegetation composition and conditions in the region, were also recorded. Global Positioning System (GPS) co-ordinates at each site were also recorded.

4.2.3. Data analysis

Normality testing was done on the dataset in the burnt and unburnt sites using the Shapiro Wilk's W test (Royston 1982). Correlations were done to test the effect of a) FRP on *E. rhinocerotis* density and cover and b) stocking density on *E. rhinocerotis* density and cover. General regression models were used to test whether the density and cover of *E. rhinocerotis* changed over time. Regressions were also used to determine a difference in the density and cover of *E. rhinocerotis* in burnt and unburnt Namaqualand Granite Renosterveld sites. A Mann-Whitney U test was performed for the data that was not normally distributed to compare the differences between the mean distance of the nearest *E. rhinocerotis* individuals in the burnt and unburnt sites. Principal Component Analysis (PCA) were used to determine the relationship between *E. rhinocerotis* and other species in Namaqualand Granite Renosterveld. All the statistical analyses were performed in STATISTICA (StatSoft, Inc., Tulsa, OK, version 8, 2007).

4.3.Results

4.3.1. How does the fire intensity (as Fire Radiative Power) affect the density and cover of Elytropappus rhinocerotis over time?

The correlation between the FRP on *E. rhinocerotis* densities and cover was r = -0.47, n = 10 and r = -0.52, n = 10, respectively, which indicate a moderate negative relationship. Overall the FRP does not have a significant effect on the density (p = 0.169) and cover (p = 0.116) of

E. rhinocerotis in the burnt vegetation. This can be explained as the density of *E. rhinocerotis* decreases the FRP increases. The same trend is observed for the *E. rhinocerotis* cover and FRP.

4.3.2. How does stocking density affect the density and cover of Elytropappus rhinocerotis in the burnt and unburnt vegetation?

From Fig. 4.3. A, it can be seen that the density of *E. rhinocerotis* decrease as the stocking density increase in the burnt sites. Whereas, the density of *E. rhinocerotis* remains constant across the different stocking densities. The correlation between the stocking density and the density and cover of *E. rhinocerotis* in the burnt (r = -0.33; p = 0.35; n = 10) and unburnt (r =0.39; p = 0.25; n = 10) sites were not significant.

From Fig. 4.3. B, it can be seen that the cover of *E. rhinocerotis* is higher in the unburnt sites than in the burnt sites across the different stocking densities. The trend line remains flat for the cover of *E. rhinocerotis* in the burnt sites. In the unburnt sites the trend line shows an increase in the cover with the different stocking densities. The correlations show that stocking densities is not statistically associated with the cover of *E. rhinocerotis* in the burnt (r = 0.20; p =0.57; n = 10) and unburnt (r =0.34; p =0.33; n = 10) sites.

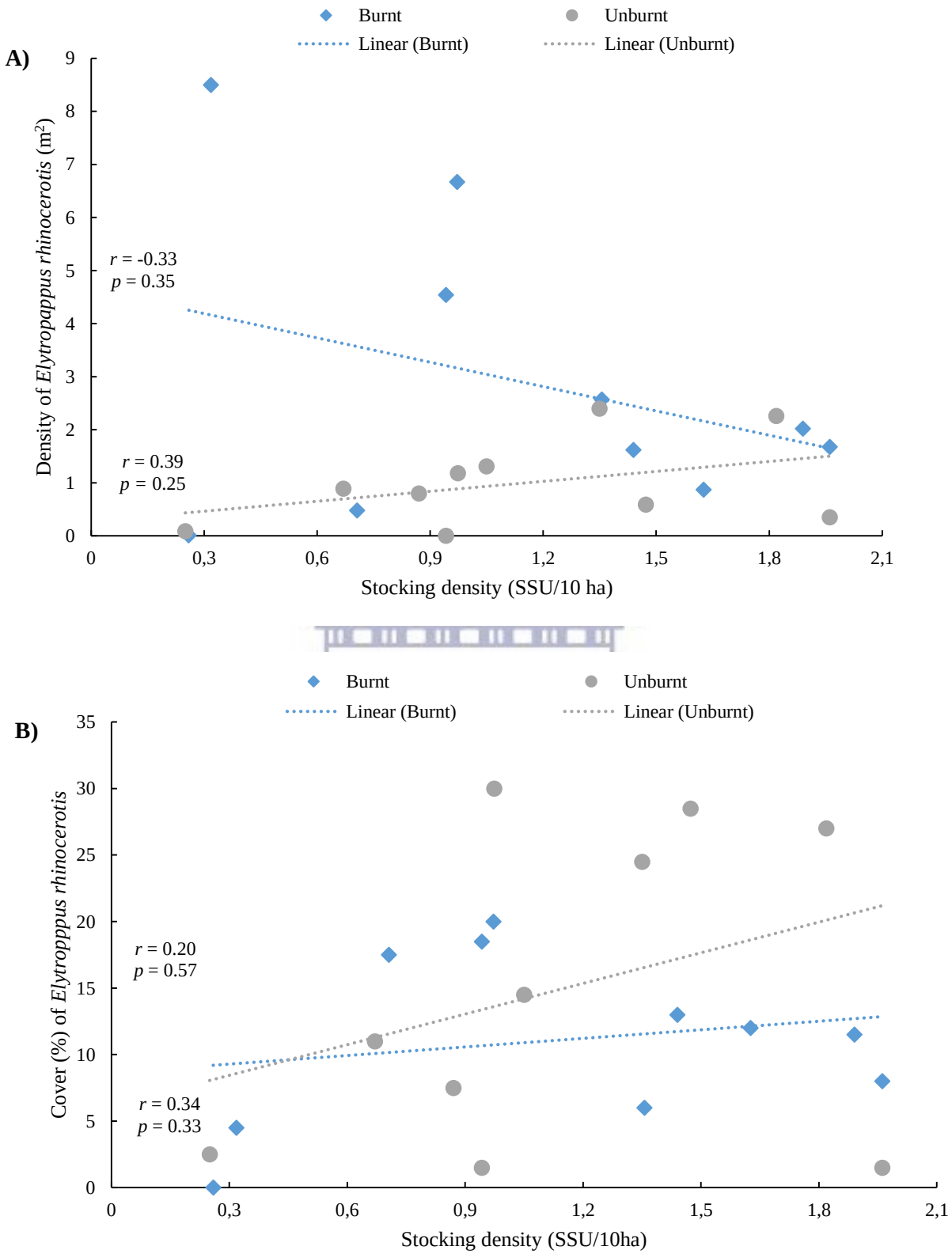


Figure 4.3: Pearson correlation showing the effect of stocking density (SSU/10 ha) on the a) density and b) cover of *Elytropappus rhinocerotis* in burnt and unburnt renosterveld vegetation.

4.3.3. Does the density and cover of Elytropappus rhinocerotis change over time? And how will it compare to the unburnt vegetation?

The highest densities of *E. rhinocerotis* were 8.5 m^2 and 6.67 m^2 and were recorded at two and 14 years post-fire sites, respectively (Fig.4.4. a). The lowest density was 0.01 m^2 recorded after 15 years post-fire. From the regression it can be concluded that the density of *E. rhinocerotis* does not change overtime ($F_{1, 8}$= 0.70; p = 0.43) with a negative and weak regression of R^2 = 0.08. The can be due to the high variability in *E. rhinocerotis* densities found in the different sites. The difference in *E. rhinocerotis* densities also did not change significantly (p = 0.64) overtime.

The cover of *E. rhinocerotis* gradually increased from 6 % to 20 % after 1 to 14 years post-fire, in that order. For most of the burnt site the % cover of *E. rhinocerotis* was > 10 % (Fig. 4.4 b). The % cover of *E. rhinocerotis* did not change significantly overtime ($F_{1,8}$ = 0.15; p = 0.70) with a weak regression of R^2 = 0.01. The cover difference in *E. rhinocerotis* did not change significantly (p = 0.70) overtime. It can be seen that the cover of *E. rhinocerotis* is higher in the burnt sites compared to the cover difference of *E. rhinocerotis*. Thus, this indicates that a few larger shrubs in the unburnt sites increases in % cover.

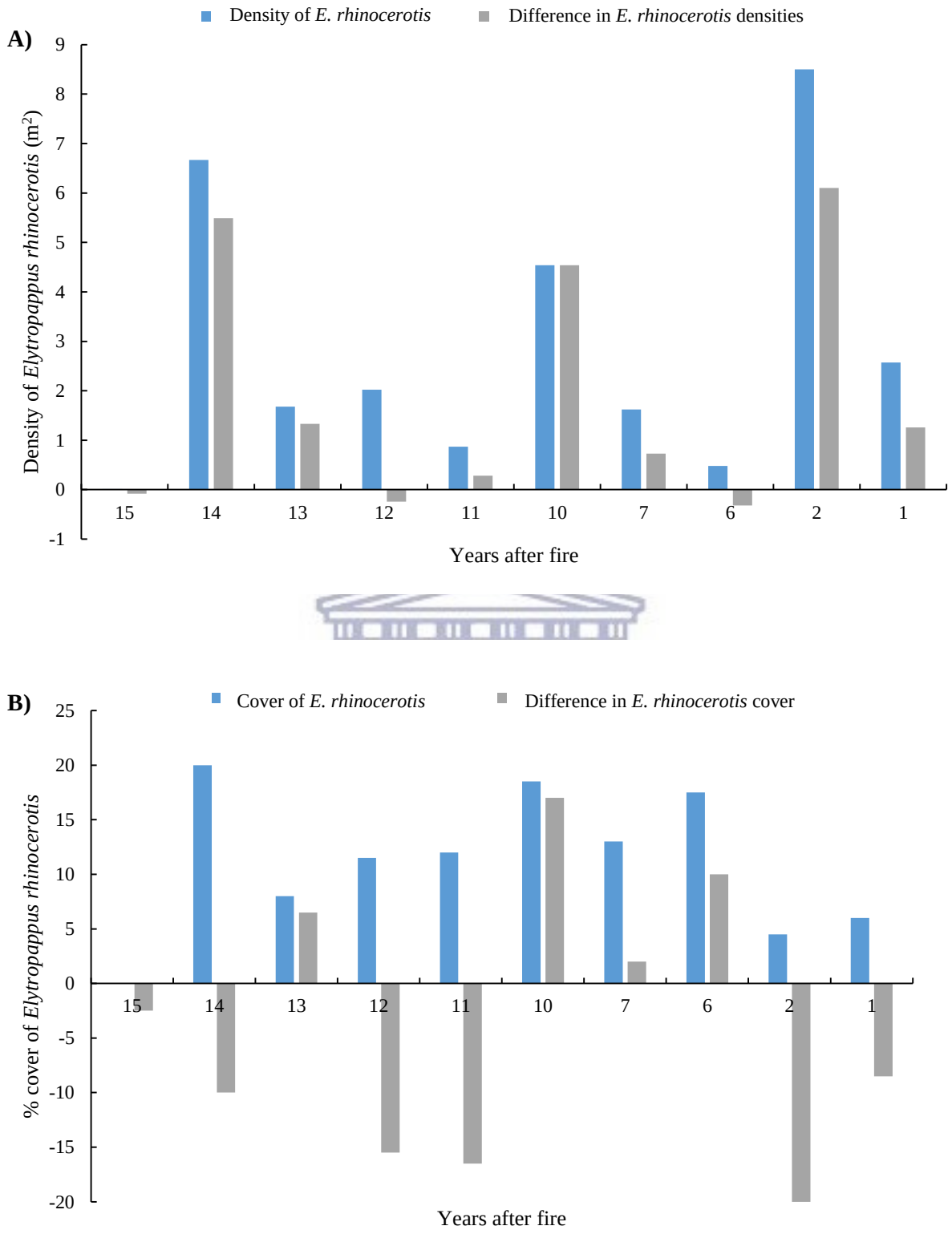


Figure 4.4: Post-fire change in (a density and cover (b of *Elytropappus rhinocerotis* and the in these variables in Namaqualand Granite Renosterveld overtime.

Table 4.1: Mean distance of nearest *Elytropappus rhinocerotis* individuals in Namaqualand Granite Renosterveld burnt and adjacent unburnt sites

Sites	Type	N	Mean	Mann-Whitney U	Z-score	P-value
Hoorngaat	Burnt	100	380.17	356.5	3.505	**0.000****
	Unburnt	100	158.94			
Skaaprivier	Burnt	100	10.245	3293.5	-4.169	**0.000****
	Unburnt	100	40.845			
Xharras 2	Burnt	100	24.93	2340.5	-6.4982	**0.000****
	Unburnt	100	102.63			
Die Vlak	Burnt	100	38.02	4302	-1.7054	0.088
	Unburnt	100	56.51			
Toringberg	Burnt	100	79.19	4415	1.4293	0.153
	Unburnt	100	77.53			
Groenkloof	Burnt	100	42.55	4778.5	-0.5412	0.588
	Unburnt	100	47.32			
Xharras 1	Burnt	100	65.55	3624	3.362	**0.000****
	Unburnt	100	45.04			
Kliphoek	Burnt	100	60.2	4889.5	-0.27	0.780
	Unburnt	100	51.76			
Populierskloof	Burnt	100	33.14	3142.5	4.5386	**0.000****
	Unburnt	100	23.07			
Xharras 3	Burnt	100	32.345	3899	-2.6902	**0.007***
	Unburnt	100	45.59			

Significance is shown at *p<.01, **p<.001

From Table 4.1, it can be observed that the average distance of *E. rhinocerotis* between individuals varies greatly across the different sites. This pattern is seen between burnt and adjacent unburnt sites. The results indicated significant differences in the mean distance between *E. rhinocerotis* individuals in the burnt and unburnt. It can also be seen that half of the significant sites, the mean distances were higher in the burnt sites whereas the other half in the unburnt sites. These sites are namely Hoorngaat (U = 3565.5; p = 0.001), Skaaprivier (U = 3293.5; p = 0.001), Xharras 2 (U = 2340.5; p = 0.001), Xharras 1 (U = 3624; p = 0.001), Populierskloof (U = 3142.5; p = 0.001) and Xharras 3 (U = 3899; p = 0.001).

4.3.4. Which species are associated with Elytropappus rhinocerotis and does it change the dynamics of Namaqualand Granite Renosterveld?

The principal component analysis (PCA) (Fig 4.5) depicts the relationship between the density and cover of *E. rhinocerotis* and various variables across the different burnt and unburnt sites. Axis 1 and 2 from the PCA explains 54.73 % of the variation in the data set. The Pearson correlation matrix (not shown) showed only two significant (with an alpha = 0.05) correlations; between the densities of *E. rhinocerotis* and unpalatable species (r =0.73), and unpalatable species and decreasers (r =0.67). From the PCA it can be seen that there is a relationship between the cover of *E. rhinocerotis* and the stocking densities. Negative relationships are seen between *E. rhinocerotis* at Hoorngaat in the burnt sites (depicted by B0) and between stocking densities and unpalatable species.

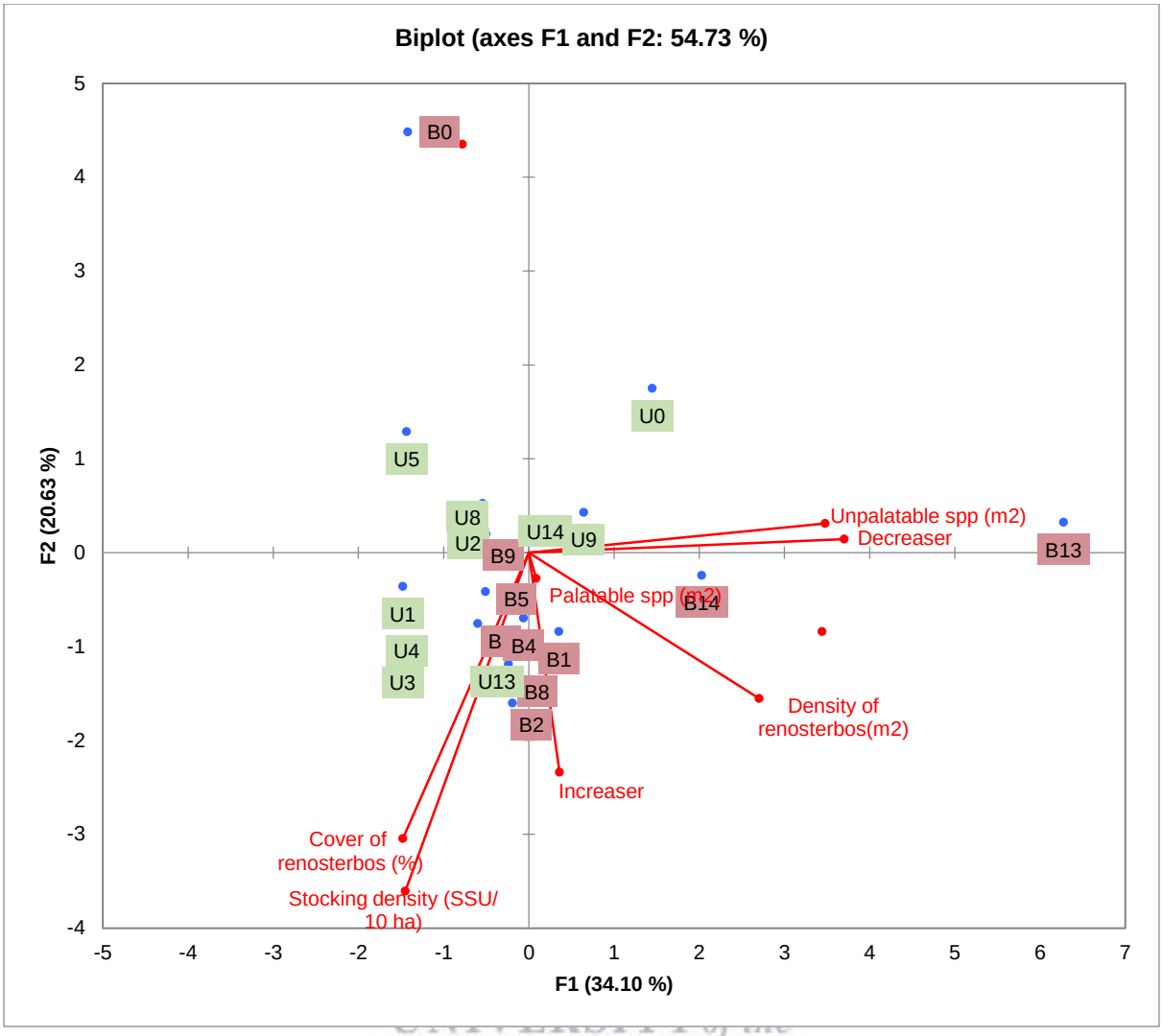


Figure 4.5: PCA depicting the various burnt and unburnt vegetation from 2000 – 2015 and the effect of various parameters on the density and cover of *Elytropappus rhinocerotis*. B = burnt sites and U = unburnt sites; the number indicates the year of the fire event.

Axis 1 and 2 explains 76.68 % of the variability in the vegetation parameters measured during this study. This PCA (Fig. 4.6) depicts the post-fire succession (over 15 years) of plant species in Namaqualand Granite Renosterveld burnt sites and their association with the dominant *E. rhinocerotis*. Species differ in their palatability (F1) and life form (F2). Annual species are fast growing and they generally tend to be present during the early (0-2 years post-fire) successional stage (shown as ES on PCA). These pioneer species are highly palatable and is not associated with *E. rhinocerotis*. The later (3-6 years post-fire) successional stage (LS) is dominated by palatable perennial species which are not associated with *E. rhinocerotis*. These sites tend to

have a moderate slopes and rock cover. The mature stage (M) is cluster with species which are closely associated with *E. rhinocerotis* and facilitates in each other's growth. This stage (> 7 years post-fire) is mostly dominated by unpalatable species, hence the dominance of *E. rhinocerotis* during this stage. In Namaqualand Granite Renosterveld the mature stage has a high grazing intensity.

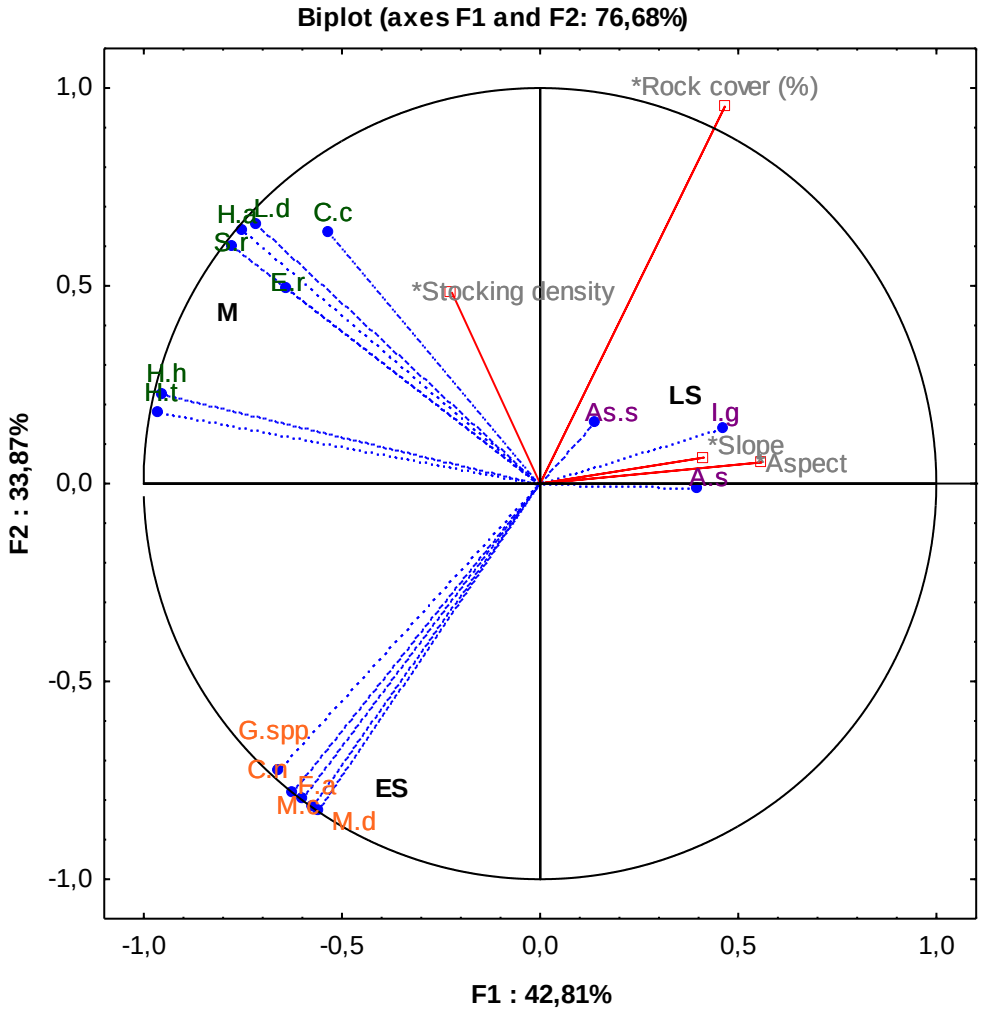


Figure 4.6: Principal component analysis bi-plot showing the association of *Elytropappus rhinocerotis* with species palatability (F1) and species life form (F2) over time in Namaqualand Granite Renosterveld. Species depicted in peach illustrates the Early successional stage *(*G.spp= grass spp*;* C.n= *Crassula natans;* F.a= *Felicia australis;* M.c= *Mesembryanthemum crystallinum;* M.d= *Massoia depressa),* while those in purple represents the Later successional stage *(*As.s= *Aspalathus spinosa;* I.g= *Ischyrolepis gossypina;* A.s= *Anthospermum spathulatum)* and the Mature stage *(*C.c= *Chrysocoma ciliata*; L.d= *Leobordea digitata;* H.a= *Hermannia amoena;* S.r= *Stachys rugosa;* E.r= *Elytropappus rhinocerotis;* H.h= *Helichrysum hamulosum;* H.t= *Helichrysum tinctum)* is shown in green.

4.4.Discussion

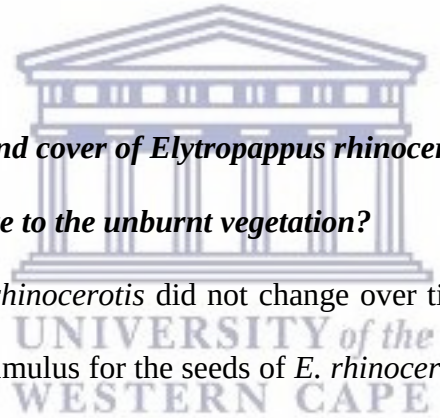
4.4.1. How does the fire intensity (as Fire Radiative Power) affect the density and cover of Elytropappus rhinocerotis in Namaqualand Granite Renosterveld?

In this study the FRP did not affect the density and cover of *E. rhinocerotis* in Namaqualand Granite Renosterveld**.** *Elytropappus rhinocerotis* occurred in low densities in areas where the FRP was > 100 MW. This was an interesting finding as these cool fires reduced *E. rhinocerotis* but it did not kill the shrub. When the FRP was very cool (< 60 MW) the cover remained high, because these fires were not hot enough to completely kill *E. rhinocerotis*. These sites with a low FRP are less intense, as these fires normally occur during the early dry season and as a result fires are smaller than those which occur later during the dry season and are more intense and larger fires (Yates *et al*. 2008 and Williams *et al*. 1998). At Xharras 3 the fire was hot enough to kill *E. rhinocerotis* as only a few burnt branches remained (personal observation). Furthermore, this fire continued for five days and burnt 922 ha (see Chapter 3 for further discussion) but by the time sampling begun a few seedlings had emerged, hence the low density and cover recorded. Fires with higher FRPs might burn larger parts of the landscape, as they are likely to be extinguished by night –time weather conditions, moist fuels, or topographical barriers. However, the results showed that a high FRP tend to burn small areas. As human land uses increase so does the FRP decrease (Archibald *et al*. 2010). These findings concur with Levyns (1926) in which she concluded that *E. rhinocerotis* rapidly colonise disturbed areas. Furthermore, fire not only kills *E. rhinocerotis* but it also allows the species to spread further in disturbed areas (Levyns 1956). Once the tap root has been developed it allows *E. rhinocerotis* to access ground water as it is about 6 m deep and this allows the species to grow and survive in this semi-arid rangeland (Scott and Van Breda 1938). Archibald *et al*. (2010) found that for most of Southern African vegetation types the fire size and intensity remains constant throughout the fire season, with forests even showing a decreasing trend in fire

intensity (Archibald *et al*. 2010). In the Kamiesberg the FRP is distributed very heterogeneous within the study area. In the arid savannas of the Eastern Cape, fire intensities ranging from cool (925 kJ s^{-1} m^{-1}) to extremely intense (3, 326 kJ s^{-1} m^{-1}) had not effect on the grass sward recovery in the first and second growing season after fires (Trollope 2007). In the Kamiesberg the effect of fire intensity on *E. rhinocerotis* in Namaqualand Granite Renosterveld is complex as each site has a specific Environmental variation associated with topography (aspect, slope and steepness) which aid in these fires (Sherriff and Veblen 2006). Because the FRP did not significantly affect the density and cover of *E. rhinocerotis* for this study. I would assume factors such as microhabitats, physical conditions, and grazing history and rates would explain the differences found.

Microhabitats are specialised communities on specific substrates (Curtis 2013) which occur across the study sites. These microhabitats exist because of the structural changes at each post-fire site which influences the microclimate and the distribution of resources such as nutrients and moisture (Higgins *et al*. 2007). These changes in turn have cascading effects on biodiversity with some species responding to microclimate and resource availability as shown by Higgins *et al*. (2007). Each site has its own microclimate and physical condition which either affects *E. rhinocerotis* positively or negatively. Slope is one of the topographical features which affects the distribution of species, vegetation composition and structure, and their microclimates (Bennie *et al*. 2008). This is due to the variation in incoming solar radiation (Paudel and Vetaas 2014). A study done by Scherrer and Körner (2011) showed that topography can have an influence in the different temperature over a short horizontal distances in mountainous regions. Thus even though slope orientation was not tested for this study, it was evident that different slopes had some influence on the abundance of *E. rhinocerotis*. The abundance of *E. rhinocerotis* was higher at gentle to moderate slopes (such as Skaaprivier and Groenkloof) whereas at Hoorngaat which has a steep slope hardly any *E. rhinocerotis* shrubs

were recorded. In higher mountainous areas in the Kamiesberg *E. rhinocerotis* did not occur. This could be due to grazing as Anderson *et al*. (2010) found that steep rocky uplands are grazed on less because they are inaccessible. Furthermore, at Kliphoek *E. rhinocerotis* occurred in low density but this was a moderate slope. Again, grazing is the reason for these low densities as this study site has a low stocking rate. The cover at Kliphoek was reasonably high cover, therefore this finding showed that *E. rhinocerotis* has a dense canopy cover providing shade to the understory species. As a result, at sites with low stocking rates it would be expected that *E. rhinocerotis* is less abundant because it would require some degree of disturbance to thrive as shown by Levyns (1926). These findings result due to the environmental range of each site and the effect various factors has on these sites (Anderson *et al*. 2010).



4.4.2. Does the density and cover of Elytropappus rhinocerotis change over time? And how will it compare to the unburnt vegetation?

The density and cover of *E. rhinocerotis* did not change over time in Namaqualand Granite Renosterveld. Fire acts as a stimulus for the seeds of *E. rhinocerotis* to germinate as found by Levyns (1926). In the burnt sites more individuals of *E. rhinocerotis* were recorded and the cover was higher at these burnt sites which indicates that larger shrubs are found at these sites. *Elytropappus rhinocerotis* seedlings rapidly colonise disturbed area as the highest density was recorded during the second post-fire year. This result is in accord with Levyns (1926) findings that *E. rhinocerotis* seedlings tend to be higher in post-burn area. The low density of *E. rhinocerotis* in the first year post-fire is expected as seeds only germinate after one year. In the Kamiesberg shade is not an inhibiting factor for *E. rhinocerotis* seedlings and they are able to establish themselves between the bushes.

Fire not only maintains vegetation diversity but also increases spatial heterogeneity (Van der Merwe and Van Rooyen 2011). This is seen from the results as the density of *E. rhinocerotis* varies across the post-fire study sites. Grazing by livestock further allows the species to spread (Levyns 1956). At Populierskloof, the stocking density is below the recommended rate but the highest density of *E. rhinocerotis* occurs at this site. This is followed by Skaaprivier and Groenkloof which are overstocked. In addition to grazing, the reason for the high abundance of *E. rhinocerotis* is also due to the fact that these sites have moderate slopes, as discussed previously. The results depict a successional decline in *E. rhinocerotis* over time in older and mature renosterveld communities. At Hoorngaat (15 years post-fire) and Kliphoek (six years post-fire) only a few individuals were recorded. These sites have relatively low grazing intensity and as a result the populations of *E. rhinocerotis* die off and creates new available space and a release in competitive pressure (Todd and Hoffman 1999). *Elytropappus rhinocerotis* can adapt to a wide range of conditions (Levyns 1956); however, in relatively undisturbed areas such as Hoorngaat and Kliphoek, the plant cannot survive (Levyns 1926; Levyns 1929). Since these sites are not heavily disturbed by grazing and they occur on steep rocky uplands. In order for *E. rhinocerotis* to survive and occur in high abundance at these sites, it will have to be disturbed in some way. Levyns (1929) found that *E. rhinocerotis* will not be able to reproduce in undisturbed areas and will give way to other species overtime. Open spaces create opportunity for ephemeral species which are characteristically smaller, short-lived and has a higher reproductive response, giving them the competitive advantage to colonise open spaces (Grime 2001). For this study, opportunities for species such as *Eriocephalus microphyllus*, *Ischryrolepis grossypina*, *Osteospermum* spp. and *Ruschia* spp. were created and they were able to outcompete *E. rhinocerotis*. The stocking densities varied across the study sites but it has no significant influence on the density of *E. rhinocerotis*. The seedlings of *E. rhinocerotis* does not decrease in quantity in the unburnt vegetation and their

abundance remains relatively high. This is also due to the grazing at these sites which allow *E. rhinocerotis* individuals to persist in this landscape. Heavy grazing often favours unpalatable species in area and reduces the population of various woody and succulent shrubs in the Kamiesberg (Anderson and Hoffman 2007). However, at heavily grazed sites such as Xharras 1, Toringberg, Die Vlak and Xharras 2 *E. rhinocerotis* did not occur in high densities but the cover shows that the canopy of *E. rhinocerotis* is relatively dense at these sites. Heavy grazing in the Kamiesberg is very patchy as certain areas are more heavily grazed than others due to the distribution of water points and stock posts (Hoffman and Todd 2000). The effect of fire on *E. rhinocerotis* is dependent upon the environmental and physical conditions at each site. Therefore, long term effects of fire and changes in vegetation succession is also largely dependent on biotic and abiotic factors.

The results showed significant differences between the nearest *E. rhinocerotis* shrubs in the recently burnt sites. This result was expected as seedlings have not yet matured properly, hence the gap between shrubs. In semi-arid regions grazing is an important driver of plant interactions (Howard *et al*. 2012). Thus, this explains the differences in distances between shrubs for the other burnt and unburnt sites (such as Xharras 1, Xharras 2, Skaaprivier, and Hoorngaat). The fact that *E. rhinocerotis* shrubs are far apart could be due to species recolonizing in these open patches which were created by grazing as shown by Howard *et al*. 2012. Similarly, Moinde (1998) found that trampling and herbivory reduced the size of clumps and increased the inter – shrub space. This in turn increased the proportion of bare ground which ultimately created opportunities for the seeds of *Galenia africana* to recruit. In addition, competition from other species could also be a reason. A study at Tierberg on nearest-neighbour data (Esler and Cowling 1993) found that competitive interactions between *Pteronia* species, grown under nursery conditions, varied with soil pH and calcium conditions. The interaction between non-succulent *Pteronia pallens* dominated under moist condition, but was outcompeted by

succulent shrubs seedlings (*Ruschia spinosa*) at low moisture level (Milton *et al*. 2003). Competition for water within shrub communities may determine spacing of various perennial plants (Milton *et al*. 2003). Understanding the root distribution and competition of plant species under various conditions is important to comprehend the value of plant species as indicators for distinguished lands of grazing value only or as crop production (Scott and van Breda 1936). It is evident from the results that *E. rhinocerotis* show a strong nurse plant effect. This result is similar to Mooney and Hobbs (2000) which illustrated that plant facilitation are dominant processes in semi-arid shrublands.

4.4.3. Which species are associated with Elytropappus rhinocerotis and does it change the dynamics of Namaqualand Granite Renosterveld?

In this study the abundance of *Elytropappus rhinocerotis* is extremely variable at each renosterveld community. The reason for this is the canopy dominance of *E. rhinocerotis* can vary from one fire cycle to the next which means that the competitive dominance is also variable as described by Vlok and Yeaton (1999). The results showed that in the absence of *E. rhinocerotis* perennial species richness are high (see appendix 1). However, this trend declines overtime once the cover of E. rhinocerotis increases. Initially in the early post-fire successional stage a 'window period' is created, in which understory species may be able to establish (Vlok and Yeaton 2000a). Fast growing annual species often decline over this successional gradient due to woody, resprouting, perennial species (Tilman 1982). The shrub cover of *E. rhinocerotis* could potentially have created mosaics of fertile patches across the landscape. And during Scott and van Breda (1936) study, farmers have indicated that the dominance of *E. rhinocerotis* is an indication of fertile soil. The fact that *E. rhinocerotis* can penetrates the soil to a depth of 6 m (Scott and van Breda 1936), aerates the soil and brings up solutes from the lower soil layers supported this theory. Thus, deep-rooted perennials have the ability to outcompete colonizers

of open ground (Milton *et al*. 2003). During the later succession stage in renosterveld (3- 6 years post-fire) woody palatable species such as *Anthospermum spathulatum*, *Aspalathus spinosa* and *Ischyrolepis grossypina* tend to dominate. As a result, of the physical attributes of each site and grazing intensities as shown by Anderson *et al*. (2010). At sites which are lightly grazed palatable shrub are able to increase in abundance as described by Walton (2006). Once the grazing intensity increases in renosterveld, an increase in *E. rhinocerotis* and various woody perennials, especially unpalatable species will emerge. The relative importance of *E. rhinocerotis* may be seen during the mid-post-fire successional stage as these woody perennials increase in abundance and becomes dominant in these sites. However, the abundance changes along the successional gradient, as the density tend to decline while the cover steadily increases (Guo 2001). The long term effects of fire favours the increase in the cover of unpalatable shrubs such as *E. rhinocerotis, Crysocoma ciliata*, and *Dodonaea angustifolia*. *Elytropappus rhinocerotis* has a very low forage value and is not desired by livestock (Kotze *et al*. 2010). Todd and Hoffman (1999) found that heavy grazing pressures in Leliefontein have contributed to the decline of indigenous perennial grasses directly and as a result of this practise the fuel load was also indirectly reduced.

The results showed that species are able to establish in the absence of *E. rhinocerotis*. This is similar to Mountain Fynbos in which proteas created a gap for species to establish. Moreover, proteas have the ability to supress graminoids which are vigorous competitors (Vlok and Yeaton 1999; 2000b). Thus, as Curtis (2013) asked the question "could *Elytropappus* be the 'protea' of renosterveld?" The results show that *E. rhinocerotis* could be the 'protea' of renosterveld. The reason for this is, *E. rhinocerotis* is one of the few shrubs which can outcompete grasses and in this way shades out vigorous competitors. In addition, in the mature renosterveld, *E. rhinocerotis* showed strong associations with annual and perennial palatable species such as *Helichrysum tinctum, Helichrysum hamulosum*, and *Leobordea digitata*.

Elytropappus rhinocerotis can increase the species richness of understory species and maintain species composition in Namaqualand Granite Renosterveld. This result is similar to Vlok and Yeaton (1999) study on the importance of proteas. They concluded that overstory protea has an important role in maintaining species richness in fynbos. *Elytropappus rhinocerotis* seems to have a facilitative role as a nurse-plant or refuge site protecting graminoids, geophytes, forbes and dwarf shrubs from abiotic conditions, exposure or grazing and trampling as explained by Walton (2006) and Allsopp *et al*. (2014). The idea of *E. rhinocerotis* having a role as a nurse-plant dates back to work done by Levyns (1929). Levyns (1929) regarded *E. rhinocerotis* as a potential 'problem plant' and noted: "Although the renoster bush is its principle constituent, several shrubs and other composites, and many geophytes are associated with it." The findings of this study concurs with Howard *et al*. (2012) in which they state that shrub encroachment is often thought to lead to land degradation and that these communities are ecologically poor in species. Thus, a nurse-plant such as *E. rhinocerotis* is of ecological importance in maintaining vegetation composition and structure in Namaqualand Granite Renosterveld. From the results it can be concluded that management of renosterveld, especially in arid and semi-arid regions needs to focus on the ecological role of *E. rhinocerotis* before eliminating it from the ecosystem.

4.5. Conclusion

Semi-arid regions exhibit temporal variability in vegetation dynamics. In Namaqualand Granite Renosterveld fire initially result in the spread of *E. rhinocerotis* but it is not the only factor affecting its abundance (Walker *et al*. 1981). The grazing, microclimates, physical and abiotic factors contributes to the survival and spread of *E. rhinocerotis*. Over time species compete with *E. rhinocerotis* for available space and resources, but its deep tap root allows it to withstand a variety of moisture and temperature fluctuations (Scott and van Breda 1937). In

this study it was seen in areas with low grazing *E. rhinocerotis* was not as abundant compared to other sites. *Elytropappus rhinocerotis* is a vigorous competitor and is able to outcomes various species in Namaqualand Granite Renosterveld because it responds positively to a disturbance. Furthermore, from the result it does seem that *E. rhinocerotis* enhances the biodiversity of various species. As after 10 years *E. rhinocerotis* created space for palatable annual species to reproduce. This shrub seems to have a vital role as a nurse-plant in renosterveld communities. As strong species (both palatable and unpalatable) associations were found in this study. The dynamics around the system is complex because if *E. rhinocerotis* is to be removed from the systems it is unknown whether it will be replaced by palatable grasses or competing unpalatable species.

5. Chapter five: Post-fire plant diversity and vegetation recovery of Namaqualand Granite Renosterveld

5.1.Introduction

Mountain Upland Renosterveld occur in small pockets across the Greater CFR (GCFR) (Bergh *et al*. 2007; Parker and Lomba 2009). This vegetation shows strong karroid affiliations (Van der Merwe and Van Rooyen 2011) and occupies an intermediate zone between the Fynbos biome and Succulent Karoo biome (Mucina and Rutherford 2006). Mountain Renosterveld is found on less fertile soils at higher altitude and is more intact compared to its lowland counterpart (Lomba and Parker 2009) which occurs on relatively fertile soils and is more suitable for intensive agriculture (Curtis 2013). Namaqualand Granite Renosterveld is recognised as one of two (the first being Kamiesberg Granite Fynbos) distinct vegetation types restricted to the Kamiesberg Upland (Helme and Desmet 2006). In the Kamiesberg, renosterveld has been extensively used for grazing and more than 20% of Namaqualand Granite Renosterveld has been transformed by agricultural activities (i.e. overgrazing and cropping) (Helme and Desmet 2006). This vegetation supports the second highest number of endemic plant species in the Kamiesberg. As in other renosterveld types, it supports a diversity of geophytes, especially in recently burnt areas (Helme and Desmet 2006). The disturbance by fire and grazing can generate different patches across the Kamiesberg, contributing to a shifting mosaic that presumably enhances biodiversity (Fuhlendorf *et al*. 2006). Landscapes mosaics of different fuel structures contributes to the heterogeneous fire regime which provides different habitats for both floral and faunal diversity. These mosaics will result in patches of different stand ages which in turn contributes to landscape diversity (Uys *et al*. 2004; Keeley *et al*. 2012).

Arid and semi-arid regions are very often characterized by occasional and event-driven changes in species composition and these occur in response to rare or extreme events (Walker 1993; Wiegand *et al*. 1995; Wiegand and Milton 1996). Different processes (e.g. grazing intensity, fire, and environmental conditions) determine plant diversity at various spatial scales (Crawley and Harral 2001). Various semi-arid rangelands in South Africa are frequently subjected to unplanned veld fires during the dry summer season and driven by dry season winds (Breedt *et al*. 2013). These unplanned veld fires area often ignited naturally by lightning (Helme and Desmet 2006) or anthropogenically such as collecting wild honey during the dry summer season (see Chapter 3; Simons 2015). Fire is an important driver of biodiversity in many ecosystems (Tierney and Watson 2009) and this ecosystems process affects diversity at community, local and regional scale (Keeley *et al*. 2012). Furthermore, fire can contribute to the continuation of diversity; namely that fire increases species richness by avoiding competitive exclusion and increases spatial heterogeneity (Van der Merwe and Van Rooyen 2011). Anthropogenic changes to the fire regimes affect biodiversity while unpredictable fire events could be harmful to many fire intolerant species (Avitabile *et al*. 2013). Semi-arid rangelands are heterogeneous landscapes with high spatial and temporal variability in resources and plant diversity (Samuels *et al*. 2016). In South Africa these rangelands have been used for grazing for about 2 000 years ago by the KhoenKhoen and San who were herders and hunters and they practised a nomadic lifestyle (Saayman *et al*. 2016). Livestock grazing can have various impacts on rangelands as they alter plant species composition, reduce vegetation cover and accelerate erosion (Saayman *et al*. 2016). Positive impacts of livestock grazing include increase to plant diversity under light grazing whereas negative impacts due to heavy grazing includes loss of plant cover, variation in diversity and compositional shifts (Anderson and Hoffman 2007). All these factors can contribute to a reduction in range conditions (Cupido 2005) with associated lower plant species diversity. Overgrazing can reduce the overall

stability and productivity of the system, with major economic implications (Todd and Hoffman 1999). Renosterveld vegetation can be described by measurable attributes (i.e. vegetation height, cover, species richness, soils, palatability, etc.) that are relevant to grazing use and conservation (Milton 2007). Understanding the influence of fire and grazing on ecosystems are important for both conservation and production (Bond and Archibald 2003). Post-fire vegetation usually attracts more grazing animals than the pre-fire vegetation due to the destruction of unpalatable vegetation (Archibald and Bond 2003). Furthermore, animals are attracted to the fresh regrowth which has high forage quality (Walton 2006). Grasses and herbs tend to be more prominent in these post-fire vegetation as they have better nutritional quality and are more palatable (Snyman 2003; Breedt *et al*. 2013). Intense grazing lead to a decrease in species richness and ultimate rangeland degradation (Snyman 2015) and highly palatable vegetation that is grazed tends to have lower fire potential. Therefore, it could be expected that grazing following burning should reduce flammability more than either grazing or burning on their own (Breedt *et al*. 2013). In order to manage arid shrubland plant communities for sustained animal production and species conservation, an understanding of their dynamic behaviour is require (Wiegand and Milton 1996).

Understanding the way in which each driver affects and changes renosterveld dynamics is important for both pastoralists which uses the vegetation daily and for biodiversity to determine which species might be of conservation concern. The chapter aimed at understanding the effect of post-fire vegetation recovery and to which extent this affected Namaqualand Granite Renosterveld rangeland conditions in combination with grazing in the Kamiesberg Uplands. The chapter addressed the following questions:

5.1. How will plant species diversity, vegetation cover and structure change in burnt and unburnt areas over time?

5.2. Is growth form diversity affected in Namaqualand Granite Renosterveld?

5.3. What is the effect of stocking density on species richness over time?

5.4 How does fire and grazing affect biodiversity and forage productivity over a temporal scale?

5.2. Methods

5.2.1. Study area

The study area was Leliefontein village grazing area in the Kamiesberg Uplands, refer to Chapter two for a full description of the study area. The study sites are all located on Namaqualand Granite Renosterveld (Mucina and Ruherford 2006) and used primarily as rangelands for livestock grazing. The burnt study sites have diverse fire histories ranging from 2000 to 2015, as described in Chapter three.

5.2.2. Data collection

The study was designed to describe the post-fire vegetation dynamics of Namaqualand Granite Renosterveld in terms of its diversity, structure, cover, grazing pressure, conservation and agricultural purpose. These variables were studied in 10 burnt and 10 adjacent unburnt Namaqualand Granite Renosterveld sites. The unburnt sites act as the control for the study as these sites have not been burnt over the past 25 years.

5.2.2.1.Diversity, structure and cover data collection

Field sampling processed between the months of March, May and September in 2014, with the last collection in September 2015. Species diversity was assessed using a 10 × 10 m site (Guo 2001). Refer to Chapter four for a detail description of this method. The species diversity

for each site was calculated using the Shannon-Wiener Index (H'). This diversity measure was calculated as follow:

$$H' = -\sum_{i=1}^{s} p_i \ln p_i$$

Where H' = Shannon-Wiener diversity index

S = total number of species in the community (richness)

p_i = the proportion of species *i* relative to the total number of species

The Simpson's Diversity index is a measures diversity which takes into account the number of species present, as well as the relative abundance of each species. As species richness and evenness increases, so diversity increases. The Simpson's diversity index was calculated as follow:

$$D = 1 - \frac{\sum n_i (n_i - 1)}{N (N - 1)}$$

Where D = Simpson's diversity index

n_i = number of individuals of the *i*th species

N = Total number of individuals of all species

The temporal changes in post-fire vegetation were assed using structure and cover obtained from Simons (2013). The descending point method according Roux (1963) was used to collect vegetation structure and cover data in burnt and adjacent unburnt sites. Refer to Chapter four for a detailed description of the vegetation cover data collection. According to the descending

point method, once a plant species was struck the height and width, of the plants at every 1 m interval along transect, were measured to determine the vegetation structure (Fig. 5.1).

Figure 5.1: The descending point method used to collect cover and structure data in Namaqualand Granite Renosterveld.

Plant species were identified in the field and further positive identifications were done using the Namaqualand Flower Guide (Le Roux 2015), consulting with ARC rangeland ecologists, and plant taxonomist at the UWC herbarium. Plant species were assigned to categories of life forms (annual and perennial) and growth forms (shrubs, graminoids, succulents, herbs and trees). The physical attributes (i.e. slope, aspect, and rockiness) at each site were recorded.

5.2.2.2.Stocking density data collection

Refer to Chapter four for a detailed description.

5.2.2.3.Ratio of increaser and decreaser species data collection

Along a 100 m line transect at each 10 m intervals plant species growing within a 1 m radius of *E. rhinocerotis* were identified, counted (species numbers and abundance) and recorded as described in Chapter four. This method was done in each burnt and unburnt site. Furthermore, species were classified based on their ecological status (Decreaser and Increaser perennial species) as defined by O'Connor *et al*. (2010), Trollope *et al*. (2014), Snyman (2015) and consulting with rangeland ecologist and experts. Species were classified according to their reaction to grazing (see appendix 2). As no definition currently exist for the ecological status of renosterveld species, the follow definitions were given after rangeland ecologist currently working in Namaqualand:

Decreaser species (D) - species which predominate in good rangeland condition but declines in abundance once the rangelands are over- or underused. These decreaser species are generally palatable to livestock, especially after a fire disturbance (e.g. *Ehrharta calycina, Ficinia indica* and *Leobordea digitata*).

Increaser species (I) - species which increases when the rangelands are over- or underused and thrive in disturbed areas. These increaser species are generally unpalatable to livestock (e.g. *Anisodontea bryoniifolia, Chrysocoma ciliata* and *Elytropappus rhinocerotis*).

5.2.3. Data analysis

The Shapiro Wilk's W test (Royston 1982) was used to test for normality within the data set. Dependent t-tests were performed to test changes in a) diversity (Shannon-Wiener and Simpson's indices), b) vegetation cover (%), c) vegetation structure, d) growth form species richness and e) between the different growth forms in all burnt and unburnt sites. Linear regressions using general regression models (GRM) were performed to test the effect of stocking densities (SSU / 10 ha) on species richness in both burnt and unburnt sites. Regressions were used to test if years after fire predicted changes in increaser/ decreaser species ratio and species richness over time. Further statistical tests were done to test whether 0-7 years and 10-15 years after fire had significant effects on increaser and decreaser species ratio and species richness in Namaqualand Granite Renosterveld. Separate regressions were performed to test the changes in annual and perennial species diversity over time. Dependent t-tests were performed to test for differences in the diversity of annual and perennial species. Separate regressions were done to test the effect of stocking density (SSU / 10 ha) on life and growth form species richness. All the statistical analysis was performed in STATISTICA (StatSoft, Inc., Tulsa, OK, version 8, 2007).

5.3.Results

5.3.1. Changes in plant diversity, vegetation cover and structure in burnt and unburnt Namaqualand Granite Renosterveld over time.

The Shannon-Wiener (H') and Simpson's (D) diversity indices are relatively high and constant across the studied sites (Fig 5.2.a and 5.2.b). In the burnt sites for both indices, diversity peaks between 2.3 and 2.2 (Shannon-Wiener) and 0.89 and 0.86 (Simpson's) for years 6 and 7 respectively. At years 10 and 15 post-fire diversity at these two sites are higher comparted to the unburnt sites. Overall the diversity at the unburnt sites remained constant across the study

sites. Thus, the dependent t-test showed no statistically significant differences in the Shannon-Wiener diversity t (9) = -1.9; p = 0.08 and Simpson diversity t (9) = -1.8; p = 0.36 between the burnt and unburnt sites.

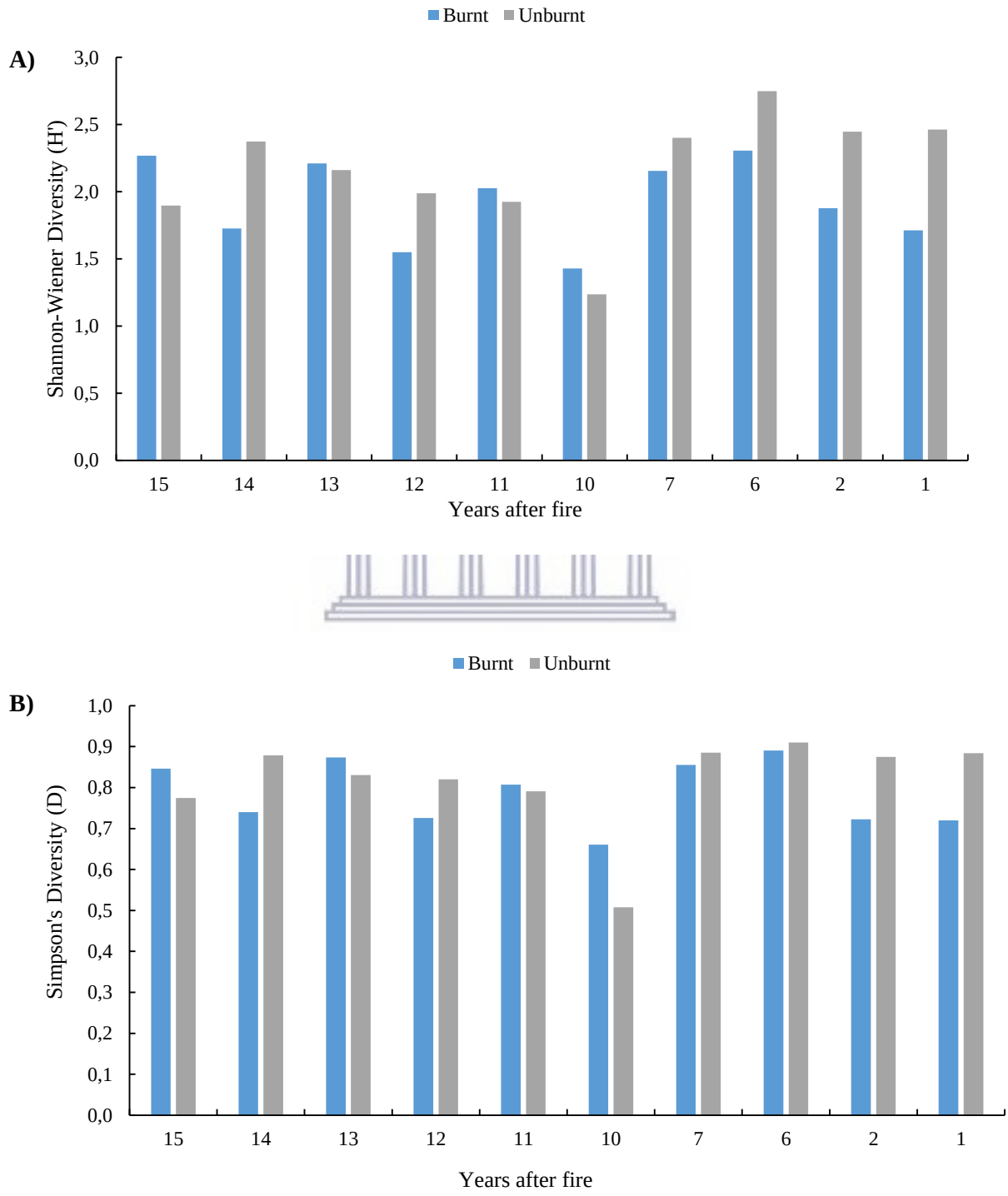


Figure 5.2: The change in a) Shannon-Wiener diversity and b) Simpson's diversity, c) burnt and unburnt in Namaqualand Granite Renosterveld.

The vegetation cover (%) and structure (cm) both increased over time (Fig 5.3.a and Fig 5.3.b) and peaks between 10 and 11 years post fire. The cover of the unburnt sites was higher when compared to the burnt sites but it was not significantly different t (9) = -1.5; p = 0.17. The average vegetation height (cm) fluctuated across the study sites for both burnt and unburnt areas. It can be seen from Fig. 5.3.b that the vegetation heights were the highest at the oldest burnt site (15 years after fire). This was even higher than the adjacent unburnt site. However, the height did not differ significantly t (9) = -2.0; p = 0.08 across the burnt and unburnt areas.

A separate regression was done to test the change in annual and perennial species diversity over time. Perennial diversity did not change significantly overtime, whereas annual diversity changes significantly over time ($F_{1,8}$ = 5.87, p = 0.04, R^2 = 0.3). The annual and perennial diversity were compared to the unburnt sites insignificant results were found.

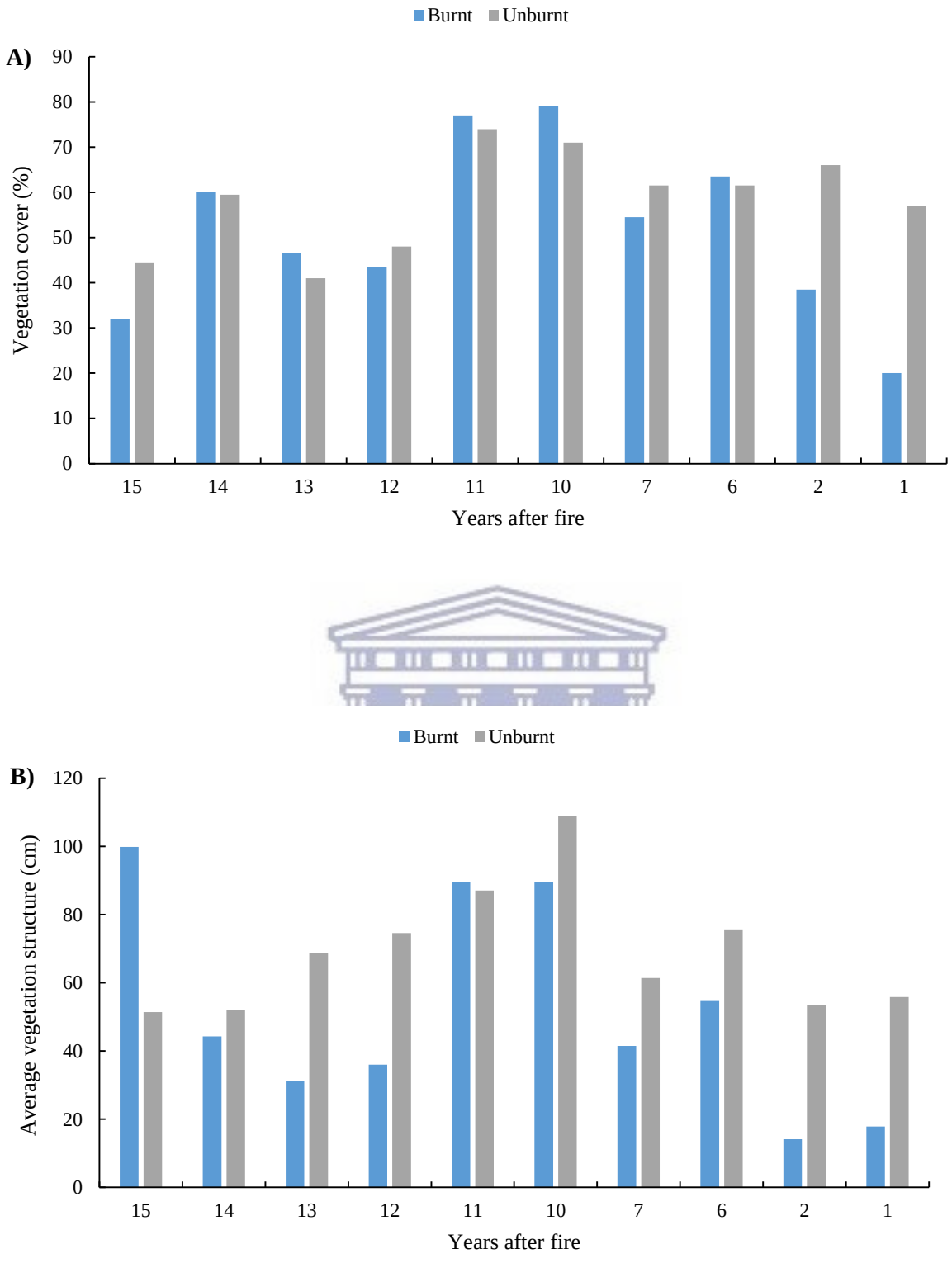


Figure 5.3: The vegetation a) cover and b) mean structure in burnt and unburnt Namaqualand Granite Renosterveld.

5.3.2. Post-fire changes in growth species richness.

In Fig. 5.4 it can be seen that the species richness of shrubs, succulents and herbs are higher in the unburnt sites. The graminoids and trees are relatively low in both burnt and unburnt sites. However, there were no statistically significant difference between the shrubs (p = 0.11), graminoids (p = 0.27), succulents (p = 0.09), herbs (p = 0.78) and trees (p = 0.05) in the burnt and unburnt sites. Further testing, showed that there were significant differences between the different growth forms in both burnt and unburnt sites (see appendix 3). And that shrubs differed significantly (p = 0.01) than the rest of the growth forms in both burnt and unburnt areas.

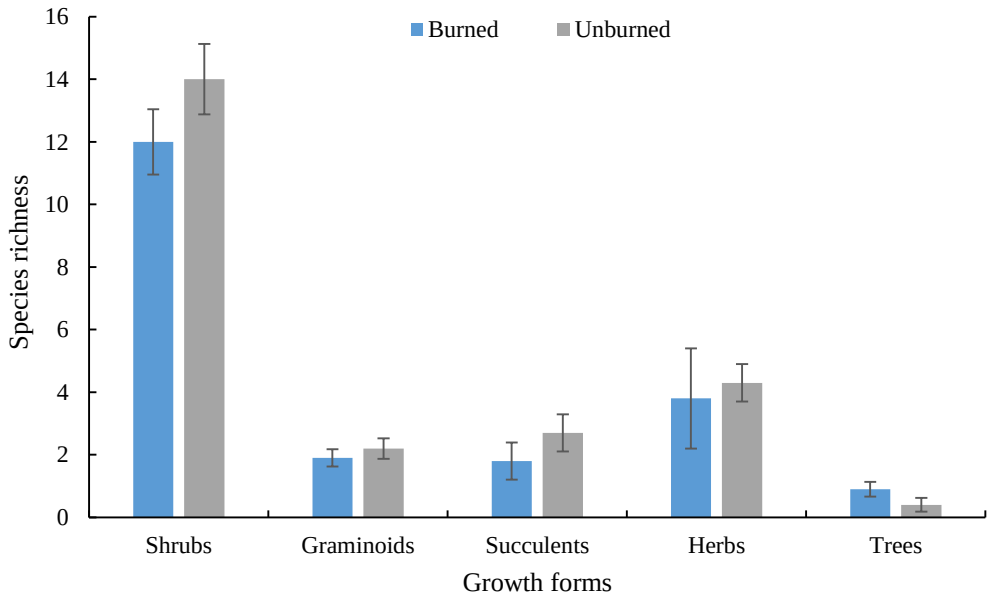


Figure 5.4: Mean (±SE) species richness of growth forms (shrubs, graminoids, succulents, herbs and trees) in burnt and unburnt 100 m^2 sites.

5.3.3. Effects of stocking densities on species richness over time.

Stocking density has a significant effect on species richness (Fig.5.5) in the burnt area ($F_{(1,8)}$ = 7.14; R^2 = 0.47; p = 0.02). However, in the unburnt areas stocking density does not affect species richness ($p > 0.05$). The effect of stocking density on growth form species richness was tested separately. Stocking density had a significant effect on shrubs ($F_{(1,8)}$ = 5.41; R^2 = 0.40; p = 0.04) and perennial species ($F_{(1,8)}$ = 9.24; R^2 = 0.47; p = 0.01) in the burnt sites. The insignificant results show that plant species are not solely affected by grazing in renosterveld.

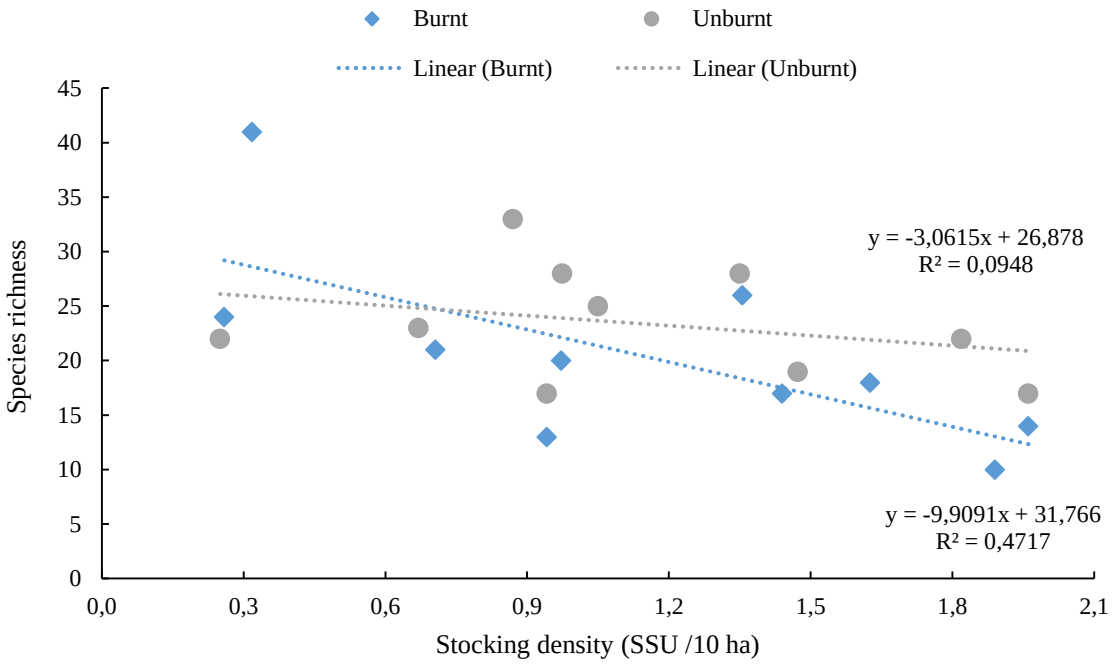


Figure 5.5: Relationship between species richness and different stocking densities (SSU/ 10 ha) in burnt and unburnt areas.

5.3.4. The implication of fire on conservation and agriculture for livestock grazing in Namaqualand Granite Renosterveld.

The ratio of increaser and decreaser species are low in the first two years post-fire. During this stage more decreaser species occur in the renosterveld communities (Fig.5.6). Following this after at seven years this ratio reaches its peak of 3.7 and increaser species escalate. After 10 years the ratio between increaser and decreaser species fluctuates. The relationship between the increaser and decreaser species ratio and species richness are inversely related and the years after fire does not statistically effect the ratio of increaser and decreaser species and species richness overtime ($F_{(1,8)} = 0.30$; $R^2 = 0.04$; $p = 0.59$).

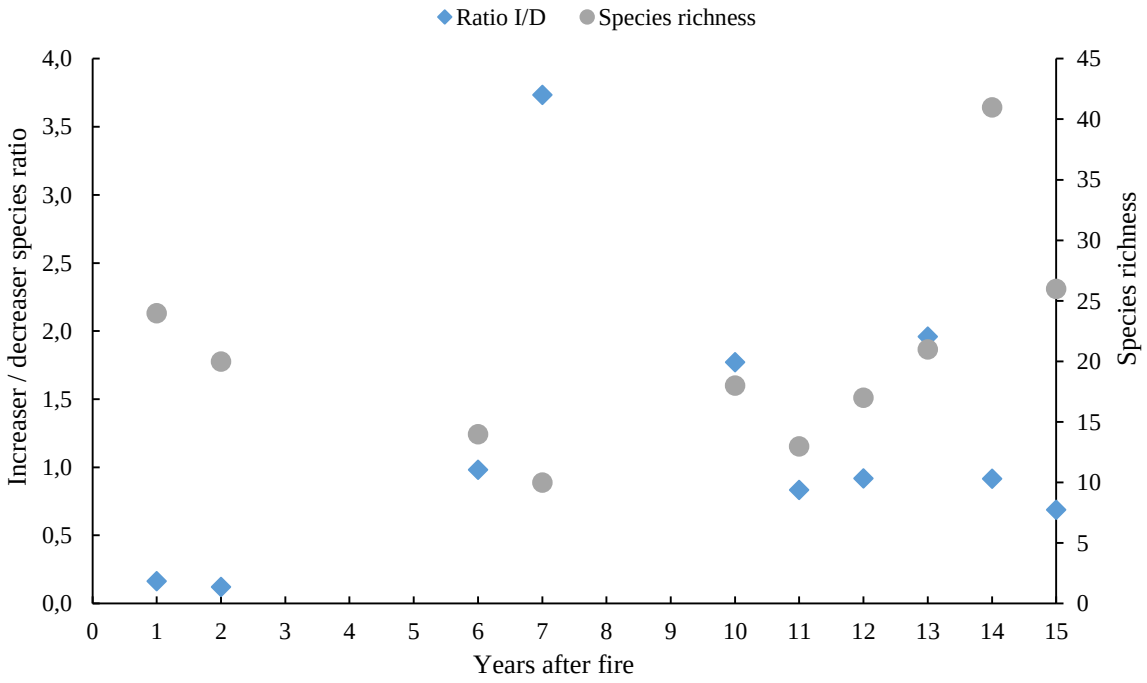


Figure 5.6: The ratio of increaser and decreaser and species richness in burnt Namaqualand Granite Renosterveld vegetation.

From Fig. 5.7.a, it can be seen that the ratio of increaser and decreaser species increaser over from 0 to seven years post-fire. This ratio peaks at seven years post-fire. The regressions for

the increaser and decreaser species ratio were both strong and positive ($R^2 = 0.7$), but it was not significantly affected ($p = 0.17$) by time (0-7 years post-fire). The results show that there was a statistically significant decline in species richness ($F_{(1,8)} = 58.32$; $R^2 = 0.96$; $p = 0.01$) over 0-7 years post fire.

In Fig. 5.7.b, the regression showed that the ratio of increaser and decreaser species declined, whereas the species richness increased overtime. However, insignificant results were found for increaser and decreaser ratio ($p = 0.42$) and species richness ($p = 0.13$).

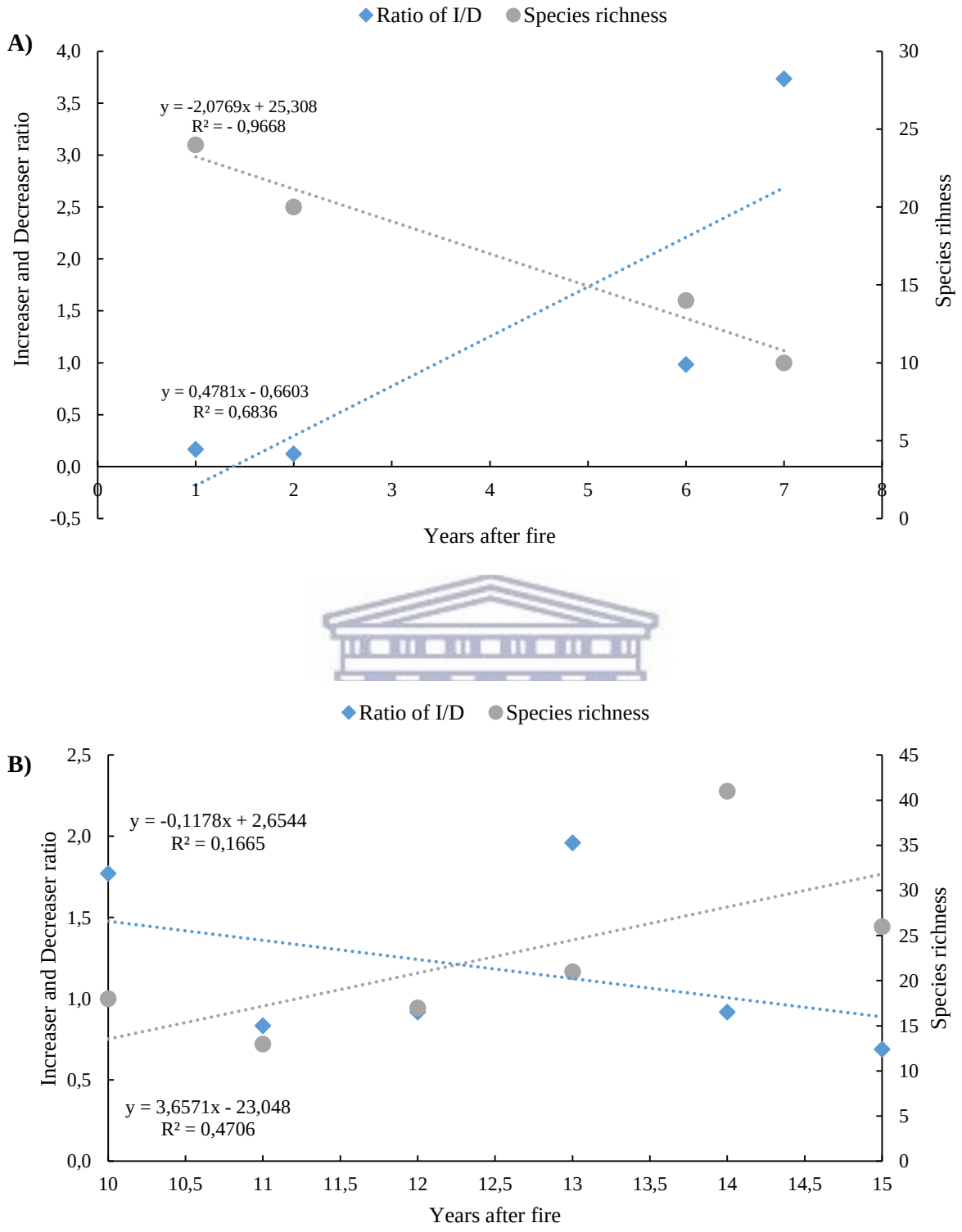


Figure 5.7: The a) short term (0 -7 years) and b) long term (10 -15 years) effects of years after fire on the increaser and decreaser species ratio and species richness in Namaqualand Granite Renosterveld.

5.4.Discussion:

5.4.1. The post-fire effects on species diversity, vegetation cover and structure in Namaqualand Granite Renosterveld.

Mountain Renosterveld communities display rapid vegetation recovery after fires as I found vegetation re-establishment within the first four to 10 months following fire. Van der Merwe and Van Rooyen (2011) also found vegetation cover to re-establish after nine months in Mountain Renosterveld in the Roggeveld. Rapid vegetation recovery after fire is characteristic of fire-prone Mediterranean-type ecosystems (Keeley *et al*. 2005; Capitanio and Carcailett 2008; Keeley *et al*. 2012). The results show that the Shannon-Wiener and Simpson diversity indices are moderately high within the first two years post-fire due to the high abundance of annual species. The results are supported by similar finding by Potts *et al.* (2003); Keeley *et al*. (2005); Conlisk *et al*. (2016) which reported a peak in species diversity during the first year's post-fire with many annual species characteristically present in Mediterranean ecosystems. Similar to renosterveld recovery, Eastern Australia mulga shrublands also flourish with high percentages of annuals after occasional good rainfall (Williams 2002). The fast growing annual species also known as "fire annuals" (see, Keeley *et al*. 1981) only germinate after fire. In the Kamiesberg, these annuals (locally referred to as "*opslag*") include *Salvia dentate*, *Leobordea digitata*, *Hermannia amoena*, *Mesembryanthemum crystallinum*, *Massonia depressa* and *Aspalathus spinosa* and grass species. These post-fire successional pioneer species can be further classified as seeders and resprouters species and as a result not all of these pioneer species will be replaced over the successional stages in Renosterveld. As *Cotula barbata*, *Crassula natans*, *Felicia australis*, *Helichrysum tintcum,* and *Stachys rugosa* will re-establish, after about 6 to 12 months post-fire. Eventually, overtime some of the above mentioned species are replaced by woody perennials which are slow growing, longevity, competitive persistence and has the ability to regenerate within mature strands as explained by

Knuckey *et al*. (2016). Species which retain these traits are major drivers of long-term succession following fire in resource-limited ecosystems (Reese *et al*. 2001; Knuckey *et al*. 2016). Post-fire nutrients are high within the first season (Breedt *et al*. 2013) and is available for a short duration as nitrogen availability decreases after nine months and phosphorous levels returning to pre-fire levels after only four months (Cowling *et al*. 1997). The fluctuation in species diversity across the study sites are primarily due to the elimination of fast growing annuals species which are shaded out by the slow growing woody species and environmental conditions. In semi-arid mulga shrublands temporal variations in species diversity is also associated with seasonal conditions (Williams 2002). Renosterveld displays complex successional trends as plant diversity recorded at 6, 13 and 15 years after fire remains high. This is due to low *E. rhinocerotis* individuals recorded at these sites. The absence of unpalatable *E. rhinocerotis* shrubs allows succulents, graminoids and other shrub species to recover in the open space between plants (see Chapter 4). This is similar to Mountain Renosterveld in the Roggeveld where the level of species richness remained high from years 3 to 10 after fire as found by Van der Merwe and Van Rooyen (2011). It is clear that fire favours some species in renosterveld, but it is difficult to determine which species are fire-dependent, fire-sensitive and those eliminated by fire (see appendix 4). The reason for this can be explained by the role of each plant or plant groups, as they change over time as a result of changes in the physical and vegetation conditions as explained by Knuckey *et al*. (2016). Fire creates space for many species and stimulates their germination. Additionally, it may possibly play a role in maintaining some species in renosterveld, although there are many species which are not dependent on fire (such as *Phylica rigidifolia*, *Anthospermum spathulatum* and *Aspalathus spinosa*) given their post-fire recruitment and increasing abundance with time since fire. It is difficult to determine if these species might be less abundant in unburnt vegetation as grazing might also benefit in their dominance as found by Guo (2001).

The difference in species diversity can also be attributed to the fact that numerous microhabitats occur across the study sites which exist as specialised communities on specific substrates. These microhabitats exist because of the structural changes at each post-fire site which influences the microclimate and the distribution of resources such as nutrients and moisture (Higgens *et al*. 2007; Curtis 2013). These changes in turn have cascading effects on biodiversity with some species responding to microclimate and resource availability as shown by Higgens *et al*. (2007). Furthermore, Peeters and Butler (2014) found that microhabitats provide and support a relatively high number of endemic species in semi-arid mulga shrublands.

While plant diversity remained fairly stable overtime, vegetation cover continuously increased across the study sites but fluctuated between 12 – 15 years post fire. This is due to the less inhibiting effect of the relatively open vegetation canopy as described by Rutherford *et al*. (2011). The high vegetation cover recorded at Groenkloof (10 years post-fire) and Toringberg (11 years post-fire) is attributed due to good rains received during these years. Whereas, the low cover recorded after 12 years is due to drought conditions of very low to almost no rain (B. Links, personal communication). The apparent decline in vegetation cover after 15 years is a result of the physical attributes such as the steep slope and high percentage of rock cover (see Chapter 4) but also the site occurring on an ecotone between renosterveld and succulent Karoo.

Fire leaves black scars across the burnt landscape, which creates the impression of a markedly altered vegetation (Noy-Meir 1995). Vegetation structure is a key indicator of long-term vegetation change in semi-arid ecosystems (Mooney *et al*. 1980; Keeley *et al*. 1981; Trodd and Dougill 1998). The results indicate that changes in vegetation structure are temporarily and mostly perennials usually persists in the years after fire. The height of the post-burn vegetation was below 50 cm within the first two years due to the removal of dry plants and litter by fire

as explained by Noy-Meir (1995). In addition, across the majority of the study sites the heights of the new vegetation were lower in the burnt areas than in the adjacent unburnt areas. The difference in height were hardly noticeable in years 11, 14 and 15 years. To an extent larger shrubs and trees manage to survive fire by resprouting from the bases but these species are severely reduced in height, while some shrubs show a drop in height in the old vegetation due to partial diebacks with increasing age (e.g. *E. rhinocerotis*) while others (e.g. *Ischyrolepis gossypina* and *Euryops lateriflorus*) continue to increase in height. Graminoids and herbs do not show much differences in height for the different areas. The vegetation structure varies greatly across the study sites and this indicates that another mechanism is influencing these areas. To a large extent the unburnt vegetation has higher species composition compared to the burnt sites. Moreover, the high vegetation structure can be explained due to the removal of woody perennial species which compete with other species for shade, increased humidity or soil stability as found by Hunter and Aarssen (1988); Belsky (1992). These unburnt sites are mainly disturbed and maintained by livestock grazing at different intensities as it continues to reduce the green leaf material during the growing season and to slow down the rate at which it accumulates as explained by Noy-Meir (1995). Over their successional stages, burnt sites returned to their woody and dense state but the species were not necessarily unpalatable.

5.4.2. The effect of stocking density on growth form composition and plant species richness

Post-fire areas are susceptible to selective heavy grazing which in turn can alter the recovery of plant species (Jacobs and Jangle 2008). In post-burn sites the stocking density had a significant effect on the species richness. Due to the nutritional value of post-fire areas livestock are often attracted to these areas with their new plant growth (Bond and Van Wilgen

1996). Thus, unpalatable species increase at the loss of more palatable species which occur after fire and assemblage in lightly to mild grazed sites which is similar to Cowling-Hilton-Taylor (1999) findings. According to Boucher (1995) the exclusion of fire combined with grazing or bush cutting may reduce diversity in renosterveld. However, the results of my study show that variation in grazing (i.e. light, moderate and high) increased and maintained species richness in older burnt and unburnt renosterveld vegetation. Similar results were reported by Walton (2006) which described that under moderate grazing diversity and cover of grass and geophytes on renosterveld fields over a period exceeding 30 years in the absence of fire increased. Even in dry prairie remnants, burning and removing exotic species and woody vegetation appears to enhance and maintain plant diversity as shown by Kraszewski and Waller (2008). Thus, removing livestock from the rangelands will not ensure that palatable perennial shrubs will replace equally unpalatable perennial shrubs in the natural rangeland which has not burnt such as succulent karoo as reported Milton (1994).

Furthermore, the results show that stocking densities did not have an effect on growth form composition and plant species richness in the unburnt sites. This finding is supported by previous studies done by Todd and Hoffman (1999); Petersen *et al*. (2004); Allsopp *et al*. (2007); Anderson and Hoffman (2007) which reported that there is not a significant relationship between species richness and grazing intensity in Namaqualand. The insignificant results in the unburnt sites is due to the long history of heavy and exclusion of fire. It would seem that the evolutionary history of renosterveld could have resulted in the vegetation becoming resistant to high grazing pressures in renosterveld rangelands in the Kamiesberg. This is supported by Papanastasis *et al*. (2002) study in the Psiloritis mountain of Crete, a Mediterranean system, which found that the ecosystem has become resilient to heavy and continuous grazing pressure due to a balanced structure with a large variety of plant growth forms which has adapted to a long evolutionary history associated with animals. The results show that the indigenous

perennial grass component of renosterveld shrublands has declined and there has been a widespread dominance of the unpalatable perennial shrubs such as *E. rhinocerotis* and *C. ciliata*. Similarly, to Australian mulga communities with high grazing pressure reduces the perennial grass cover which decreases the natural fire frequency and increases woody shrub cover (Peeters and Butler 2014). In rangelands with a long history of grazing for instance Leliefontein, the selective pressure of grazing animals has fluctuated over time, allowing the development of different "grazing-tolerant" species (as mentioned above) to adapted to high or low grazing intensities as described by Cingolani *et al*. (2005); Todd and Hoffman (1999). Therefore, heavy grazing does not necessarily lead to a reduction in plant species richness (Rutherford and Powrie 2013). The unburnt and older burnt sites may appear to be degraded due to the abundance of unpalatable species, but they are not unresponsive to variations in grazing intensity as species richness are maintained at a high level in moderate to heavily grazed areas as explained by Saayman *et al*. (2016).

Livestock grazing is not the only factor affecting patterns of variation in growth form composition and plant species richness in Namaqualand Granite Renosterveld. In the Kamiesberg climatic conditions such as low rainfall and drought events affect ephemeral flushes during the wet season (Anderson and Hoffman 2007), which is characteristic of Namaqualand landscapes during winter (Milton and Wiegand 2001). In the event where an area is experiencing prolonged dry period, it will result in the disappearance of ephemeral seedbanks and the inability of seeds to germinate and establish themselves (Milton and Wiegand 2001). During 2003 and 2015 (sampling period) the Kamiesberg experienced drought events and this may have resulted in some seeds not being able to germinate which account for the low numbers of ephemeral (annual and geophytes) species recorded over the sampling period. Thus, a balance between fire frequency and grazing would promote the persistence of the shrub component. Mature unpalatable shrub species can be expected to produce more flowers and

seeds then palatable species whose seed production is reduced by browsing (Milton and Dean 1990). Thereby hindering rangeland recovery which depend on annual rain rather than a persistent seed bank for regeneration (Milton 1995) which is unimportant to perennial shrubs in the Succulent Karoo (Saayman *et al*. 2016).

5.4.3. Long-term effects of fire on biodiversity and forage productivity in Renosterveld rangelands

Fire has been regarded as a natural process (Breedte *et al*. 2013; Trollope *et al*. 2014) which play an essential role in maintaining biodiversity (Pausas and Keeley 2009; Keeley *et al*. 2012) but is not necessarily needed for livestock production (Trollope *et al*. 2014). Rangeland conditions can improve if fire is excluded (Trollope *et al*. 2014) as seen in the unburnt and older (>10 years) burnt sites where species richness is higher and more decreaser species occur. Even though species richness fluctuates with decreaser and increaser species ratio, in Namaqualand Granite Renosterveld the results show that increaser species such as *E. rhinocerotis* and *C.ciliata* increased in abundance in burnt and grazed areas. Similarly, Kraszewski and Waller (2008) found that in dry prairie remnants in South-central Wisconsin *Rhus glabra* increased in the burnt sites and has been shown to increase in shoot density following fire. Whereas *Juniperus virginiana* increased on areas which lacked management interventions. Changes of this nature makes the vegetation unattractive to livestock grazers as vegetation becomes dense, difficult to access and can provide cover for predators (B. Links, personal communication).

The general notion regarding communal rangelands is that they are overstocked and as a consequence degraded (Allsopp e*t al*. 2007). Although some burnt and unburnt areas had high stocking rates (such as Populierskloof, Toringberg, Die Vlak and Xharras) the results show that

these rangelands were not degraded. Thus, the results contradict literature regarding communal rangelands. Even though natural scientist (Todd and Hoffman 1999; Allsopp 1999; Anderson and Hoffmann 2007) have reported on ecological changes associated with grazing in the Leliefontein village; there is no evidence to prove that this system has become less productive over the past 50 year as shown by Benjaminsen *et al*. (2006) and Allsopp *et al*. (2007). Due to the fact that renosterveld often displays a uniform grey appearance owing to the abundance *E. rhinocerotis* it creates the illusion of a homogenous vegetation with low diversity (Curtis 2013). Currently, in renosterveld rangelands more species tended to be decreaser than increasers in older burnt and unburnt vegetation. Therefore, thriving or winning under the current grazing regime. Older burnt and unburnt vegetation are mostly disturbed by grazing in renosterveld rangelands. In this way the disturbance by grazing opens space for decreaser species to recolonise over time in renosterveld.

In Leliefontein rangeland biodiversity is maintained by resting the veld (Samuels 2013). Pastoralist move their livestock seasonally to lower altitude villages such as Tweerivier to evade the cold temperatures, access seasonal forage and to rest their pastures in Leliefontein as found by Samuels (2013). In Leliefontein village pastoralist move just before the start and immediately after the onset of the cold season. Recently burnt areas are also allowed to rest for approximately 6-12 months depending on rainfall. As livestock will only be allowed on the rangelands after the winter months. During those six months of rest the veld is dominated by poisonous plants such as *Moraea herrei*, *Moraea miniata* and *Moraea pallida*, as well as other *Moraea* spp. These species are usually amongst the first green plants to appear after fire as found by Botha *et al*. (2013). This makes the plant attractive to animals especially younger animals (such as lambs in Leliefontein) that have not experienced *Moraea* spp. before. The presence of these species is often an indication that the veld is unavailable for grazing (Simons 2015). Spatial heterogeneity of vegetation across the rangeland provides pastoralist with a

range of grazing sites to select for their livestock. Therefore, farmers can maintain relatively high stocking rates due to the veld being rested seasonally.

It is important to understand in which way fire and grazing affects conservation and livestock production (Bond and Archibald 2003; Kirkpatrick *et al*. 2011). The results indicate that Namaqualand Granite Renosterveld should not be burnt sooner than seven years to increase forage productivity and to eliminate or reduce increaser species. Seeing as renosterveld is situated in an agricultural system burning should be done in favour of forage productivity. These increaser species make the system unproductive which puts farmers at a disadvantage. Even though it can be seen from the results that the species richness peaks after 14 years post-fire but the composition changed over time. Also, more decreaser species occur in older burnt sites but the quality of forage is not the same as that of the younger plants (found in the initial post-fire sites) even if it's from the same species. From a conservation perspective burning renosterveld after seven years will not decrease species richness as these species will be restored after 10 years.

Allowing the veld to recover for about seven years is the middle point to benefit both forage production and biodiversity. This result is similar to semi-arid mulga plant communities in central Australia which has a fire return interval of 7 to 10 years to reduce fuel loads in the vegetation as found by QPWS (2013). Fire can restore heterogeneity within a rangeland and alter grazing patters in such a way that it enhances the local biodiversity (Anderson *et al*. 2006; Fuhlendorf *et al*. 2009) as in the case of Namaqualand Granite Renosterveld. The same principle is applied to mulga communities which can maintain different landscape mosaics in burnt and unburnt patches. This is done to limit the frequency of fires and the potential impacts of damaging unplanned veld fires as described by QPWS (2013) and Peeters and Butler (2014). Regardless of whether renosterveld is burnt or grazed, species richness will remain constant

overtime (Simons 2013). As the number of species present are not of much importance, but rather which species are present as described by Todd *et al*. (1998). All the species identified for this study are least concerned and are therefore not species of conservation concern.

5.5. Conclusion

The results showed that the variation found in species diversity, vegetation cover and structure, growth and life form species richness were primarily driven by environmental variation across the landscape. This study showed that even though increaser species dominate Renosterveld rangelands up to seven years. The species richness does not decline and remains stable overtime even in older Renosterveld over 15 years. The disturbances by grazing, ploughing and climate conditions open space for decreaser species to re-colonise. Consequently, renosterveld patches will have to be assed differently to determine its biodiversity and community value as illustrated by Milton (2007). Fire in Namaqualand Granite Renosterveld resulted in the creation of different mosaics. The interaction between fire and grazing often promotes landscape heterogeneity as described by Martin and Sapsis (1992). Thus, different burnt areas which has different post-fire ages and has different patches of heterogeneity. This in turn results into different forage quality across the rangelands. Renosterveld rangelands are not degraded due to heavy grazing on recently burnt sites or even older burnt sites. Heterogeneity across the Kamiesberg landscape could have been established due to historical pyrodiversity (fire regime) which has supported and maintained diversity as explained by Martin and Sapsis (1992) using the 'Pyrodiversity begets biodiversity' hypothesis. This study has established a middle point of seven years to benefit both forage productivity and biodiversity in terms of conservation for Renosterveld in the Kamiesberg.

6. Chapter six: Study synthesis

6.1. A state-in-transition model for Namaqualand Granite Renosterveld

My findings in understanding Namaqualand Granite Renosterveld in terms of assessing the fire regime, succession of *E. rhinocerotis* and post-fire vegetation recovery overtime has achieved the overall aims of this thesis.

Renosterveld is a complex vegetation type which has multiple structural states, including shrublands, grasslands, lawns and herblands as described by Radloff *et al*. (2014). To date there exists an uncertainty as to whether the present unpalatable shrubland state might have replaced a historical grassland state of renosterveld (Radloff *et al*. 2014). Numerous models (Cowling *et al*. 1986; Rebelo 1995; Krug 2004; Walton 2006; Radloff *et al*. 2014) have proposed ways in which various drivers will interact and change the composition of renosterveld.

The results from this study provides evidence for the development of a state-and-transition model for Namaqualand Granite Renosterveld in communal rangelands (Fig. 6.1). This model illustrates different transitions between vegetation states for Namaqualand Granite Renosterveld. Drivers in the form of stocking densities, physical and climatic factors and ecological processes trigger the transition from one state to another. These transitions may occur over different time scales and are not necessarily reversible. The states are based on the vegetation characteristics (plant species richness, plant cover, vegetation structure and the ratio between increaser and decreaser species) as discussed in the thesis.

I found that the post-burnt sites generally comprise a mixture of shrubs and herbs such as *Aspalathus spinosa*, *Asparagus* spp., *Cotula barbata*, *Hermannia amoena*, *Leobordea digitata* and *Mesembryanthemum crystallinum*. This is the first alternative state (AS-1) known as the shrub and herb dominated herblands. This is one of the most desirable states for livestock due

to the herbaceous components, which have good fodder value, and are not found in the older stable sites.

In post-burn sites which are heavily grazed, a mixed shrub and graminoids dominated shrubland with species such as *Elytropappus rhinocerotis, Pelargonium* spp., *Ficinia indica, Ficinia nigrescens* and *Ischryrolepis gossypinus* emerges. This is the another alternative state (AS-H3) as heavy grazing prevents the establishment of seedlings of palatable plants, thus only the graminoid seedling unpalatable to small stock will be able to survive.

My results showed that under moderate grazing a mixed herb and succulent dominated shrubland with species such as *Felicia australis, Lessertia* spp., *Leipoldtia* spp. and *Ruschia* spp. will emerge. This is also an alternative state (AS-M2), with barely any grasses occurring in these sites. Following this state, herbs are outcompeted and the vegetation moves towards other alternative states (as seen from AS-M3, AS-H3, and AS-H4). These alternative states are all shrub dominated shrublands with species such as, *Aspalathus spinosa, Anthospermum spathulatum Chrysocoma ciliata, Elytropappus rhinocerotis, Eriocephalus microphyllus, Euryops lateriflorus,* and *Stachys rugosa* dominating. Since most of these species are unpalatable to livestock, herders move their animals away and the stocking densities are reduced either to moderate grazing (under former heavy grazing) or low grazing (under former moderate grazing densities. As these shrublands are mostly dominated by unpalatable shrubs they are able to grow and thus both lineages of alternative states converge into a tall shrub dominated shrubland. This stable states (SS-1) are not desirable to farmers. However, over time (in the absence of fire) these stable state communities will have high ephemeral abundance during winter due to the removal of unpalatable species which create large open spaces (Anderson *et al.* 2010).

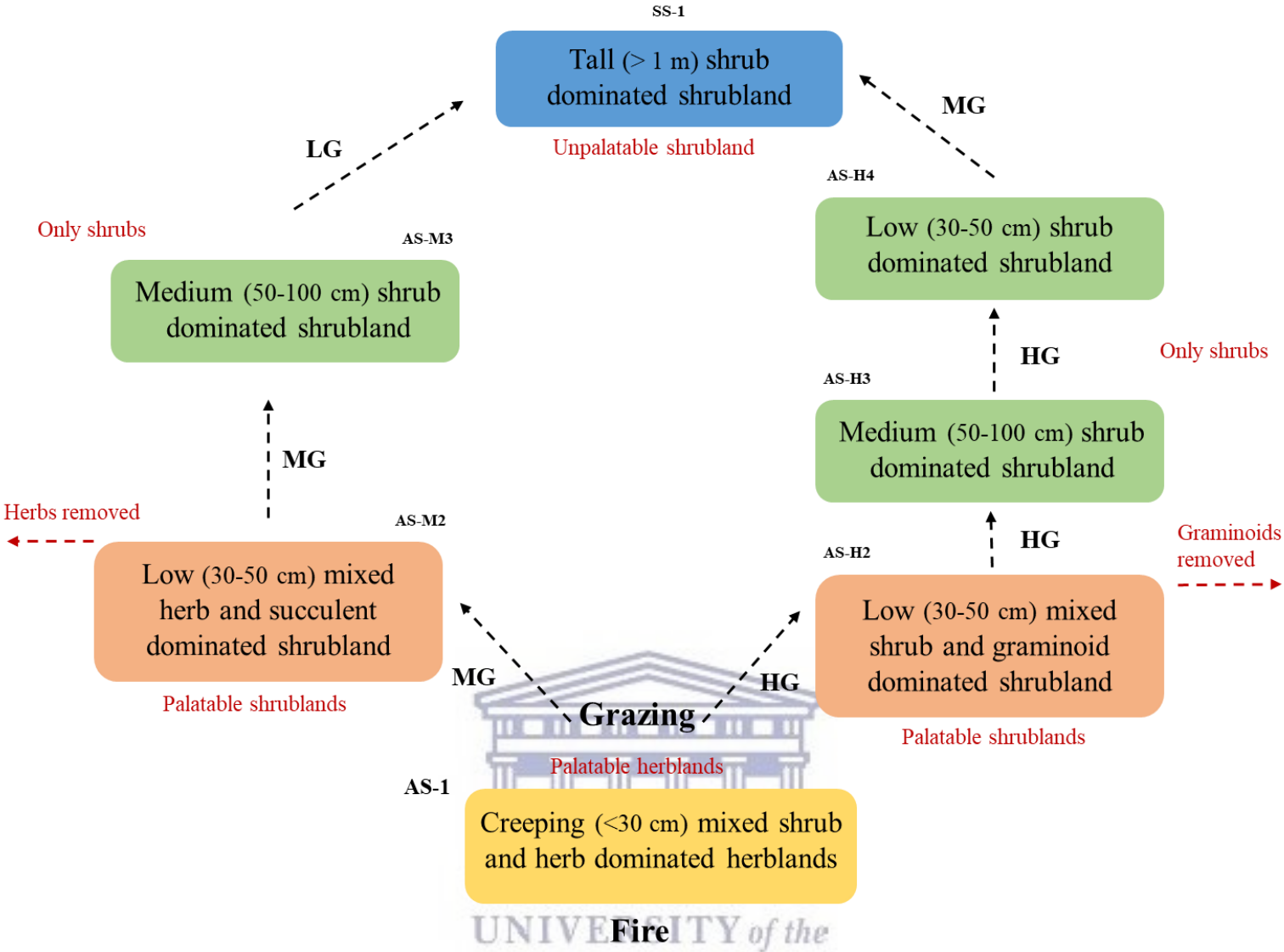


Figure 6.1: A state –and- transition model illustrating the effects of fire and grazing for Namaqualand Granite Renosterveld. HG=Heavy grazing intensity, MG= Moderate grazing intensity and LG= Low grazing intensity. Vegetation states are illustrated as AS=Alternative state and SS=Stable state; the letter following the hyphen indicates the grazing level and the number indicates the successional year (e.g. AS-M2).

6.2.A rangeland condition model relevant to farmers and conservationists

Since the study was conducted in an agricultural landscape, a conceptual model based on rangeland condition and biodiversity is also presented.

The Namaqualand Granite Renosterveld rangelands are influenced by two grazers one being livestock which can be selective and fire that is non-selective. From Fig. 6.2, it can be seen that fire, an abiotic grazer and will consume large amounts of plant material at different fire

intensity and in a preferred season. Furthermore, fire thrives on woody and unpalatable increaser species that fuels the fire. After the fire, there is now a higher ratio of palatable decreaser species to increaser species. Decreaser species start to decline and unpalatable increaser species survive as grazing commence on these sites after the fire. The balance between decreaser and increaser species is controlled mainly by the intensity of grazing and how long grazing has commenced on the post-fire site (Fig. 6.2.) As a change in species composition from decreaser to increaser dominated in Namaqualand Granite Renosterveld emerge, i.e. unpalatable shrub dominated shrubland, the condition is often regarded by farmers as unproductive. Furthermore, from a conservation perspective the species richness is high at this state but it's mostly dominated by *E. rhinocerotis* and *C. ciliata*. Therefore, this is also not a preferred state in terms of biodiversity.

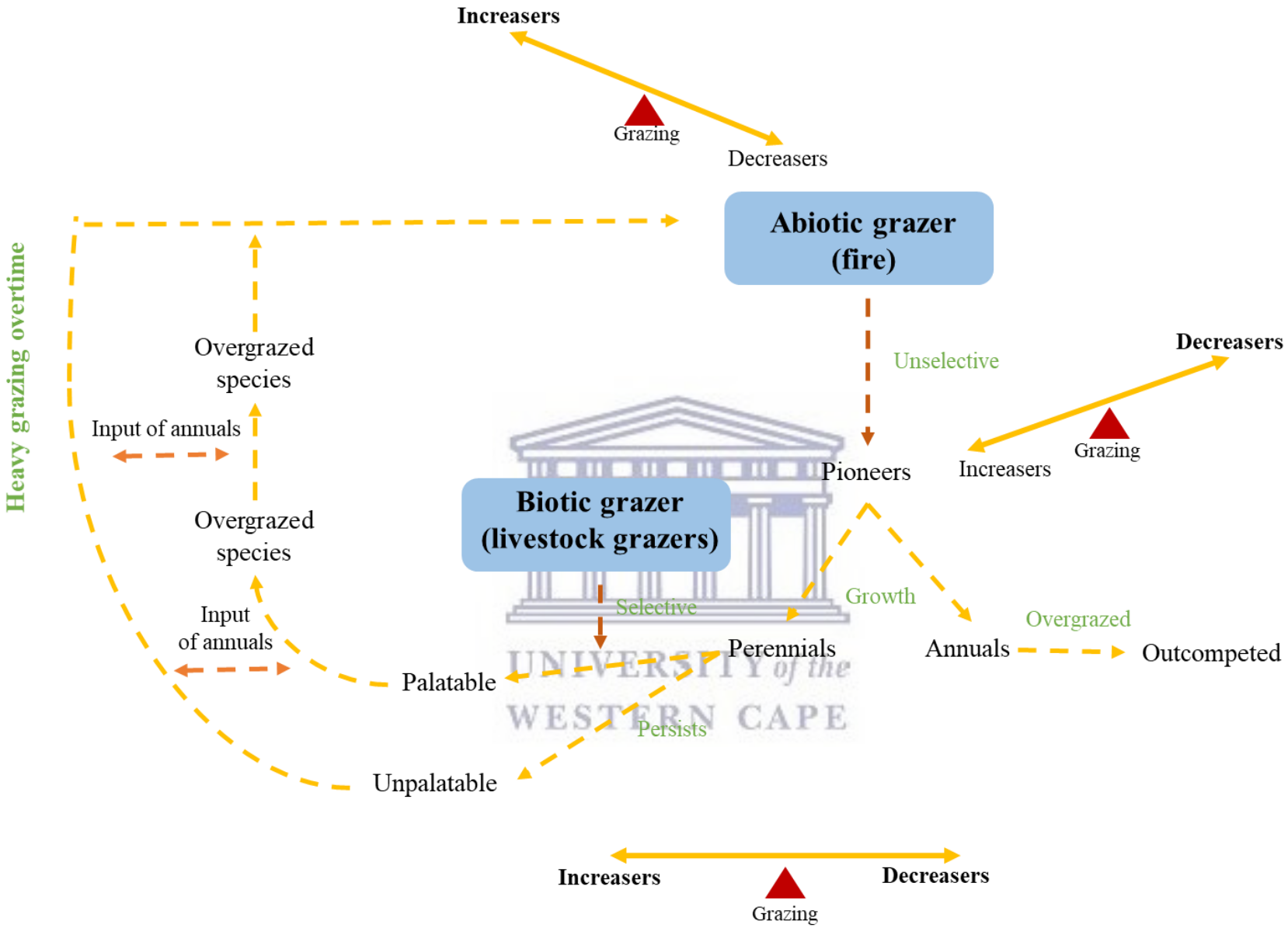


Figure 6.2: Conceptual model for the rangeland condition and biodiversity in relation to fire and grazing in Namaqualand Granite Renosterveld.

6.3. Dynamics of *Elytropappus rhinocerotis* (renosterbos)

As farmers rest their veld and move their animals seasonally, thus, altering the grazing patterns, farmers increase heterogeneity within the rangelands. Even though *E. rhinocerotis* replaces various palatable decreaser species over time, is not as dominant as it appears on the landscape. As after 10 years *E. rhinocerotis* creates a suitable microclimate for a few decreaser species to reproduce and as a result this indicates that older renosterveld is in a good rangeland condition (Chapter 5). As renosterveld is situated in an agricultural system, burning should be done in favour of forage productivity and burning renosterveld every seven years is ideal for forage productivity. Although decreaser species are dominant in older (> 10 years post-fire) sites, these sites do not have very high forage quality due to the change in ratio between increaser/decreaser species. Burning after seven years will also not have a negative effect on biodiversity, and may even favour local biodiversity.

6.4. Namaqualand Granite Renosterveld fire regime

Analysing the fire regime of Namaqualand Granite Renosterveld provided an understanding of the fire behaviour and its ecological role. Unplanned veld fires are occurring at an ecologically acceptable frequency and season. The fire regime of the Kamiesberg is quite broad as it incorporates both social and ecological elements. Fires cannot be eradicated from the Kamiesberg as it would be diminishing to the unique biodiversity. However, burning for wild honey harvesting could have more knock on effects on farmers (in terms of decrease in fodder availability) and animal biodiversity (e.g. decline in bees' populations, reptiles and small mammals). Specifically, areas which have been subjected to repeated fires are areas of concern. Public awareness and vigilance with regard to fires, specifically human ignited fires, has to increase. In so doing early identifications of fires could be detected before their spread becomes

too dangerous. Over time various authors have estimated the fire frequency of renosterveld (Table 6.1).

Table 6.1: Fire return intervals for Renosterveld vegetation in South Africa.

Authors	Year	Fire return period	Tenure	Herbivores
Cowling and Holmes	1992		All areas	Yes
Jacobs and Jangle	2008	3-5 years	All areas	Yes
Kraaij and Van Wilgen	2014		All areas	Yes
Cape Nature	2013	3-10 years	Commercial farming	Yes
Rebelo	1992	3-10 (up to 40) years	All areas	Yes
Pooley	2015	5-10 years	National Park	No
Forsyth *et al.*	2010	5-20 years	All areas	Yes
Kraaij	2010	6-7 years	National Park	Yes
Simons	**2017**	**7 years (current study)**	**Communal farming**	**Yes**
Jacobs and Jangle	2008		All areas	Yes
Helme and Rebelo	2005	10-15 years	All areas	Yes
Curtis	2013		Conservation reserve	Yes
Forsyth and Van Wilgen	2008	17.4-37.4 years	National Park	No

Although more research has been done in Lowland Renosterveld studies done by Walton (2006); Newton and Knight (2010), Curtis (2013) has highlighted that the vegetation in general requires further research. Mountain Renosterveld particularly in communal areas appears to be the 'black sheep' of renosterveld. as fewer studies have been done in terms of understanding its ecology. Most of the research conducted on renosterveld has also focused mainly on land use patterns and pressures such as grazing but excluded fire (Todd and Hoffman 1999; Anderson and Hoffman 2007; Allsopp *et al*. 2007; Anderson *et al*. 2010; Anderson and Hoffman 2011; Samuels 2013; Paulse 2013; Samuels *et al*. 2016). As a result, Mountain Renosterveld is mostly viewed as overgrazed and heavy grazing and a possible conservation

concern for this vegetation type. The dominance of *E. rhinocerotis* further perpetuate this notion of a homogenous vegetation. However, after the winter season the landscape is almost totally covered in ephemeral plant species.

6.5.Limitations of study

Mountain Renosterveld has not been studies as in-depth as Lowland Renosterveld and as a result literature was limited the former. In addition, literature on fire in renosterveld has also been limited. It is also likely that several fires have been overlooked due to NASA not picking up some of these fires, which could have had a low intensity and flame length. Also, the fire record is not entirely complete as there are a few uncertainties in the records about the sizes or dates of some fires. Consequently, these fires were not associated with weather conditions on the day of the fire. A limitation, particularly pre- 2000, was the inadequacy of wildfires reported and recorded. As no fire event was recorded for the areas. A suggestion would be to train a community member/s to compile a logbook thoroughly regarding all fire events (e.g. date, time, cause, flame direction, and climate conditions especially the rain after the fire).

Due to financial constraints, sampling immediately after fires were impossible. As a results, no immediate post –fire sites (0-5 months) were not sampled. At most sampling was only done during spring after the first rain. Again I would recommend training community members to compile a plant data base in terms of recording the names of species immediately after fire and at which stage these species disappears. Furthermore, noticing the species after the winter season before spring.

6.6.Recommendation for management

6.6.1. Fire management of Namaqualand Granite Renosterveld

Understanding the importance of fire for the conservation and management of renosterveld is important for biodiversity and ecosystem delivery in the Kamiesberg. Also to ensure that this fire-prone environment is safe and secure for the community, fauna and flora. Fires should be used as a management tool as it is relatively inexpensive and can be used to manipulate the vegetation (Bond and Van Wilgen 1996). Unfortunately, the Kamiesberg does not have a fire management plan. In the Kamiesberg long-term experiments will have to be conducted for practical fire management, as rangelands often have high fire frequencies. Currently the area is burning at a rate which the desired biological effect is achieved and the vegetation can sustain. If Namaqualand Granite Renosterveld is burnt too frequently (< 4.2 years) then adverse consequences could follow. Semi-arid ecosystems can be described as constantly variable vegetation mosaics changing in response to the ecological driving forces such as rainfall, grazing intensity and fires (Scholes and Walker 1993; Belsky 1994). Due to the complexity and slow growing nature of the vegetation in the Kamiesberg, too frequent fires should not be allowed. Namaqualand Granite Renosterveld has different stable states due to both fire and grazing, and neither of these two disturbances should be ignored in the management of Namaqualand Granite Renosterveld.

It is also important to note that each study site is highly variable and complex as topographical (aspect and slope), biological (variable climate) and management (grazing intensity) factors affects them in different ways. Simons (2013) showed that the vegetation returns to a dense and unpalatable state (initial state) after 12-year post-fire. Thus, a management plan should focus on these denser areas, ensuring that they burn under controlled environments to make sure that the vegetation is beneficial to both local biodiversity and communal farmers for their

livestock in terms of fodder productivity. Considering the current fire regime fires should only be applied in areas where they have been suppressed > 10 years. As man-made fires do not burn the entire area. The vegetation is slow growing thus fires should not be applied to areas which have not recovered.

Fires should be applied for:

- Livestock production as this is dependent on the quality and quantity of plant fodder, which can be influenced by manipulating the fire regime (Bond and Van Wilgen 1996).
- To reduce and control the encroachment of unpalatable shrubs of undesirable plants (Trollope 2004).
- Reducing the fire hazard and prevent intense wildfire by removing moribund material (reduce fuel-build up).

These fires can be implemented at different frequencies, intensities and seasons.

Some of the constrains in using fire as a management practice could be

- Ecological: fires might have undesirable effects on the vegetation composition if applied at the incorrect season, frequency or intensity.
- Safety: Most of the fires in the Kamiesberg are wildfires. The danger in these fires escaping and burning closer or even in the Leliefontein village could not only damage property but loss of lives as well.
- Temporal: the seasonal differences in weather can result in conditions not suitable to fires and can be extremely dangerous (large uncontrolled fires)

Unplanned human ignitions remain the biggest ignition sources. This is due honey harvesting; there is a great need to train children on how to harvest honey safely and sustainably. Fire training is imperative in the area. Training community members, could not only create jobs but

this can prevent the spread of large fires as they will be able to detect these fires early enough and would be able to control them.

6.7.Future research

Future fire research topics in relation to Namaqualand Granite Renosterveld dynamics have been identified.

1. Short-term effects of an unplanned veld fire on soil characteristics and nitrogen availability in Namaqualand Granite Renosterveld.
2. Long-term effects of unplanned veld fires on biochemical quality of soil organic matter in Namaqualand Granite Renosterveld.
3. The influence of Patch Mosaic Burning on small mammals in Namaqualand Granite Renosterveld: testing the “pyrodiversity begets biodiversity” paradigm.
4. The impacts of fire and grazing on the recruitment and survival of small mammals in Namaqualand Granite Renosterveld.

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Appendices

Appendix 1: List of plant species associated with *E. rhinocerotis* across all study sites in Namaqualand Granite Renosterveld. B= Burnt area and U=unburnt area.

Plant species	Hoorngaat (2000)		Skaaprivier (2001)		Xharras 2 (2002)		Die Vlak (2003)		Toringberg (2004)		Groenkloof (2005)	Xharras1 (2008)		Kliphoek (2009)		Populierskloof (2013)		Xharras 3 (late 2013)	
	B	UB	B	UB	B	UB	B	UB	B	UB	*B	B	UB	B	UB	B	UB	B	UB
Anisodontea bryoniifolia			25								10		10			364		5	
Anthospermum spathulatum	2	5	1		18				69	7	1	21	6	2			18		19
Arctotis fastuosa			2	10			2									1			
Aspalathus Spinosa			29	1			2		28			6		4	1		3		1
Chaetobromus involucratus																		82	9
Chrysocoma ciliata		128	4	16		29	57	15		1	3	15	62	3	48	21	21		7
Cliffortia ruscifolia			3		2	31			26			6	6	2	12		7		
Conicosia elongata												6		6					
Cotula barbata				1			6									200	4	22	
Crassula natans						18	1								1	4		98	
Crassula spp.		2									1								
Didelta spinosa											6				2				
Diosma acmaeophylla	8				10	2		1	1								15		17
Dodonea angustifolia	4						1				1			24			1		
Ehrharta calycina		5		4				6				1		3	2	7			
Elytropappus rhinocerotis							11		9			35		9			10	52	5
Eriocephalus brevifolius									2										2
Eriocephalus microphyllus	30	7	1			54	1	4	15	15		5	5	3			32		88
Euphorbia mauritanica		12																	

Euryops lateriflorus		1			1	2		1	2	24						3	8		1
Felicia australis							30											201	
Ficinia indica					108	8	2						5		26				
Ficinia nigrescence					38								14						
Fingerhuthia africana												5							
Galenia africana		4																	
Grass spp.			2		6	53						20	16	5			10	5	14
Haworthia arachnoidea									1										
Helichrysum hamulosum	21	23	26	15	40	50	29	34	52	102	84	27	7	39		35	14	3	52
Helicrysum tinctum																		20	
Heliophila variabilis																		2	
Hermannia amoena			6															10	
Hermannia disermifolia		3																	
Hessea breviflora											20								
Hyobanche sanguinea							1	12							1		1		1
Indigofera nigromontana														10					
Ischyrolepis gossypinus		4						8	41				15	29			43		5
Karroochloan schismoides			15	3										29	252	1		3	10
Lachenalia mutabilis				1			1									10		2	
Lachenalia violacea																		1	2
Leipoldtia spp.														7					
Leobordea digitata			4		76	3			2	2		24	10		1	7			
Leucospermum rodolentum												3	18						
Leysera gnaphalodes														5	5				
Lycium cinereum					2									6					
Massona depressa				8					1			1						1	
Mesembryanthemum crystallinum															1	4		12	
Moraea miniata									7									42	
Moraea serpentina				1												1			1

Moraea tortilis																		19	
Muralthia spinosa			3			1			19		4					1			
Osteospermum pinnatum	2	3																	
Othonna spp.																			1
Oxalis namaquana				6											18				
Oxalis spp.					3	52	1	53	3							12	2	3	
Passerina truncata											2	11	3			5		12	
Pelargonium spp.													2	1					
Pelargonium incrassatum											5					2		5	
Phylica rigidifolia		8									2	5	3			13			
Polygala leptophylla		1							2										
Pteronia divaricata															3				
Pteronia glaborata															1			5	
Pteronia incana	2													18					
Pterygodium hallii															1				
Roepera morgsana				2				1						13		3			
Romulea citrina						12													
Romulea kamisensis																		6	
Ruschia robusta	3	275					1		3		2	3	20			5			3
Ruchia spp.		9							2										
Salsola kali																		6	
Salvia dentata			1			1	11	1											3
Sarcocaulon l'heritieri											17								
sedge spp.					16	1		4											
Selago divaricata				8	4					2		3	3	2		12			
Senecio cinerascens						1	1	6					5						
Spiloxene serrata				13															
Stachys rugosa	4		37		3			68							11				
Stipagrostis ciliata															5				9

Struthiola ciliata							4								23		
Struthiola leptantha		4		36	10	3	3	16			3			2	19		
Sutherlandia frutescens	2								1								
Tribolium utriculosum	2		6						11						1		5
Tripteris sinuata			1						18								
Viscum capense	2		1							1			7				
Wahlenbergia spp.																4	
Wiborgia spp.												22					
Wildenowia incurvata				23						204	1				9		

* No individuals of *E. rhinocerotis* were found in the unburnt vegetation at Groenkloof

Appendix 2: List of perennial species and their ecological status in the Kamiesberg. Asterisks reflect in which sites species were found: B= Burnt area, U=unburnt area, C= found in burnt and unburnt areas.

Ecological status	Burnt	Unburnt	Both
Decreaser species			
Perennial graminoids			
Chaetobromus involucratus			*
Ehrharta calycina			*
Ficinia indica			*
Ficinia nigrescence			*
Fingerhuthia africana	*		
Schismus schismoides			*
Stipagrostis ciliata			*
Perennial shrubs			
Agathosma capensis	*		
Anthospermum spathulatum			*
Arctotis fastuosa			*
Aspalathus Spinosa			*
Chrysanthemoides monilifera	*		
Cliffortia ruscifolia			*
Didelta spinosa			*
Diosma acmaeophylla			*
Eriocephalus brevifolius			*
Eriocephalus microphyllus			*
Hermannia disermifolia		*	
Indigofera nigromontana	*		
Leobordea digitata			*
Muraltia spinosa			*
Passerina truncata			*
Pelargonium incrassatum			*
Pelargonium spp.			*
Polygala leptophylla		*	
Pteronia divaricata	*		
Pteronia glaborata			*
Struthiola ciliata			*
Struthiola leptantha			*
Sutherlandia frutescens			*
Tripteris sinuata			*
Wiborgia monoptera	*		
Perennial succulents			
Roepera morgsana			*
Ruschia robusta			*

Ruschia spp.			*	
Salsola kali	*			
Increaser species				
Perennial graminoids				
sedge	*			
Tribolium utriculosum				*
Ischyrolepis gossypina				*
Wildenowia incurvata				*
Perennial shrubs				
Anisodontea bryoniifolia				*
Chrysocoma ciliata				*
Elytropappus rhinocerotis				*
Euryops lateriflorus				*
Galenia africana			*	
Helichrysum hamulosum				*
Helicrysum tinctum	*			
Hermannia amoena	*			
Lysera gnaphalodes				*
Monsonia spinosa	*			
Othonna spp.			*	
Phylica rigidifolia				*
Pteronia incana				*
Pterygodium hallii	*			
Salvia dentata				*
Senecio cinerascens				*
Stachys rugosa	*			
Perennial succulents				
Euphorbia mauritanica			*	
Perennial trees				
Dodonea angustifolia				*
Unknown status				
Perennial shrubs				
Leipoldtia spp.	*			
Leucospermum rodolentum				*
Lycium cinereum	*			
Osteospermum spp.				*
Selago divaricata				*
Perennial succulents				
Haworthia arachnoidea	*			

Appendix 3: The difference between growth forms in the burnt (B) and unburnt (UB) renosterveld sites. *N* = 10 for each site.

Growth forms	Mean	Std.Dv.	N	Mean diff.	Std.Dv. Diff.	*t* -value	df	*p* - value
B_Shrubs	12.0	3.3	10	-2.0	3.7	-1.7	9	0.12
UB_Shrubs	14.0	3.6						
B_Shrubs	12.0	3.3	10	9.8	2.8	11.0	9	**0.00****
UB_Graminoids	2.2	1.0						
B_Shrubs	12.0	3.3	10	9.3	3.3	8.8	9	**0.00****
UB_Succulents	2.7	1.9						
B_Shrubs	12.0	3.3	10	7.7	3.9	6.2	9	**0.00****
UB_Herbs	4.3	1.9						
B_Shrubs	12.0	3.3	10	11.6	3.7	9.9	9	**0.00****
UB_Trees	0.4	0.7						
B_Graminoids	1.9	0.9	10	-12.1	3.4	-11.3	9	**0.00****
UB_Shrubs	14.0	3.6						
B_Graminoids	1.9	0.9	10	-0.3	0.8	-1.2	9	0.28
UB_Graminoids	2.2	1.0						
B_Graminoids	1.9	0.9	10	-0.8	1.9	-1.4	9	0.21
UB_Succulents	2.7	1.9						
B_Graminoids	1.9	0.9	10	-2.4	2.1	-3.7	9	**0.01***
UB_Herbs	4.3	1.9						
B_Graminoids	1.9	0.9	10	1.5	1.3	3.7	9	**0.00****
UB_Trees	0.4	0.7						
B_Succulents	1.8	1.9	10	-12.2	2.9	-13.1	9	**0.00****
UB_Shrubs	14.0	3.6						
B_Succulents	1.8	1.9	10	-0.4	1.8	-0.7	9	0.49
UB_Graminoids	2.2	1.0						
B_Succulents	1.8	1.9	10	-0.9	1.5	-1.9	9	0.09
UB_Succulents	2.7	1.9						
B_Succulents	1.8	1.9	10	-2.5	2.8	-2.8	9	**0.02*****
UB_Herbs	4.3	1.9						
B_Succulents	1.8	1.9	10	1.4	2.3	1.9	9	0.08
UB_Trees	0.4	0.7						
B_Herbs	3.8	5.1	10	-10.2	4.4	-7.3	9	**0.00****
UB_Shrubs	14.0	3.6						
B_Herbs	3.8	5.1	10	1.6	5.0	1.0	9	0.34
UB_Graminoids	2.2	1.0						
B_Herbs	3.8	5.1	10	1.1	5.2	0.7	9	0.52
UB_Succulents	2.7	1.9						
B_Herbs	3.8	5.1	10	-0.5	5.7	-0.3	9	0.79
UB_Herbs	4.3	1.9						

B_Herbs	3.8	5.1	10	3.4	5.3	2.0	9	0.07
UB_Trees	0.4	0.7						
B_Trees	0.9	0.7	10	-13.1	3.4	-12.3	9	**0.00****
UB_Shrubs	14.0	3.6						
B_Trees	0.9	0.7	10	-1.3	1.2	-3.5	9	**0.01***
UB_Graminoids	2.2	1.0						
B_Trees	0.9	0.7	10	-1.8	1.8	-3.3	9	**0.01***
UB_Succulents	2.7	1.9						
B_Trees	0.9	0.7	10	-3.4	2.1	-5.2	9	**0.00****
UB_Herbs	4.3	1.9						
B_Trees	0.9	0.7	10	0.5	0.7	2.2	9	0.05
UB_Trees	0.4	0.7						

Significance is shown at *p<.01, **p<.001, ***p<0.05

Appendix 4: List of species recorded in each 100 m^2 in the Kamiesberg. Species are arranged according to a) response to fire and b) growth forms. Asterisks reflect in which sites species were found: B= Burnt area, U=unburnt area, C= found in burnt and unburnt areas. The species recovery strategies (R= resprouter and S= seeder) after fire are included.

Species	B	U	C	Recovery strategy
Fire sensitive species				
Perennial dwarf shrubs				
Chironia baccifera	*			
Hirpicium alienatum		*		
Perennial graminoids				
Pentaschistus spp.	*			
Stipa capensis	*			S
Stipagrostis ciliata		*		
Wildenowia incurvata		*		
Perennial herbs				
Albuca cooperi	*			S
Babiana dregei			*	
Ficinia indica		*		
Gazania leiopoda	*			
Hessea breviflora		*		
Hesperantha pauciflora	*			S
Lachenalia mutabilis			*	
Leysera gnaphalodes		*		
Moraea herrei	*			S
Moraea miniata	*			S
Moraea tortilis			*	S
Moraea schlechteri		*		S
Moraea serpentina	*			S
Moraea spp.	*			S
Oxalis spp.			*	S
Romulea citrina	*			S
Pelargonium incrassatum		*		R
Pelargonium spp.		*		R
Spiloxene serrata		*		S
Perennial semi-parasite shrub				
Viscum capense			*	
Perennial shrubs				
Anisodontea bryoniifolia			*	S
Asparagus spp.			*	R
Chrysanthemoides monilifera			*	S
Didelta spinosa		*		
Diospyros austro-africana			*	R
Eriocephalus brevifolius		*		R
Euryops lateriflorus			*	S

Gomphocarpus cancellatus	*			
Gomphocarpus fruticosus	*			
Haworthia arachnoidea		*		
Helichrysum hebelepis			*	
Indigofera nigromontana		*		
Indigofera spp.	*			
Lebeckia sericea		*		R
Leipoldtia spp.		*		
Lessertia frutescens	*			R
Leucospermum rodolentum			*	R
Polygala leptophylla	*			
Pteronia spp.	*			
Salsola kali		*		
Salvia dentata			*	S
Selago divaricata		*		S
Senecio cinerascens			*	S
Seriphium plumosum		*		S
Struthiola ciliata			*	S
Tetragonia spp.	*			S
Tripteris sinuata		*		S
Perennial trees				
Searsia undulata			*	R
Fire-tolerant species				
Perennial dwarf shrubs				
Hermannia amoena	*			S
Hermannia disermifolia		*		S
Phylica rigidifolia			*	
Selago divaricata	*			
Perennial graminoids				
Grass			*	
Ischyrolepis gossypina			*	S
Sedge			*	S
Perennial herbs				
Leobordea digitata			*	S
Massonia depressa			*	
Perennial shrubs				
Anthospermum spathulatum			*	R
Aspalathus spinosa			*	R
Chrysocoma ciliata			*	R/S
Cliffortia ruscifolia			*	R
Diosma acmaeophylla			*	
Elytropappus rhinocerotis			*	R
Eriocephalus microphyllus			*	R
Helichrysum hamulosum			*	R/S

Nylandtia spinosa			*	R
Passerina truncate			*	
Pentzia incana			*	
Pteronia divaricata			*	R
Pteronia incana			*	
Stachys rugosa			*	R/S
Struthiola leptantha				R
Perennial trees				
Dodonea viscosa	*			R/S
Searsia pyroides		*		R