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***Long-term vegetation change in the Cape of Good Hope  
Section of Table Mountain National Park, in response to  
climate, fire and land use***

Robyn Faye Powell

Supervised by Prof. Timm Hoffman and Assoc. Prof. Lindsey Gillson





University of Cape Town

Faculty of Science

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***Long-term vegetation change in the Cape of Good Hope Section  
of Table Mountain National Park, in response to climate, fire and  
land use***

**Robyn Faye Powell**

Supervisor: Prof. Timm Hoffman (Plant Conservation Unit, UCT)

Co-supervisor: Associate Professor Lindsey Gillson (Plant Conservation Unit,  
UCT)

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# Abstract

Climate change, fire and land use are known threats to biodiversity and are predicted to have a major impact on the Cape Floristic Region of South Africa over the next fifty years. This study investigated long-term (1966-2012) changes in the vegetation of the Cape of Good Hope Section of Table Mountain National Park in response to 20<sup>th</sup> century changes in climate, fire and land use. Climate variables and fire history were assessed over the last 100 years using linear regression, segmented regression and GIS analyses. Seventy eight repeat photographs were taken at stratified points set out across the reserve and changes in total vegetation cover and the cover of major growth forms (broad leaved shrubs, proteoids, ericoids and restioids) within three main vegetation types (Dune Asteraceous Fynbos, Restioid Fynbos and Proteoid Fynbos) were assessed. Transition matrices were used to investigate the extent of vegetation change likely over the next fifty years. In a separate study changes in the number of individuals as well as the cover of several Proteaceae species (*Mimetes fimbriifolius*, *Leucospermum conocarpodendron* and *Leucadendron* spp.) were also investigated using the repeat photographs. Results from linear regression analyses showed that the majority of climatic variables did not change significantly over the course of the 20<sup>th</sup> century with the exception of maximum temperature and relative humidity. Maximum temperature increased significantly by 0.36 °C per decade, while relative humidity decreased by -1.9%. The number of fires occurring over a five year period was greater for the period before 1965 than the period between 1966 and 2010. Since 1975, however, fire frequency has not increased significantly. Changes in land cover classes (sparse, low, medium and high) derived from a comparison of aerial photographs between 1945 and 2008 indicated that there has been no significant change in vegetation cover at CGH although the number of separate patches of different land cover classes has decreased. Comparisons of the repeat ground photographs from 1966 showed that the change in vegetation cover varied considerably between sites across the reserve in response to fire and a range of different biotic and abiotic disturbances. In many cases there was very little difference between the two photographs (1966-2012) in percentage vegetation cover and growth form composition.

Dune Asteraceous Fynbos sites appeared particularly vulnerable to local disturbances, such as invasion of mole rats (*Bathyergus suillus*), anthropogenic impacts and even dune remobilization. Sites within Restioid Fynbos, however, showed very little change in cover, even when they differed in post fire age. Changes within this vegetation types were experienced primarily as changes in the dominance of key growth forms which were sometimes linked to the removal of alien invasive species and the expansion of seepage areas. Changes within Proteoid Fynbos sites were strongly correlated with difference in post fire age which explained most of the variation at these sites. Markov transition matrices project a decline in broad leaved shrubs with a corresponding increase in ericoid shrubs in Dune Asteraceous Fynbos. In other vegetation types (Restioid Fynbos and Proteoid Fynbos) key growth forms remained relatively unchanged when projected into the future. Populations of emergent Proteaceae shrubs appeared stable across the reserve, with site-specific increases or decreases. Fire frequency was a significant predictor of percentage cover change of *Leucadendron spp.* but did not influence the number of individuals of *Mimetes fimbriifolius* and *Leucospermum conocarpodendron* in a population over the period of study. Although no significant trend in rainfall was detected in this study, some aridification effects, linked to increases in temperature, might be expected due to increased water stress on plants which in turn might be expected to slow vegetation recovery after fire. The results of this study, however, do not support this hypothesis of aridification and no consistent decline in vegetation cover was recorded across all vegetation types. These findings support the conclusions of other plot-based studies at the CGH which suggest that the vegetation is stable at a meta-community scale. Future monitoring of vegetation change is essential in detecting further changes in the vegetation of CGH in response to climate, fire and land use.

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# Chapter I: Introduction

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## **The Cape Floristic Region**

The Cape Floristic Region (CFR) is situated in the south western part of South Africa and is a unique and dynamic region. It is regarded as one of six floral kingdoms in the world (Good, 1974; Takhtajan, 1986) and contains high species diversity as well as hosting a number of endemic plant and animal species (Cowling et al., 1997; Myers et al., 2000). Although covering less than 5% of the land area of South Africa, it is a botanically diverse region with approximately 9030 vascular plants, or about 44% of South Africa's flora (Goldblatt and Manning, 2002). The fynbos biome lies within this region and supports a diverse and highly endemic flora (69%) with many habitat specialists. The diverse composition of families and genera is also noteworthy (Bond and Goldblatt, 1984; Goldblatt, 1997; Goldblatt and Manning, 2000). There are three main families that characterise the fynbos biome: Proteaceae, Ericaceae and Restionaceae (Cowling 1995) although the Asteraceae, Fabaceae and Iridaceae are the most speciose families in the biome (Goldblatt and Manning, 2000). These dominant families contain many species of medicinal importance, as well as of commercial and cultural value (Goody, 1993). The fynbos is also dominated by certain growth forms with the presence of fine-leaved sclerophyllous shrubs being the most prominent. There is also a significant number of geophytes and a low presence of trees in this biome (Goldblatt and Manning, 2002).

## **Threats to the Fynbos Biome**

The three main drivers of vegetation change in the Cape Floristic Region are climate, fire and land use. Each of these drivers influence changes in vegetation composition and cover in isolation as well as sometimes in combination.

## ***Climate Change***

Climate change increasingly poses a major threat to ecosystems globally, causing species range shifts, population declines and extinctions (Hughes, 2000; McCarty, 2001; Parmesan and Yohe, 2003; Root et al., 2003). Climate change is also expected to be a major threat to the biodiversity in the CFR (Rutherford et al., 1999; Midgley et al., 2002, 2003). Projections for the CFR indicate that the winter rainfall region of southern Africa will be one of the regions in the world most affected by climate change (Midgley et al., 2005). By 2050 temperatures are expected to have increased by between 1.5 to 3.5 °C (Haensler et al., 2010) and annual rainfall totals to have decreased by up to 20% (Midgley et al., 2005). Climate change is predicted to have a significant impact on fynbos vegetation with models predicting extensive range contractions of some species. Under extreme scenarios, a loss of up to 30% of the Cape Floristic Region's biodiversity is expected (Midgley et al., 2003; Midgley et al., 2005). These projections portray a negative impact of climate change on the fynbos biome, which could lead to a significant reduction in diversity or extinction of some species.

The fynbos biome exists in a relatively restricted winter rainfall region bounded by oceans to the west and south and by arid environments to the north and east. A shift in a species' climate envelope might lead to range contraction and local extinction as the potential for species to shift their ranges is low. Also, many fynbos plants are pollinated by insects, birds and small mammals (Graham and Grimm, 1990; Warren et al., 2001), and if the species' range shifted, then the distribution of the animals that pollinate it may not shift as quickly or not at all. This could result in a failure of pollination and seed set which could lead to a decline in populations over time (Parmesan and Yohe, 2003). The possibility of a drier and hotter South Western Cape in accordance with future climate change projections therefore poses a significant threat to the fynbos biome. Aridification in the region could further contract the fynbos biome as surrounding arid areas may expand.

## ***Change in Fire Regime***

Fire is an essential and important factor of disturbance in the history and biology of fynbos (Cowling et al., 1997). As fire is a natural disturbance in fynbos, the majority of species have adapted to fire. Fire plays a major role in the life-cycle of many fynbos species as they rely on it to release seed (Mutch, 1970). Some Proteaceae species have serotinous cones, which when burnt, release seeds, that are then dispersed by wind (Lamont et al., 1991; Keeley and Fotheringham, 2000). Germination in some species of Proteaceae, Ericaceae and Restionaceae has shown to be triggered by smoke (Brown and Botha, 2004). As fire is so strongly linked to germination of the many species in the fynbos and is required to burn moribund material, the fire regime is of high importance. Both infrequent and too frequent fires can threaten the biome. For instance, if fires are too frequent, seedlings of species will not be able to grow, flower and release seeds and will as a consequence of this not be able to re-establish the next generation. Conversely, as many species require fire to germinate, infrequent or no fire could also significantly affect fynbos species negatively. Alterations in fire intensity and fire seasonality can also influence species composition and survival in the fynbos biome (Bond and van Wilgen, 1996; Keeley, 2009). The fire regime will also likely be altered with the expected aridification. A drier and hotter winter rainfall region will result in more frequent and possibly more intense fires in the future (Forsyth and van Wilgen, 2008).

## ***Land Use***

Land use is considered to be one of the major drivers of global biodiversity change in terrestrial ecosystems (Sala et al., 2000). Destruction and degradation of ecosystems globally is mostly due to agriculture, invasive alien plants and urbanisation. These land use practices are the main cause of past and present species endangerment and extinction worldwide (Czech and Krausman, 1997; Wilcove et al., 1998). In the fynbos biome, these land use practices result in habitat transformation and have also been identified as major past, present and future threats to biodiversity (Rebelo, 1992; Richardson et al., 1996; Rouget et al., 2003; Latimer et al., 2004). With increasing human population, these threats are expected to put even more pressure on the fynbos biome,

its species and ecosystem processes, especially the lower-lying regions which are more in demand for agriculture and urbanisation.

## **Objectives and Key Questions**

Given the threats to fynbos vegetation mentioned above, there is an important role for long-term studies that record and monitor changes in fynbos vegetation, such as the work being carried out by the South African Environmental Observation Network (SAEON). Long-term records provide an important means to assess the impacts of changes in key drivers such as climate, land use and fire on fynbos vegetation over historical time periods. An understanding of long-term trajectories of change can also be used to make predictions about future change.

However, long-term data sets for the fynbos biome are rare. Most information about the biome has been derived from short-term field studies typically associated with student dissertations. Some longer term research programmes, such as the Fynbos Biome Project (Kruger 1978) operated for a few decades in the 1970s and 1980s but these have not been continued. One of the most important means of assessing long-term change is to resample previous vegetation surveys of an area such as was undertaken by Privett (1998) and Hall (2010) who resampled Hugh Taylor's 1966 survey plots at the Cape of Good Hope reserve. Another approach, which has not been undertaken in the fynbos biome is to use historical photographs to document vegetation change. This provides a useful tool for understanding long-term change often at temporal scales of decades or longer (Webb et al., 2010).

The overall objective of this study was to document the extent, nature and rate of vegetation change in the Cape of Good Hope reserve using repeat photography and to relate these changes to the key drivers of climate, fire and land use. Changes in total vegetation cover as well as in key growth forms and species (e.g. emergent Proteaceae) formed part of this investigation.

Specific questions and anticipated outcomes include the following:

**i) How have the main drivers of change (climate and fire) in the Cape of Good Hope reserve changed over the last 100 years?**

It has been widely reported that temperatures have increased over the course of the 20<sup>th</sup> century across southern African while changes in rainfall are equivocal (Haensler 2010, Midgley et al. 2005). Fire frequency has increased on the Cape Peninsula since the 1970s (Forsyth and van Wilgen 2008) and is expected to increase both as a result of anthropogenic ignitions and increased aridification as a result of climate change.

**ii) How has total vegetation cover changed over a 46 year period in relation to post fire age?**

An assessment of the difference in post fire age provides a critical lens through which to compare and interpret vegetation change in the fynbos biome. Using this approach previous studies in the Cape of Good Hope reserve have reported stability in the vegetation at a meta-community level (Thuiller et al., 2008) although recent analyses suggest a loss of diversity in response to climate change (Hall 2010). With an increase in temperature and no significant change in annual rainfall amounts it is possible that vegetation cover in similar post-fire age stands would be lower in 2012 when compared to 1966 and more rocks and bare soil would be exposed.

**iii) How has the dominance of key fynbos growth forms (broad leaved shrubs, proteoids, ericoids and restioids) changed over a 46 year period in relation to post fire age and how might they change in the future?**

Change in the abundance of key growth forms provide some measure of the trajectory of environmental change. For example, West et al., (2012) have shown that shallow-rooted ericoid shrubs are susceptible to drought impact while obligate reseeding proteoid shrubs are negatively affected by fire frequency (Enright and Goldblum, 1999). Broad leaved shrubs,

especially those associated with coastal thickets should be relatively stable over time. Restioid growth forms, especially those which dominate seepage areas should decline under conditions of increased aridification. Changes in growth forms, however, are strongly influenced by the post fire age of vegetation (Rutherford et al. 2011) and this should be taken into account when assessing growth form change in fynbos vegetation. Furthermore, the future for fynbos vegetation under a range of climate change scenarios has been described as catastrophic by some researchers (e.g. Cowling et al., 2003; Midgley et al. 2003). Given that the rate of change in the growth forms under study can be observed over years to decades, it is possible, using transition matrices, to provide some guidance as to the likely future changes, at least over the short to medium term period of years and decades for the next time step. This approach has numerous assumptions not least of which is the assumption that the main drivers of growth form change, such as climate, fire and land use, will not depart beyond the critical threshold conditions needed for the continued persistence of the majority of species and growth forms in the region. Such threshold conditions could include moisture availability (West et al., 2012) or pollinator presence (Pauw., 2007) which have implications for the growth, survival and reproduction of fynbos species.

**iv) How have populations of large reseeding Proteaceae shrubs changed over a 46 year period? And what is their response to climate and fire?**

With anticipated aridification and increases in fire frequency, the populations of large reseeding Proteaceae are expected to decline across the reserve. Although many Proteaceae require fire to germinate, the fire return interval is expected to become shorter in the future which would impact on germination and the establishment of new propagules. Evidence from climate change projections indicate that obligate reseeding Proteaceae will be particularly susceptible to the combined impact of aridification, a reduction in the fire return interval and land use change

(Midgley et al., 2002; Bomhard et al., 2005; Forsyth and van Wilgen, 2008; Wilson et al., 2010). Scale is important when determining the main driver that affects populations. Climate is a regional driver of change and if it is driving changes in Proteaceae populations, we would expect a relatively uniform trend across all populations throughout the reserve. However, fire occurs at a more local scale and affects relatively small patches of the reserve at a time. If fire is the main driver of change, site specific changes would be more commonly observed.

## Thesis Outline

This thesis is separated into an introductory chapter, three main data analysis chapters and a final conclusion chapter which expands on the conservation implications of the main findings. Chapter I introduces the main threats to the fynbos biome and provides the main objectives and key questions of the study. Chapter II provides a description of the study area and history of land use practices at the Cape of Good Hope reserve. Chapter III presents an analysis of long-term change in climate, fire and land cover change within the reserve. Each of these drivers are analysed and described separately as they are different factors which are expected to have an impact on the vegetation of the reserve. Chapter IV draws on 78 repeat photograph pairs derived from Hugh Taylor's 1966 original imagery and which were re-photographed in 2010 to assess the extent, nature and rate of vegetation change found in major growth forms. The growth forms assessed were broad-leaved shrubs, proteoid shrubs, ericoid shrubs and restioids. Observing and analysing how vegetation cover has changed over a 46 year period is a proxy for understanding broad environmental changes in the reserve over this period. In Chapter V, I analyse the change in abundance of large reseeding Proteaceae species (*Mimetes fimbriifolius*, *Leucospermum conocarpodendron* and *Leucadendron* spp.) between 1966 and 2012 using repeat photographs. Many of these species provide Cape Point with its unique character and managers of the reserve are concerned that these species have declined over the last several decades. The thesis concludes (Chapter VI) with a brief synthesis of the main findings and outlines future research directions. It also

provides an assessment of the value of repeat photography in long-term fynbos biome research.

# Chapter II: Study Site Description and History of Land Use at Cape of Good Hope

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## **Location and Topography**

The Cape of Good Hope Section of Table Mountain National Park (referred to as “Cape of Good Hope” or the “reserve”) lies in the south western part of the Table Mountain National Park and includes the most south-westerly tip of South Africa (Figure 1). The reserve forms part of the Cape Folded Belt with quartzitic sandstones the dominant surface rock over most of the reserve. There is a low plateau rising from an altitude of 60 m with the highest peaks in the east. These peaks reach an altitude of 360 m and are comprised of Judas Peak, Paulsberg and Vasco da Gama Peak (Privett, 1998). The reserve forms a point towards the south and is surrounded by the Atlantic Ocean. The west coast is more exposed and rugged while the east coast faces towards False Bay and is thus more sheltered from the heavy Atlantic swells which impact on the coastline. The interior of the reserve is characterised by a large level area with scattered seepage sites where water is trapped and marsh conditions are prevalent throughout the year (Privett, 1998). The soils in the area are nutrient poor and are shallow, sandy and well-drained. There are small areas where the soils support a high level of organic matter, but these areas are not common throughout the reserve (Privett, 1998).



**Figure 2.1: Map of the Cape of Good Hope showing different areas (mostly old farms) in the reserve.**

## **Land Use**

Understanding land use and its impact on the vegetation in the Cape of Good Hope Section of Table Mountain National Park (hereafter referred to as the reserve) is a challenging and interesting task. The area has been home to San hunter-gatherers, Khoi-khoi nomadic pastoralists and settler farmers, as well as numerous fauna over the years. A somewhat dated, but nonetheless comprehensive study done by Opie (1967), sums up the history of the reserve. The area is now owned by South African National Parks (SANParks) who manage it in accordance with National Park management practices. This region, however, has a fascinating history and many different land use activities have taken place in the reserve over time. This section describes the history of land-use in the reserve and puts this in a wider context of settlement and human history of the Cape Peninsula.

### ***Early Humans on the Cape Peninsula***

For early humans, the Cape Peninsula was a difficult place to survive with scant provisions of food, shelter from the elements and protection from predators. In the summer, droughts were common, with few sources of fresh perennial water. Mountain fynbos vegetation also does not provide an abundance of edible fruits or other easily accessible sources of nutrition, unlike other vegetation types and areas (Opie, 1967). This however, has not deterred hominids from inhabiting the area for the past half a million years (Deacon 1983, Deacon 1992). There is also evidence that fire has been used by humans in this region for the last 100 000 years with the existence of formal hearths found in stone-age dwellings, dating from the beginning of the Late Pleistocene (Deacon, 1983).

Nomadic pastoralists, Khoi-khoi had established themselves in the Cape about 2000 years ago (Schweitzer, 1979). Evidence of Khoi-Khoi in the reserve was found in a shelter at Smitswinkel Bay where Poggenpoel & Halkett, (1990) found sheep and cattle bones. Khoi-khoi pastoralists often moved livestock, in this case sheep and cattle, with them and are known to have burnt land for improved grazing for their livestock (Neumann et al.,

2011). These practices could have increased the presence of fire in the reserve, thus affecting the landscape. Vegetation composition may have been altered somewhat as well as vegetation cover.

### ***The Portuguese Explorers***

During the 15<sup>th</sup> century, Portuguese sailors were sent to the east as there was a great demand for products from India (Opie, 1967). The sailors navigated around the coast of Africa to guide them to their destination. Bartolomeu Diaz was the first European explorer to circumvent the coast of the Cape in 1487 (Opie, 1967). His journey however was not easy with huge roller waves and high winds that pounded his light ship and therefore named the Cape “Cabo Tormentas” (the Cape of Storms). Later, in 1488, the King changed it to “Cabo da boa Esperança” – the Cape of Good Hope, as reaching India seemed possible (Botha, 1926). On arrival in the Cape of Good Hope, Dias erected a padrao, which is a large stone cross inscribed with a coat of arms. To this day however, no trace of it has been found and a navigation beacon above Bordjiesdrif serves as a memorial (Opie, 1967). The sea route around Africa to India was opened by Vasco da Gama in 1497 and ships passed the Cape regularly. Only in 1652, 149 years later, did the Peninsula have European inhabitants again when Jan van Riebeeck landed in the Cape. Towards the end of the 17<sup>th</sup> century the Dutch East India Company established a winter harbour in Simons Bay (Opie, 1967).

### ***Farmers of the Cape of Good Hope***

In 1738, the first Europeans colonised the area of Cape of Good Hope, when stock farmer, Jurgen Schuster was granted the area of “Wilschutsbrand” (Privett, 1998). His grazing rights stretched from Red Hill to the west coast of the reserve. European settlement in the reserve was slow, with only three families residing in the area in 1786. The second British occupation of the Cape in 1806 triggered the settlement of many more farmers in the Cape of Good Hope. Simon’s Town was established as their naval base and with that an increase in demand for produce available locally (Opie, 1967; Privett, 1998). People were encouraged to burn the bush and subsequently farm the

land. Indigenous fauna of the Cape inhabited the area, such as Grysbok, Zebra, Red Hartebeest, Chacma Baboon, Leopard, Duiker, Bush Buck, Cape Buffalo, Klipspringer, Ostrich and Hippopotomus. Farmers in the area ruthlessly shot and killed much of the fauna (Opie, 1967). Cattle were allowed to move into neighbouring farms and fishermen had access to the sea coast. Horses were very important during this period, as a journey on horseback was a third of the time it took by ox wagon; thus a large number of horses were present in the area. The vegetation was burnt to generate new shoots on which the horses fed. Vegetable farming was practiced on the banks of the Klaasjagers River. *Leucospermum conocarpodendron* and *Mimetes fimbriifolius* was harvested and used for firewood (Opie 1967). Between 1855 and 1885, the southern portion of the reserve (Cape Point to Buffelsfontein) was used for breeding horses, cattle, pigs and ostriches (Hallinan, 1992). During the Anglo Boer war (1899-1902), 6,000 horses were grazed in the reserve and their feed was supplemented by locally grown barley (Greene, undated). The grazing around Schoestervallei was good and farmers permitted residents of Simon's Town to graze their cattle for an annual rental.

Deposits of ferricrete, found on the farm Rooi Hoogte, were used to keep roads in a good condition. Lime, used for building homesteads, was obtained from two local kilns at Bordjiesdrif and Buffels Bay. Due to the scarcity of water during summer, farmers built their homesteads near perennial springs and often practiced transhumance by sending their cattle to Saldanha Bay when these drought periods occurred (Report upon lands in the District of Simonstown, 1821). "Kreupelhout" (*Leucospermum conocarpodendron*) was utilised by the farmers and mainly used for wood and for tanning hides (Adamson & Salter, 1950).

Strong winds and coarse sandy soils made farming difficult in this area. Low depressions and seepages contained peaty black acidic soil. However, this acidity was overcome by adding calcium carbonate from the lime kilns. Slaves were used to till these soils, while cattle grazed on the surrounding poorer soils. Barley, tomatoes, potatoes and wheat were grown in these patches, whilst fresh meat and dairy products were sold at the market (Report on lands in the District of Simonstown, 1821). The farms were mostly at

a subsistence level and never reached commercial importance or standards during the period 1880-1960. The history of farmers and farming practices is extensive and may have had a significant impact on the reserve. The impact on the fauna in the reserve is the most obvious, with farmers killing a large number of the indigenous fauna that inhabited the reserve. The fire regime in the reserve may also have been affected, and fire frequency probably increased. This was due to the regular burning conducted by the farmers to produce new shoots for their livestock to graze on. The change in the fire regime as well as other disturbances such as wood harvesting and general clearing of vegetation would have had a substantial impact on the vegetation in the reserve. Vegetation cover throughout the reserve may have decreased and the threat of alien species introduced with the presence of fodder for livestock.

The Cape of Good Hope was also used for military purposes. Mobilisation and training exercises were conducted throughout the reserve between World War I and World War II (Hallinan, 1992). Exercises involved the use of 60-pounder guns and shells of these can still be found across the reserve. Personnel set up temporary camps between Teeberg and Klaasjagersberg in the northern part of the reserve and observation posts were established at Cape Point, Da Gama Peak and Olifantsbos (Privett, 1998). These activities would have had a direct impact on the immediate surroundings, resulting in a decrease of vegetation cover but may have also led to an increase in fire frequency in the reserve.

### ***Founding and Creation of Cape of Good Hope Reserve***

From around 1880 farmers at the Cape of Good Hope could not compete with the commercial and successful vegetable farms established in the Cape Flats. The farms in the Cape Flats had more fertile soils and the Cape of Good Hope farmers could not supply sufficient produce to compete. As stock farming in the reserve never reached commercial productivity, many farmers abandoned their farms and allowed it to revert to natural vegetation during this period. A few of the farmers with larger farms continued subsistence farming. The Smith brothers owned a farm located in the southern part of the reserve, just west of Buffels Bay. They allowed the general public to visit the reserve and make use of sites around Buffels Bay for camping and fishing. In 1930, the Smith

brothers made a decision to put the farm up for purchase. Much pressure was put on Cape Town Municipality to purchase this farm, as it provided a recreational area that was often used by the public. The Municipality decided against purchasing the farm and it remained for sale. Later, in 1939 a large portion of the farm "Blaauberg" owned by the Hare family was donated to the Divisional Council. The Council then purchased three bungalows (Pegram's, Taylors and Newmans) in the reserve (Opie, 1967). Owning these properties in the surrounding area swayed the decision of the Divisional Council of the Cape and they purchased the farm from the Smith brothers. The council also applied for the Crown land surrounding Bobbejaanshoek and subsequently obtained this land. During this period many farmers sold their farms so as to get return on their unprofitable land. Farms purchased included Olifantsbos, Kromme Rivier and the bulk of Klaasjagersberg, sold by Mr. Mimicki in 1941 (Opie, 1967).

In 1942, the Divisional Council was granted all Crown land south of the main road (road running from the entrance to the point), with the Crown only retaining the mineral rights (Opie, 1967). During the period 1943-1963 the Council bought up the land in the area, with much difficulty as many farmers had been in the area for generations and their widows did not want to sell land. Often, only after these people had passed away was the farm purchased (Opie, 1967). The final farm, Wildschuts Brand, was bought in 1965 when Mrs Malherbe sold it to the Council. The reserve was then eventually established and a fence was erected around the reserve.

### ***Personal Observation and Study by Abbot***

Prof. C.W. Abbot was part of the Faculty of Agriculture at the University of Natal (now UKZN). He resided in Pietermaritzburg and often contributed to education alongside missionaries at the Methodist church (Buckley, 1997).

Abbott (1969) stayed in the cottage 'Alpha Marina' from 1918-1922 situated at Buffels Bay and revisited the reserve in 1969. The main changes he perceived to have occurred in the reserve over the 50 year period were as follows:

- There was a decrease in dassies. In 1918 they were very abundant and in 1969 there were few or none.
- There were fewer baboons.
- There was an increase in plant growth. In 1918 there were a few milkwoods (*Sideroxylon inerme*) and fine quick grass (*Cynodon dactylon*), but nothing like the grove that was present in 1969.
- The Buffels river valley (area between Smith's farm and cottage) was much more open and sandy in 1918.
- There had been a great increase in sea kelp at Buffels Bay from 1918 to 1969.

In 1969 the reserve had Bontebok (*Damaliscus pygargus*), Baboon (*Papio ursinus*), Ostrich (*Struthio camelus*), Springbok (*Antidorcas marsupialis*), Zebra (*Equus quagga burchellii*), Wildebeest (*Connochaetes taurinus*), Eland (*Taurotragus oryx*) and possibly Vaal Rheebok (*Pelea capreolus*). Abbott (1969) suggested the following species as possible introductions: White Rhino (*Ceratotherium simum*), Cape Mountain Zebra (*Equus zebra*), Warthog (*Phacochoerus africanus*), Buffalo (*Syncerus caffer*), Sable (*Hippotragus niger*), Gemsbok (*Oryx gazella*), Waterbuck (*Kobus ellipsiprymnus*), Kudu (*Tragelaphus strepsiceros*), Impala (*Aepyceros melampus*), Blesbok (*Damaliscus pygargus phillipsi*), Nyala (*Tragelaphus angasii*), Steenbok (*Raphicerus campestris*), Black-backed Jackal (*Canis mesomelas*).

There has evidently been an increase in fauna in the reserve. Some animals have been removed from the reserve while others have been reintroduced. The increase in the number of animals in the reserve probably had a negative impact on the vegetation of the reserve, with increased levels of grazing and trampling as well as the formation of game paths throughout the reserve. The fire regime could indirectly also have been affected by the increase in animal numbers through a decrease in vegetation cover.

## ***Cape Of Good Hope Nature Reserve (1965-1998)***

The reserve was officially proclaimed in 1965 and covered 7 750 hectares. Ownership of the reserve belonged to the Cape Divisional Council and it was therefore managed in accordance with Cape Divisional Council policies. There were two main eras of management strategies. The first occurred from 1965 to 1975 and second after 1975 (Privett, 1998).

Pre-1975, a fence was erected across the reserve, along the boundary between the original farms of 'Blaauberg' and 'Buffelsfontein'. This fence split the reserve in two sections which were managed separately (Privett, 1998) (see Figure 43 in Appendix B). Fire was suppressed throughout the reserve during this period. The northern section of the reserve was managed for the conservation of the flora of the reserve, whereas the southern section was managed as a game reserve, primarily with fauna as a priority. Management practices such as bush cutting, ploughing and the planting of exotic grasses was carried out in the southern section in order to provide sufficient grazing for the animals (Millar, 1970). The introduction of herbivores included Springbok (*Antidorcas marsupialis*), Burchells Zebra (*Equus quagga burchellii*), Hartmans Zebra (*Equus zebra*), Fallow Deer (*Dama dama*), Red Hartebeest (*Alcelaphus buselaphus*), Black Wildebeest (*Connochaetes gnou*), tortoises (Testudinidae), Eland (*Taurotragus oryx*) and Bontebok (*Damaliscus pygargus*). The last two species were introduced as early as 1946 (Privett, 1998).

The fence between the northern and southern sections of the reserve was taken down in 1974, which marked the start of the second era of management. The removal of the fence was in accordance with a new policy against keeping game populations within defined boundaries (Anon 1980). The policy of fire suppression was also changed in this year to a prescribed burning programme. Vegetation was burnt on a ten year rotation and a plan to remove alien plants was developed. In 1987, prescribed burning was altered to a variable frequency burning plan where blocks were burnt in young (6-10 years), intermediate (12-20), and old (>20 years) vegetation between January and April at a ratio of 1:3:1 respectively for each of the vegetation ages (Anon, 1990). Wildfires were

also permitted to burn and occurred regularly in the reserve. A plan based on adaptive interference burning was put into place in 1992 and was practiced until 1998 (Privett, 1998), when the reserve was taken over by SANParks. The fires that were actively put out were fires that threatened infrastructure or young vegetation. Old vegetation was burnt on the advice of recognised fynbos ecologists (Privett, 1998).

Alien vegetation is a major threat to biodiversity and species such as *Acacia saligna*, *A. cyclops*, *A. longifolia* and *Pinus pinaster* are the main invasive species in the reserve. In 1974 a 10 year plan was drawn up to eradicate alien species in the reserve (Privett, 1998). Alien vegetation, however, continued to increase, with surveys conducted in 1976 and 1980. The main reduction in alien cover was as a result of a large fire burning the northern portion of the reserve in 1986. From then until 1988, the focus was on the mechanical clearing of the regrowth of these species. Privett (1998) found in his survey that nearly all alien stands had been eradicated, with the exception of a stand in the very south of the reserve.

### ***Cape Of Good Hope Section Of Table Mountain National Park (1998-Current)***

With the change of government in 1994, the Cape Divisional Council was disbanded and the Western Cape municipalities were established. During this time, the reserve was given over to SANParks to manage as part of the Table Mountain National Park in accordance with the National Conservation Strategy.

The primary objective of SANParks is to manage the reserve to maintain natural patterns and processes of the land and seascapes through a focus on the three main areas of biodiversity, ecological restoration and fire management (SANParks, 2008). Biodiversity is managed by developing inventories of all terrestrial, marine and aquatic systems as well as developing habitat-specific management strategies to prevent extinction of rare, endemic or threatened plant species. After the removal of alien species from an area, ecological restoration of degraded sites is put into place, through methods of collection, propagation, reintroduction and appropriate use of fire (SANParks, 2008). Fire is also an

important tool in management and there is currently a policy of wildfire prevention, protection and suppression as well as prescribed burning. Control of wildfires is achieved by maintaining fire breaks (SANParks, 2008). A strategy of post fire recovery is also important and this includes maintaining a fire record of all fires, restricting public access in sensitive or dangerous areas as well as monitoring vegetation recovery (SANParks, 2008). Prescribed burning is also practiced where vegetation is moribund, to ensure that overall biodiversity is not negatively affected through old individuals dying at a site and no germination of new species due to a lack of fire. The overall aim in terms of fire management is to maintain a mosaic of vegetation communities of different ages to ensure a rich biodiversity in the reserve (SANParks, 2008). Alien management is also an integral part of the management plans of SANParks, with active removal of alien species an on-going practice (SANParks, 2008).

This section deals with the land use history of the reserve and understanding this is essential to identify potential impacts on the vegetation of the reserve. It relates to the research by describing land use practices that have occurred throughout the reserve, at the appropriate spatial and temporal scale. Land use impacts may have residual impacts on the vegetation which need to be understood if the correct interpretation of the changes is to be made. For example, climate and fire conditions immediately prior to the two sampling periods have an effect on vegetation cover and composition and need to be considered in a discussion of the results.

# Chapter III: Key Drivers of Vegetation Change in the Cape of Good Hope

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## **Introduction**

Climate and fire are key drivers of vegetation change in the fynbos biome. Projections of future climate are for warmer and drier conditions in the Winter Rainfall Region of South Africa, but there are few analyses of historical climate and fire history data and how these long-term drivers impact the vegetation of the fynbos biome. Understanding long-term trends in climate and fire provides a more extensive record on which to base future trajectories. In this chapter, I investigated the long-term trends in several climate variables as well as the fire history of the reserve. These trends are used in subsequent chapters to understand and interpret the potential response of vegetation change in the reserve since 1966. I also used an analysis of repeat aerial photography to investigate broad-scale changes in vegetation cover between the period 1945 and 2008. This provided a general overview of the change in different land cover classes in the reserve.

## **CLIMATE**

The Cape of Good Hope Section of Table Mountain National Park experiences a Mediterranean-type climate with summer droughts and maximum rainfall in the winter season (April to September) (Tyson and Preston-Whyte, 2000; Midgley et al., 2003; Chase and Meadows, 2007; Haensler et al., 2010). This precipitation occurs as a result of cyclonic cold fronts associated with north westerly winds. Orographic rainfall is also common, with moisture derived from the Indian Ocean and warm Agulhas Current being transported to the southern mountains throughout the year (Midgley et al., 2003). Rainfall is variable between years ( $400\text{-}700\text{ mm}\cdot\text{y}^{-1}$ ) and this variability is related to the Antarctic Annular Oscillation (Reason and Rouault, 2005), South Atlantic sea surface

temperatures (Reason et al., 2002; Reason and Jagadheesha, 2005) as well as to the El Niño Southern Oscillation (ENSO) (Phillipon et al. 2011).

Climate is a key driver of vegetation change as it is directly linked to plant growth, reproduction and mortality. Variables such as temperature, rainfall, humidity and wind-run, influence environmental conditions as they affect available moisture. By 2050 annual temperatures are projected to increase in the South Western Cape by 1.5 – 3.5 °C (Midgley et al., 2005). This is likely to bring about an increase in plant water stress (Midgley et al., 2003). The projected 20% decrease in annual rainfall for the South Western Cape (Haensler et al., 2010; Midgley et al., 2005) will also lead to less moisture being available for plants. This combination of increased temperature and lower moisture will increase drought stress. If severe enough this could also increase plant mortality and ultimately lead to the local extinction of vulnerable species. Studies have shown that drought can have variable impacts on species in the CFR (West et al., 2012). Moisture availability is also influenced by wind speed and humidity (Roderick et al., 2009), particularly in a wind prone region such as the Cape of Good Hope. Wind also affects plant stability and can lead to the uprooting of vegetation and particularly the abrasion of seedlings by sand particles. Excessive and continuously strong winds can also result in “blow outs” i.e. the re-mobilisation of extensive dune areas. The interaction between these variables and vegetation cover is important in predicting how future climate changes will affect vegetation in the CFR (Hoffman et al., 2011). Climate change models predict extensive range contractions of many fynbos species as well as the extinction of up to 30% of the Cape Floristic Region’s (CFR) species under extreme scenarios which occur with great increases of temperature and decrease in rainfall, as well as an increase in frequency and intensity of drought (Midgley et al., 2003; Midgley et al., 2005; Hewitson et al., 2005). Therefore, an understanding of historical trajectories of climate and vegetation change could help in understanding the trajectory and impacts of future climate changes on the vegetation and biodiversity of the CFR.

The objectives of this section were the following:

- i. To analyse trends and patterns in long-term temperature, rainfall, wind speed and humidity data from the weather station at Cape Point and to compare these against the findings from other studies in the winter rainfall region. In particular, the hypothesis of an increase in aridification (resulting from a decrease in rainfall and an increase in temperature) which is anticipated for the Western Cape (Midgley et al. 2005), is investigated.
- ii. To produce a standardised precipitation index of mean rainfall for the period 1900-2010.

These climate variables were chosen as they are expected to have the greatest impact on growing conditions and survival of vegetation in the region.

## Methods

Data for maximum temperature, minimum temperature, rainfall, wind speed and humidity at the reserve was requested from the South African Weather Service (SAWS) for the longest time period available for each variable. The weather station is located at Cape Point, close to the lighthouse at the following coordinates, 34°21'26''S, 18°29'49''E. The data were separated into three different time series: summer (Oct-Mar), winter (Apr-Sep) and annual (Jan-Dec), to show graphical trends for different seasons as well as for the year overall. Trend lines were calculated for each data series using linear regressions and  $R^2$  values as well as significance which were calculated in MS Excel (2010). F-values were obtained for the respective linear regressions. A segmented regression was produced in MS Excel (2010) for wind speed, using SegReg 1.0 (2011) to identify break points in the data. This software has been used in a number of studies, including botanical and hydrological studies (e.g. Medzegue et al., 2007; Kokutse et al., 2010; Quinlan et al., 2012). The programme enables users to perform segmented regressions on ordered data sets, by identifying breakpoints in the record. A segmented regression was performed on the wind speed record only as this was the only climate variable which showed a clearly defined increasing/decreasing trend in the data over time.

Anomalies relative to the annual mean values for each of the time periods specific to the variables were also calculated to determine periods of extremes for temperature, wind run and humidity, and the patterns were shown with bar graphs. A summary table was created showing the linear equation,  $R^2$  value, P-value, significance and the extent of decadal change for each variable.

The rainfall data were further analysed using a standardised precipitation index (SPI) (Mckee et al., 1993; Edwards and McKee, 1997) to determine the years of above average rainfall (wet periods) and years of below average rainfall (drought periods). A standardised precipitation index (SPI) is a probability index that represents anomalous wet periods and dry periods. It standardises the probability of the observed precipitation and can be calculated for any precipitation total observed in the rainfall record (Guttman, 1999). Details of how the SPI is calculated and a programme to carry out the calculation may be found at: <http://drought.unl.edu/MonitoringTools/DownloadableSPIProgram.aspx>

## Results

Data for minimum and maximum temperature was only available from 1918, whereas rainfall data were available from 1900. Comprehensive wind speed and humidity data were available from 1965 and 1978 respectively. For some of the variables, there were inconsistencies with the data and some of the monthly means were missing or inaccurate. In these cases, the median value for the year or season was used as a replacement value so as not to bias the results towards a non-value or outlier (as would be the case with deleting the value or using the mean).

### Mean Maximum Temperature

Both the linear regression (Figure 3.1) as well as the bar graph, showing the pattern in the anomalies over time (Figure 3.2), indicate that there has been a significant ( $p < 0.01$ ) increase in maximum temperature for the period 1930-2010. There was no pattern of cyclical change, but rather a linearly increasing trend of approximately 2.2 °C, over the 80-year period with an  $R^2$  value of 0.7778 for the annual time series.

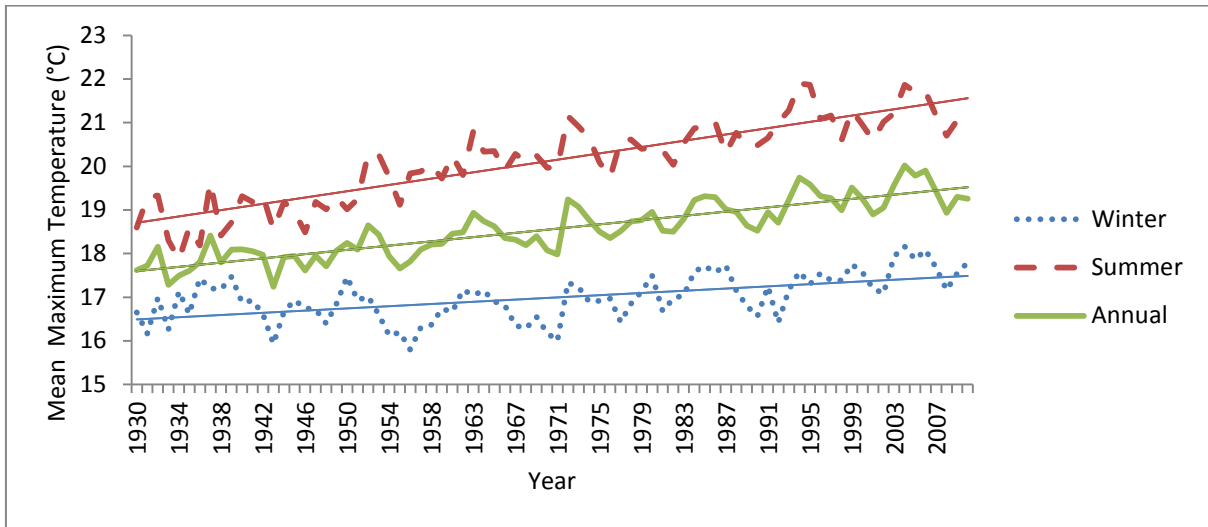


Figure 3.1: Linear regression for mean maximum temperature showing changes for different time series.

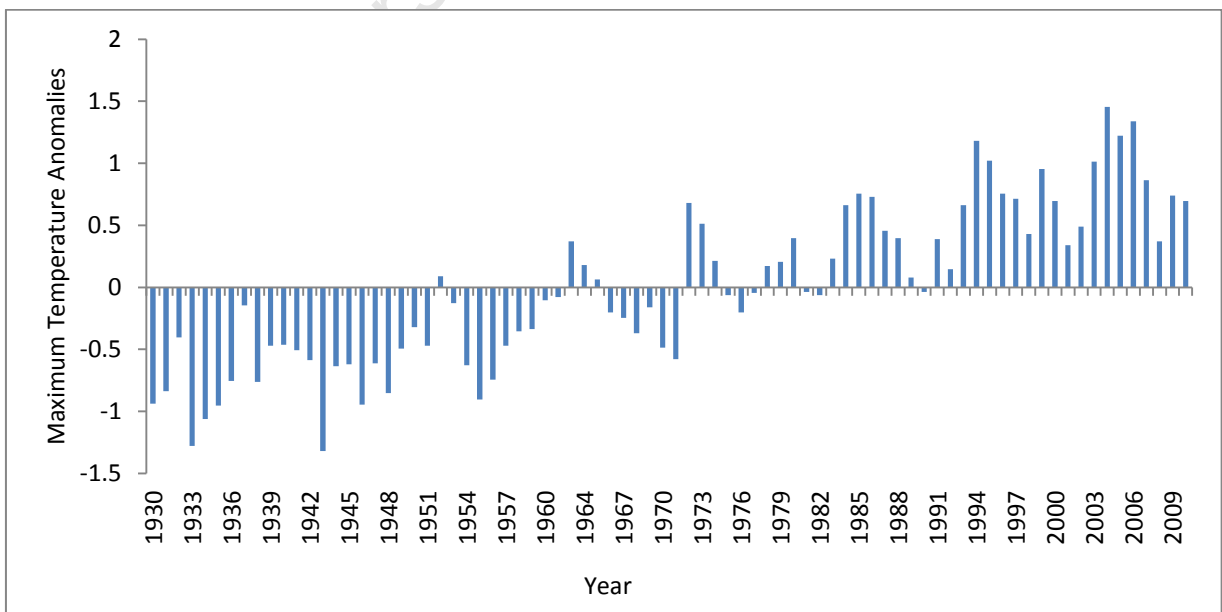
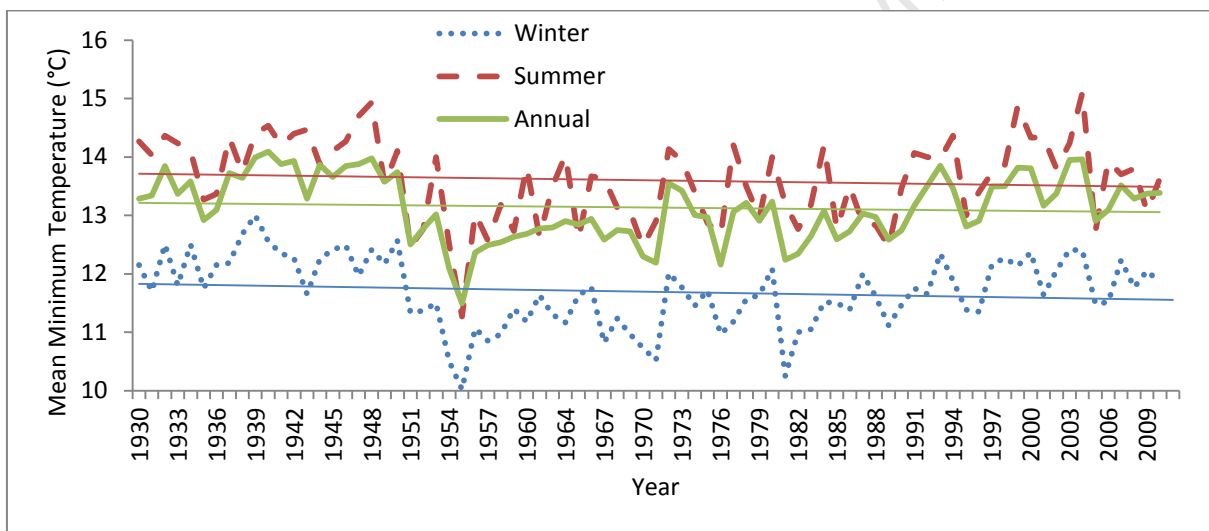


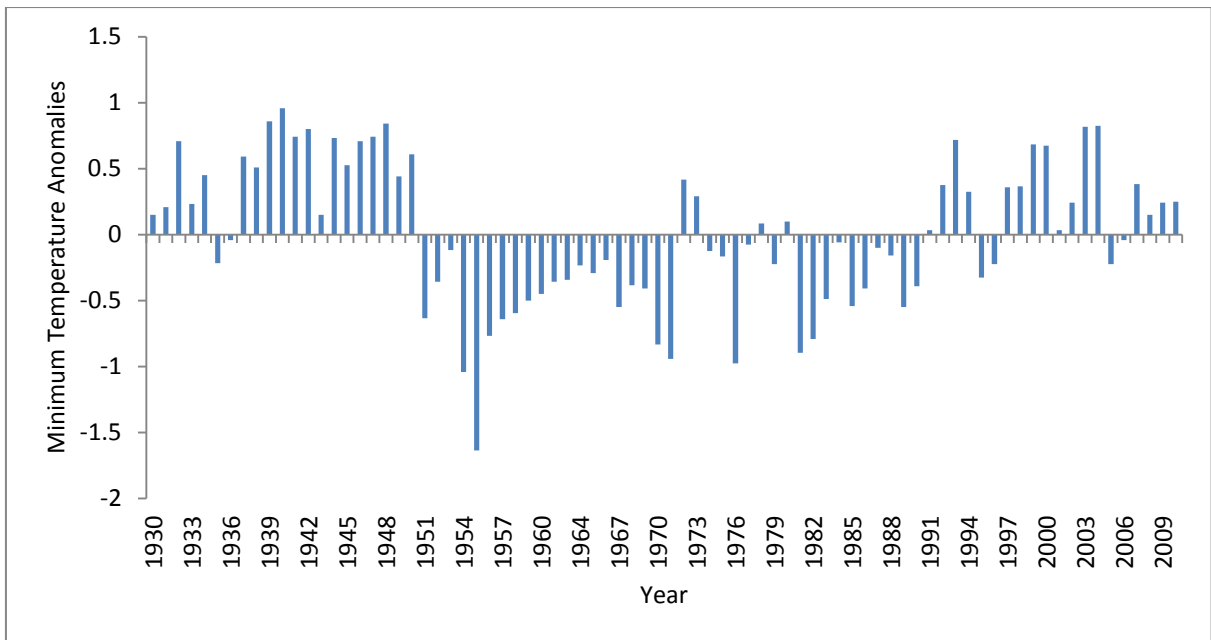
Figure 3.2: Annual maximum temperature anomalies showing difference from the mean annual maximum temperature.

### **Mean Minimum Temperature**

The linear regression ( $R^2=0.0074$ ) for mean minimum temperature shows no significant ( $p>0.5$ ) linear increase or decrease in seasonal or annual values for the period 1930-2010 (Figure 3.3). The bar graph for the annual anomalies shows that there was a cooler period from 1951-1990, with the coolest part being from 1951-1970 (Figure 3.4). Since 1990 mean minimum temperatures have been more similar to the temperatures that prevailed during the period 1930-1950.



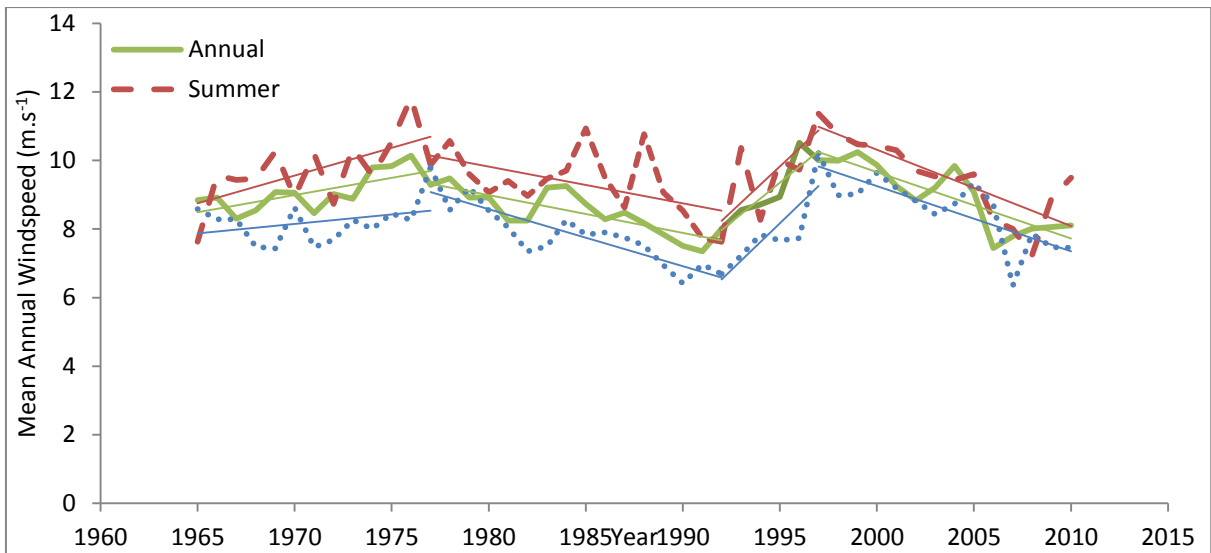
**Figure 3.3: Linear regression for mean minimum temperature showing changes for different time series.**



**Figure 3.4: Annual minimum temperature anomalies showing difference from the mean annual minimum temperature.**

### ***Mean Wind Speed***

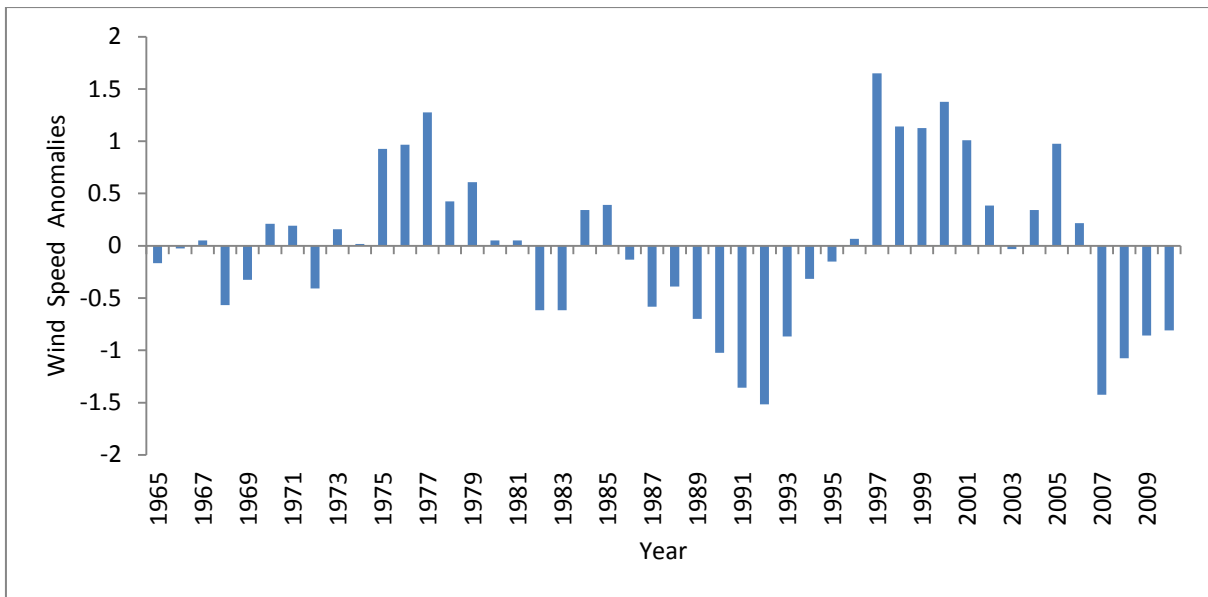
Mean wind speed values have exhibited two periods of increase (1965-1976 and 1992-1997) and two periods of decrease (1977-1991 and 1998-2010) over the length of the recorded time series. This is evident in the segmented linear regression analysis, (Figure 3.5, Table 3.1) as well as in the anomalies (Figure 3.6).



**Figure3. 5: Segmented linear regression for mean wind speed showing changes for different time series.**

**Table3.1: Summary of segmented regression statistics (\* P<0.5, \*\* P<0.01).**

Segmented Regression	Linear Equation	R <sup>2</sup> value	Significance
Annual <sub>1</sub>	$y=0.1005x-189$	0.503	**
Annual <sub>2</sub>	$y=-0.1102x+227.16$	0.6523	**
Annual <sub>3</sub>	$y=0.4617x-911.67$	0.8258	**
Annual <sub>4</sub>	$y=-0.1941x+397.83$	0.7321	**
Summer <sub>1</sub>	$y=0.1617x-309.03$	0.389	*
Summer <sub>2</sub>	$y=-0.1071x+221.9$	0.2883	*
Summer <sub>3</sub>	$y=0.5286x-1044.7$	0.5134	*
Summer <sub>4</sub>	$y=-0.2231x+456.47$	0.6722	**
Winter <sub>1</sub>	$y=0.0558x-101.72$	0.1149	NS
Winter <sub>2</sub>	$y=-0.1677x+338.63$	0.7658	**
Winter <sub>3</sub>	$y=0.5443x-1077.7$	0.7005	*
Winter <sub>4</sub>	$y=-0.1901x+389.47$	0.6212	**



**Figure 3.6: Annual wind speed anomalies showing difference from the mean annual wind speed.**

### ***Mean Relative Humidity***

Since 1978 at the start of the record, humidity has declined significantly ( $p < 0.01$ ) in summer and winter as well as annually (Figure 3.7). A decline of approximately 6% over 32 years has been experienced with an annual  $R^2$  value of 0.6242. The anomalies show the same trend (Figure 3.8), with a decline over the period. Positive anomalies were present from 1978-1990 and from 1991-2010. Overall values have generally been lower than the annual mean.

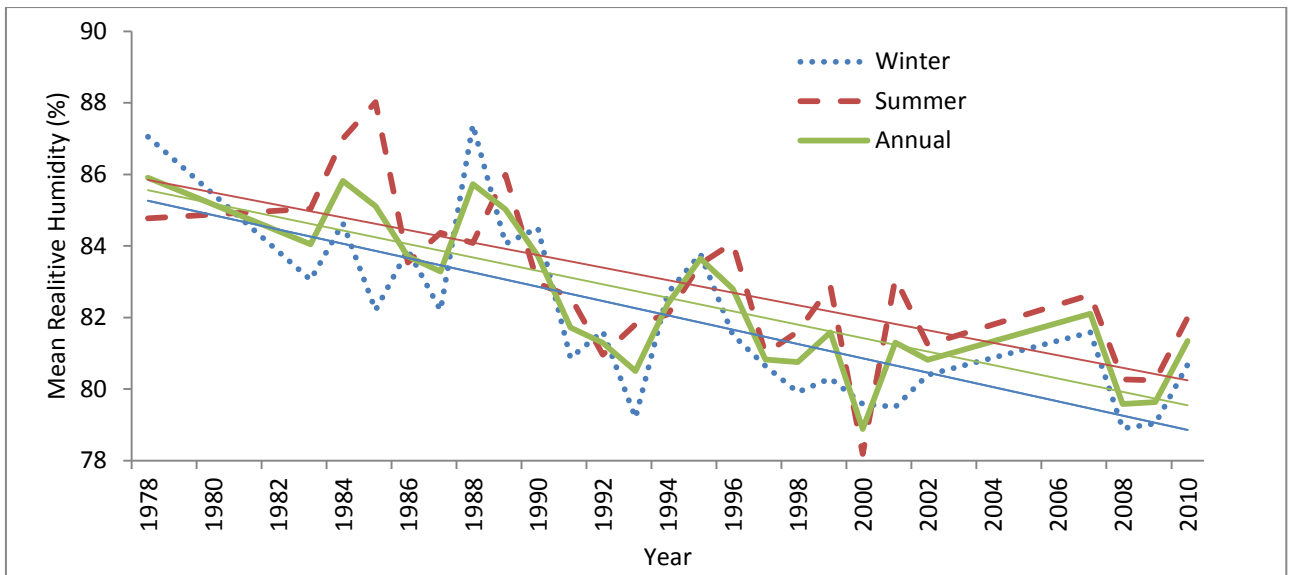


Figure 3.7: Linear regression for mean humidity showing changes for different time series.

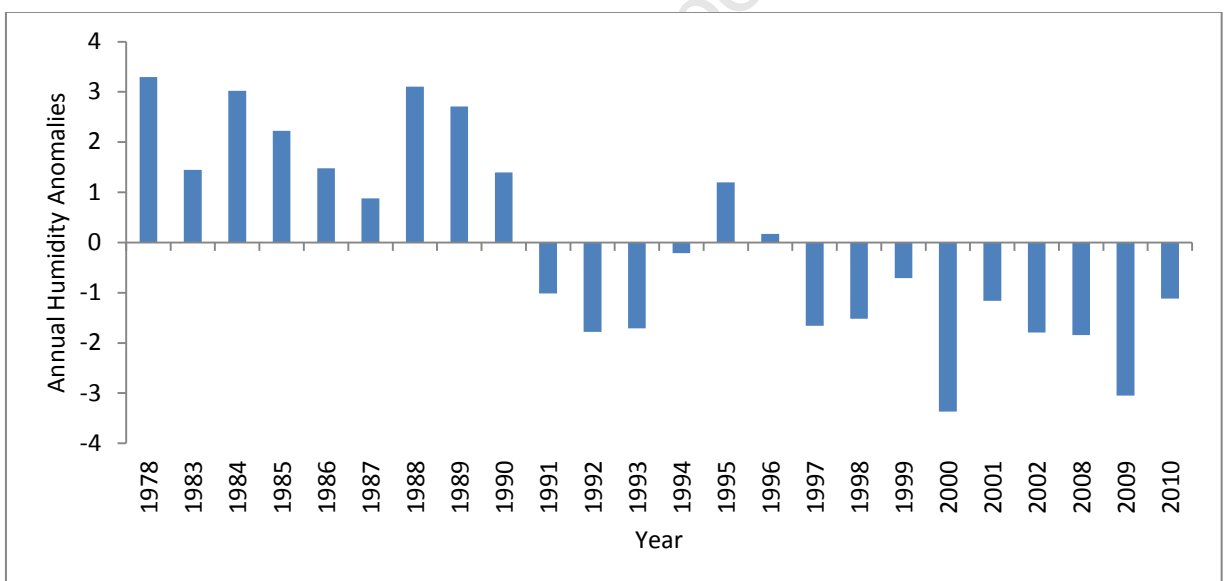


Figure 3.8: Annual humidity anomalies showing difference from the mean annual humidity.

### Mean Rainfall

The time series for total rainfall shows an insignificant ( $p > 0.05$ ) increasing trend for winter and annual totals of approximately 230 to 260 mm and 370 to 390 mm respectively. Summer rainfall also appears not to have changed significantly over time ( $p > 0.05$ ) (Figure 3.9). Since more than 70% of rain falls in the winter months, trends in

annual rainfall correspond closely with those for the winter period. The Standardised Precipitation Index (SPI) (Figure 3.10) shows numerous dry and wet periods from 1931-2010 while the extended wet period that took place in the 1950's is noticeable. Since 1980 dry periods have alternated relatively frequently with wet periods.

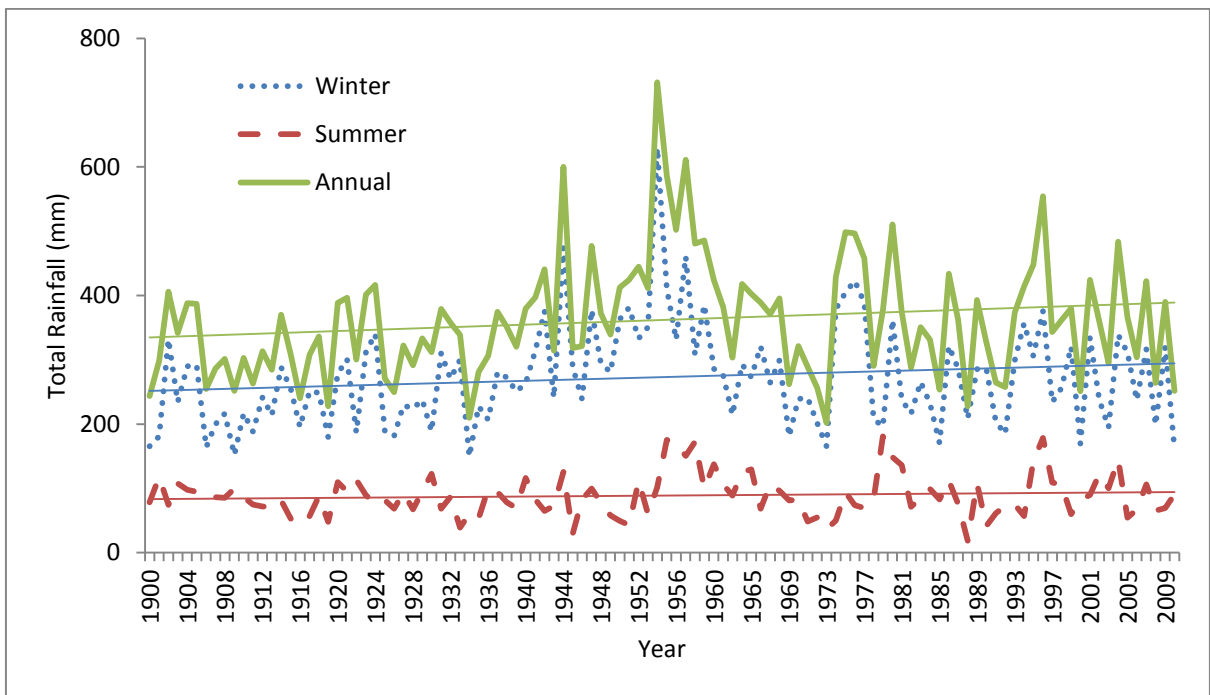


Figure 3.9: Linear regression for mean rainfall showing changes for different time series.

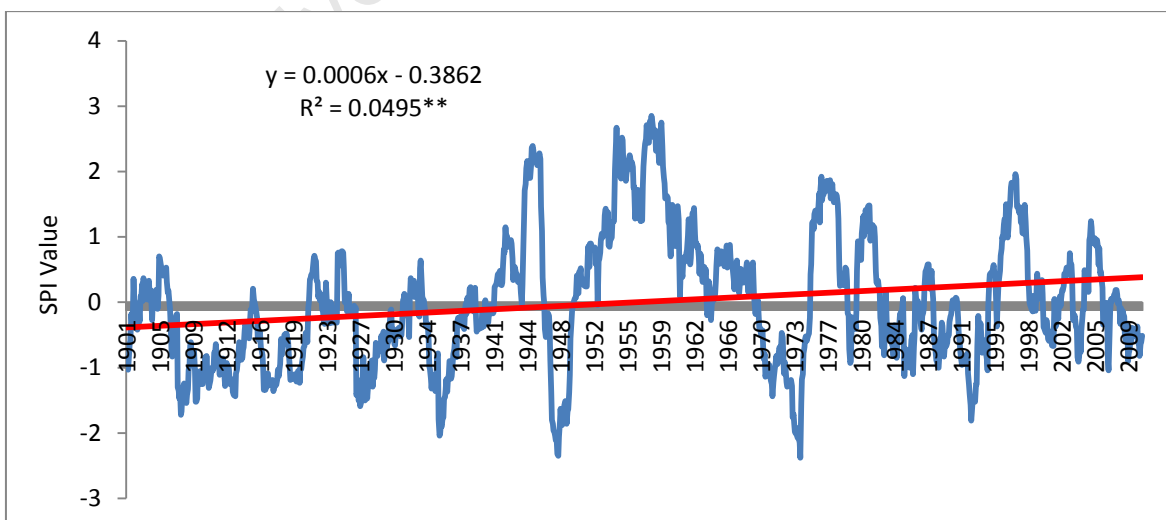


Figure 3.10: Standardised Precipitation Index (SPI) for mean annual rainfall at the reserve.

### **Summary of Climate Variables**

Table 3.2 is a summary of the statistical outputs from the linear regressions run for maximum temperature, minimum temperature, humidity and rainfall. It shows the linear regression of each category (annual, summer, winter) for the variables, as well as the R<sup>2</sup> value, P-value, significance and decadal change.

Only some variables showed significant change (i.e. maximum temperature, windspeed, and humidity). R<sup>2</sup> values range from 0.0074 (Mean minimum annual temperature) to 0.7778 (mean maximum annual temperature). Mean maximum temperature and total rainfall show positive trends whereas mean minimum temperature and humidity have negative trends.

**Table 3.2: Summary of climate variables linear regression statistics.**

Variable	Linear Equation	R <sup>2</sup> -value	P-value	Significance	Decadal Change
Mean Maximum Temperature: Annual	y=0.0361x+18.669	0.7778	p<0.01	**	0.361 °C
Mean Maximum Temperature: Summer	y=0.0243x+17.576	0.7565	p<0.01	**	0.243 °C
Mean Maximum Temperature: Winter	y=0.0126x+16.482	0.3074	p<0.01	**	0.126 °C
Mean Minimum Temperature: Annual	y=-0.002x+13.216	0.0074	p>0.05	NS	
Mean Minimum Temperature: Summer	y=-0.0028x+13.714	0.0082	p>0.05	NS	
Mean Minimum Temperature: Winter	y=-0.0034x+11.831	0.0175	p>0.05	NS	
Mean Relative Humidity: Annual	y=-0.1876x+456.64	0.6242	p<0.01	**	-1.876%
Mean Relative Humidity: Summer	y=-0.1751x+432.18	0.4727	p<0.01	**	-1.751%
Mean Relative Humidity: Winter	y=-0.2001x+481.09	0.5407	p<0.01	**	-2.001%
Total Rainfall: Annual	y=0.8138x+504.28	0.0248	p>0.05	NS	
Total Rainfall: Summer	y=0.1038x+83.286	0.0094	p>0.05	NS	
Total Rainfall: Winter	y=0.4018x+246.5	0.0267	p>0.05	NS	

## Discussion

Twentieth century trends in the Cape of Good Hope climate variables are similar to those reported elsewhere for the Western Cape. For example, the increase of 2.2 °C in mean maximum temperature since 1930 corresponds with trends reported previously (Midgley et al, 2005; Warburton et al., 2005; Hoffman et al, 2011). While the critical experiments are currently lacking, an increase in maximum temperatures of such magnitude could significantly influence moisture availability, plant stress and even survival of drought-sensitive species (Rutherford et al., 1999; Midgley et al., 2003; Walther et al., 2002; West et al., 2012). If temperatures continue to increase at the current rate, there is an even greater risk that species' tolerance limits will be exceeded and that plant mortality will increase significantly (Rutherford et al., 1999; Walther et al. 2002). Furthermore, the decrease in humidity, from approximately 86% to 80% is strongly linked to air temperature and is likely to decline even further if temperatures increase. This has important implications for plant function as well as other ecological processes. Fire frequency and intensity are also likely to be affected (Midgley et al., 2003; Midgley et al., 2005).

In agreement with several other studies in the Western Cape (Midgley et al., 2005; Hewitson et al., 2006; Haensler et al., 2010; Hoffman et al., 2011) both winter and annual rainfall have not changed significantly in the Cape of Good Hope over course of the 20<sup>th</sup> century. However, the increase in annual rainfall of 5 mm per decade, which amounts to an increase of about 50 mm over 100 years, might have important consequences for moisture availability and plant growth. The extended wet period that took place in the 1950's is closely linked with the cooler mean minimum temperatures recorded for that time (Haensler, 2010). These cooler temperatures could be as a result of the increased rainfall as well as an increase in cloud cover across the reserve. This increased moisture availability is also likely to have influenced plant biomass in the reserve, which would in turn alter fire intensity and perhaps also fire frequency. The alternating wet and dry periods in more recent years are closely tied to ENSO effects (Phillipon 2011).

The last decade has shown a decline in wind speeds (from 11 m.s<sup>-1</sup> to 7.8 m.s<sup>-1</sup>) which corresponds with findings of Roderick et al., (2007) and Hoffman et al., (2011), but long-term trends show alternating increasing and decreasing trends. Rayner, (2007) stated that wind patterns are influenced by large-scale climatological change as well as local phenomena. As local conditions have not changed around the weather station for Cape Point (i.e. the weather station has remained at the same location and the surrounding area has always been a tourist site), these changes are probably the result of large-scale changes in atmospheric processes. A potential explanation for the decrease in wind speeds is due to the changing temperature gradient between polar and tropical regions, which in turn changes the strength of air movement between them, therefore decreasing wind speed (Midgley et al., 2005). The alternating pattern of wind speed may have had corresponding alternating positive and negative responses on the vegetation of the reserve. During periods of wind speed increase, there is likely to be a negative impact on vegetation cover as wind speed affects moisture availability (Roderick et al., 2009). High wind speeds are also important in the mobilization of dune systems with negative consequences for dune vegetation. Alternatively, when wind speeds decrease, moisture availability could be enhanced although the higher temperatures associated with stilling could have important implications for leaf and whole plant physiology (Hoffman et al., 2011).

The data analyses here are the recordings of one weather station at Cape Point, and therefore the findings cannot be scaled up to the whole CFR. Primarily, the Cape Point weather station is in a unique position on a narrow peninsula. While it is buffered from temperature change by the sea it is also exposed to severe storms and high winds, and therefore may experience different trends from more inland or upland areas of the CFR. A larger-scale, more regional study including more weather stations such as those undertaken by Midgley et al. (2005) and Hoffman et al. (2011) may better show the effects of climate changes on the vegetation of the CFR at the regional scale. It is also important to note that linear regressions highlight the broad changes, specifically

between the two time periods and do not highlight the subtle variability in the record. Furthermore, in terms of the effects of increasing temperature on range distribution, the vegetation of the reserve occurs at the southern latitudinal limit of the CFR. Because it is at the “leading” climate edge it is likely that the impacts of increasing temperature and declining humidity might be first detected in the vegetation of the reserve.

## **FIRE**

Fire is an essential and important factor of disturbance in the biology of fynbos (Dansereau, 1957; Ahlgren and Ahlgren, 1960; Spurr, 1964; Daubenmire, 1968; Cowling, 2001; Keeley et al., 2012). Many fynbos species are adapted to fire as a medium for the release of seeds and for germination (Mutch, 1970). For instance, some Proteaceae species have serotinous cones which, when burnt, release seeds, that are then dispersed by wind (Lamont et al., 1991; Keeley and Fotheringham, 2000). Also, it has been shown that some seed germination can be triggered by smoke (Brown and Botha, 2004). Smoke germination has been observed in Proteaceae species (e.g. *Leucadendron xanthaconus*, *Serruria villosa*, *Protea repens*), Ericaceae species (e.g. *Erica plukenetti*) and some Restionaceae genera (e.g. *Elegia* and *Chondropetalum*), (Brown and Botha, 2004). Fire also leads to the removal of dead plant material, which creates gaps for seedlings to establish (Keeley and Fotheringham, 2000). This removal of dead individuals is very important in fynbos as unlike other communities, fynbos species experience very low levels of herbivory as well as slow decomposition rates (van Wilgen et al., 1992). Therefore, without a fire, moribund vegetation could dominate the landscape, and thus not allow seedlings to establish successfully due to lack of space and available resources as well as a reduction in light at the soil surface. This in turn, could lead to a great decline in species diversity and perhaps even local extinction.

The term fire regime refers to different components and attributes of fire. These attributes include frequency, seasonality and intensity (Bond and van Wilgen, 1996; Keeley, 2009). These components of the fire regime can shape the vegetation of the

landscape by altering the vegetation structure and composition of the area (Rutherford et al., 2011). For instance, a very intense fire could also impact vegetative material below the soil, by burning underground storage organs and roots of plants, thus making recovery much more difficult than if the fire had only burnt the above-ground vegetation. Differences in fire regime can thus lead to differences in plant communities and structure and could perhaps lead to extinction of species (Gill and Bradstock, 1995; Charrette et al., 2006; Fisher et al., 2009). In this study, fire frequency is the only component of the fire regime assessed due to the quality of the data available.

Understanding fire frequency is critical, because if the fire frequency is too high, the vegetation may be threatened (Keeley, 1995), as species may not have enough time to recover from the impact of the fire before the next fire burns an area. Time is needed after a fire for plants to grow, flower and set seed. This process of recovery often occurs in a relatively predictable manner in which different species dominate the community at different times in relation to post-fire age (Hoffman et al. 1987; Cowling et al., 1997). If fires are too frequent, the later post-fire succession species will not have a chance to establish and a dominance of early post-fire succession, or pioneer species, will occur. This could lead to the local extinction of late succession species. Because of their protective bark, some adult individuals of the large reseeding Proteaceous shrubs such as *Leucospermum conocarpodendron* and *Mimetes fimbriifolius* are able to survive fires. However, if fires are too frequent, the juveniles will be killed and this will impact on population structure and long-term viability (Midgley et al., 1998).

Infrequent or no fire is also detrimental to fynbos biodiversity as a biome with a low decomposition rate needs to be burnt to allow space and nutrients for new individuals once the old individuals have become moribund. Fire is also a trigger for seed germination for many fynbos species. A decline in species richness has been recorded for fynbos older than 10 years (Hoffman et al. 1987; Cowling and Pierce, 1988; Cowling and Gxaba, 1990; Privett et al., 2001) and thus, regular fires every 8-15 years are needed to maintain diversity. The accumulation of old dead plant material aids this process and encourages burns at these sites.

Fire in fynbos is a natural phenomenon, caused by lightning strikes and falling rocks (Edwards, 1984). However, not all fires that occur in this vegetation are natural and many fires are anthropogenic in origin. Schwilk et al., (1997) found that the general fire return interval for the Cape Floristic Region was between 15-25 years. Fire frequency can increase due to increasing human populations that cause a higher number of ignitions. In California, for instance, fire frequency is five times greater in the north than the south solely due to the greater human population (Haiman, 1973; Keeley, 1982, Freedman, 1984; Minnich, 1989; Minnich and Dezzani, 1991). An increase in fire frequency is also expected with corresponding climate change projections. A predicted increase in temperature and decrease in rainfall for the South Western Cape could lead to hotter and drier environmental conditions, which are more conducive for the start and spread of runaway fires (Bessie and Johnson, 1995; Rambal and Hoff, 1998; Piñol et al., 1998; Millán et al., 1998; Midgley et al., 2003; Simmons et al., 2004; Flannigan et al., 2007; Cochrane and Barber, 2009; Forsyth and Van Wilgen, 2008). The combination of increases in human ignitions and climatic fire-prone environments could lead to a great increase in fire frequency, which would have negative impacts on the vegetation and biodiversity of fynbos.

Fire is used as a tool in managing the vegetation in conservation areas and understanding fire is therefore essential to managing an area for its best potential. Prescribed burning is used to manipulate the landscape to meet conservation targets. Often, a combination of controlled as well as uncontrolled fires burn through a landscape. For instance, in the Kruger National Park, a prescribed burning policy formed part of the management plan for the protected area (du Toit et al., 2003). However, during this period, wildfires still occurred and thus both prescribed fires and wildfires were burning the vegetation of the park, which increased the heterogeneity of the vegetation (du Toit et al., 2003). This resultant pattern of fire in the landscape is referred to as the fire mosaic, as fire is seldom spread evenly across an area both on a spatial and temporal scale and a heterogeneous mosaic of patches of different postfire age is likely to be beneficial to biodiversity (Driscoll et al., 2010). It is very important in terms of management to understand these

fire mosaics as it influences community structure and composition. Managers need to be able to control burns and therefore face the problem of reducing the risk of damaging wildfires, while simultaneously ensuring that fire continues to play its vital role in maintaining healthy ecosystems (Biswell, 1989; Keith et al., 2002; Keeley and Fotheringham, 2006). Thresholds of potential concern (McLoughlin et al., 2011) could be put into place to monitor the impact of the fire regime on the vegetation to allow managers to gauge the effectiveness as using fire as a tool for management in the reserve.

Fire in Cape of Good Hope reserve is a long-standing and current form of disturbance. Before the area was established as a reserve in 1965, people farmed on the land. This had a major impact on fire frequency as farmers would burn so as to provide grazing for their livestock (Taylor, 1969). Since the establishment of the reserve, fire frequency has changed. Initially, there was a policy of fire suppression as it was thought that fire would be harmful to the animals. In 1987, a programme that varied the frequency of burning was established. Vegetation was burnt in blocks and at different ages. These stages included young (6-10 years), intermediate (12-20 years) and old (>20 years) stands of vegetation. Burning usually took place between January and April and wildfires were also allowed to burn (Anon, 1990). In 1992, however, a plan of adaptive interference was adopted, whereby only fires that threatened infrastructure or young vegetation were controlled and old vegetation was burnt on the expert opinion of recognised fynbos ecologists (Privett, 1998). This management strategy is still employed today (SANParks, 2008).

The main objective of this section of the chapter was to understand the history of fire in the reserve, specifically related to fire frequency. Two sets of data were used. The first was presence/absence data for the period 1940-1974 (Privett, 1998) while the second data set was comprised of GIS layers of fire scars, obtained from SANParks' Cape Research Centre for the period 1975-2010 (Forsyth and van Wilgen, 2008). The latter data set was richer as it gave fire frequency and extent and thus was used for analyses

from 1975. These parameters were not available for the earlier fire records prior to 1975 (Privett 1998).

The specific aims of this section of the chapter were to:

- i. Determine the patterns of fire occurrence for the period 1940-1974, using fire presence/absence data from Privett (1998). Fire frequency was expected to increase over time, because of an increase in the frequency of anthropogenic ignitions recorded elsewhere on the Cape Peninsula (Forsyth and van Wilgen, 2008).
- ii. Observe patterns in spatial fire frequency for the period 1975-2010, using data from the Cape Research Centre. It was expected that the northern part of the reserve would have burnt more as it is closer to the source of human ignitions, thus increasing the chance of burning.
- iii. Determine the relationship between fire frequency and area burnt per year, for the period 1975-2010. A positive relationship would suggest that the area burnt is a function of the number of fires in the reserve. A negative relationship would suggest that a few, large fires burn a greater area than many small fires. No relationship would suggest that area burned is independent of the number of ignitions.

## **Methods**

### ***Analysis of Presence/Absence Data (1940-2010)***

To determine the trend in fire occurrence for the period 1940-1974, fire presence/absence data were used from Privett (1998) and was based on whether a fire occurred in the year or not. This data set was compiled by estimating the post-fire age for each of 100 plots, by counting the number of nodes and internodes present on the branches of Proteaceae shrubs found in the plot (Hall, 1959). This record extends from 1940-1996. However, only records from 1940-1974 were used, as from 1975, more detailed fire frequency data were available for the reserve from the Cape Research Centre. Fire presence/absence was also determined using the Cape Research Centre

data for the time period 1975-2010. The running average for a five year period was calculated for the two time steps. A line graph was produced that showed the calculated running average of presence of fires per year against winter and summer total rainfall in two separate graphs. This was done to illustrate how the average number of fires present per year changed over time and to show the relationship with rainfall. Summer and winter rainfall was compared separately as it was thought that each would have a different relationship. Summer rainfall was more likely to correspond to climatic conditions leading to more fire prone weather, while winter rainfall would be a proxy for biomass and thus indicate amount of fuel available to burn.

### ***Analysis of Spatial Fire Frequency and Extent (1975-2010)***

The fire data from the Cape Research Centre was in GIS format and showed the occurrence and spatial extent of every fire that occurred across Table Mountain National Park for the period 1975-2010 (Forsyth and van Wilgen, 2008). Each fire was represented by a polygon and a separate layer represented each year. ArcGIS (ver. 9.3) was used to view these layers and to show fires that only occurred within the Cape of Good Hope Section of Table Mountain National Park. These layers were then clipped to the reserve boundary using Taylor's (1969) vegetation map which was digitized for these purposes. This produced layers of fires that were recorded in the reserve for the period 1975-2010. The separate layers were then all converted from vector to raster format and a count field (equal to one) was added to the attribute table of each layer. The union tool was then used to add the layers together to obtain a fire frequency layer for different polygons across the reserve, indicating the combined number of fires for each polygon. Finally, using the raster calculator tool, the frequency of fire for the entire reserve, for the period 1975-2010, was calculated and a map produced.

### ***Analysing the Relationship between the Number of Fires and Area Burnt***

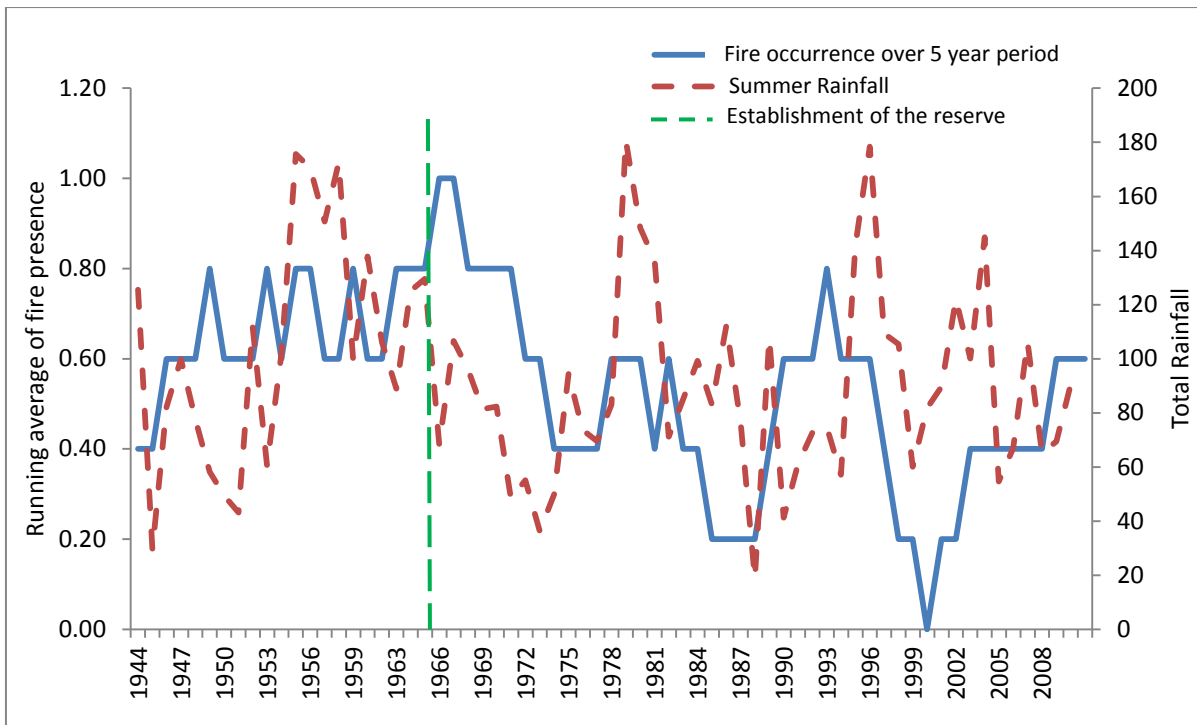
To calculate the area burnt per year, the clipped fire layers created in the previous method was used. The 'calculate area' tool in ArcGIS (ver. 9.3) was used to calculate the areas of polygons that occurred in the same year. These areas were then added to

obtain a value for area burnt in that specific year. This was done for all years for the period 1975-2010. Fire frequency for each year was determined by manually counting the number of fire occurrences for that year and was also done for each year in the period 1975-2010. The relationship between these two variables was illustrated by producing a combined graph where area burnt per year was represented by bars and fire frequency by points. A linear regression was also produced to determine the strength of the relationship between fire frequency and area burnt per year.

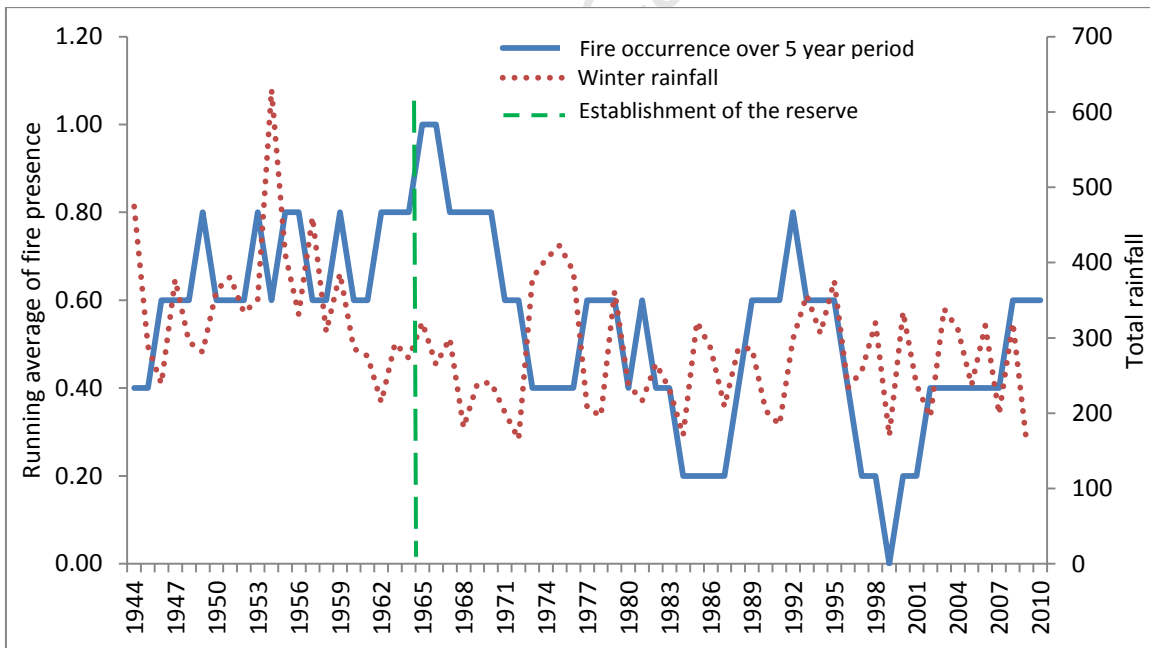
## Results

### *Fire Presence/Absence (1940-2010)*

There has been a decrease in the presence of fires in the reserve since 1940, with the most marked point around 1965 (Figure 3.11 and Figure 3.12). Prior to 1965, there were mostly between three and four fires present over five years, whereas after 1965, this dropped to between one and three. The period when fires were most frequent was between 1961 and 1966 when there was a fire in every one of the years over this five year period. Fires also occurred frequently between 1987 and 1994. Fires were least frequent over the five year period prior to 1999 when no fires occurred in the reserve. There was no relationship between the number of fires and either summer ( $n=36$ ,  $y=3.125x+95.706$ ,  $R^2=0.0165$ ,  $p=0.455$ ) or winter ( $n=36$ ,  $y=-1.1184x+274.68$ ,  $R^2=0.0005$ ,  $p=0.895$ ) rainfall totals (Figure 3.11 and Figure 3.12).



**Figure 3.11:** Line graph showing running average of fire presence in a year against the total summer rainfall. The dotted vertical line indicates the establishment of the reserve.



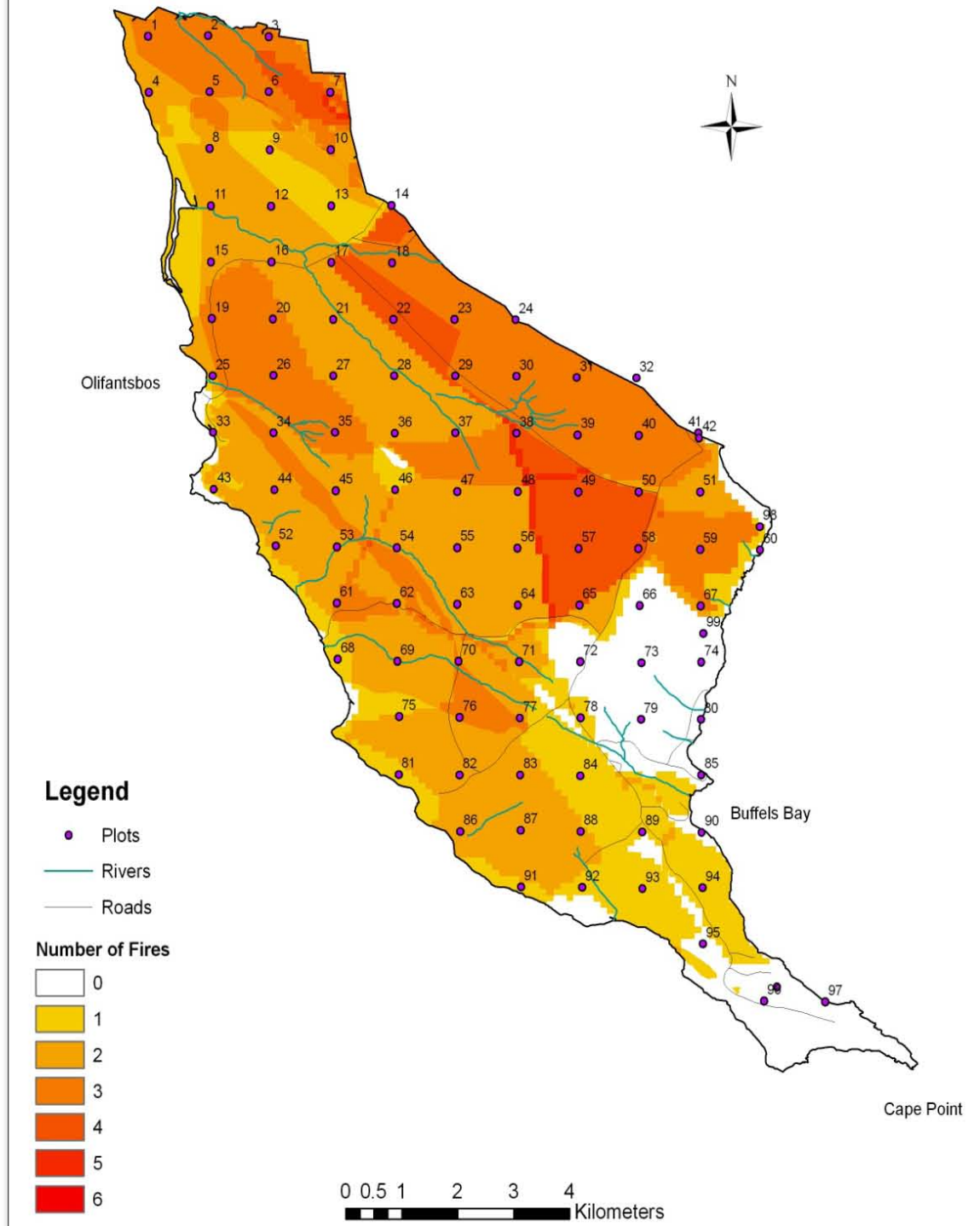
**Figure 3.12:** Line graph showing running average of fire presence per year against the total winter rainfall. The dotted vertical line indicates the establishment of the reserve.

### ***Spatial Fire Frequency and Extent***

The north eastern part of the reserve has also burnt more than the south western part. The south east and southern parts of the reserve, including the areas between Paulsberg and Buffels Bay and the area around Cape Point, have not burnt since 1975. A number of small fires have occurred in the northern part of the reserve, with larger, more extensive fires burning further south. The maximum number of times that any one area has burnt over the period 1975-2010 is six, although this is only for 3% of the reserve (approximately 230 ha), whereas approximately 3500 ha (45%) of the reserve has experienced three fires (Figure 3.13).

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## Fire Frequency in the Cape of Good Hope (1975-2010)



**Figure 3.13: Map showing the pattern and distribution of fire frequency in the Cape of Good Hope (1975-2010).**

## ***Relationship between the Number of Fires and Area Burnt (1975-2010)***

Results of a linear regression show that between 1975 and 2010 the total area burnt was not related to the number of fires ( $n=26$ ,  $R^2=0.0226$ ,  $p>0.05$ ). In some years (e.g. 1989) a small number of fires burnt a large area while in others (e.g. 1991) many fires burnt a small area only (Figure 3.14). The largest area burnt was in 1986, when three fires occurred and the second largest area burnt was in 2007 when four fires burnt in the reserve.

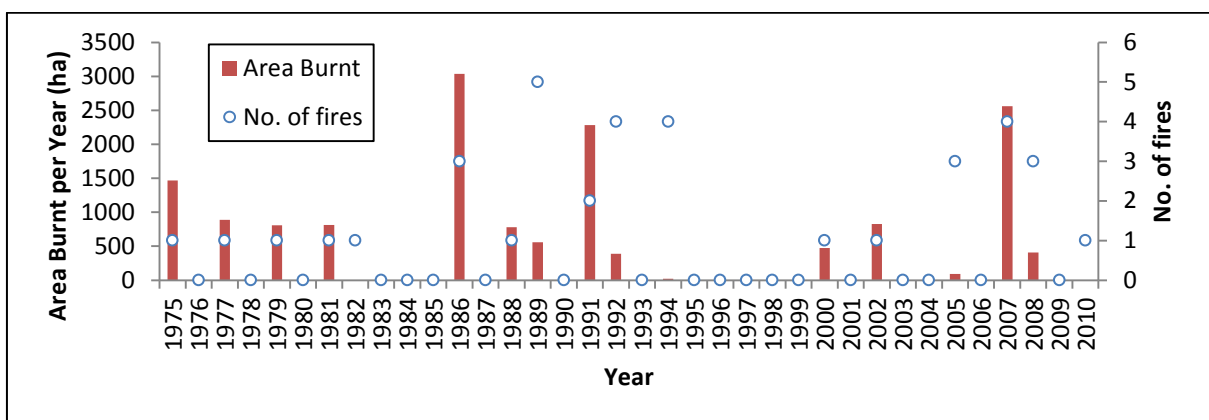


Figure 3.14: Graph showing the relationship between the number of fires and area burnt per year (ha) for Cape of Good Hope for the period 1975-2010.

## **Discussion**

### ***Presence/Absence of Fire (1940-2010)***

The running average of fire presence over a five-year period shows that fire presence in the reserve has been decreasing, specifically from the mid-1960's. This is not what was expected, particularly since there has been an increase in temperature over the last 50 years in the Western Cape (Midgley et al., 2003; Simmons et al., 2004; Flannigan et al., 2007; Hoffman et al. 2011). One explanation for this result, however,

is that since the establishment of the reserve in 1965 it has been managed as a single unit, whereas prior to that, farmers were managing their farms separately and as they saw fit (Opie, 1967) (see study area description). This, as well as changes in management practices within the reserve, may have led to this decrease in the presence of fires.

The period of high occurrence of fire from 1986-1994, could be explained by the management strategy in place at that time. In 1987 a block burning programme was put into place, with vegetation systematically burnt according to its age (Privett, 1998). This led to an increase in the number of fires that occurred in the reserve, and these fires may have also become uncontrolled and spread to wildfires. In 1992, the block programme was replaced with an adaptive interference management plan. These factors also led to the increased fire frequency for the period 1975-2010. This led to a lower occurrence of fire after this period (Privett, 1998). The poor relationship between fire presence and rainfall (total and seasonal) suggests that on its own rainfall does not influence the incidence of fire in the reserve.

### ***Spatial Fire Frequency and Extent (1975-2010)***

Humans often start fires, and with increasing human presence, the potential for fires to occur increases (Henderson 1964; Haiman, 1973; Keeley, 1982; Freedman, 1984 Minnich, 1989; Minnich and Dezzani, 1991). A greater fire frequency was found along the roads, specifically the main road, which is the road most travelled by visitors within the reserve. This road is the direct route to Cape Point, which is the main tourist attraction in the reserve. With the large number of people making use of this road, the number of potential ignitions, from cigarette butts or other litter thrown out of car windows increases, potentially leading to more fires. This could explain the greater fire frequency along the main road compared to other parts of the reserve.

The northern part of the reserve has shown a greater fire frequency than the southern part of the reserve. An explanation for this is that the northern boundary of the reserve borders on the edge of the settlement of Scarborough. The presence of a residential

area in such close proximity to the reserve increases the potential for fires to occur, with humans acting as a source of ignitions (Minnich, 1989; Minnich and Dezzani, 1991).

The south eastern and southern parts of the reserve, the area around Paulsberg to Buffels Bay and Cape Point, have not burnt since 1975. This is probably due to the suppression and action against fire moving into these areas, as these areas are frequented by droves of tourists. A fire moving into the area could be a threat to visitors as well as to the recreational facilities that have been provided for the visitors. However, local factors such as the particular composition of the vegetation as well as the proximity to the coastline might also be responsible for the absence of fire in this region.

The maximum fire frequency for this period was found to be six. The section that has burnt six times, is a track and has probably been purposely burnt to maintain a fire break between management units and to keep the area clear of vegetation for recreational users. The majority of the reserve however, has been burnt three times over a 36 year period, which is an average of one fire every 12 years. This seems to be appropriate in terms of accepted fire frequencies for the fynbos biome and corresponds with findings and recommendations from other studies (Cowling and Pierce, 1988; Cowling and Gxaba, 1990; Schwilk et al., 1997; Privett et al., 2001).

### ***The Relationship between Number of Fires and Area Burnt (1975-2010)***

Many small fires occurred in the reserve over the period 1975-2010, often with only one fire occurring in a year. Only under exceptional circumstances have fires burnt extensive parts of the reserve over this period. This variation in fire regime can result in a mosaic of fire which could have a positive impact on biodiversity (Driscoll et al., 2010).

Fire frequency is an important component of the fire regime as it dictates how and to what extent the vegetation is altered and affected (Biswell, 1989; Keith et al., 2002; Keeley and Fotheringham, 2006; Rutherford et al., 2011). In this study, it was found that fire frequency has not increased since the 1970s which is contrary to other findings for

Forsyth and van Wilgen's (2008) analysis of the entire Cape Peninsula and other parts of the fynbos biome (Rutherford et al., 2011). However, as environmental conditions change with predicted changes in climate, more marked trends in fire frequency could be expected. This would then have a greater negative impact on vegetation change and could lead to a decline in biodiversity for the Cape of Good Hope reserve (Gill and Bradstock, 1995; Charrette et al., 2006; Fisher et al., 2009). The value of Hugh Taylor's (1969) plots as well as the photographs presented in this study will increase with time as they provide one of the few permanent records of long-term change in fynbos vegetation.

## **LAND COVER CHANGE**

The study of long term vegetation change is essential in understanding how ecological processes function over time. Vegetation changes can give an indication of how environmental factors have changed and how vegetation has responded to these changes. The use of aerial photographs to analyse changes in vegetation attributes has been widely used and are useful as they often can provide information across a long time sequence as well as offering good resolution for large areas (Mast et al., 1997; Carmel and Kadmon, 1998; Ekhardt et al., 2000; Fensham et al., 2002).

Although aerial photography has been used extensively, there have been no key studies that have assessed vegetation cover change in the Western Cape. One study was conducted in Australia found that after a wet period in 1970, vegetation cover increased (Fensham et al., 2002). Other studies have looked at structural and successional changes in vegetation, such as trends in woody vegetation cover in Kruger National Park and found that it was in fact decreasing due to frequent fire and the impact of elephant populations.

The main objective of this section of the chapter was to:

- i. Classify and compare land cover types over a 63 year period (1945 - 2008) using aerial images.

## Methods

Two sets of aerial images were used from two time periods, 1945 and 2008. The 1945 series was chosen, as it is the oldest available, good quality set of aerial photographs. The 2008 images are the most recent aerial images available and thus a 63-year time period was compared. Unfortunately, the aerial photographs that were taken between 1946 and 2007 are of very poor quality and it was therefore decided not to use them as it would have been difficult to identify cover classes on the images.

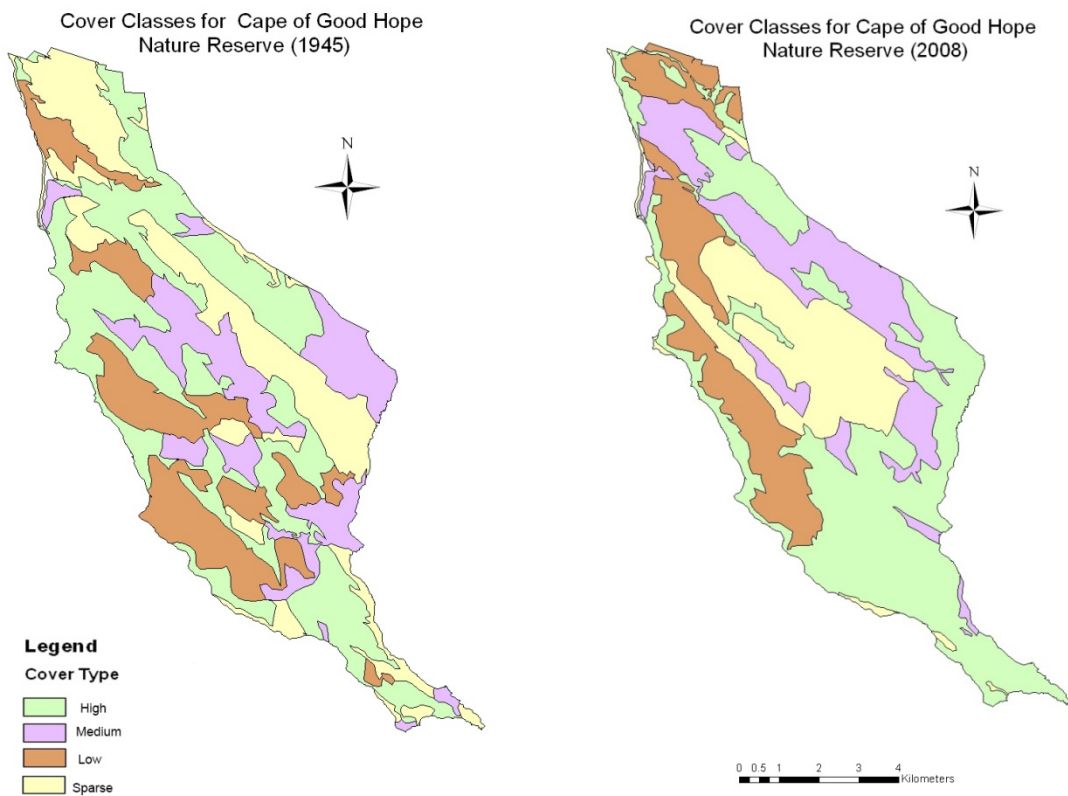
ArcGIS (ver 9.3) was used to analyse the aerial images. The aerial image was added to ArcGIS as a layer. The vegetation map (Figure 1) of the reserve (which was digitised from Taylor's (1969) vegetation map) was merged and used as the base polygon. This layer was made transparent with a border and laid over the aerial image. A scale of 1:15 000 was chosen providing the appropriate amount of detail at a landscape scale. Cover classes were defined (Table 3.3) with the corresponding class values. The different cover polygons were digitised and cut out of the base polygon. Cover classes were then assigned in the attribute table as sparse (0-5 %), low (5-25%), medium (25-50%) and high (50-100%) vegetation cover class values. This was done for both the 1945 and 2008 images and maps of the different cover classes were produced (Figure 3.15).

## Results

**Table 3.3: A description of the different cover classes and their associated percentage cover values.**

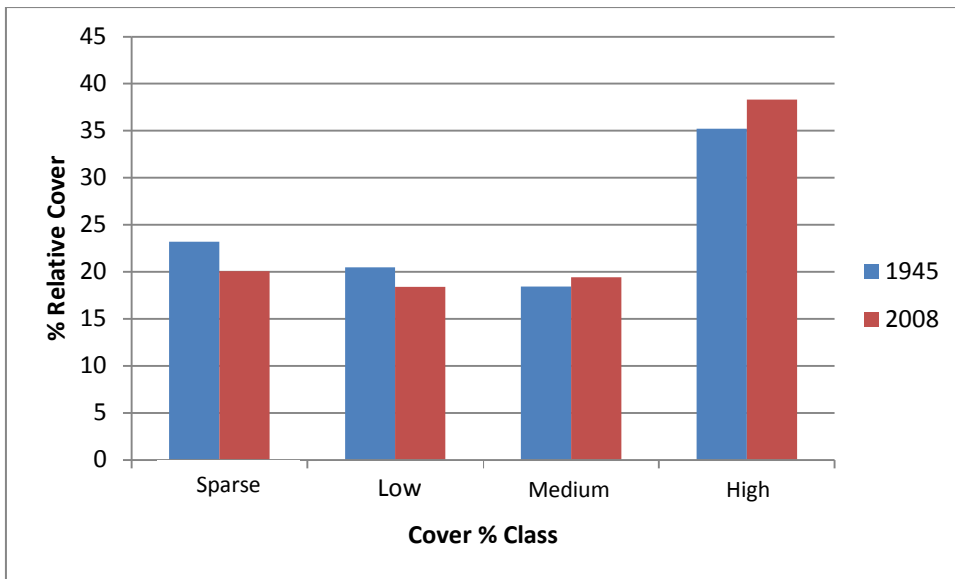
<b>Cover Class Value</b>	<b>Category</b>	<b>Description</b>
0-5%	Sparse	Bare ground to sparse vegetation cover, usually associated with sandy habitats, referred to as sparse
5-25%	Low	Low vegetation cover, usually associated with rocky habitats, referred to as low
25-50%	Medium	Low to moderate vegetation cover associated with sandy, rocky and non-rocky habitats, referred to as medium
50-100%	High	Moderate to high vegetation cover associated with sandy, rocky and non-rocky habitats, referred to as high

When comparing the two maps (Figure 3.15), it is apparent that the high cover class has increased in area since 1945. The different cover classes also appear less fragmented, with 16 patches in 2008 relative to 40 patches in 1945 indicating a general homogenisation of cover classes across in the reserve.



**Figure 3.15: Map representing the different cover classes as digitised by aerial images from 1945 and 2008.**

Figure 3.16 shows the specific changes in relative cover of the different cover classes. There has been an increase of 3.3% in the high cover class with a corresponding decrease in the sparse and low cover classes between 1945 and 2008. The medium cover class has also increased slightly by 1.4 %. These changes however, are not significant ( $n=4$ ,  $\chi^2=0.786$ ,  $p>0.05$ ).



**Figure 3.16: Graph showing changes in cover from 1945 to 2008 at Cape of Good Hope Reserve, classified by the different cover classes.**

## Discussion

Evidence from an analysis of aerial photographs between 1945 and 2008 suggests that the four different land cover classes identified in this study have not changed significantly across the reserve. The number of patches of discrete land cover classes, however, has decreased between 1945 and 2008. This could have resulted from the management practice at the time, with farmers managing their land separately and as they saw fit. These practices were not necessarily focused on good conservation practice but rather optimised the land for their livestock. In some areas vegetation cover was very low, particularly in 1945. Subsequent to the establishment of the reserve, overall vegetation cover has not decreased significantly and become more homogenous as a result of the reserve being managed as a single management unit.

## Summary of Drivers of Change

The most important change in the climate of Cape Point has been the significant increase in maximum temperature of 2.3 °C from 1930 to 2010 and the decline in relative

humidity from 86 to 82% since 1978. None of these other climate variables measured showed a significant change. These results mostly confirm the trajectories of other studies in the winter rainfall region (e.g. Haensler et al., 2010; Hoffman et al., 2011) although the non-significant change in minimum temperature is somewhat anomalous in this regard. The fire record showed that the number of fires at the Cape of Good Hope reserve has decreased since 1940 which contrasts with the findings of Forsyth and van Wilgen (2008) for the Table Mountain National Park. However, the results of the latter study were for the whole of the Cape Peninsula and not exclusively for Cape Point, which may explain the conflicting results. The occurrence of fire is not spatially uniform across the reserve. Since 1975, when the collection of spatially-explicit fire records was introduced, the northern section of the reserve appears to have burnt more frequently than the southern section. A broad-scale aerial photograph analysis of change in four land cover classes between 1945 and 2008 indicated that there has been a decline in the degree of fragmentation of the land cover classes across the reserve. The effect of this on species diversity, fire regimes and a range of ecosystem processes such as seed dispersal, pollinator movements, competition, nutrient cycling and hydrology is extremely complex and poorly studied. Changes in climate, fire and broad land cover classes will be used to interpret the findings from subsequent chapters which provide a more detailed investigation of the change in vegetation cover, growth form composition and emergent Proteaceae individuals over the period 1966-2012.

# Chapter IV: Photographic Evidence of Long-term (1966-2012) Vegetation Change in the Cape of Good Hope Section of Table Mountain National Park

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## Introduction

The Cape of Good Hope Section of Table Mountain National Park (hereafter referred to as “Cape of Good Hope” or “the reserve”) lies within the Cape Floristic Region (CFR). The flora in this region has a high diversity of species and hosts a large number of endemics (Goldblatt and Manning, 2002). It is regarded as a major biodiversity hotspot in the world (Cowling et al., 1997; Myers et al., 2000; Joppa et al., 2011). Understanding what maintains diversity in this system, and how total vegetation cover and the cover of major growth forms change over time in response to key drivers, is crucial to conserving it (Sousa, 1984; Glenn-Lewin; van der Maarel, 1992, Thornicke et al., 2001).

The cover and composition of vegetation in most communities within the fynbos biome is determined primarily by soil type, climate, fire and land use (Deacon et al., 1992; Meadows, 2006). Since soil type is relatively uniform within the Cape of Good Hope (Taylor, 1969; Hall, 2010), the focus in this chapter is on the influence of climate, fire and land use on vegetation and growth form cover. The potential influence on the vegetation of elevated levels of atmospheric carbon dioxide and nitrogen, due to anthropogenic impacts, is also discussed briefly. These drivers have important implications for vegetation cover and composition and could influence future vegetation dynamics in the region (Midgley et al., 1995; Poorter, 1998; Midgley et al., 1999)

## ***Climate***

Climate is a major determinant of overall vegetation health and plant survival since physiological processes within individuals are influenced by temperature, available moisture and wind speed amongst several other factors (Campbell and van der Meulen, 1980; Jungerius and van der Meulen, 1997; Roderick et al., 2009). The winter rainfall region of Southern Africa has been reported to be one of the regions in the world which will be most affected by climate change (Haensler et al., 2010), with temperatures expected to increase and rainfall expected to decrease (Midgley et al., 2005). Projected changes by 2050 are for a rise in temperature of between 1.5 and 3.5 °C and a decrease in annual rainfall of 41 mm which is 20% lower than current levels (Haensler et al., 2010; Midgley et al., 2005). Changes in the wind regime could also alter the vegetation of the CFR, with increases in wind speed leading to an increase in evaporation, a decrease in available water and potentially, an associated decrease in vegetation cover. Conversely, a decrease in wind speed could potentially result in a decrease in evaporation and an increase in vegetation cover (Hoffman et al., 2011).

Climate change is predicted to have a significant impact on the cover and composition of fynbos vegetation, with models predicting extensive range contractions of species. Under extreme scenarios, a loss of up to 30% of the Cape Floristic Region's (CFR) species is expected (Midgley et al., 2003; Midgley et al., 2005) as a result of an increase in drought stress (Miller et al., 1983; Midgley et al., 2003; Jacobsen et al., 2009) although recent models are considerably more conservative in their projections (Driver et al. 2012). These negative impacts could potentially result in a dramatic loss of vegetation cover as well as a shift in the composition of fynbos vegetation, seemingly towards more arid-adapted species and less diverse plant communities (Midgley et al., 2003; de Dios et al., 2007; Midgley et al., 2007).

## ***Fire***

Fire is a major driver of vegetation change in most Mediterranean-type ecosystems (Zedler, 1995; Bond and van Wilgen, 1996; Bond et al., 2003a; Keeley, 2009; Rutherford et al., 2011). Fire can reduce vegetation cover significantly by burning above and sometimes even below ground vegetation (Keeley and Fotheringham, 2000). Fire can also determine which species dominate a site as it is linked to the germination of many species (Mutch, 1970; Bond et al., 1984; Lamont et al., 1991; Midgley et al., 1998; Keeley and Fotheringham, 2000). The fire regime in a region and how it changes over time can thus alter the species composition of the area (Driscoll et al., 2010; Rutherford et al., 2011).

After a fire has burnt an area, there is a process of post fire succession (Taylor, 1969; Hoffman et al., 1987; Rutherford et al., 2011; van Wilgen, 2012; Keeley et al., 2012). This succession involves an increase in vegetative cover as well as a shift in the dominance of families and growth forms (e.g. annuals, geophytes, restioids, ericoids and proteoids). A general pattern for the process might be described as follows. For the first year after a fire, the site is often dominated by a flush of annuals, geophytes and nitrogen-fixing shrubs within the family Fabaceae (Cowling and Pierce, 1988). Restioid and graminoid growth forms may also dominate this early successional period, particularly in lowland fynbos vegetation (Hoffman et al. 1987). As vegetation recovers, however, the dominance of restioids decreases and ericoid shrubs start to become more dominant. Often, the dominance of tall, broad leaved proteoid shrubs coincides with that of shorter, small-leaved ericoid shrubs (Bond et al., 1984). However, in old (>25-30 years) post fire age veld, the presence of proteoid shrubs often decreases and ericoid shrubs become the most dominant growth form (Taylor, 1969; Cowling et al, 2007; Keeley et al., 2012). While general patterns are discernible, post-fire successional processes are complex and vary considerably between sites and are influenced by soil or moisture conditions as well as by stochastic processes (Connell & Slatyer 1977; Noble & Slatyer 1980; Pickett et al. 1987). For instance, often a moist, marsh-like environment can host long-lived restioids species and will never support a dominance of ericoid shrubs (Bond et al., 1992; Capitanio and Caraillet, 2008; Rutherford et al., 2011).

## ***Land Use***

Land use, particularly cultivation and herbivory, has an impact on vegetation cover and is a leading cause of species loss in many fynbos communities (Cowling, 1995). Although the reserve has a history of farming, it was never very intensive. The land is not suitable for cultivation and only small-scale subsistence agriculture was practised in the past. The reserve was established in 1965 and appropriate conservation practices were carried out from then onwards (Privett, 1998). Based on an analysis of aerial photographs (Chapter III) between 1945 and 2008, the impact of land use on the vegetation of the Cape of Good Hope since the establishment of the protected area appears not to have had a negative effect on the cover of vegetation in the reserve.

## ***Atmospheric CO<sub>2</sub> Concentration***

Anthropogenic climate change is associated with increased levels of atmospheric CO<sub>2</sub> (Ziska et al., 2009; Oliver et al., 2009; Oishi et al., 2009; Keenan et al., 2011). This is predicted to have a positive impact on vegetation biomass in many southern African biomes, since elevated CO<sub>2</sub> is thought to favour the establishment and growth of woody species in particular (Bond et al., 2003b). While no definitive evidence has been found that fynbos species respond to elevated levels of carbon dioxide (Midgley et al., 1995; Poorter, 1998; Midgley et al., 1999) more research is needed in this regard.

## ***Evaluating Change in Fynbos Vegetation***

Within fynbos vegetation, total vegetation cover, growth form cover (e.g. restioid, ericoid, proteoid) as well as the abundance of different species changes over time in response to major disturbance events such as fire and drought (Fisher et al., 2009; Driscoll et al., 2010; Rutherford et al., 2011; West et al., 2012). The nature, extent and direction of such changes can be used as proxies for shifts in the underlying environmental conditions (Campbell, 1986) or disturbance regimes (Volk and Yeaton, 2000; Fisher et al., 2009) at a site or within a landscape. For example, an increase in the abundance of species such as *Osmitopsis asteriscoides*, and *Berzelia abrotanoides* and growth forms (e.g. restioids or medium-high ericoid-leaved shrubs) which are usually

associated with wetlands or seepage areas would suggest that more water has become available at such a site over time (McDonald et al., 1996). Alternatively, a decline in, or local extinction of species at a site within the re-seeding Proteaceae guild, for example, would indicate that fires have been too frequent and that individuals have not been able to flower and set seed between fire events (Gill and Bradstock, 1995; Keeley, 1995; Charrette et al., 2006; Fisher et al., 2009).

Difference in post-fire age in the vegetation observed at two different time periods is a critical factor to keep in mind when assessing the nature and extent of change in fynbos vegetation (Daubenmire, 1968; Cowling, 2001; Cowling et al., 1997). Successional trajectories are strongly influenced by post-fire age (Rutherford et al., 2011) although subsequent disturbance events (e.g. drought, small-mammal diggings) can deflect such trends and pathways (Mokotjomela et al., 2009). It is reasonable to assume, however, that in the absence of major environmental change or shifts in the disturbance regime, total plant cover and the cover of major growth forms at a site should be similar for vegetation of the same post-fire age. However, if environmental conditions deteriorate at a site, for example, because of higher temperatures, greater evaporative demand and lower rainfall or from intense disturbance by wind or animals overall cover should decline as fewer individuals and species will be able to survive under such conditions (Midgley et al., 2003; Jacobsen et al., 2009). Conversely, increased rainfall and less evaporative demand and perhaps also the effects of elevated atmospheric levels of CO<sub>2</sub> should result in an increase in vegetative cover at a site (Poorter, 1998; Midgley et al., 1999). Shifts in the abundance of different species, growth forms and reproductive guilds (e.g. obligate reseeders versus resprouters) thus provide additional information for the interpretation of long-term changes in the prevailing environmental conditions or disturbance regimes.

## ***Prior Research on the Vegetation of the Cape of Good Hope***

Previous studies on the vegetation of the Cape of Good Hope have focused on vegetation description and change in diversity over time. Taylor's (1969) comprehensive survey of 100, 10 m x 5 m plots, and accompanying photographs, has provided the baseline against which subsequent changes have been measured. Using the same approach, Privett (1998) relocated and resurveyed 81 of Taylor's (1969) original plots. His results showed that there was considerable turnover in species within a plot but that overall there was little change in species richness at a meta-community level (Thuiller et al., 2007). For example, while 80.3% of species were recorded as having disappeared from at least one of the re-sampled sites, 73.9% were considered as colonisers, or species that were present in a plot in 1996 but not in 1966. Furthermore, 62 new species (15.6%) were recorded in 1996 while almost twice as many species (124 or 29.4%) were not recorded at any of the 81 sites at this time (Thuiller et al., 2007). The very high turnover in species (e.g. 74% of the sites exhibited >50% turnover) was attributed to stochastic variability in the fire regime and immediate post-fire weather conditions. Density-dependent interactions were considered unimportant determinants of the remarkable temporal constancy observed at the meta-community level as were the particular biological traits of species and growth forms (e.g. resprouter versus reseeder).

Using the same approach as Privett (1998), Hall (2010) located and resampled 26 of Taylor's original plots. His main finding was that species richness had decreased by almost 25% over the 44-year period. This decrease was attributed to changes in fire regime, grazing pressure, alien species and anthropogenic climate change. It was also found that coastal vegetation types were very stable whereas inland, fynbos-associated sites were highly dynamic and unstable in terms of species diversity. Fire history was identified as a major driver, with sites with similar fire ages showing little difference and conversely, sites with large difference in fire age exhibiting large changes in species richness.

## ***Objectives***

Previous research in the reserve has focused on changes in plant richness within 10x5 m plots over time. The current study, however, used Taylor's (1969) collection of historical photographs of each of his 10x10m plots to assess changes in the vegetation of the Cape of Good Hope Section of Table Mountain National Park at a broader landscape scale. The main objectives of this study were to:

- i. Document the change in total vegetation cover in relation to post-fire age between sites and within three major vegetation types (Dune Asteraceous Fynbos, Restioid Fynbos, Proteoid Fynbos) for the period 1966-2012 as determined from an analysis of repeat photograph pairs. These vegetation types were derived from a synthesis of existing vegetation classification analyses for the reserve (see Appendix D).
- ii. Determine changes over the 46-year period in four main growth forms (restioids, ericoids, proteoids and broad leaved shrubs) which dominate the three vegetation types of the reserve in relation to post-fire age.
- iii. Use Markov transition matrices, based on the historical changes in the four main growth forms, to project the major trends in the vegetation of the reserve over the next 50 years.

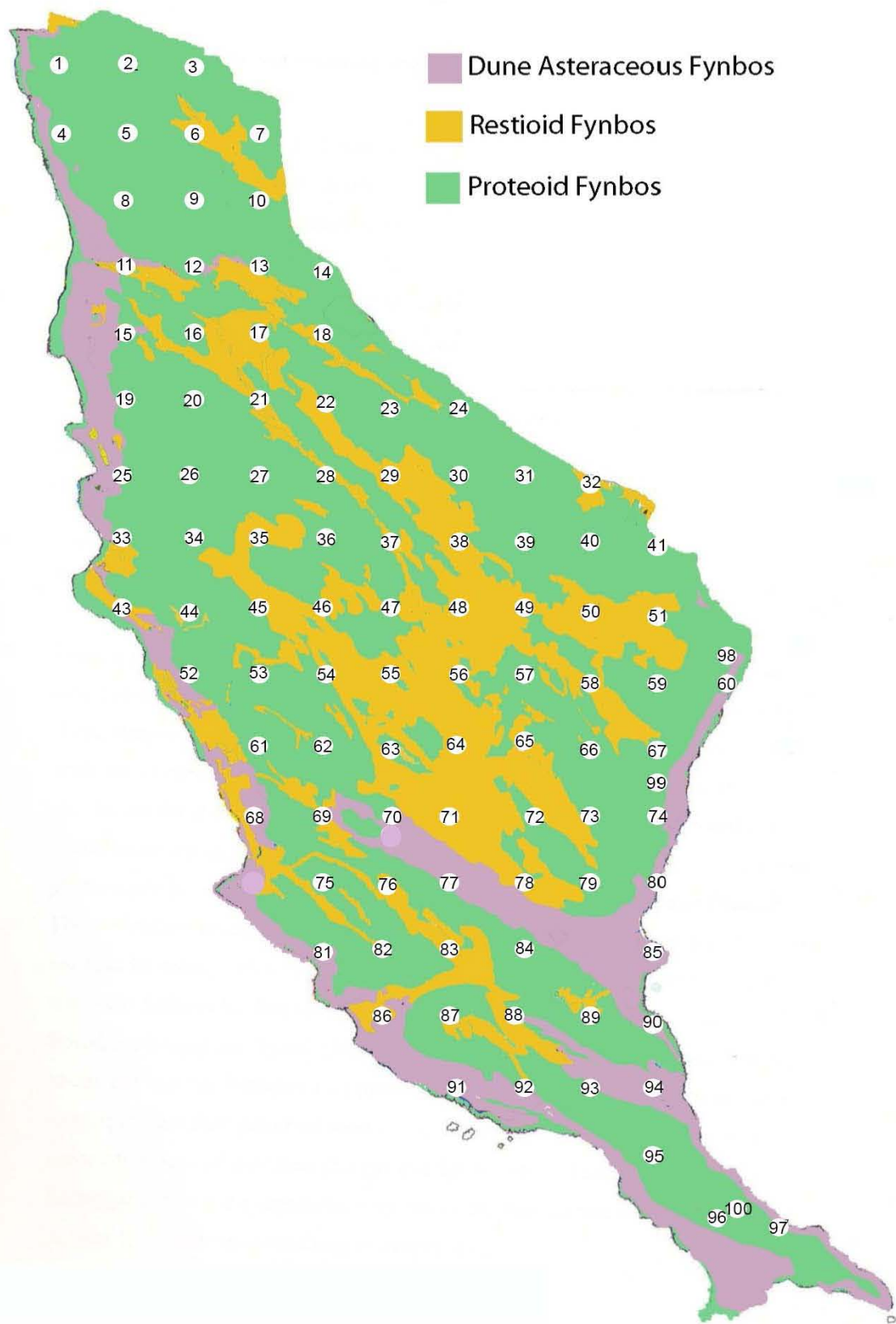
## **Methods**

### ***Site classification***

In order to analyse major changes in vegetation over time, study sites were grouped into vegetation types using available vegetation classifications for the Cape of Good Hope produced by Taylor (1969), Privett (1998) and Mucina and Rutherford (2006) (see Appendix D). Hugh Taylor created his vegetation map in 1966 from the species lists he compiled in the field at each of his 100, 10 x 5 m survey plots. He classified his vegetation types manually, using standard phytosociological techniques available at the time (Daubenmire 1968). Privett (1998) resampled Taylor's survey plots and updated the

species lists to reflect their abundance within the plots in 1996. He conducted a TWINSpan analysis that defined groups for the different sites based on species abundance and produced a map of the vegetation of the Cape of Good Hope (Appendix D, Figure 2). Mucina and Rutherford's (2006) classification places the vegetation of the Cape of Good Hope within a broader national perspective. Their map is based on a synthesis of the results from Taylor (1969), Privett (1998) and other contributing projects conducted in the region.

Each of these studies used different methods to derive their classifications and two or sometimes three names exist for each of the vegetation types. Privett (1998) used quantitative tools at an appropriate community-level of scale to group sites in the reserve. Because his classification of vegetation types has been used by subsequent researchers (Thuiller et al., 2008) it was considered most suitable for this study. However, Privett's (1998) treatment of sites associated with coastal thicket and Dune Asteraceous Fynbos was too finely disaggregated. Sites associated with these coastal communities were more generally treated by Taylor (1969), in a classification scheme supported by Mucina and Rutherford's (2006) concept of these vegetation types. Based on a consideration of the different classification schemes sample sites were grouped into three main vegetation types (Figure 4.1), namely Dune Asteraceous Fynbos, Wet Restioid Fynbos (referred to hereafter as Restioid Fynbos) and Mesic Oligotrophic Proteoid Fynbos (referred to as Proteoid Fynbos).



**Figure 4.1** Vegetation map showing spatial distribution of combined vegetation types and the location of Taylor's (1969) 100, 10x5 m plots (after Privett, 1996 and Mucina and Rutherford, 2006).

## ***Repeat photography***

In addition to his species data for 100, 10x5 m plots at the Cape of Good Hope reserve, Hugh Taylor also usually took a photograph of most of his plots. Eighty-six of his original photographs, which all date to 1966, are currently available in SANBI's photographic archive collection and in his thesis (Taylor, 1969). For the purposes of this study, these photographs were all scanned as high resolution TIFF images either from the original negatives, or when these were not available, from prints in his thesis. Seventy-eight of these photographs were relocated in the field (Appendix A). At each of these sites, the image was rephotographed using three different cameras. These were a Mamiya 645 medium format camera loaded with Ilford FP4Plus 120 black and white film, a Minolta X70 35 mm camera filled with Fujichrome Velvia 35 mm colour slide film and a 23 megapixel digital camera (Canon 5D Mark II). While only the digital images were used in the analysis the black and white and colour slide film was archived together with the data sheets completed for each site in the Plant Conservation Unit's repeat photographic archive. The landscape that was in the field of view in the image was divided into different landforms (e.g. level plain, seasonally-waterlogged marsh, rocky outcrop, etc.) and notes were recorded in the field on the major differences in the vegetation within each landform between the two time periods. The vegetation at each site was also surveyed by walking through the landscape within the main field of view. The dominant plant species that were found in the area were photographed and percentage cover was visually estimated by consensus between two ecologists for each of these species in the field (Appendix C). The dominant species were also grouped within major growth forms including annuals, geophytes, graminoids, restioids, ericoid shrubs, proteoid shrubs, broad-leaved (thicket) shrubs and leaf succulents and a cover estimate for each growth form was determined in the field.

## ***Comparison of Average Post Fire Age and Difference in Post Fire Age***

Fire records were collected from Privett's thesis (1998) as well as from SANParks' Cape Research Centre (Forsyth and van Wilgen, 2008). The record from Privett (1998) extended from 1940-1996, and was compiled from estimates of the post-fire age for each of 100 plots, by counting the number of nodes and internodes present on the branches of Proteoid shrubs found in the plot (Hall, 1959). This source of information was used in determining the fire record for the period 1966-1974 but from 1975 more accurate fire frequency data were available from the Cape Research Centre (Fire Section, Chapter 2).

Post fire age is the number of years after a plot has burnt and is calculated as follows:

**Post fire age** (*e.g. 14 years*) = **year image was taken** (*e.g. 1966*) – **year when fire occurred** (*e.g. 1952*)

Post fire age of each of the plots was calculated for the original image (1966) and the repeat image (2012). The average post fire age was calculated for each of the vegetation types (Dune Asteraceous Fynbos, Restioid Fynbos, Proteoid Fynbos) from the fire record for 1966 and 2012. A student t-test was then conducted on the data to see if there was any significant difference in average post fire age between the two years for each of the vegetation types.

Difference in post fire age is a proxy for expected change in vegetation cover and was calculated as follows:

**Difference in post fire age** = **post fire age of site**<sub>(repeat 2012)</sub> – **post fire age of site**<sub>(original 1966)</sub>

A frequency distribution histogram was produced of the difference in post fire age for all the sites within 10 year time step intervals ranging from -30 to 50. Difference in post fire age was also displayed on the figures showing % change in vegetation cover.

## ***Analysis of Percentage Change in Vegetation Cover Using Repeat Photographs***

The original and repeat photographs were matched as accurately as possible using Photoshop CS5.1 (2011). A grid of 260 points (20 x 13 points) was then laid over the original photo and the repeat photo. A number of grid sizes (100 – 500 points) was tested on a photograph and a grid size of 260 points was selected. This grid size was chosen as it clearly showed changes in the landscape when analysed. Smaller grid sizes did not record all the changes and larger grid sizes were time-consuming to analyse and did not provide greater accuracy in determining the major trends in vegetation cover and composition. A growth form class for restioid, ericoid, proteoid and broad leaved shrubs was assigned to each intersecting point on the grid. If the point fell on sand or rock this was scored as a separate cover class. Null values were also classified for entities such as the sky, sea and roads. These points were disregarded and removed from the 260 points. The proportion of each cover class (expressed as a percentage) in the original photo and repeat photo was calculated respectively using the following formula:

$$\text{Proportion of cover class} = \frac{\text{no. of points of cover class}}{\text{total no. of points (260-no. of null points)}} \times 100$$

Values for each of the cover classes in the original photograph were subtracted from values in the repeat photograph to provide an estimate of the percentage change in cover. Values for each of the sites within each of the three vegetation types were represented in separate bar graphs. It was recognised that the relationship between post fire age and vegetation cover is not linear and the data were transformed accordingly (Appendix E). This transformation did not deviate from the original figures and it was decided to use the original figures (not transformed) in this thesis for simplicity and for better comparison of the results. Scatter plots with the 95% confidence intervals indicated, were also produced to show the relationship between difference in post fire age and percentage vegetation change.

Similarly, the percentage change in cover of the main growth forms (restioids, ericoids, proteoids and broad leaved shrubs) were calculated in the three vegetation types. The average change within different post fire age categories was calculated and graphed in a histogram so as to better display trends.

### ***Transition Matrix Projections***

A Markov transition matrix was used to assess trends and predict changes in the dominance of key growth forms for the period 1966 to 2058 (Balzter, 2000; McAuliffe, 1988; Horn, 1975). The transition matrix is based on Markov chains developed by the Russian mathematician Markov in 1907. A Markov chain represents a system of elements transitioning from one state to another over time. The conditional probabilities  $P(X_t = j | X_s = i) = p_{ij}(s,t)$  are called transition probabilities of order  $r = t - s$  from state  $i$  to state  $j$  for all indices  $0 \leq s \leq t$  with  $1 \leq i, j \leq k$ . They are denoted as the transition matrix  $P$ . For  $k$  states  $P$  has the form:

$$P = \begin{bmatrix} p_{11} & p_{12} & \dots & p_{1k} \\ p_{21} & p_{22} & \dots & p_{2k} \\ \dots & \dots & \dots & \dots \\ p_{k1} & p_{k2} & \dots & p_{kk} \end{bmatrix}$$

At time 0 the initial distribution of states in  $P(X_0 = i) = p_i(0) \quad \forall i \in \{1, \dots, k\}$

The state probabilities  $p_i(t)$  at time  $t$  are estimated from the relative frequencies of the  $k$  states, resulting in the vector

$$p(t) = (p_1(t), p_2(t), \dots, p_k(t))$$

Denoting the  $v$ -th observed state with  $i_v$ , a stochastic chain fulfilling the following equation is a first order Markov chain:

$$P(X_{t+1} = i_{v+1} | X_t = i_v, X_{t-1} = i_{v-1}, \dots, X_0 = i_0) = P(X_{t+1} = i_{v+1} | X_t = i_v) \\ \forall v \geq 2, \forall i_0, i_1, \dots, i_{v+1} \in \{1 \dots k\}$$

Predictions of future state probabilities can be calculated by solving the matrix equation:

$$p(t) = p(t-1).P$$

(Balzter, 2000).

Proportions and probabilities of change within key growth forms were calculated from percentage proportion cover values recorded from the grid which was laid over the landscape photographs. These were represented in stacked columns. This method assumes linearity and only accounts for disturbances that have occurred during the two time periods.

## **Results**

### ***Correlation between Field Estimates and Photograph Analysis***

The relationship between the field estimates of total vegetation cover and growth form cover and those derived from quantitative assessments from the photographs was significant ( $p < 0.001$ ) for all growth forms and for total vegetation cover with  $R^2$  values between 0.67 and 0.95 (Figure 4.2). None of the slopes differed significantly from a slope of 1 ( $p > 0.05$ ) suggesting that observations derived from an assessment of the photographs were a realistic reflection of the cover estimates in the field.

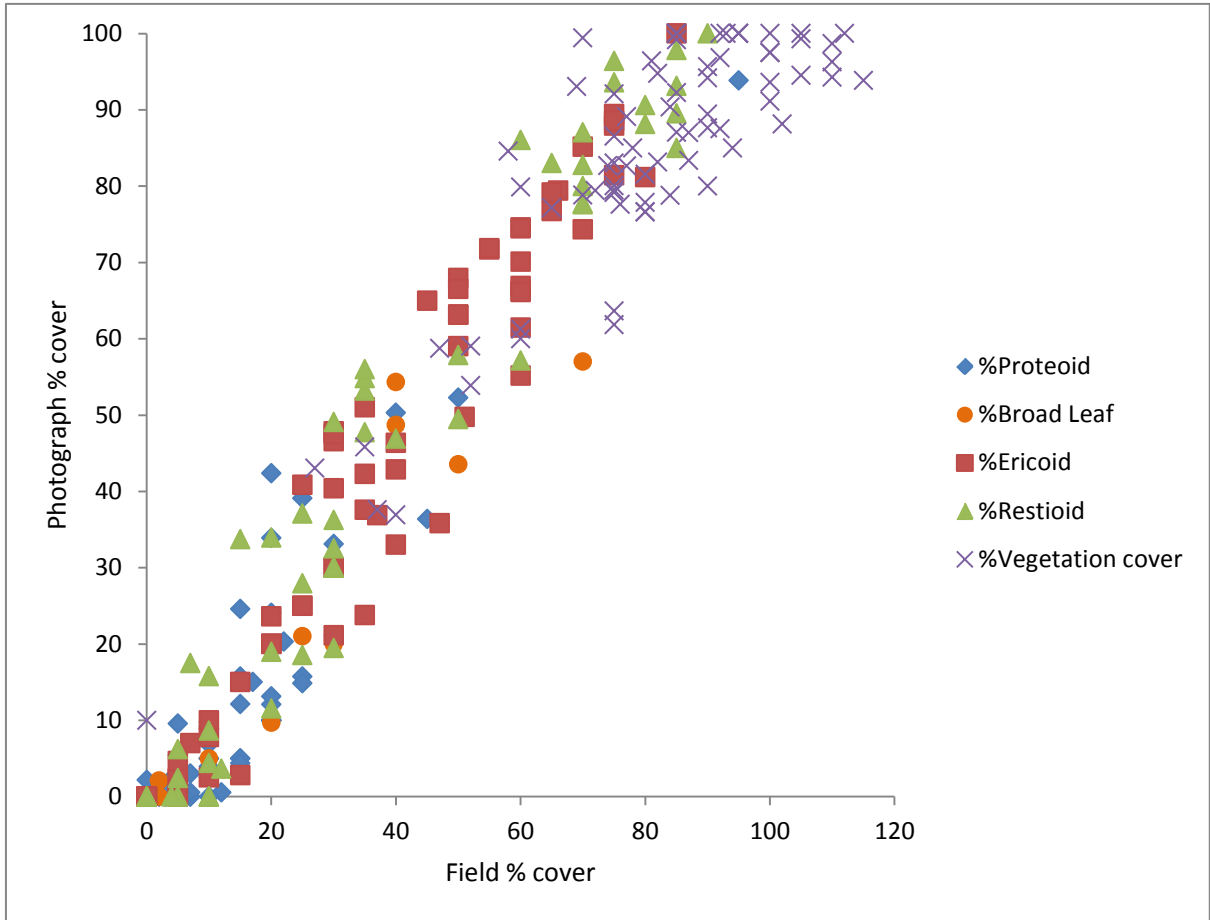


Figure 4.2: Linear regression of the relationship between growth form cover values derived from field estimates and from those derived from an analysis of photographs.

### ***Comparison of Average Post Fire Age and Difference in Post Fire Age***

From Table 4.1 it is evident that there is no significant ( $p > 0.05$ ) difference between the average post fire age in 1966 and 2012.

**Table 4.1: Comparison of average post fire age (PFA) between 1966 and 2012. A t-test shows that there is no significant difference between the average post fire age in 1966 and 2012 for sites within the different vegetation types**

Vegetation Type	n	Average post-fire age (1966)	Average post-fire age (2012)	t-test statistic	Significance and p-value
Dune Asteraceous Fynbos	12	24.8	35.3	-1.57	NS (p=0.15)
Restioid Fynbos	25	10.5	11.2	0.69	NS (p=0.50)
Proteoid Fynbos	40	12.4	17.2	0.61	NS (p=0.46)

There is a normal distribution of difference in post fire age across categories with the majority of sites (18 out of 78) in all vegetation types exhibiting differences in post fire age of between 0-10 years (Figure 4.3). No records exist for the 31-40 years post fire age category. Sites within Dune Asteraceous Fynbos were scattered across a wide range of post fire age categories while sites within Restioid Fynbos differed between -20 and +30 years. Sites within Proteoid Fynbos were normally distributed amongst the post fire age categories with the majority of sites (11 out of 39) being 11 to 20 years older in post fire age in the repeat photograph as compared to when the original photographs were taken.

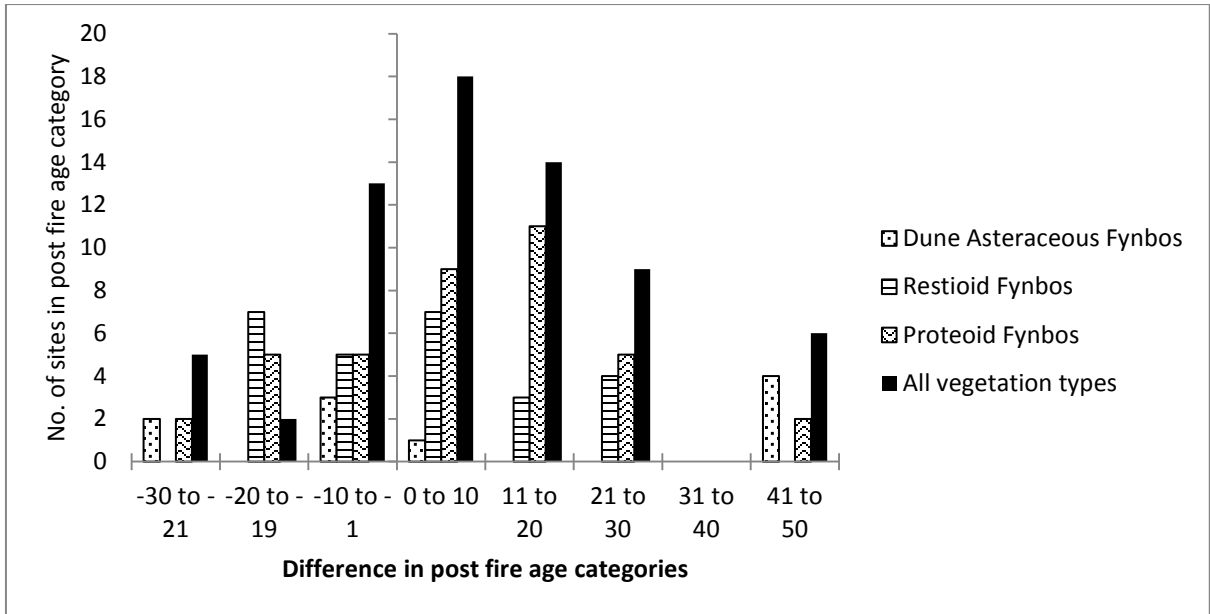


Figure 4.3: Frequency distribution of difference in post fire age in three vegetation types and for all vegetation types in the Cape of Good Hope.

### *Dune Asteraceous Fynbos*

The relationship between post fire age and % vegetation cover for Dune Asteraceous Fynbos was not significantly different for sites in 1966 and 2012 ( $n=10$ ,  $F=2.52$ ,  $p=0.09$ ) (Figure 4.4).

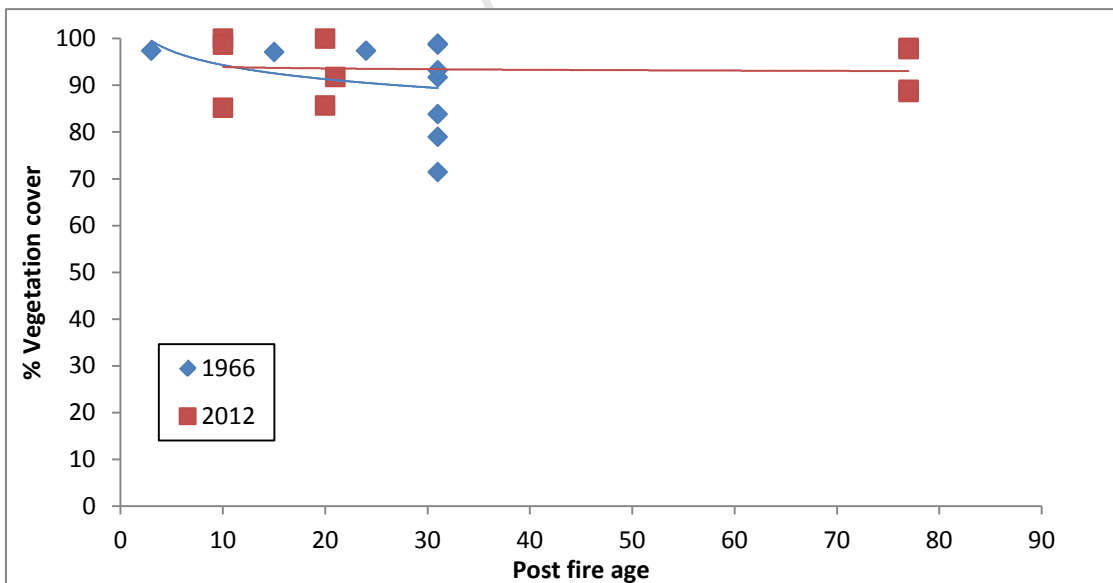
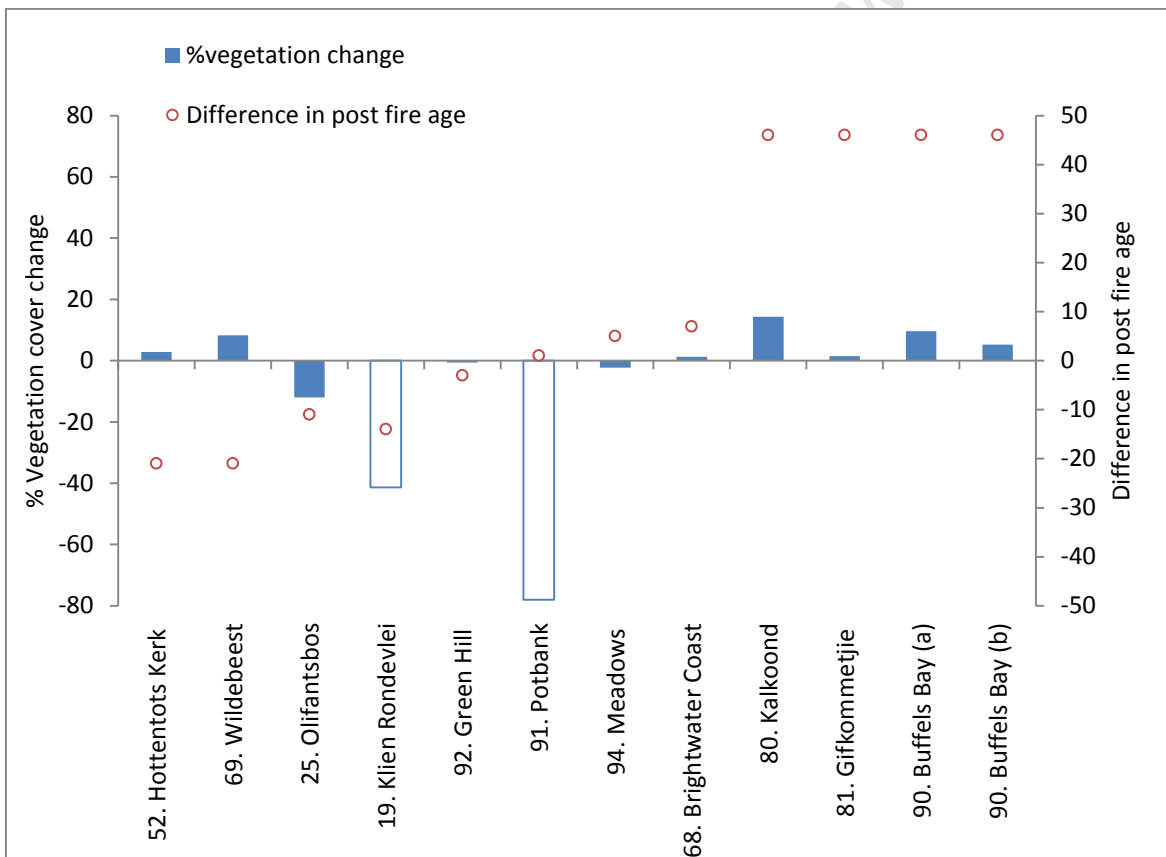


Figure 4.4: The relationship between post fire age and % vegetation cover in Dune Asteraceous Fynbos sites in 1966 ( $R^2=0.133$ ,  $p>0.05$ ) and 2012 ( $R^2=0.004$ ,  $p>0.05$ ).

More than half of the sites within Dune Asteraceous Fynbos showed changes in vegetation cover of  $\pm 10\%$  or more (Figure 4.5). A rapid decrease in vegetation cover of -41% and -80% was observed at the sites Klein Rondevlei and Potbank respectively (Figure 4.6). These sites have been disturbed by mole rats (Klein Rondevlei) and by the remobilization of the dune system (Potbank) along the coast. There was also an observed decrease in vegetation cover at Olifantsbos, Green Hill and Meadows, but to a lesser degree. The remaining seven sites show an increase in vegetation cover, with the greatest increase of 15% recorded at Kalkoond. All sites with a difference in post fire age of 40 years or more had a positive increase in cover.



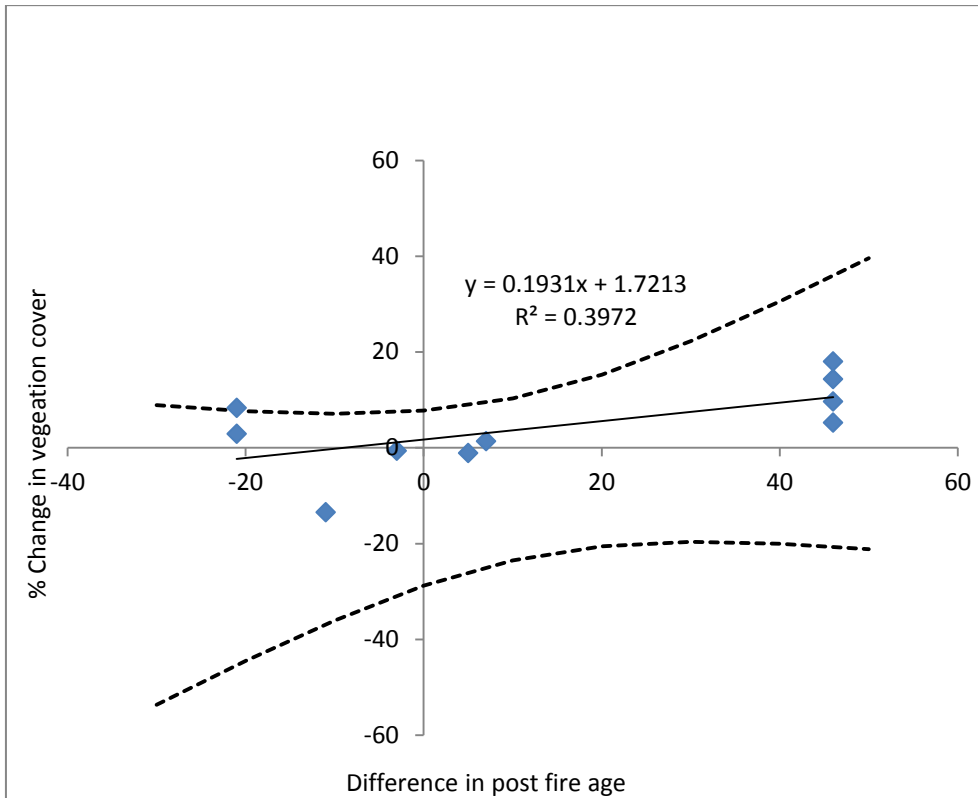
**Figure 4.5: Percentage vegetation cover change (1966-2012) for 12 sites within Dune Asteraceous Fynbos. Difference in post fire age for each site is also displayed on the graph. Outliers in the data (severely disturbed sites) are represented by clear bars.**



**Figure 4.6: Potbank (No. 965) A.1966 B. 2012. This site reflects a re-mobilization of the dune system which has removed all of the vegetation cover that was present in 1966. The original site was dominated by *Juncus kraussii* and herbaceous species such as *Chironia decumbens* and *Samolus valerandi*. There are a few remnant vegetation patches on the dune ridges on the left of the image, which are dominated by *Morella cordifolia* and a few grasses such as *Cynodon dactylon*, *Stenotaphrum secundatum* and *Ehrharta calycina*. These remnant patches appear to be quickly inundated by mobile sands.**

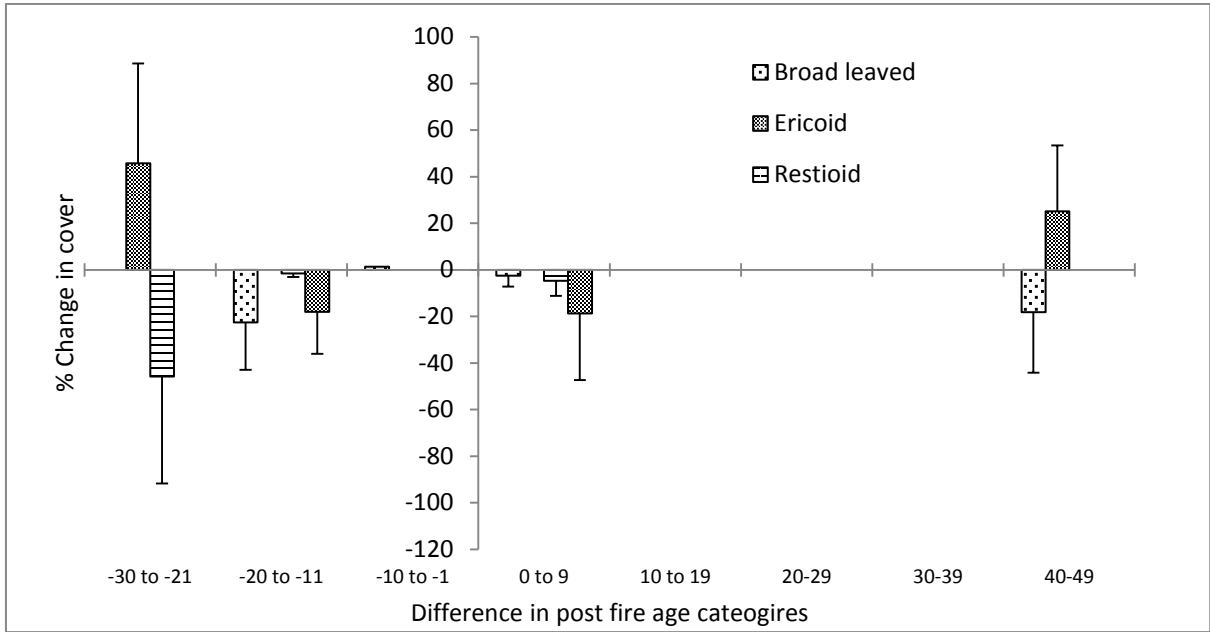
When all Dune Asteraceous Fynbos sites were considered together there was no relationship between the difference in post fire age and percentage change in vegetation cover ( $n=12$ ,  $R^2=0.1574$ ,  $p=0.2$ ). However, two sites (Klein Rondevlei and Potbank) lie outside the 95% confidence limits and both show a lower vegetation cover relative to the difference in post fire age. These differences in vegetation cover occur as the result of

increased mole rat activity (Klein Rondevlei) and dune re-mobilization (Potbank). When these two sites were excluded (Figure 4.7) from the analysis the relationship between the difference in post fire age and percentage change in vegetation cover approached significance at the 95% level ( $n=10$ ,  $R^2=0.3972$ ,  $p=0.05$ ).



**Figure 4.7: Scatterplot showing the relationship between difference in post fire age and % change in vegetation cover for 12 sites within Dune Asteraceous Fynbos (1966-2012). The linear regression as well as the 95% confidence limits is also displayed. Outliers (Klein Rondevlei and Potbank) have been excluded from the analysis.**

The percentage change in cover of the three dominant growth forms (broad leaved shrubs, ericoids and restioids) in Dune Asteraceous Fynbos (Figure 4.8) indicates a wide response with few of the changes linked to difference in post fire age. The most significant switch in growth form cover occurred at Wildebeest where the dominance of restioids in 1966 was largely replaced by ericoids in 2012 (Figure 4.9).



**Figure 4.8: Average percentage change between 1966 and 2012 (n=12 sites) in vegetation cover for Broad leaved, Ericoid and Restioid growth forms with difference in post fire age categories for Dune Asteraceous Fynbos. One standard deviation of the data is represented by the bars.**

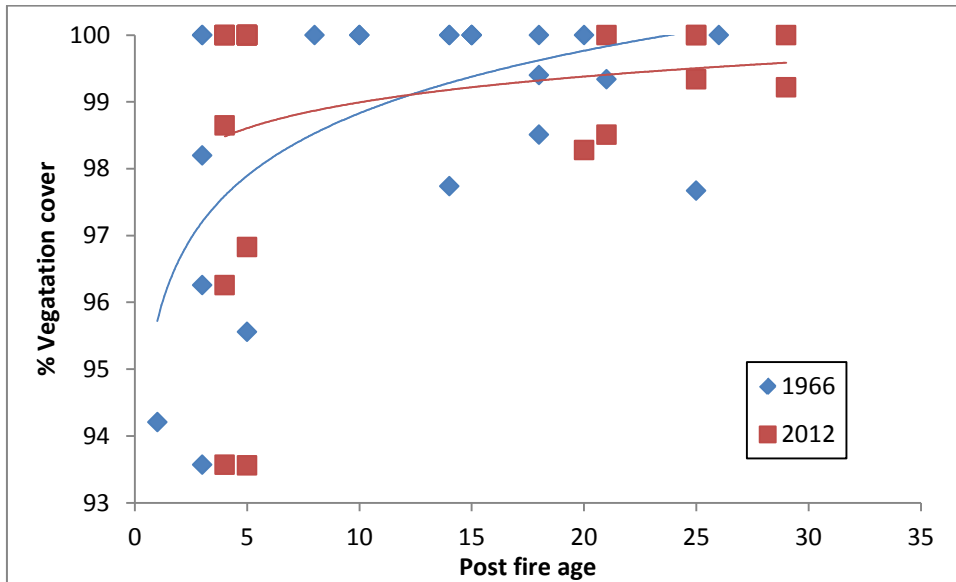
University of Cape Town



**Figure 4.9: Wildebeest (No. 945) A 1966 B: 2012. Overall there has been significant reduction in restioids which have been replaced by a diverse mix of ericoids and broad leaved shrubs, including some dune thicket specialists, such as *Pterocelastrus tricuspidatus* and *Maytenus oleoides*. The relatively open restioid veld is now a dense fynbos and dune scrub community.**

## *Restioid Fynbos*

The relationship between post fire age and % vegetation cover for Restioid Fynbos was not significantly different for sites in 1966 and 2012 ( $n=23$ ,  $F=1.05$ ,  $p=0.45$ ) (Figure 4.10).



**Figure 4.10: The relationship between post fire age and % vegetation cover in Restioid Fynbos in 1966 ( $R^2=0.366$ ,  $p>0.05$ ) and 2012 ( $R^2=0.053$ ,  $p>0.05$ ).**

There has been very little change in total vegetation cover in Restioid Fynbos sites (Figure 4.11). This stability in cover is shown clearly at Teefontein South (Figure 4.12). All sites with a negative value for vegetation cover change were also younger in post fire age in 2012 relative to 1966, with the exception of one site (Berzelia Seepage). At this site the difference in vegetation cover between the two time periods is explained by the removal of alien plants which dominated the site in 1966. The site with the greatest increase in cover within Restioid Fynbos was at Mimicki's. This increase occurred because of the removal of alien plants within a seepage area which resulted in a large increase in the cover of ericoid shrubs.

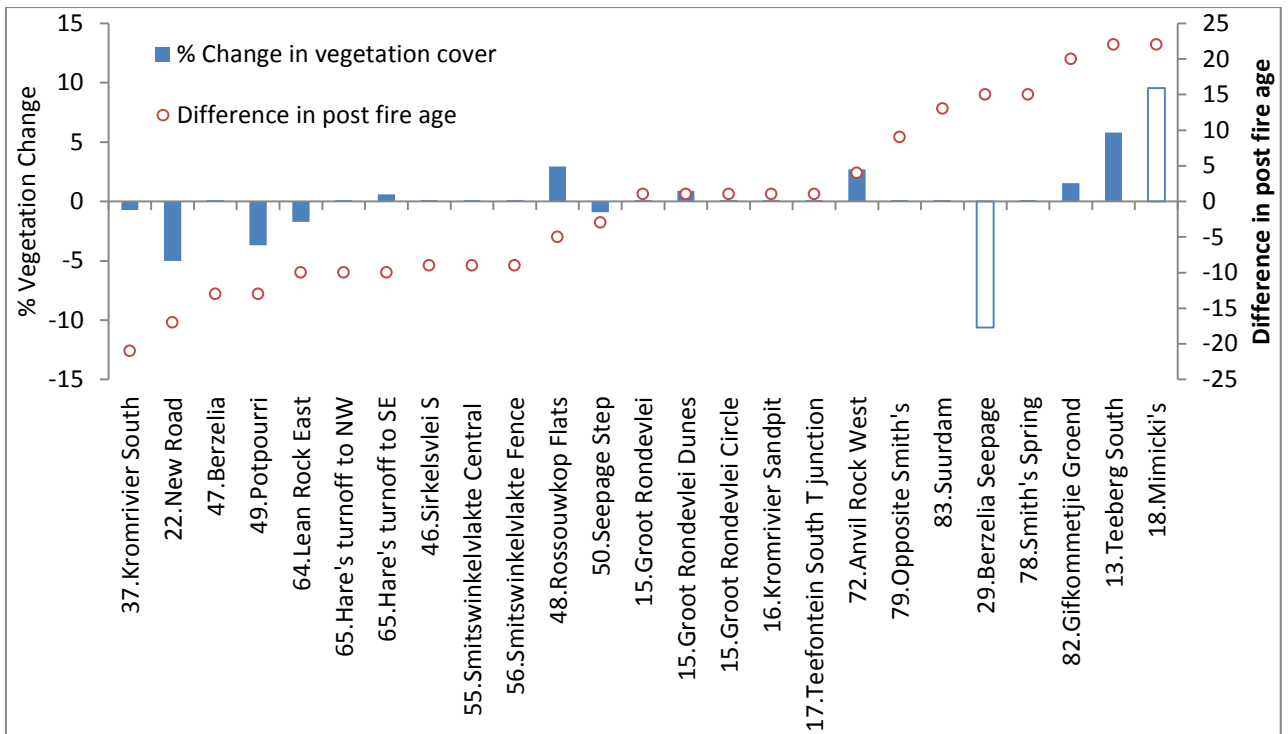


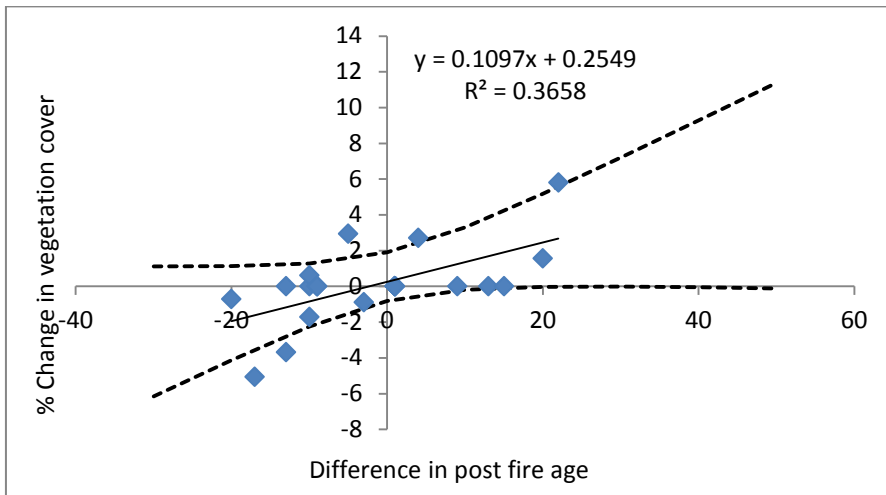
Figure 4.11: Percentage vegetation cover change (1966-2012) for 25 sites within Restioid Fynbos. Difference in post fire age for each site is also displayed on the graph.

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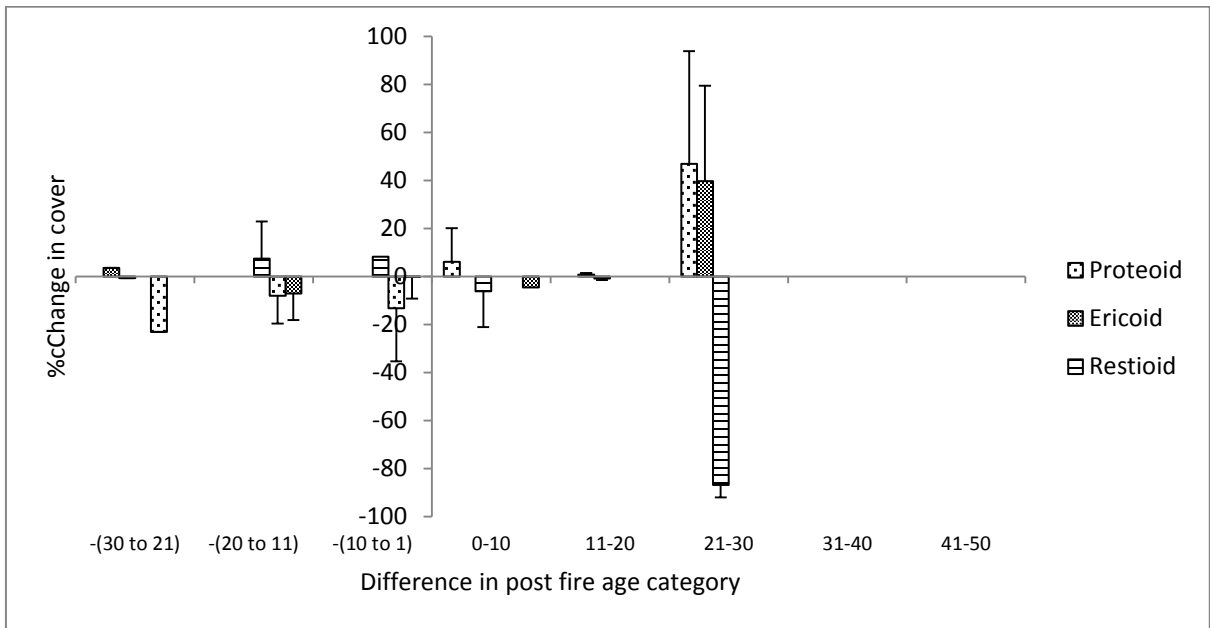
Figure 4.12: Teefontein South (No. 754) A; 1966 B: 2012. This site appears to have remained very stable over the 45 year period. The dominance of *Restio bifurcus* and *Restio quinquefarius*, is evident in the original photograph as well as in the repeat image. Vegetation cover is high and similar when comparing the two images.

The relationship between difference in post fire age and percentage change in vegetation cover in Restioid Fynbos was not significant ( $n=25$ ,  $R^2=0.1343$ ,  $p=0.07$ ) when all sites were considered. However, when the Berzelia Seepage and Mimicki's sites were removed from the analysis (Figure 4.13), the relationship was highly significant ( $n=23$ ,  $R^2=0.3658$ ,  $p=0.002$ ). Both of these outliers were dominated by alien trees in the 1966 photographs and were cleared of these elements prior to the 2012 survey.



**Figure 4.13: Scatterplot showing the relationship between difference in post fire age and % change in vegetation cover in 25 sites within Restioid Fynbos (1966-2012). The linear regression as well as the 95% confidence limits are also displayed. Outliers (Berzelia Seepage and Mimicki's) have been deleted from the analysis.**

The percentage change in cover of the dominant growth forms in Restioid Fynbos was generally small except for sites with the greatest positive difference in post fire age (21-30 years) (Figure 4.14). These sites, particularly Teeberg South (Figure 4.15) and Mimicki's, were characterised by a large reduction in restioids and an increase in proteoids and ericoids. Such transformation in physiognomy, however, was relatively rare within this vegetation type and in the case of Mimicki's at least, was related to the removal of alien plants from an upstream seepage area.



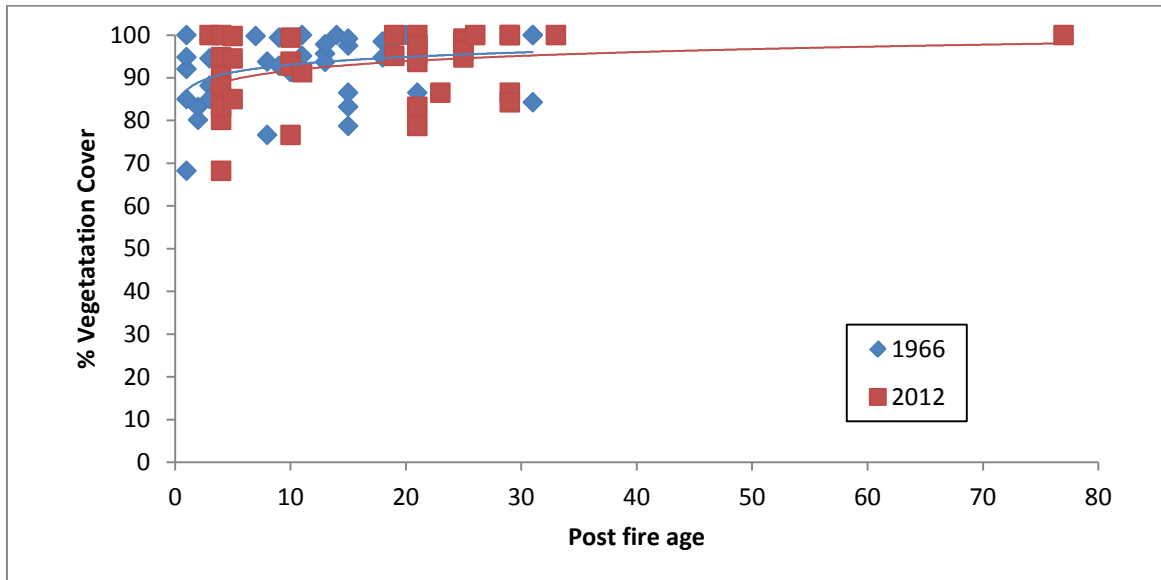
**Figure 4.14: Average percentage change (1966-2012) in vegetation cover for functional groups Proteoid, Ericoid and Restioid within difference in post fire age categories in 25 sites within Restioid Fynbos. One standard deviation of the data is represented by the bars.**



Figure 4.15: Teeberg South (No. 749) A: 1966 B: 2012. The foot slopes of Teeberg are now completely dominated by *Protea lepidocarpodendron*. There is a similar cover of restioids in the foreground between the two time periods but dwarf asteraceous shrubs have declined in abundance. The alien species in the early photograph as well as the *Leucospermum conocarpodendron* individuals have disappeared since 1966.

## *Proteoid Fynbos*

The relationship between post fire age and % vegetation cover for Proteoid Fynbos was not significantly different for sites in 1966 and 2012 ( $n=38$ ,  $F=1.11$ ,  $p=0.37$ )(Figure 4.16).



**Figure 4.16: The relationship between post fire age and % vegetation cover in Proteoid Fynbos sites in 1966 ( $R^2=0.12$ ,  $p>0.05$ ) and 2012 ( $R^2=0.109$ ,  $p>0.05$ ).**

The percentage change in vegetation cover varied across Proteoid fynbos sites (Figure 4.17). Sites that were younger in post-fire age in 2012 than in 1966 generally had lower cover than sites that were older in post-fire age when the original photograph was taken. Two sites (Duiwelsvlei and Point 282) stand out as being considerably lower in vegetation cover in 2012 than when they were originally photographed in 1966 (Figure 4.18). For many sites there was very little difference in vegetation cover between the two time periods such as at Die Boer. Sites that have shown a greater than 15% increase in cover include Above Circle Drive, Klein Blouberg, Bonteborg Nek and Kanonkop Distant.

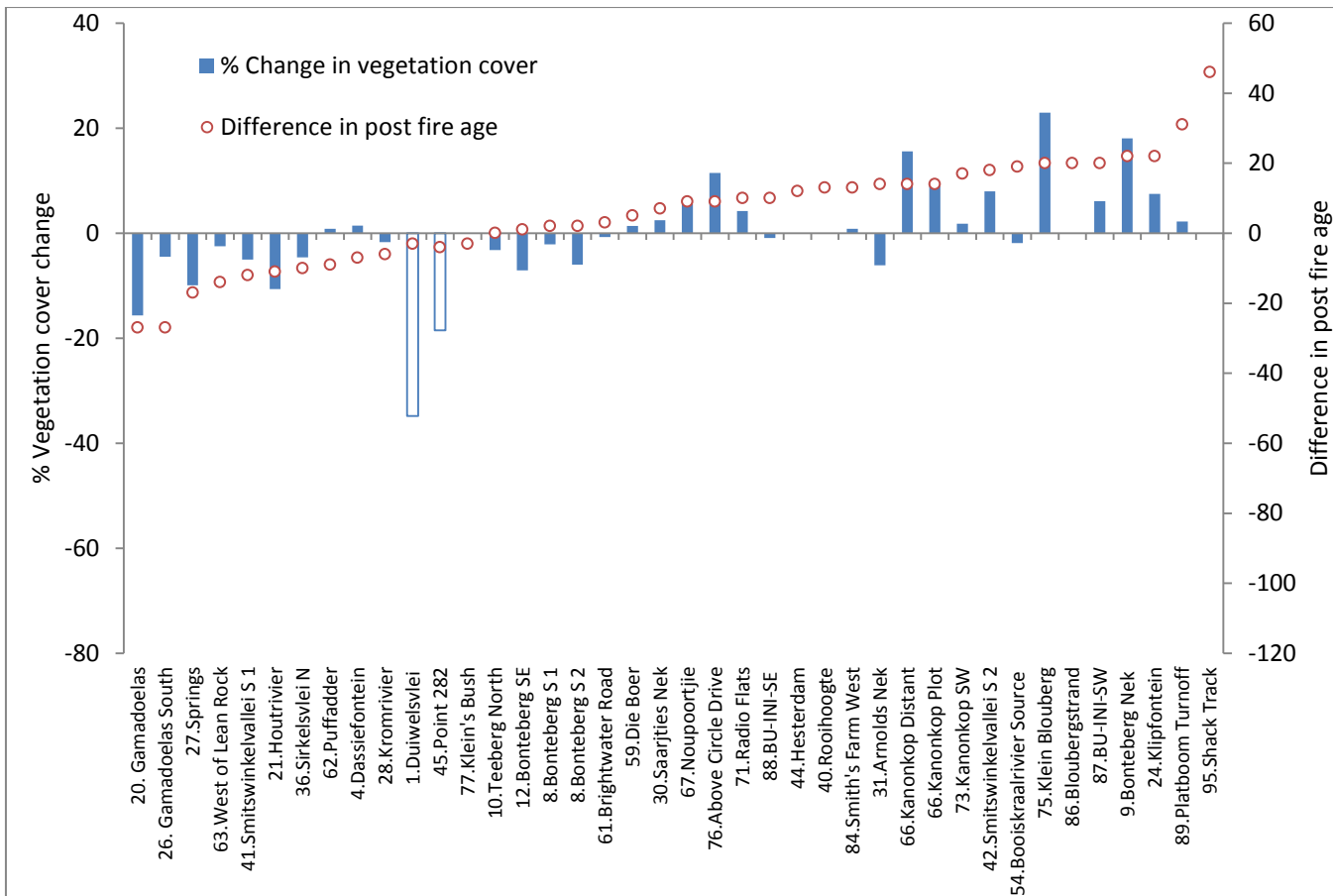


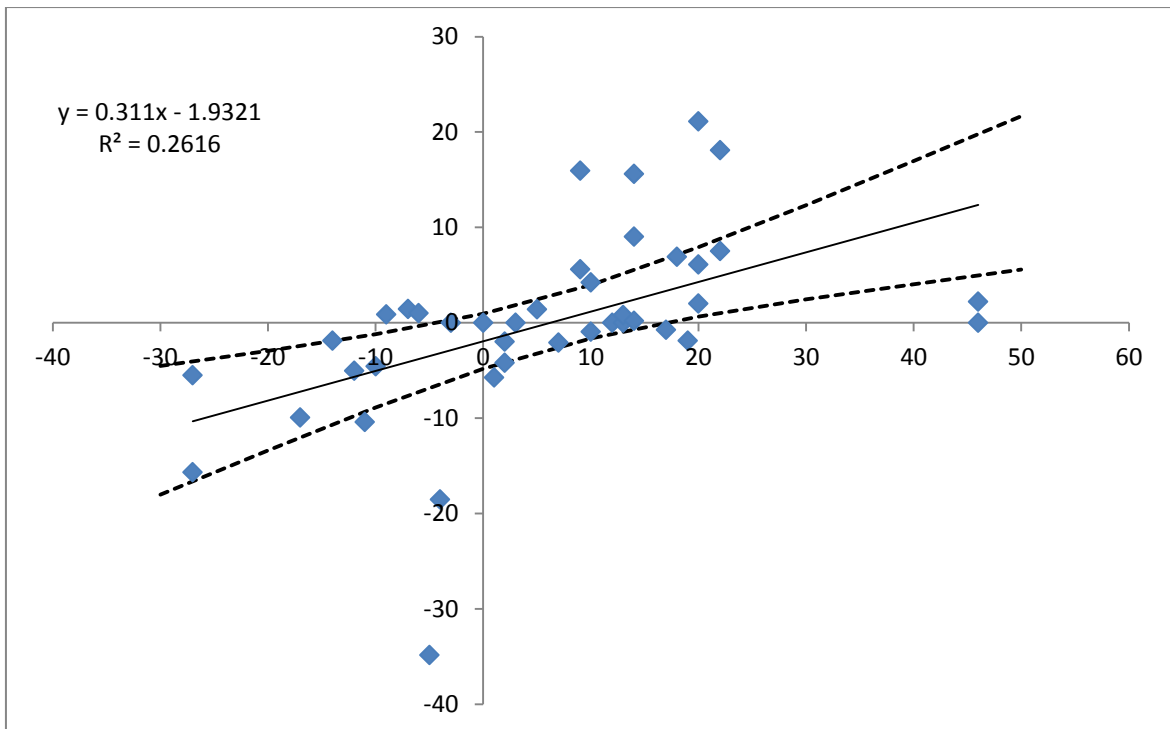
Figure 4.17: Percentage vegetation cover change (1966-2012) for 40 Proteoid Fynbos sites. Difference in post fire age is also displayed on the graph. Outliers in the data are represented by clear bars.



**Figure 4.18: Duiwelsvlei (No. 742). A: 1966 B:2012. The most obvious change is the reduction in vegetation cover and increased visibility of rocks, as a result of the recent fire which occurred in 2007. Post fire recovery has been slow and dead skeletons still appear in the landscape. The general physiognomy of the site however appears similar when comparing the two images, with a co-dominance of ericoids and restioids.**

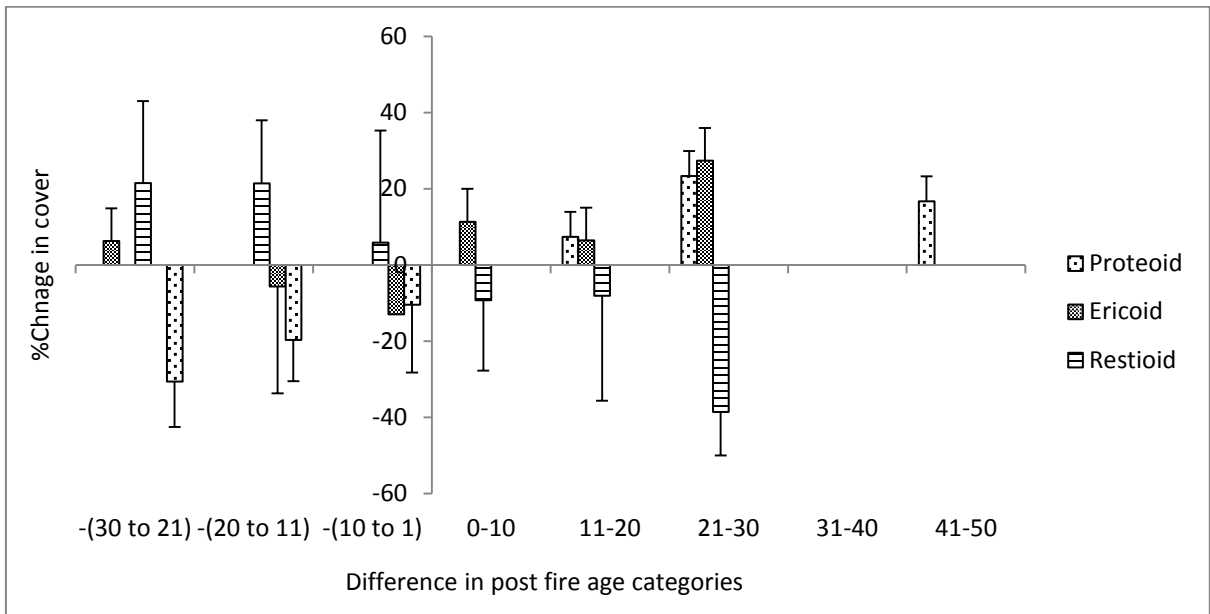
There was a strong relationship between difference in post fire age and percentage change in vegetation cover for the Proteoid Fynbos sites ( $n=40$ ,  $R^2=0.2616$   $p<0.001$ ) (Figure 4.19). The two sites that showed the greatest decline in vegetation cover relative to the difference in post fire age were Duiwelsvlei (Figure 4.18) and Point 282. Sites that showed a low vegetation cover for a high difference in post fire age were Shack Track and Platboom Turnoff. There were also five points that show a greater vegetation cover relative to the difference in post fire age.

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**Figure 4.19: Scatterplot showing the relationship between difference in post fire age and % change in vegetation cover for 40 sites within Proteoid Fynbos (1966-2012). The linear regression as well as the 95% confidence limits are also displayed.**

There is considerable variability within plant growth forms of the Proteoid Fynbos sites. A pattern of post fire succession was evident, in which restioids dominated within sites that were younger in post fire age in 2012 relative to 1966. However, as the difference in post fire age became more positive, restioids declined and ericoid and proteoid growth forms were more dominant in the landscape (Figure 4.20 and Figure 4.21).



**Figure 4.20: Average percentage change in vegetation cover (1966-2012) for restioid, ericoid and proteoid growth forms within difference in post fire age categories in 40 sites within Proteoid Fynbos. One standard deviation of the data is represented by the bars.**

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Figure 4.21: Klein Blouberg (No. 949) A: 1966 B: 2012. The major differences between the images can be attributed to the difference in post fire age which in this case is more than 30 years. The presence of low herbaceous cover and the high visibility of rocks in the 1966 photograph reflects the fact that the area had been burnt a year prior to when the original photograph was taken. In 2012 there was a dense fynbos stand, co-dominated by a range of eriocoid shrubs and emergent proteoids. While *Leucospermum conocarpodendron* and *Mimetes fimbriifolius* co-occur, the former appear to be more abundant, at least in the field of view.

## Markov Transition Matrices

The proportion of ericoids greatly increased in the Dune Asteraceous Fynbos sites between 1966 and 2012 (Figure 4.22). If current rates of increase were to continue over the next 46 years then projections derived from the Markov transition matrices suggest that the ericoid growth form will dominate Dune Asteraceous sites by 2058. Restioids currently contribute a relatively low proportion to vegetation cover in the landscape and at current rates of change could disappear completely from this vegetation type by 2058. Broad leaved shrubs also showed a rapid decline in cover from 1966 to 2012. If the same rate of change that occurred between 1966 and 2012 continues then this growth form will comprise less than 2% of the vegetation cover by 2058.

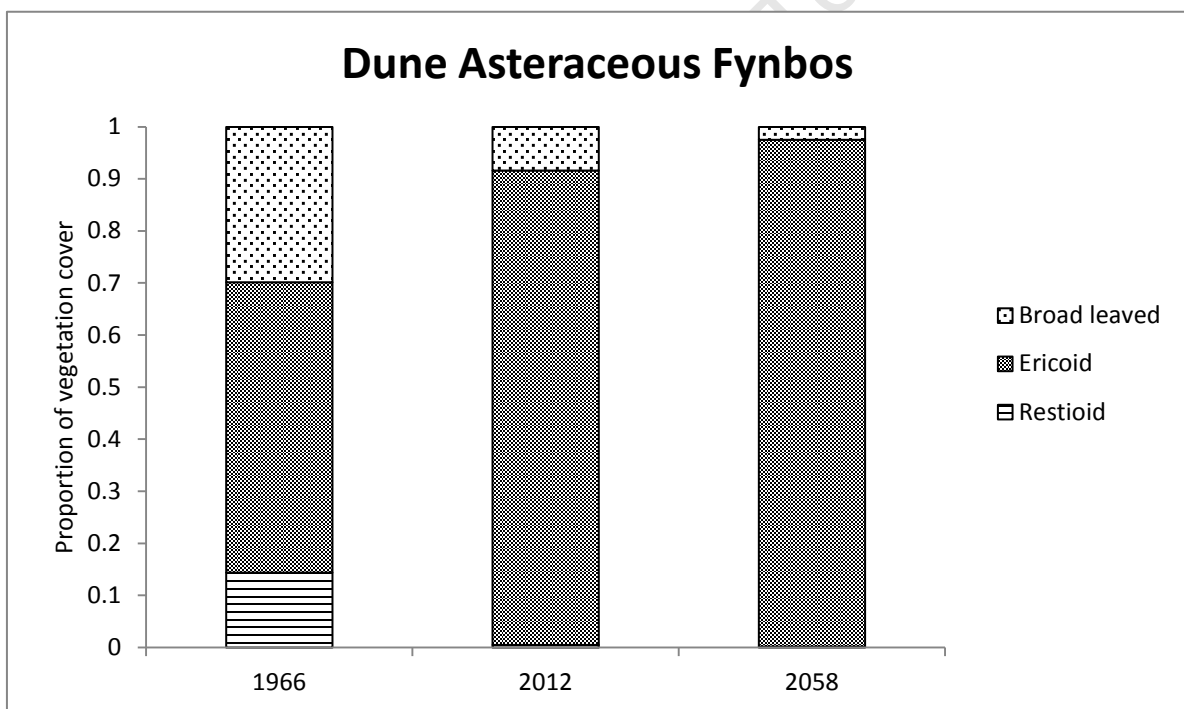
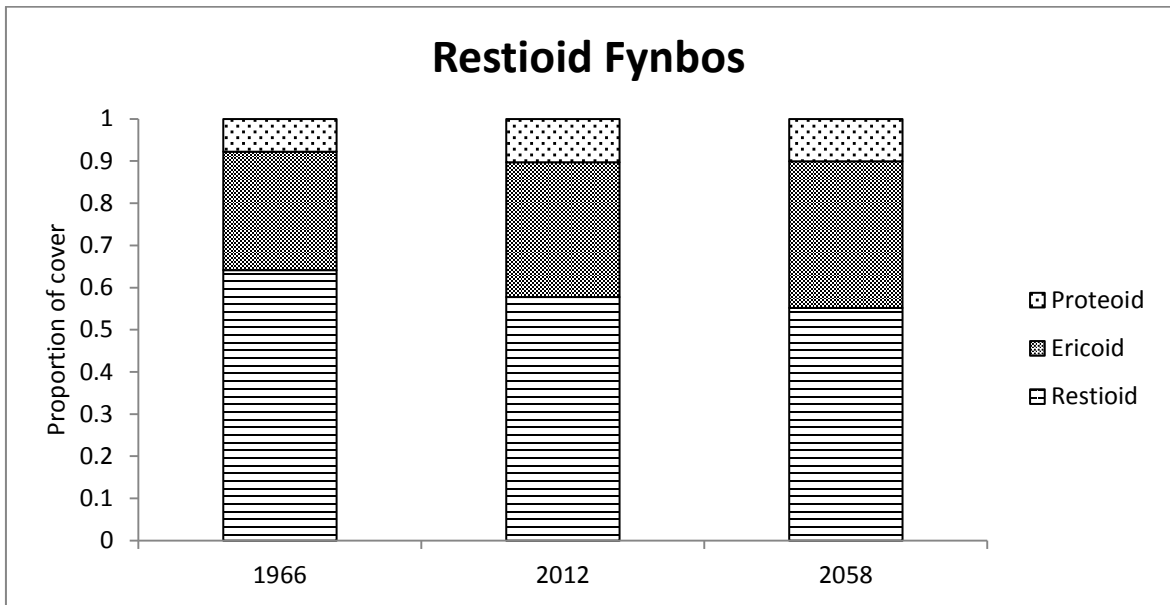


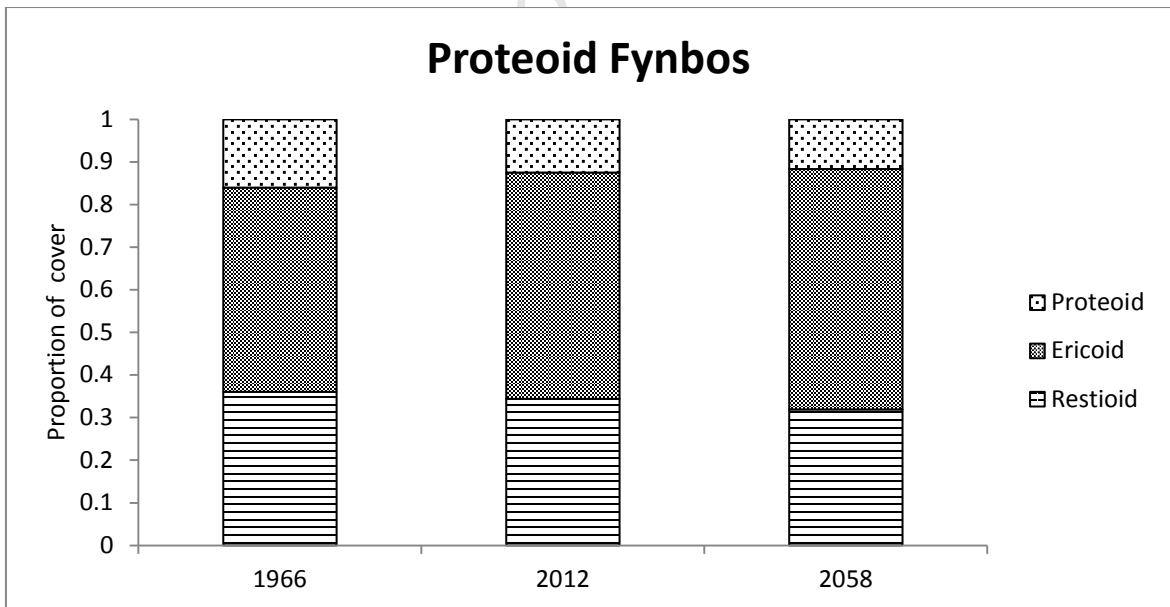
Figure 4.22: Markov transition matrix displaying projections (1966-2058) for changes in the dominant plant functional groups within Dune Asteraceous Fynbos sites.

The projections for growth forms within Restioid Fynbos sites showed very little change over the nearly 100 year time frame. If the rates of change which were evident between 1966 and 2012 in the different growth forms continue then restioids are likely to decrease, and ericoids and proteoids increase slightly in cover between 2012 and 2058 (Figure 4.23).



**Figure 4.23: Markov transition matrix displaying projections (1966-2058) for changes in dominant functional groups for Restioid Fynbos sites.**

If current trends continue, projected changes for Proteoid Fynbos sites indicated a slight increase in ericoids and a slight decrease in proteoid and restioid growth forms over the period from 2012-2058 (Figure 4.24).



**Figure 4.24: Markov transition matrix displaying projections (1966-2058) for changes in dominant functional groups for Proteoid Fynbos sites.**

## Discussion

### Historical changes in the Vegetation of the Reserve

#### *Changes in Total Vegetation Cover*

After obvious disturbance-affected outliers were removed from the analysis, the percentage change in total vegetation cover was significantly related to difference in post-fire age in all three vegetation types. There is no indication in the data that the increase in temperature measured at Cape Point over the 20<sup>th</sup> century has impacted on vegetation cover by slowing down recovery after fire, for example, through decreased moisture availability. If this were the case then mature fynbos stands (>10 years post-fire age) should have had less cover and more exposed rock relative to similar-aged stands in 1966. Successional processes should also have proceeded more slowly and vegetation of young fynbos stands (<10 years post-fire age) should also have taken longer to recover relative to those measured in 1966.

While these findings from the repeat photograph analysis support the general conclusion from plot-based studies of stability at the meta-community level (Thuiller et al. 2008) there were interesting and radical changes in cover and composition at several sites within all vegetation types. In some instances sites had significantly less cover in 2012 than in 1966 after the difference in post-fire age had been taken into account while in other instances sites had significantly more cover in 2012 when compared to 1966. Sites with more cover than predicted from the relationship between the difference in post-fire age and % vegetation change and which occurred above the 95% confidence limit line, generally showed an increase in biomass through the increase of large Proteaceae species such as *Mimetes fimbriifolius*, *Leucospermum conocarpodendron* and *Protea lepidocarpodendron*. The latter species appeared particularly 'mobile' in the landscape. At some sites it was observed to expand its distribution across a hill slope while at others (e.g. Teeberg South) it was absent in 1966 but completely dominated the site in 2012 and

appeared to have colonised the site from relatively distant source populations not immediately visible in the original photograph. The dynamic nature of this species could only have been determined from an analysis of historical photograph and would not have been apparent in the 10x5 m plots surveyed by Privett (1996) and Hall (2011) and analysed by Thuiller et al. (2008).

Some sites also showed a significant increase in cover over time apparently in response to the clearing of alien vegetation and the increase of moisture at the site. For example, at Mimicki's the removal of alien species has resulted in a community shift from restioid- and alien-dominated vegetation to ericoid- (especially *Berzelia abrotanoides*) and proteoid-dominated vegetation. This finding may be attributed to an increase in moisture availability due to the removal of alien species. In general, however, seepage sites within all vegetation types either showed very little change in total vegetation cover or showed an increase in cover and biomass (particularly of *Berzelia abrotanoides*) over time. This observation further rejects the hypothesis that the warming trajectory, apparent in the region, has impacted on the vegetation.

In some instances sites had significantly less cover in 2012 than in 1966 after the difference in post-fire age had been taken into account. No single driver could account for these reductions in cover which appeared to be site-specific and local in nature. However, the most important drivers responsible for a significant reduction in vegetation cover fell into one of four categories: (1) an increase in fire frequency, (2) an increase in dune mobility, (3) an increase in biotic activity, and (4) the physical removal of alien vegetation by park personnel.

The impact of an increase in fire frequency on vegetation cover was particularly evident at two sites, Duiwelsvlei and Point 282, within Proteoid Fynbos. Duiwelsvlei is situated close to the settlement of Scarborough on the northern border of the reserve and has experienced 3 fires in the last 20 years. Although Point 282 is situated centrally in the reserve it has experienced two fires within a 10 year period. The frequent incidence of fire is the best explanation for the reduction in vegetation cover at these two sites and

has been highlighted by Forsyth and van Wilgen (2008) as a major concern for the diversity and composition of Fynbos vegetation on the Cape Peninsula.

Of all the sites examined in this study, the loss of vegetation cover over time is greatest at Potbank where there has been an 80% reduction in cover since 1966. While the history of this site has not been examined in detail the dune system in the region has been re-mobilised either by an increase in wind strength and intensity (perhaps in association with fire) or a change in wind direction. Such changes at this site have meant that the vegetation has been uprooted and blown away leaving an active and mobile dune system in the region (Campbell, 1929; Jungerius and van der Meulen, 1997). This phenomenon has been observed at other sites along the western coast of the reserve such as at plot 20 (Gamadoelas) and at plot 43 (Thomas. T. Tucker). The changes in the landscape at the latter site have been so great that it was not possible to re-locate the original photograph.

The impact of biotic disturbances on the reduction of vegetation cover was greatest at Klien Rondevlei where dune mole rats (*Bathyergus suillus*) have colonised the site after the 2008 fire. The disturbance from soil turnover and the uprooting of plants and consumption of geophytes (Mokotjomela et al., 2009) has led to a significant loss in vegetation cover over time. The long-term impact of this disturbance is unknown but is likely to diminish as vegetation cover increases with post-fire age. Other biotic disturbances include the impact of grazing ungulates, particularly Bontebok (*Damaliscus pygargus*), although these were not identified as being significant in this study.

The site known as Berzelia Seepage has also shown a significant decline of about 15% in vegetation cover when compared to the original photograph. This can be explained by the removal of alien plants (*Pinus pinaster* and *Acacia saligna*) from the site since 1966. The original photograph showed that the site was dominated by alien trees which have been manually removed and eradicated from the site. This removal of these species has resulted in removal of biomass and thus a decline in vegetation cover as determined

from a comparison of matched photograph pairs. This decline is however not a negative impact, but rather a positive result.

### ***Changes in the Cover of Dominant Growth Forms in Relation to Post Fire Age***

Changes in the four main growth forms (broad leaved shrubs, proteoids, ericoids and restioids) in relation to post-fire age show different responses in the three different vegetation types. Although sample size was too small (n=12) in Dune Asteraceous Fynbos to detect clear patterns, both restioids and broad leaved shrubs appear to have declined with increasing difference in post-fire age while ericoids increased in cover in this vegetation type. There was considerable variation between sites, however, with a few on the eastern side of the reserve (e.g. Buffels Bay) showing an increase in cover of broad leaved shrubs such as *Sideroxylon inerme*.

In both Restioid and Proteoid Fynbos changes in the dominant growth forms followed 'classic' post-fire succession trajectories with restioids declining and ericoids and proteoids increasing in cover with time since fire (Hoffman et al., 1987; Rutherford et al., 2011; van Wilgen, 2012; Keeley et al., 2012). While there was considerable variation between sites, the pattern of growth form change with post-fire age was particularly clear in Proteoid Fynbos. This suggests that long-term studies of vegetation and growth form change in Fynbos vegetation need to be mindful of the difference in post-fire age when comparing the response in growth forms and species over time.

### **Future Trajectories in the Vegetation of the Reserve Derived from Markov Transition Matrices**

Markov transition matrices provide a useful way to understand vegetation dynamics and how vegetation composition might change over time (McAuliffe, 1988). However, they have significant limitations in environmental management (Pontius et al., 2004) primarily because they are sensitive to changes in the disturbance regime, which in fynbos vegetation is primarily defined by the frequency, seasonality and intensity of fire (Bond

and van Wilgen, 1996). Despite these limitations predictions from transition matrices are useful in determining broad changes over time in rank dominance of dominant growth forms (Balzter, 2000). Conservation managers in particular need some broad indication of historical and potential future trajectories in major growth forms so as to inform their monitoring and intervention activities.

Major trends in the dominant growth forms of the reserve over time are most noticeable in the Dune Asteraceous Fynbos. Results from the Markov transition matrices suggest that restioids and broad leaved shrubs have decreased in cover and ericoids have increased substantially in cover since 1966. Although the results of the transition matrices should be interpreted with caution, future projections for the next approximately 50 years indicate that if current trends continue then both restioids and broad leaved shrubs will be reduced to minor components of the floras of this vegetation type and ericoids will dominate Dune Asteraceous Fynbos vegetation. A change in the disturbance regime, however, could have significant implications for the future trajectories of this vegetation type.

While restioids appear not to have been a large component of this vegetation type in 1966, the loss of broad leaved shrubs within Dune Asteraceous Fynbos sites is a cause for concern. These species (e.g. *Sideroxylon inerme*, *Maytenus oleoides*) are rarely impacted by fire but are sensitive to human disturbance such as that associated with the development of parking areas and footpaths. These findings reflect those of Rickard et al., (1994), who found that pedestrian and vehicle traffic greatly reduced vegetation height and cover of dune vegetation. The degree of protection given to sensitive groves or clumps of broad leaved shrubs (such as at Buffels Bay where the cover of broad leaved shrubs has increased) can have important impacts on their response over time. Further investigation is needed to assess the full impact of human disturbance on broad leaved shrubs in the Cape of Good Hope reserve and should be an urgent priority for reserve managers. However, in some instances, such as at Potbank, the impact of large-scale abiotic disturbances cannot be anticipated and are difficult to manage.

In contrast to Dune Asteraceous Fynbos, changes in the cover of dominant growth forms in both Restioid and Proteoid Fynbos were far less pronounced over the study period. If current trends continue and disturbance regimes remain broadly the same, then the cover of the dominant growth forms within these two vegetation types is also not projected to change substantially over the next 50 years. The cover of restioids will decrease slightly while the cover of ericoids will increase slightly in both vegetation types. Proteoids will not change substantially in either of the two vegetation types and will continue to occupy about 10% of the total vegetation cover in the reserve.

## **Is There Evidence of Impacts of “Aridification” on the Vegetation of the Reserve?**

Climate change projections for southern Africa suggest that much of the Fynbos biome will become more arid during the course of the 20th century with increasing temperatures and decreasing rainfall anticipated (Midgley et al., 2005). Temperatures are expected to increase between 1.5 and 3.5 °C by 2050 (Midgley et al., 2005). Annual rainfall is expected to decrease by up to 41 mm per annum which is 20% lower than current levels (Haensler et al., 2010). Fire frequency is also expected to increase in response to increased temperature (Forsyth and van Wilgen, 2008). This in turn is predicted to lead to a concomitant loss in vegetation cover and a shift in species and growth form dominance. Although this catastrophic narrative of a ‘shrinking’ biome has recently been revised (Driver et al. 2012), it continues to influence research and policy initiatives in the region. Furthermore, in Chapter III the historical data showed a clear warming trend, with decreased humidity. Thus, although no significant trend in rainfall was detected, some aridification effects (due to increases in temperature) might be expected because of increased water stress on plants. This in turn might be expected to slow recovery after fire, but the results of the current chapter show no such results and all vegetation types appear resilient to the degree of climate change experienced thus far. This is particularly interesting since a more detailed analysis of the rainfall record indicated that the 10 years before the 1966 survey were significantly ( $n=10$ ,  $z=1.64$ ,  $p=0.005$ ) wetter by 25% than the 10 years before the 2011/12 survey. The average

(±stdev) annual rainfall amount for the period 1955-1965 was 475.4(±125.3) mm while for the ten years prior to the 2011/12 survey the value was 356.0(±76.3) mm. Rainfall also declined before both periods although there was no significant difference in the rate of decline ( $n=10$ ,  $F=2.693$ ,  $p>0.05$ ).

The Markov chains project relatively minor changes in the major growth forms of the reserve although the impact of land use on Dune Asteraceous Fynbos might be significant. These findings indicate that fynbos vegetation is resilient to the changes in climate experienced thus far across the reserve, although it is probably unwise to assume that vegetation will respond linearly to climate change (Pearson et al., 2002). It is more likely that a threshold in response to warming might occur with significant, non-linear declines in vegetation and growth form cover emerging over time.

# Chapter V: Change in the Populations of Reseeding Proteaceae Species between 1966 and 2012

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## Introduction

The Cape Floristic Region (CFR) is a global biodiversity hotspot and one of six floristic kingdoms in the world (Cowling et al., 1997; Myers et al., 2000). Anthropogenic climate change is a major threat to this region, and will likely cause species range shifts, as well as population declines and extinctions (Hughes, 2000; McCarthy, 2001; Parmesan and Yohe, 2003; Root et al., 2003). A study conducted by Bomhard et al., (2005), showed that up to 30% of Proteaceae species would be uplisted to a more threatened IUCN status. The South Western Cape was more threatened than other areas, with up to a 92.5% of species becoming threatened or extinct with future changes in climate. However, even under less severe conditions, range shifts and losses of Proteaceae could change considerably, leading to a greater sensitivity of already imminent threats such as habitat transformation and alien species (Opdam and Wascher, 2004; Hannah et al., 2005). Low altitude regions along the coast already host many threatened species and therefore these areas should be prioritised for conservation (Rebelo, 1992; Richardson et al. 1996).

This study focuses on large Proteaceae species, specifically *Mimetes fimbriifolius*, *Leucospermum conocarpodendron*, *Leucadendron laureolum*, *L. xanthoconus* and *L. salignum* as these species were identified by the Cape of Good Hope management staff as being potentially threatened by climate change. In addition, these species are the most common large Proteaceae in the conservation area and stand out in the landscape. They thus add to the aesthetic experience of visitors and there is a desire from the side of management that they are maintained in the landscape.

*M. fimbriifolius* and *L. conocarpodendron* co-occur throughout the landscape in the reserve and are difficult to distinguish from one another from a distance (Midgley et al., 1998). Both species possess thick bark and are able to withstand and survive fire once individuals reach a certain age (Midgley et al., 1998). This is a very rare trait in fynbos vegetation, with only a few species out of the thousands in the region able to do (Midgley et al., 1998). Unlike many other large Proteaceae, the seeds of these individuals are not preyed on by ants or rodents and are wind-dispersed (Brits et al., 1993). This gives them an advantage as they are able to establish across the landscape. Although not preyed on by small fauna, in summer when flowers are in bloom, baboons (*Papio ursinus*) eat the heads of the *Leucospermum conocarpodendron* flowers.

*Leucadendron laureolum*, *L. xanthoconus* and *L. salignum*, however, are highly dependent on fire for regeneration as they are reseederers (Bond et al., 1984). These species have cones that are filled with seed and after a fire has occurred, the cones open up and release seeds (Bond et al., 1984). Fire frequency is thus a very important factor as the timing of this disturbance is critical in the survival of the population. For instance, if a fire occurs before the population is reproductively mature, and before seed reserves have been produced, the species could become locally extinct. This situation would intensify in regions where fire frequency is too high (Bond et al., 1984).

Cabral and Schurr, (2009), suggested that basic demographic studies are useful for understanding range dynamics. The current study used repeat photography to document the changes that have occurred since 1966 within separate populations of these emergent Proteaceae species. The specific objectives of this study were:

- i. To assess the extent and direction of change (i.e. increase, decrease or no change) since 1966 in the number of individuals of *Mimetes fimbriifolius* and *Leucospermum conocarpodendron* in relation to differences in post fire age between the two sampling periods.
- ii. To assess the extent and direction of change (i.e. increase, decrease or no change) since 1966 in the cover of *Leucadendron laureolum*, *L. xanthoconus* and *L.*

- salignum* (hereafter referred to as *Leucadendron* spp.) in relation to differences in post fire age.
- iii. To investigate the impact of fire frequency on the number of individuals of *Mimetes fimbriifolius* and *Leucospermum conocarpodendron* and the cover of *Leucadendron* spp.
  - iv. To test the impact of the total number of fires between sampling periods on *Mimetes fimbriifolius* and *Leucospermum conocarpdendron* individuals as well as on *Leucadendron* spp. cover.

## Methods

This study made use of landscape photographs to obtain a broad understanding of landscape-level dynamics of large reseeding Proteaceae species. The same methodology as in Chapter III was used to take the repeat photographs and process them for analysis. Because of the high resolution at which the historical images were scanned or re-photographed, adult individuals were identified in the landscape in both the original photograph and the repeat photograph by zooming in on photographs and counting individuals. The emergent Proteaceae were separated into two categories as they grow in different forms. *Mimetes fimbriifolius* and *Leucospermum conocarpodendron* grow as separately-spaced individuals while the different *Leucadendron* spp. tend to grow close to each other in clumps and are not easily separated in a photographic image. Furthermore, because *M. fimbriifolius* and *L. conocarpodendron* often co-occur in the landscape and are difficult to distinguish apart in the photographs they were grouped together and counted as a single entity hereafter referred to as *Mimetes/Leucospermum*. A total of 27 photographs were used in this analysis of *Mimetes/Leucospermum* population change while 23 photographs were used in the analysis of the change in cover of *Leucadendron* spp.

The photographs were analysed in powerpoint, using a slideshow of the matched images. Individuals were identified and marked with yellow dots in the original photograph. These dots were then selected, copied and pasted onto the repeat photograph. If the individuals were still present, the dot remained yellow. If the individual had died, the dot

colour was changed to red. Green dots were added to identify new individuals in the repeat photograph. The number of individuals were recorded and analysed using the formulas below (Figure 5.1).



Figure 5.1: Example of identification and recording of changes in number of individuals of *Mimetes/Leucospermum* between 1966 (top) and 2012 (bottom). Yellow dots in the top image denote all living *Mimetes/Leucospermum* individuals identified from the photograph in 1966. Red dots in the bottom photograph indicate those individuals that have died since 1966 while green dots indicate new individuals. Three individuals (indicated with a yellow dot) in the bottom image appears to be the same individual as in 1966.

## ***Analysis of change in *Mimetes fimbriifolius* and *Leucospermum conocarpodendron****

Individuals of *Mimetes/Leucospermum* were counted in each image and the percent mortality, percent recruitment and percent population change was calculated as follows:

Mortality = Number of individuals that were present in 1966 that are no longer present in 2012 divided by the total number of individuals in 1966.

Recruitment = Number of individuals present in 2012 that were not present in 1966 divided by the total number of individuals in 2012.

Population change = Number of live individuals in 2012 divided by the number of live individuals in 1966.

For the 'Population change' variable a value greater than 1 reflects population growth over the 46 years between the two survey periods while a value less than 1 indicates a decline in the population. To provide an indication of how each population of large reseedling Proteaceae assessed in the photographs had changed between 1965 and 2012 a value of 1 (i.e. no change) was subtracted from the population change value and presented graphically in relation to difference in post fire age.

## ***Analysis of change in *Leucadendron* spp.***

To analyse population change in *Leucadendron* spp., a grid of 260 points was laid across each photograph and points where the species intersected were recorded. The proportion of *Leucadendron* spp. was calculated in relation to the photograph grid using the following formula:

$$\% \text{ Cover change} = \left[ \frac{\text{No. of grid points individuals intersect on repeat}}{\text{Total number of grid points on repeat}} - \frac{\text{No. of grid points individuals intersect on original}}{\text{Total number of grid points on original}} \right] \times 100$$

## ***Effects of Fire Frequency***

The effect of fire frequency was analysed for both *Mimetes/Leucospermum* and *Leucadendron* spp. The total number of fires between sampling periods (1966-2010) was calculated from the fire records in Privett (1996) and from those held by the Cape Research Centre (Chapter II, Fire Section). The average change in % population change (*Mimetes/Leucospermum*) and % Cover change (*Leucadendron* spp.) was plotted against the number of fires at the site, i.e. 1 – 3 fires during the period 1966-2010.

## **Results**

### ***Changes in Mimetes/Leucospermum***

Table 5.1 provides a summary of the changes at the different sites where *Mimetes/Leucospermum* individuals were recorded as well as the post fire age for the sites.

In 1966 post fire age of the vegetation ranged from 1 to 31 years while in 2012 it ranged from 3 to 77 years. There was considerable variation between sites and because of this there was no significant difference between the two sampling periods in post fire age of the vegetation ( $n=27$ ,  $t=-1.20$ ,  $p>0.05$ ).

In 1966 the number of individuals of *Mimetes/Leucospermum* ranged from 0 to 32 and was similar to the values for 2012 when the number of individuals ranged from 0 to 29. There was no significant difference between number of *Mimetes/Leucospermum* individuals in 1966 and 2012 ( $n=27$ ,  $t=-0.58$ ,  $p>0.05$ ). In 1966 there were two sites which did not contain any *Mimetes/Leucospermum* individuals but which had recruits of these species in 2012. Conversely, three sites in 2012 did not have *Mimetes/Leucospermum* individuals despite their presence in 1966.

Values for both mortality and recruitment were highly variable between sites and ranged from zero to one in both cases. Average ( $\pm$ stdev) values were very similar for both variables ( $0.56\pm 0.41$  and  $0.58\pm 0.37$  respectively). Fifteen of the 27 sites had a population change value greater than one, while the average value for all populations was also greater than one ( $2.02\pm 2.94$ ).

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**Table 5.1: Fire characteristics at 27 sites in the reserve and the number of individuals of *Mimetes/Leucospermum* recorded at the sites between 1966 and 2012.**

Site	Post fire age (1966)	Post fire age (2012)	Difference in post fire age	No. of living individuals (1966)	No. of living individuals (2012)	Mortality	Recruitment	Population change
Bonteberg S 1	2	4	2	11	6	0.73	0.50	0.55
Bonteberg S 2	2	4	2	32	9	0.88	0.67	0.28
Bonteberg Nek	3	25	22	9	13	0.00	0.31	1.44
Teeberg North	3	3	0	6	3	0.67	0.33	0.50
Kromrivier Sandpit	3	4	1	5	1	1.00	1.00	0.20
Mimicki's	2	25	22	0	8	0.00	1.00	8.00
Gamadoelas	31	4	-27	2	5	0.50	0.80	2.50
Houtrivier	15	4	-11	6	0	1.00	0.00	0.00
Gamadoelas S	31	4	-27	10	29	0.30	0.76	2.90
Springs	21	4	-17	5	15	0.00	0.67	3.00
Kromrivier	10	4	-6	2	4	1.00	1.00	2.00
Arnolds Nek	11	25	14	4	6	1.00	0.33	1.50
Smitswinkelvallei S 1	31	19	-12	0	2	0.00	1.00	2.00
Smitswinkelvallei S 2	1	19	18	9	10	0.00	0.10	1.11
Kromrivier South	26	5	-21	3	1	1.00	1.00	0.33
Hesterdam	9	21	12	4	19	0.50	0.89	4.75
Point 282	9	5	-4	1	0	1.00	0.00	0.00
Booiskraal Source	14	33	19	1	2	1.00	1.00	2.00
Puffadder	19	10	-9	6	6	0.83	0.83	1.00
Above Circle Drive	1	10	9	2	1	0.50	0.00	0.50
Klein Blouberg	1	21	20	24	22	0.58	0.50	0.92
Gifkommetjie	31	77	46	19	9	0.58	0.11	0.47
Gifkommetjie Groend	1	21	20	7	5	0.86	0.80	0.71
Suurdam	8	21	13	8	16	0.13	0.56	2.00
Smitswinkel Farm West	8	21	13	0	14	0.00	1.00	14.00
BU-INI SW	1	21	20	4	8	0.00	0.50	2.00
Green Hill	24	21	-3	2	0	1.00	0.00	0.00
<b>Average</b>	<b>11.8</b>	<b>16.1</b>	<b>4.3</b>	<b>6.7</b>	<b>7.9</b>	<b>0.56</b>	<b>0.58</b>	<b>2.02</b>
<b>Standard deviation</b>	<b>11.0</b>	<b>15.3</b>	<b>17.4</b>	<b>7.5</b>	<b>7.4</b>	<b>0.41</b>	<b>0.37</b>	<b>2.94</b>

When one (i.e. no change) was subtracted from the population change value and graphed in relation to the difference in post fire age (Figure 5.2) there was no significant relationship between the two variables ( $n=27$ ,  $y=0.7515x+3.5261$ ,  $R^2=0.0162$ ,  $p>0.05$ ). Instead, changes appeared site-specific with some sites showing an increase in the number of individuals (Figure 5.3) and other sites showing a decrease (Figure 5.4), irrespective of the post fire age of the surrounding vegetation.

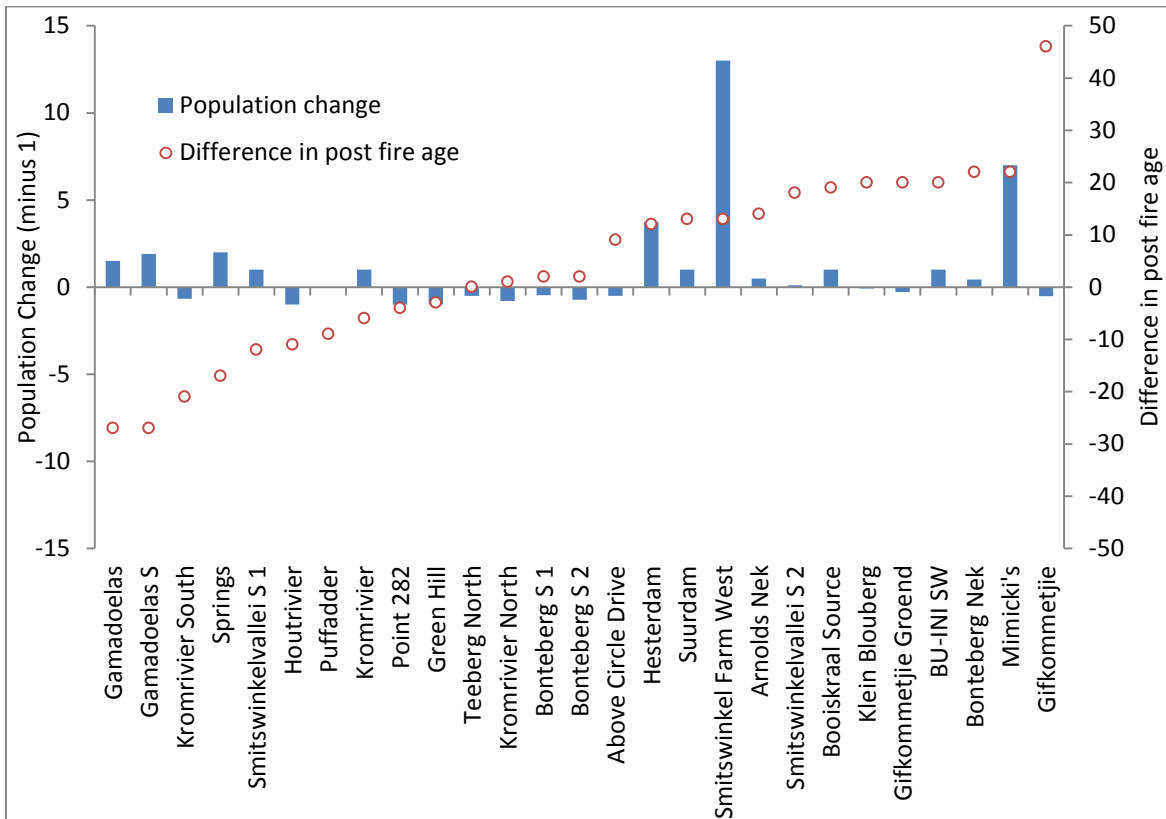


Figure 5.2: Population Change (minus 1) of *Mimetes/Leucospermum* in relation to difference in post fire age.



Figure 5.3: Hesterdam (No. 923). There has been an increase in the number of *Mimetes fimbriifolius* individuals at this site. The foreground and ridge on the far right has shown the greatest increase with a number of new individuals establishing since 1966.



Figure 5.1: Bonteberg S 2 (No. 745). There has been a significant decline in the population of *Mimetes fimbriifolius* associated with the rocky outcrop in the foreground, although they are present at the site. The presence of the dead *M. fimbriifolius* individual on the left-hand side of the frame also shows the decline in the number of individuals.

### ***Leucadendron spp. Changes***

Changes in the cover of *Leucadendron* spp. were measured at 23 sites (Table 5.2). There was a large difference in post fire age between sites which ranged from -27 to 46 years. Eleven sites showed an increase and 12 sites a decrease. There was no significant difference in the % cover of *Leucadendron* spp. in 1966 and 2012 ( $n=23$ ,  $t=0.17$ ,  $p>0.05$ ) (Table 5.2). At nine locations *Leucadendron* spp. was not visible in 1966 but was present in 2012 while at four sites no adults plants were visible in the 2012 photographs despite their presence in 1966.

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**Table 5.2: Difference in post fire age and % Population Change for *Leucadendron spp* for each of 23 sites in the reserve.**

Site	Post fire age (1966)	Post fire age (2012)	Difference in post fire age	% Cover (1966)	% Cover (2012)	% Cover Change (2012-1966)
Gamadoelas	31	4	-27	0.0	1.9	1.9
Gamadoelas S	31	4	-27	7.3	2.3	-5.0
New Road	21	4	-17	16.2	0.0	-16.2
Springs	21	4	-17	2.7	0.0	-2.7
West of Lean Rock	19	5	-14	7.7	3.5	-4.2
Berzelia	18	5	-13	15.0	0.8	-14.2
Potpourri	18	5	-13	1.9	0.4	-1.5
Houtrivier	15	4	-11	10.4	0.0	-10.4
Sirkelsvlei S	14	5	-9	22.7	0.4	-22.3
Smitswinkel Central	14	5	-9	11.5	0.0	-11.5
Point 282	14	5	-4	14.2	0.8	-13.5
Seepage Step	8	5	-3	0.0	2.7	2.7
Brightwater Road	7	10	3	0.0	2.7	2.7
Anvil Rock West	25	29	4	1.9	12.3	10.4
Die Boer	15	20	5	9.6	1.2	-8.5
Meadows	15	20	5	5.8	4.2	-1.5
Opposite Smiths	20	29	9	0.0	5.8	5.8
Radio Flats	13	23	10	0.0	9.2	9.2
Arnolds Nek	11	14	14	0.0	1.5	1.5
Kanonkop SW	12	29	17	0.0	12.3	12.3
Bonteberg Nek	3	25	22	0.0	27.2	27.2
Klipfontein	3	25	22	0.0	2.3	2.3
Platboom Turnoff	31	77	46	1.5	20.4	18.8
<b>Average</b>	16.5	15.5	-0.3	5.6	4.9	-0.7
<b>Standard Deviation</b>	7.9	16.6	17.5	6.7	7.1	11.7

There was a relatively even spread (Figure 5.4) of sites that showed an increase (Figure 5.5) and sites that showed a decrease (Figure 5.6) in cover of *Leucadendron* spp. between 1966 and 2012. There also appeared to be a good relationship between % Cover change and different in post fire age of the sites.

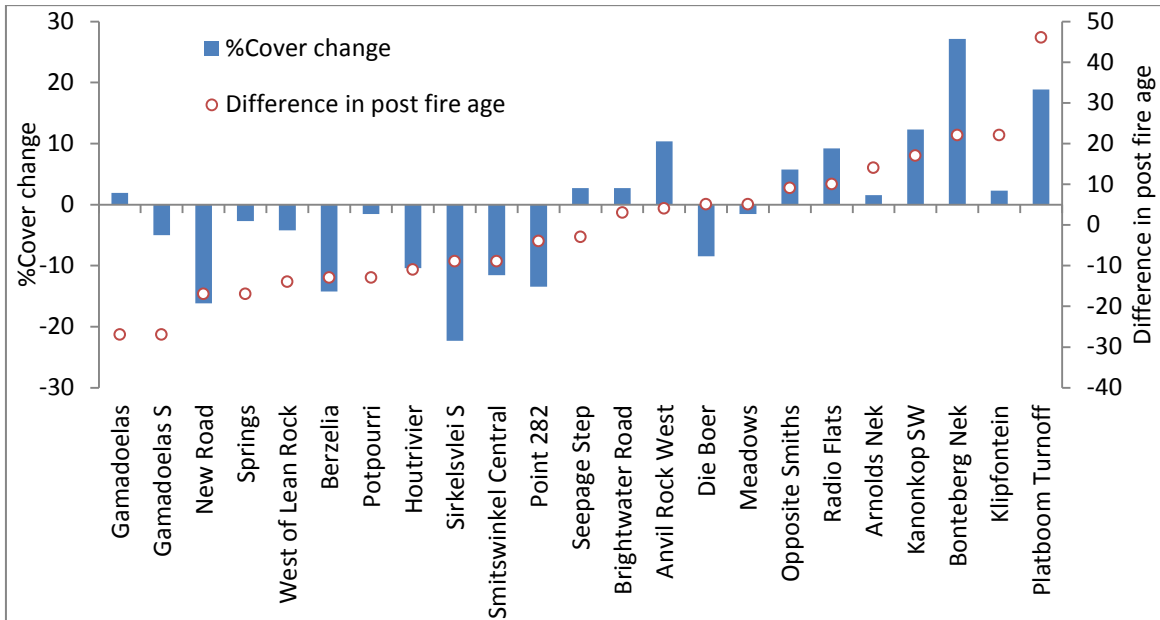


Figure 5.4: Histogram showing % Population change of *Leucadendron* spp in relation to difference in post fire age

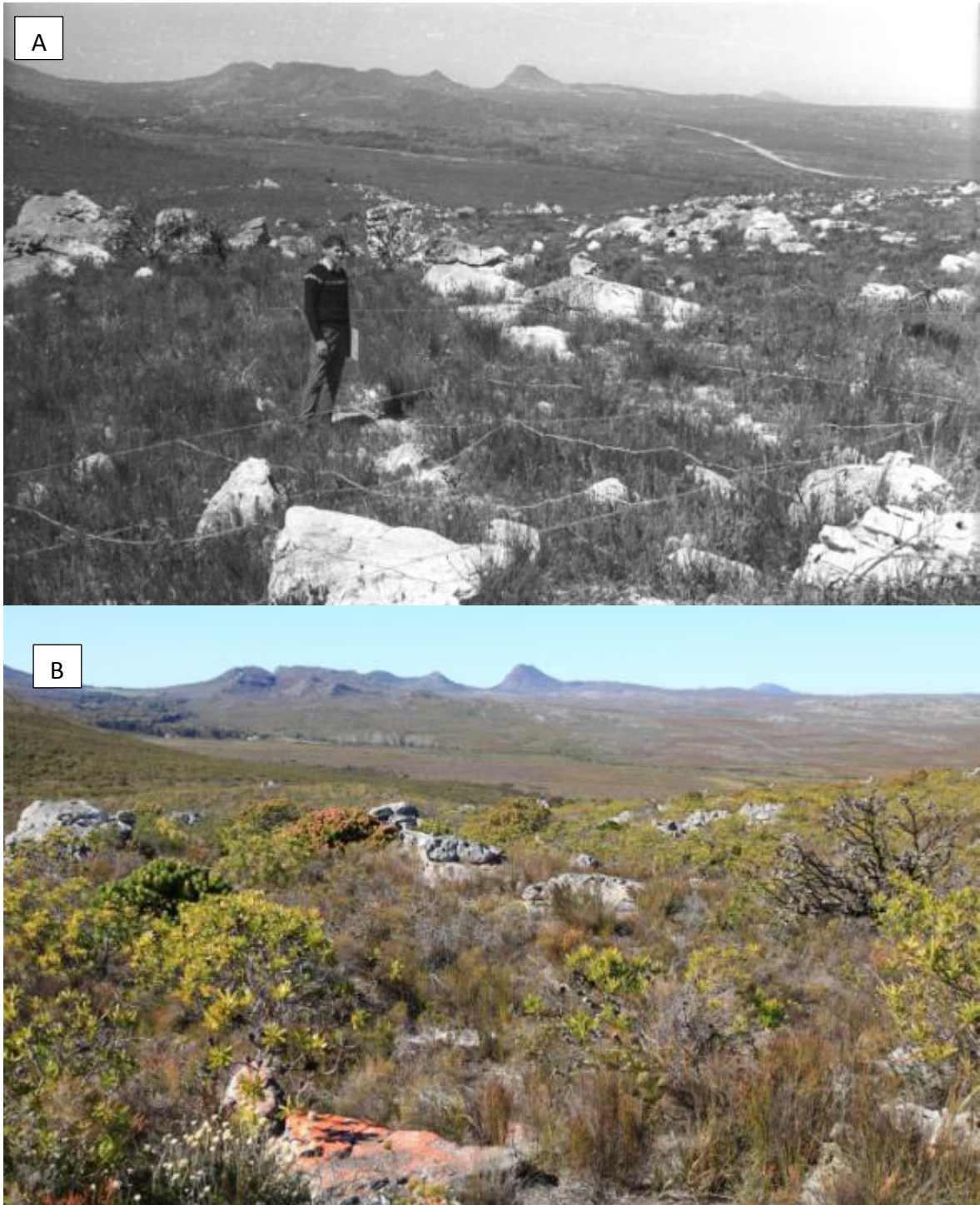


Figure 5.5: Bonteberg Nek (No. 746). In 1966, the immediate foreground was dominated by Restionaceae, Ericaceae and *Serruria villosa* (Proteaceae). A number of asteraceous shrubs were also dominant in 1966. In 2011, the immediate foreground in dominated by emergent Proteaceae shrubs *Leucadendron laureolum*, *Mimetes fimbriifolius* and the slightly smaller *Serruria villosa*.

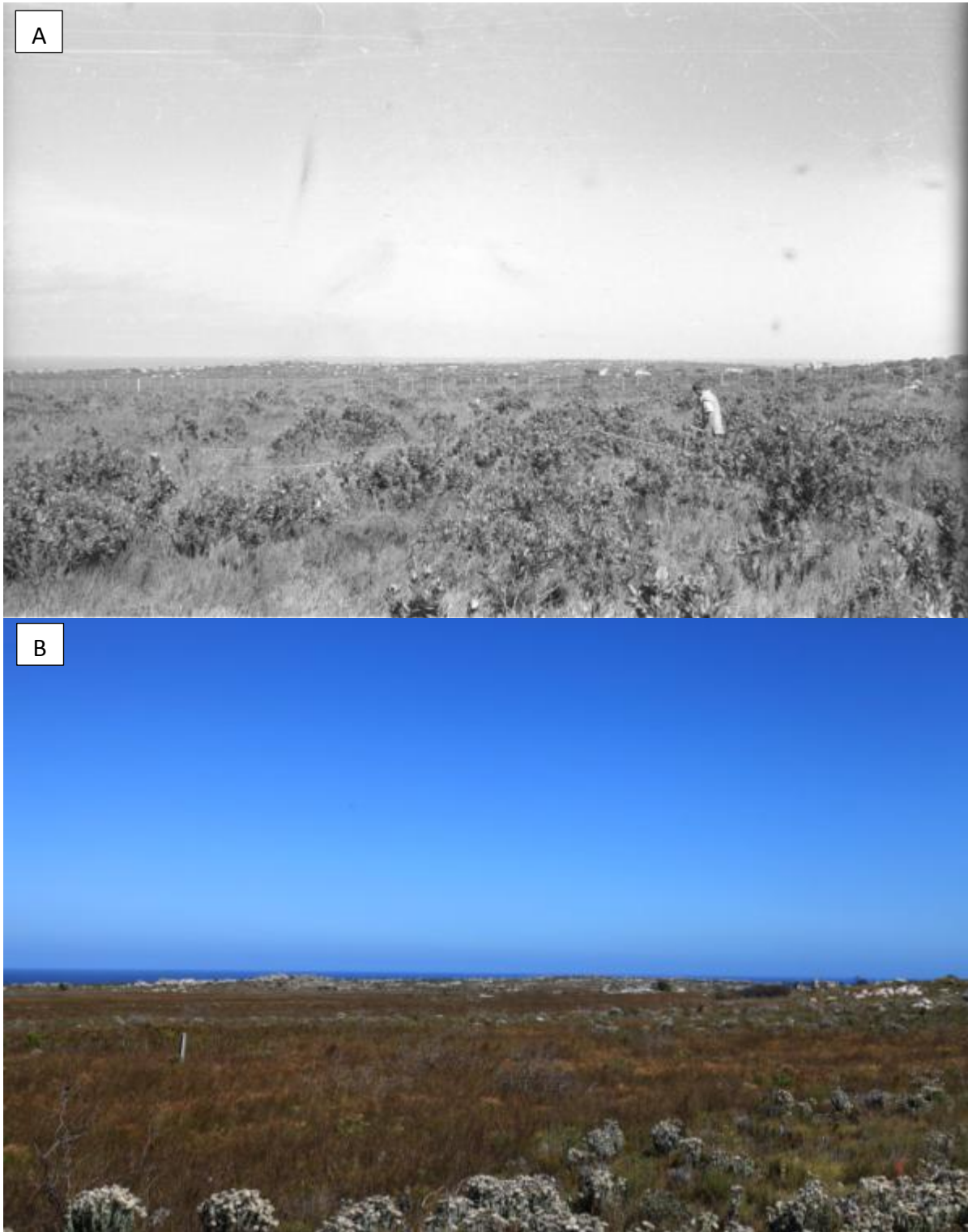


Figure 5.6: Sirkelsvlei South (No. 925). The most important changes reflect a loss of visible *Leucadendron laureolum* individuals. While individuals of this species are present at this location today they are <50cm in height as a result of the recent fire which burnt this area. With time, the vegetation will probably resemble very closely to that depicted in the 1966 image.

There was a positive relationship ( $p < 0.001$ ) between difference in post fire age and the change in cover of *Leucadendron* spp. (Figure 5.7). Similar to the trend observed in Figure 5.5, the percentage cover of *Leucadendron* spp. increased with increasing difference in post fire age.

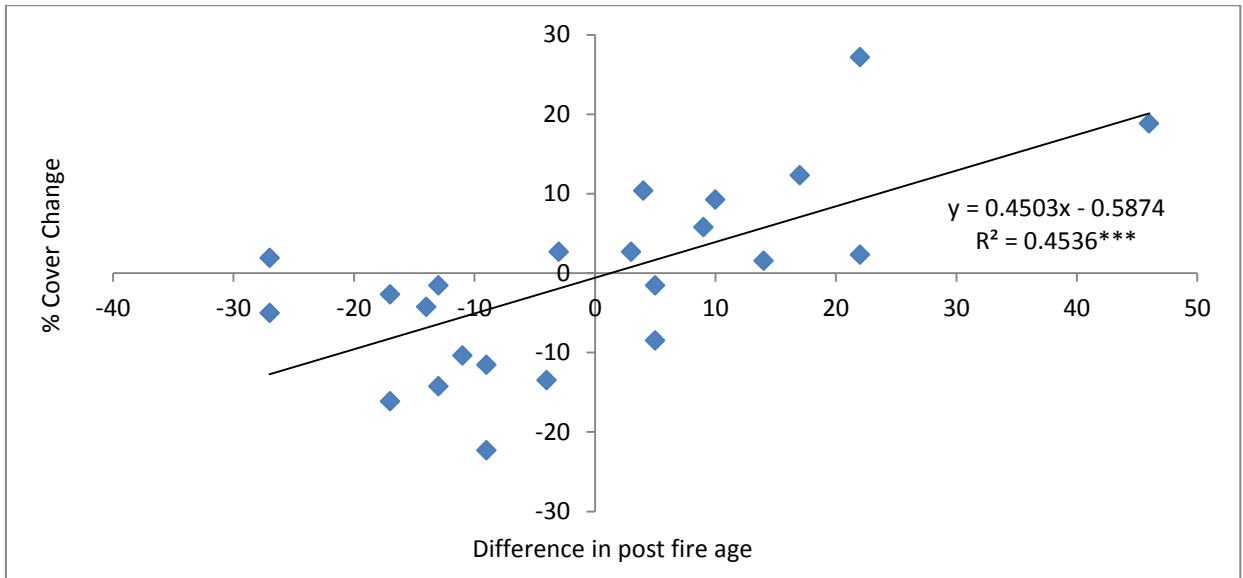


Figure 5.7: Linear regression showing the relationship between % Cover change and difference in post fire age for *Leucadendron* spp. present in 23 sites in the reserve.

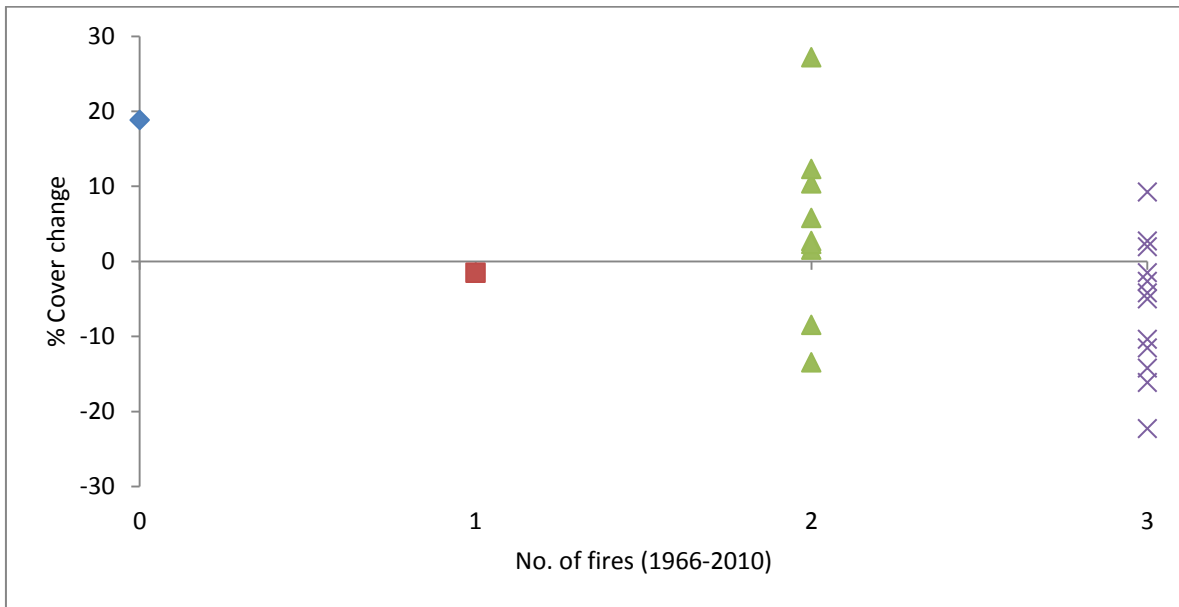
### ***Effects of Fire Frequency***

The relationship between fire frequency (expressed as the number of fires occurring between 1966 and 2012) and population change (minus 1) of *Mimetes/Leucospermum* was not significant ( $n=27$ ,  $R^2=0.021$ ,  $p>0.05$ ) (Figure 5.8). The majority of sites experience two to three fires between the sampling periods. There was only one record of a site that had not burnt at all since 1966.



**Figure 5.8: Population change (minus 1) of *Mimetes/Leucospermum* in relation to the number of fires that had burnt the area between 1966 and 2010.**

The relationship between the difference in the number of fires and the change in cover of *Leucadendron* spp. (Figure 5.9) was significant ( $n=23$ ,  $R^2=0.27$ ,  $p<0.01$ ). The majority of sites have experience two to three fires between the sampling periods. There was only one instance where a site had not burnt since 1966 and where a site had only burnt once since 1966.



**Figure 5.9: Percentage cover change of *Leucadendron* spp. in relation to the number of fires that had burnt the area between 1966 and 2010**

## Discussion

Future projections of the impact of climate change on Proteaceae suggest that the distribution of many species within the family will decrease in response to increased temperature and changing land use practices in the region (Midgley et al., 2003; Bomhard et al., 2005; Schurr et al., 2007; Higgins et al., 2008). For example, Bomhard et al., (2005) suggest that up to 29% of the 227 taxa that they assessed was affected by future climate change and as a result were uplisted, to a more threatened Red List category by 2020. In some cases species were uplisted by up to three categories higher. In their analysis, the proportion of threatened Proteaceae taxa increased by an average of 9% whereas critically endangered taxa rose from 1-7% depending on the scenario. Almost 2% of the 227 species became extinct due to the impacts of climate change on the species. Some species and regions that are currently not regarded as threatened will apparently be severely impacted by climate change and will move into the threatened category when future climates are considered.

Midgley et al., (2003) also tested the impact of climate change on the distribution of a number of Proteaceae species. Approximately 60.7% of species experienced range contractions and in five species their ranges disappeared altogether. Several species (approximately 40%), however, showed potential range expansions. Another factor tested was the degree of overlap between current and future ranges as an indicator of the ability of the species to move into and establish in the future climate envelope. Less than half of the high-risk species showed overlap between current and future distributions. This suggests that more than half of the high-risk species assessed would not be able to shift ranges and would thus potentially become extinct. Furthermore, Proteaceae are considered flagship or model taxa for the biome and other species in the region as well as outside the fynbos biome might be similarly affected (Burgman, 2002).

Few long-term studies have been conducted on population change in species within the Proteaceae. Those that have been done suggest that the demography of different species of Proteaceae are strongly influenced by the fire regime through its impact on recruitment processes (Bond et al., 1984; Lamont, 1985; Bradstock, 1990; Burgman and Lamont, 1992; Enright and Goldblum, 1999). For example, using an analysis of parent to seedling ratios, Coetzer et al., (2008), assessed long-term changes in sub populations of *Leucospermum gerrardii* in Mpumalanga. They found that local plant density appeared to be the primary factor explaining variation between populations and that disease as well as competition from surrounding pine plantations were not a threat. Another study on the demography of Proteaceae was conducted in Swartboskloof, near Stellenbosch (Le Maitre, 1998). It was found that mortality rates of *Protea neriifolia* and *Protea nitida* were higher than expected. This occurred because of a large fire event 25 years prior to the study which caused mass recruitment of these species in the immediate post-fire environment (Le Maitre, 1998). No fires had subsequently occurred in the region to trigger another recruitment event.

## ***Mimetes/Leucospermum* Changes**

Changes in the populations of *Mimetes/Leucospermum* varied considerable between sites and appeared site specific. However, overall there was no trend for either an increase or a decrease in the number of individuals across the reserve. This result is somewhat surprising given that the 10 year period prior to the 1966 photographs was 25% wetter than the 10 year period prior to the 2012 survey. The average annual rainfall for the period 1955-1965 was 475 mm while that for the period 2001-2011 was 356 mm. Given the relatively drier conditions in the latter period a slight decline in population size would have been expected if rainfall was a primary driving variable for the recruitment and mortality of these species.

The non-significant relationship between difference in post fire age and *Mimetes/Leucospermum* population change could be influenced by the fact that adults of these species possess a thick bark and are able to survive fire (Midgley et al., 1998). For example, Midgley et al., (1998), found that the proportion of live individuals of these two species was much greater than the proportion of dead individuals. This finding was reinforced by the fact that some individuals observed in 2012 were the same individuals that were present in 1966. These individuals had been able to withstand fire and survive for approximately 50 years.

Data from repeat photographs only provides information on the number of adult plants in the population. For those populations where a decrease in the number of adult plants was recorded it is also possible that seedlings may have been present in the landscape but were not visible from the photo location or were outside the field of view. Although such sites currently show a decline, in the future, these seedlings could potentially grow and lead to an increase in the number of individuals.

## ***Leucadendron* spp. Changes**

Percentage cover change of *Leucadendron* spp. was site-specific and showed no overall change in cover between 1966 and 2012. Because *Leucadendron* spp. are serotinous and their germination and establishment is influenced by fire (Bond et al. 1984), the % cover of *Leucadendron* spp. was significantly related to post fire age. Sites that did not follow the trend of increasing cover with increasing post fire age, were generally sites that did not differ in post fire age.

Although four sites appear to have 'lost' their population of *Leucadendron* spp. field inspections showed that the genus was present as seedlings (data not shown). Because of their generally small size, there were not visible in the repeat photographs. There was no evidence, therefore of local extinction at any of the sites.

## ***Fire frequency between Sampling Periods***

There was no relationship between the number of fires and the percentage change in the populations of adult *Mimetes/Leucospermum*. This supports the suggestion that these species are able to survive fires (Midgley et al., 1998) and that their population size is not influenced by fire frequency.

The significant relationship between percentage population change and fire frequency for *Leucadendron* spp. again shows the strong relationship between this species and fire. The finding that consecutive fires do have a negative impact on the cover of *Leucadendron* spp. is important as this means that it is not only the most recent fire that affects populations of these species but that fire history also needs to be taken into account when interpreting long term population changes in *Leucadendron* spp.

In summary, the results of this study suggest that large reseeding Proteaceae have not as yet been negatively affected by the observed increase in temperature recorded at Cape Point over the course of the 20<sup>th</sup> century. These differences are unsurprising for several

reasons. Firstly, the climate of the reserve is at the cooler end of the climate envelope of the fynbos biome and the impacts of climate change will probably not be detected easily at this location. Secondly, the increase in temperature at Cape Point has been relatively mild in comparison to future projections and critical thresholds might not have been reached for the species investigated in this study. If climate patterns were to continue along the same trajectories, these impacts on the Proteaceae of the reserve might become evident. It is important, therefore to continue monitoring these populations in order to be able to detect the response of these keystone species to the effects of climate and land use change.

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# Chapter VI: Conclusions

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## Introduction

This study used a repeat photography approach to analyse the extent, nature and rate of vegetation change in the Cape of Good Hope Section of Table Mountain National Park in relation to changes in key drivers such as climate, fire and land use. This is the first time that repeat photography has been used to assess vegetation change in the fynbos biome. Hugh Taylor's original photographs from 1966 were retaken from as close to the original position as possible. The repeat images were then analysed to assess the extent of vegetation change throughout the reserve.

The overall objective of this study was to document the extent, nature and rate of vegetation change in the Cape of Good Hope reserve using repeat photography and to relate these changes to the key drivers of climate, fire and land use. Changes in total vegetation cover as well as in key growth forms and species (e.g. emergent Proteaceae) formed part of this investigation. The first main objective was to assess changes in key drivers over the past 100 years. The second objective was to analyse, using repeat photography, how total vegetation cover as well as the cover of key growth forms has changed since 1966 in relation to post fire age in each of the major vegetation types of the reserve. These growth forms included broad leaved shrubs, proteoids, ericoids and restioids. Future projections based on transition matrices were also assessed for each of the major growth forms in the different vegetation types. The final objective also used repeat photography to investigate the extent of change in populations of large reseeding Proteaceae shrubs over a 46-year period. This chapter summarizes the key findings of each of the main investigations carried out in the thesis.

## **Drivers of change**

### ***Climate***

Several studies of the long-term climate record in the south western Cape have reported an increase in temperature and either a decrease or non-significant change in mean annual rainfall (Midgley et al. 2005, Warburton et al. 2005, Hoffman et al. 2011). Results from this study for one climate station located at Cape Point showed that there has been a significant increase of 0.36 °C per decade in maximum temperature. Changes in minimum temperature were not significant. Annual rainfall has also not changed significantly over the 110 year record. The trend in wind speed has alternated over time with periods of increase followed by periods of decrease. There has been a significant decline in relative humidity from 86% to 82% since 1978.

### ***Fire***

Data from a study conducted in the Table Mountain National Park suggested that between 1970 and 2008 there had been a reduction in the fire return interval from one every 32 years to one every 14 years (Forsyth and van Wilgen, 2008). However, in terms of presence of fire in the Cape of Good Hope reserve, the results from my study showed a greater number of fires that occurred, prior to 1965 when the reserve was established. Even though different management approaches to fire have been in place since the establishment of the reserve fire frequency has not changed significantly over the last 40 years.

Fire frequency across the reserve shows some spatial pattern, with areas in the northern parts of the reserve, where human presence is higher, having a greater frequency of fires. However, there are also areas in the southern portion of the reserve where fire has been suppressed for the safety of visitors to the reserve. Thus, the southeastern and southern parts of the reserve have not burnt since 1975. The average fire frequency across the reserve is three fires over a 36 year period and this was found to be acceptable in terms

of the recommendations of most research carried out in the fynbos biome (Cowling and Pierce, 1988; Cowling and Gxaba, 1990; Schwilk et al., 1997; Privett et al., 2001).

### ***Land use***

Pre-colonial populations of hunter-gatherers and pastoralists inhabited the reserve, but probably in low population numbers due to the challenging environmental conditions present in the reserve. The land use activity with the greatest potential impact that occurred in the reserve, prior to its establishment as a conservation area, was farming. Farms were established in 1810 and subsistence farming was carried out for approximately 150 years (Opie, 1967). This extensive land use history of the reserve was expected to have an impact on the vegetation in the reserve, with some decline in vegetation cover still evident around water points and seepage areas or in areas with a high prevalence of alien invasive species. The impact of herbivory by both domestic animals in the past and indigenous game primarily since 1965 was expected to be evident in the landscape. In addition, the way in which the reserve was managed immediately after its establishment in 1965 was also expected to have a significant impact on the vegetation. Between 1965 and 1975 a fence was erected and maintained across the reserve which separated the area into a grazed southern portion and an ungrazed botanical reserve in the north (Privett, 1996). However, there was no clear evidence from the repeat photographs that this impact had persisted over time although a detailed and closer inspection might find otherwise. The reserve was managed by the Cape Divisional Council until 1998 but since then has it has been managed by SANParks in alignment with national policy and adaptive conservation management plans and strategies. Results from the repeat photographic study suggest that it is difficult to discern the impact of previous land use practices on the vegetation of the reserve.

### ***Land Cover Change***

Most climate change projections suggest that the winter rainfall region will experience significant drying over the next fifty years (Midgley et al., 2005; Haensler et al., 2010). Such aridification should be reflected generally in the landscape through its impact on

vegetation cover and the exposure of rock and sand. I investigated this through the use of an aerial photographic analysis in which I identified four main cover classes. My findings were that there was no significant difference in vegetation cover over the period 1945-2008 although the reserve had changed from a patchy to a more homogeneous landscape, in terms of vegetation cover. Thus, despite the increase in maximum temperature values for Cape Point there was no indication in the aerial photograph analysis that there has been a general loss of vegetation cover or that sand and rock cover classes have increased in extent since 1945.

## **Repeat Photography and Landscape Vegetation Change (1966-2012)**

Understanding the extent, nature and rate of vegetation change is important and can give an indication of broader changes in the landscape. Vegetation cover as well as the dominance of key growth forms provides a proxy for the state of the ecosystem and how important properties might have changed over time. Trajectories of long-term historical changes can assist us when projecting into the future and predicting the potential impacts of climate change. A declining trend in vegetation cover as well as a dominance of growth forms and species associated with young post fire age vegetation was anticipated in the repeat photographs. This expectation arose because of the reported increase in fire frequency in Table Mountain National Park (Forsyth and van Wilgen 2008) as well as the increase in temperature and general trend towards a drier and hotter Cape Peninsula that has been reported (Midgley et al., 2005) and is expected in the future (Haensler et al., 2010).

Results of the landscape photographs were compared within three different vegetation types, viz. Dune Asteraceous Fynbos, Restioid Fynbos and Proteoid Fynbos. The greatest change in cover and growth form composition occurred in Dune Asteraceous Fynbos sites and the least in Restioid Fynbos sites. Proteoid fynbos sites showed a high degree of variability between sites. Dune Asteraceous sites were heavily impacted by both biotic

and abiotic disturbances leading to large changes in vegetation cover and composition. The remobilisation of previously-vegetated dune fields is an interesting finding of this study and worthy of further investigation. Restioid Fynbos sites showed very little change over the 46 year period, with vegetation cover changing by less than 5% in the majority of sites. This vegetation type appears stable even in the presence of disturbances such as fire. Sites that did show a change within Restioid Fynbos, were those that were previously impacted by alien vegetation. These sites have subsequently been cleared, resulting in a perceived loss in vegetation cover as well as an increase of moisture-associated species, presumably due to an increase in available moisture. Fire was shown to be the major driver of change in Proteoid Fynbos sites and most of the variability in cover between sites in this vegetation type could be explained by differences in post fire age.

## **Population Dynamics of Large Reseeding Proteaceae**

I hypothesised that the populations of large reseeded Proteaceae shrubs such as *Mimetes fimbriifolius*, *Leucospermum conocarpodendron* and *Leucodendron* spp. would have declined over the last 46 years in response to an increase in temperature and fire frequency. The reserve managers have also expressed concern about a perceived decline in populations of some of these species. The response of large reseeded Proteaceae to future climate change projections have also been relatively well studied (e.g. Bomhard et al. 2003) and few historical data on measured changes in population sizes are available. The relationship between fire and *M. fimbriifolius* and *L. conocarpodendron* was expected to be poor as their bark is thick and these two species can survive fires. Because *Leucodendron* spp. do not survive fire, but germinate well in post fire environments, it was expected that differences in the cover of these species would be strongly related to difference in post fire age. The results of my investigations showed that the populations of large reseeded Proteaceae were relatively stable across the reserve, but locally variable over the 46 year study period. There was a poor relationship between fire and *Mimetes fimbriifolius* and *Leucospermum conocarpodendron*, whereas

*Leucadendron* spp. appeared more dependent on time since last fire. Consecutive fires had a negative impact on the cover of *Leucadendron* spp. at a site.

## **Methodological Considerations**

### ***Climate, Fire and Land Cover***

As with every investigation there were some limitations to this study. With regard to the analysis of the drivers of change, there were a number of problems with data sources. Firstly, the climate data were based solely on the Cape Point climate station and there were a number of inconsistencies with this data. These inconsistencies, however, were accounted for and only periods with no or a few missing values only were analysed. The fire record data also showed some inconsistencies which were identified when sampling sites. For instance the 'official' fire record indicated that a fire had occurred in an area two years previously when it was obvious the site had not burnt for a number of years. Therefore, the accuracy of the fire record appeared questionable for some sites. However, this was not a frequent occurrence and when these inaccuracies were noted they were adjusted. Furthermore, the record prior to 1975 only recorded whether a fire occurred in the area or not. A more comprehensive analysis could have been conducted if a record of the number of fires (i.e. fire frequency) and an assessment of the spatial extent of each fire was available prior to 1975. In terms of the land cover analysis, it would have been advantageous to have good quality aerial photographs for the time steps between 1945 and 2008. Knowledge of the extent and nature of change at these intermediate time steps would have created a better understanding of the trend and direction of change and thus a more appropriate assessment of the trajectory of change.

### ***Repeat Photography as a Tool in Long-Term Research***

There are very few long-term studies of vegetation change in fynbos vegetation and none have used repeat photography as a primary tool. While the extent of vegetation change in other southern African biomes (e.g. Hoffman and O'Connor 1999; Rohde and Hoffman 2012) has been assessed using the techniques of repeat photography (Webb et al. 2010)

historical photographs have not been used to document the extent, nature and rate of change in vegetation of the fynbos biome.

The Cape of Good Hope has provided a model platform for long-term fynbos research. Previous studies of long-term vegetation change in the Cape of Good Hope reserve (Privett 1996; Thuiller et al. 2008; Hall 2010) have focused on species changes recorded in the 10 m x 5 m plots laid out in a systematic network by Hugh Taylor in 1966. Key findings from these studies suggest that while there is considerable turnover in species at a specific site over time the vegetation is stable at a meta-community level. Hall's (2010) study of a sub-set of Taylor's original plots suggests that there has been a reduction in species richness of almost 25%. For all of these plot-based studies it is imperative that the plot is permanently marked and that the repeat survey is conducted on exactly the same area as the original survey. A careful analysis in the field of the historical photographs of each of the plots showed that only 46% of the plots were re-located exactly on the site of the original plot. A total of 34% of the plots were not permanently marked and it was therefore impossible to re-locate them exactly while 20% of the plots were not re-located correctly. The use of the historical photographs suggests, therefore, that some of the differences in species composition and richness between the different survey dates (Thuiller et al. 2008) could have resulted from the errors associated with the re-location of the plots.

Besides providing a useful check on the accuracy of plot re-location repeat photography also provides an important record of changes in cover and dominant growth forms at a landscape level of scale. This is important for management decisions which focus more on landscape level processes evident in repeat photographs than on processes inferred from detailed plot data. As this study has shown it is possible to derive both qualitative and quantitative information from the techniques of repeat photography (Webb et al. 2010) and in so doing provide conservation managers with benchmark conditions and trajectories of change in total vegetation and growth form cover for large areas for which they have responsibility. Such information could form the basis for management interventions either, for example, through a change in fire management practices or

through the increased protection of sensitive coastal habitats. Although this method of repeat photography is time consuming, requiring a day of field work to relocate and re-photograph up to four sites, it could be used effectively for monitoring purposes. Site relocation is considerably enhanced with the aid of GPS co-ordinates and the placement of a small rock cairn on the site of the original camera station. A tripod and a high resolution camera (14 megapixel and higher) are required and some knowledge of image-processing using Adobe Photoshop is necessary to analyse the paired images after the photographs have been taken. In addition, ecological knowledge of the landscape, the disturbance regime and changes in key climate variables over time assist in the interpretation of the major changes in the vegetation over time.

The repeat photography method also had its limitations, although it did provide an interesting perspective that has not been assessed in previous studies. The main limitation with repeat photographs is that the results are biased by the dominant species in an image and therefore changes in less dominant or rare species are not recorded. A similar challenge is presented with the analysis of Proteaceae populations using repeat photographs in that the seedlings were too small to be identified and counted from the images. Although there are a number of limitations in using this approach, it provides, for the managers of the reserve, a unique and comprehensive understanding of long-term changes that have occurred in the area. It also provides evidence of how the landscape has changed over time which is the appropriate level of scale for key management decisions.

## **Management and Conservation Implications**

The reserve is currently managed by SANParks, in line with national conservation policies and management plans. Part of this management strategy involves the development of “thresholds of potential concern” (TPCs) on which to monitor and adapt management strategies in the different parks. The objectives and TPCs defined by SANParks for the Table Mountain National Park are described in Table 1. The management and

conservation implications that relate to the findings from this study will be discussed under the objectives set by SANParks.

**Table 1: Thresholds of Potential Concern (TPC) for each of four key management areas within the South African National Parks management and monitoring programme (SANParks, 2008).**

Sub-Objective	Thresholds of Potential Concern (TPC) to be developed
1. Biodiversity Representation	<b>Rare and endangered species:</b> Specific thresholds need to be set to for all rare and endangered species on the Cape Peninsula. The priority of which, will be set through workshops with scientists.
2. Ecological Restoration	<b>Alien Species Management:</b> Thresholds and conservation targets need to be set for the detection, spread, control and eradication of invasive alien species.  <b>Large Herbivores:</b> Setting indicators for large herbivore stocking rates in fenced areas of the Park.
3. Fire Management	<b>Fire Management:</b> Thresholds need to be set for the size of fires, fire frequency and fire season.
4. Threat Management	<b>Problem Animals:</b> Thresholds and conservation targets need to be set for baboon populations in order to determine their long-term survival on the Cape Peninsula.

### ***Biodiversity Representation***

Biodiversity representation, specifically related to rare and endangered species was not investigated in this study, but is well documented in other studies (Privett, 1998; Thuiller et al., 2007; Hall, 2010). This study provided an understanding of change at a landscape-level and is important to management, as landscape units provide an accessible and appropriate level of scale by which to set conservation targets for the reserve. A remarkable finding was the mobility of species across the landscape, particularly *Protea lepidocarpodendron*. Dense stands of this species have visibly migrated, especially along ridges (Figure 5, Figure 6 and Figure 8 in Appedix B). This reiterates the importance of understanding landscape-level changes, especially in a diverse ecosystem such as the fynbos biome. Tall emergent Proteaceae showed no change across the reserve and although site specific changes were recorded, overall there was little change in populations of *Mimetes fimbriifolius*, *Leucospermum conocarpodendron*, *Leucadendron laureolum*, *Leucadendron xanthoconus* and *Leucadendron salignum*..

## ***Ecological Restoration***

Currently, woody invasive alien species appears to not be a problem in the reserve, as only a few individuals were found whilst surveying and only a few dense stands primarily in the northern part of the reserve were observed. Working for Water and the management of the reserve have clearly worked hard to eradicate and minimise the invasion of alien species. The resulting changes in community composition as a result of removal of alien species and subsequent increase in moisture availability is apparent at Mimicki's (Figure 14, Appendix B). This site has experienced a shift in community composition from a restioid- and alien-dominated community to moisture-affiliated ericoid dominated vegetation. This emphasises the impact of alien invasive plants on water resources and on the indigenous flora.

The impact of large herbivores in the reserve was not apparent from the analysis in this study. There was one site (Brightwater Road, Figure 45 in Appendix B) however, that was in close proximity to an Eland midden. The presence of this Eland midden, although not in the field of view of the photograph, appeared to have an impact on the vegetation, resulting in a loss of cover. Overall, the impact of large herbivores was not observed in the results. The impact of small rodents however, had a great impact on the vegetation cover of the reserve, specifically at Klein Rondevlei (Figure 15 Appendix B). At this site, the presence of mole rats in the early post fire environment reduced the vegetation cover by 41%. These local disturbances are important to monitor.

Another management implication that is important to mention, although not part of the SANParks thresholds of concern objectives is the impact of human disturbance on broad leaved shrubs. The transition matrix for Dune Asteraceous Fynbos (Chapter IV, Figure 19), projects a decline in the dominance of broad leaved shrubs and almost complete disappearance by 2058 particularly in the more exposed western coastline of the reserve. This is an alarming trend and further monitoring should be put into place at these sites to confirm these patterns. Active restoration at these sites may be necessary before human disturbance causes severe and potentially irreversible damage to this coastal vegetation. One way to reduce the disturbance of humans is to cordon areas off from the public, as

currently they have free roam in the coastal scrub, particularly alongside the beaches. This method has been successful in the past where vegetation has recovered once rested from human disturbance, such as at Buffels Bay and Kalkoond (Figure 64, Figure 73 and Figure 74 in Appendix B), where vegetation cover has increased since 1965 due to the reduction of tourists and restriction of public movement through the vegetation.

### ***Fire Management***

Fire management in the reserve has been varied (refer to Chapter II). Fire was found to be a major driver of change in vegetation cover and growth form composition, especially in the Proteoid Fynbos sites. This has important implications for management as fire can be used as a tool to shape and transform the landscape. With more understanding and used appropriately, fire can be used to mitigate the impact of future climate change on the landscape. For instance, with changes in climate, some species are expected to dominate more than others. This would alter the growth form composition of the site or even the reserve. Understanding the influence of fire on the species, the fire regime can be used to manipulate the dominance of certain species, which would otherwise reduce biodiversity. The reserve has experienced a higher number of fires particularly along the northern border of the reserve which is adjacent to Scarborough. This could significantly impact the vegetation that lies in the northern part of the reserve by reducing post fire recovery thus leading to a lower vegetation cover as well as potentially a reduction in species persistence.

### ***Threat Management***

Although baboons are a threat to tourists in the reserve they were not studied in this thesis. Major threats to the vegetation of the reserve which were observed in this study include (1) the remobilization of dunes along the west coast of the reserve; (2) the impact of small mammals on coastal dunes; (3) the impact of tourists on the coastal vegetation; (4) the spread of alien invasive plants within the reserve. It is suggested that TPCs are developed for each of these threats and effective observation and monitoring programmes developed to assess their impact across the reserve.

## **Future research**

Although many studies have been conducted on the vegetation of the Cape of Good Hope, there is still scope for more. The most important future study would be to resurvey the plots with the correct layout so that an understanding of what changes are occurring at a species level can be clarified and confirmed. A more detailed study on the population of large reseeding Proteaceae would also be a benefit to the reserve. My study only used the landscape photographs and a long-term monitoring programme specifically focussed on these species, could assist reserve managers greatly in terms of future management plans and lead to a better understanding of the dynamics of these populations. In addition, the photographs used in this study need to be properly archived and resurveyed at regular intervals to provide a more accurate assessment of vegetation change over the next 50 years. This would be interesting as many of the current climate models forecast to 50 years in the future and whether these forecasts were in line with changes in the reserve could be determined. This study would be able to explore the nature, extent and rate of change over different time periods which would provide better projections of future changes in the reserve.

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