The Impacts of Different Methods of Alien Plant Control on the Recovery of Fynbos Ecosystems

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Abstract

Invasions by alien trees and shrubs are a serious problem in fynbos ecosystems. Dense stands of alien trees and shrubs have the ability to transform both the structure and function of the indigenous vegetation. It is for this reason that control operations are underway over large parts of the Cape Floral Kingdom. The impact of control operations itself, however, also needs to be assessed and the question of whether restoration actions (such as seed additions), are necessary after different methods of alien plant control, needs to be addressed. Very little work has been done to determine what types of restoration is needed after various methods of alien plant control, but at the Cape of Good Hope Nature Reserve control of invasive alien vegetation has been underway for decades. This therefore provided an ideal opportunity to explore the effects of the different methods of alien plant control and the types of restoration actions that may be needed.

This study investigated the effects of past management aimed at clearing invasive alien plants in South African fynbos ecosystems. The treatments included "bulldozing and aerial-spraying", "aerial-spraying", "boom-spraying" and "spot–spraying". Both an ant study and vegetation survey was conducted to test the hypothesis that fynbos is resilient to all kinds of perturbation (including the most severe alien plant management treatments employed) and that natural recovery over time is adequate to restore fynbos (without human intervention).

In Chapter 1, literature relating to invasive alien plant management in the Cape Floral Kingdom was reviewed. Another major component of this chapter included the selection of study sites. The sites were chosen based on consultations with managers, as well as with the aid of a number of maps that were derived using a Geographic Information System.

In Chapter 2 variables such as percentage canopy cover, species richness, soil pH, soil depth and functional guilds were compared between invaded and control sites. The richness in the "boom-sprayed" and one of the "spot-sprayed" treatments was reduced significantly when compared to control sites in the same locality. Multivariate analysis of the plant guilds showed that the controls are more alike in structure to each other than to the treatments, with the implication that the treatments employed to control alien vegetation are impacting on the recruitment of the indigenous vegetation.

In this study as in numerous other studies dealing with rehabilitation in mining areas as well as land assessment and monitoring programs, ants were chosen as the invertebrate indicator species of restoration success. Multivariate analysis based on species abundance has shown that the ant species formed a distinct group in the "bulldozed and aerial-sprayed site" compared to control. Here, the pitfalls that separated from the main cluster contained the invasive Argentine Ant, renown for lowering the diversity of indigenous ants in the ecosystems they are introduced into. ANOVA results also showed a decrease in diversity for this treatment site compared to its respective control. The "spot-sprayed" treatment at the Meadows also formed a separate group from the main cluster, with these pitfalls largely being associated with the presence of the seed-dispersing ant *Anoplolepis steingroeveri*. Principle components analysis based on the first principle components score indicate that the "boom-sprayed" treatment significantly impacted the ant composition, although the important seed dispersers were present. Another important finding was that ants responded to vegetation structure. Overall ants were found to be good indicators of ecosystem recovery although plants were found to be the more sensitive indicator.

This study suggests that certain methods of invasive alien clearance have been more harmful to the recovery of indigenous vegetation than others. Where treatments such as "boom-spraying" and the "foliar-spraying" treatment at the Meadows impacted most of the plant variables and also caused a change to the plant structure, species re-introductions may be needed to speed up recovery of the indigenous ecosystem.

Chapter 1: General Introduction

1. Rationale

The aim of this study was to determine the combined effects of dense alien plant invasion and past management practices used to manage invasive alien trees and shrubs on the fynbos vegetation. The study was undertaken in the Cape of Good Hope Nature Reserve section of the Cape Peninsula National Park. The working null hypothesis was that fynbos is resilient to all kinds of perturbation (including the most severe alien plant management treatments employed at the reserve), and that natural recovery over time is adequate to restore fynbos structure and diversity (without further human intervention). As a corollary, it is important to determine whether additional restoration action (such as species re-introduction) is necessary after initial control or follow-up work has been applied.

This thesis comprises four chapters. In Chapter 1, I review the relevant literature and provide a detailed description of the study sites and the management practices employed within them. This is necessary to outline the history of vegetation management and the current status of the study area.

In Chapter 2, I describe the vegetation survey carried out on each of the study sites after the different methods of alien plant management. I classify the plant species into different functional groups in order to explore whether different guilds respond differently to the various methods of control. The purpose of this approach was to provide managers and restoration practitioners with an assessment of the ability of fynbos, at different sites, to recover on its own (without human intervention) from a range of treatments, and to determine which management practices applied in the past have been most successful. The goal of the manager in this case being the efficient removal of the invader with minimal detrimental impact on the natural vegetation.

In Chapter 3, I report on an investigation into the "health" of the ecosystem at each of the study sites, using the composition/diversity of ant species as ecological indicators. Invertebrates generally occupy special niches, and therefore provide information about the area according to their presence or absence (Majer 1981). In this study as in numerous other studies dealing with rehabilitation in mining areas (Major 1983, 1985, 1992), as well as land assessment (Noss 1990) and monitoring programs (Kremen 1992, 1994), ants were chosen as the invertebrate indicator species of restoration success. Another factor making ants useful as indicator species in this study is that they are well known in the fynbos ecosystem (Bond and Slingsby 1981, 1983, 1984, Donnelly and Giliomee 1985a,b, Koen and Breytenbach 1988, Bond *et al.* 1991, Christian 2001, French and Major 2001).

In Chapter 4, I summarize the main findings of this study as well as provide some of the drawbacks or limitations encountered.

1.1. The Cape Floral Kingdom

The Cape Floral Kingdom is renowned for its exceptional beauty and rich floral heritage. Situated at the southwestern tip of Southern Africa, it spans 90 000 km^2 (less than 4 % of Southern Africa), and is one of six floral kingdoms into which botanists have divided the whole earth. This small area is home to about 9000 vascular plants (flowering plants, ferns and gymnosperms), of which approximately 70 % are endemic (Goldblatt and Manning 2000). This high level of endemism in the Cape Flora is intermediate between the higher levels of endemism in isolated island floras such as Hawaii (92.3 %) and the floristically distinctive regions within large landmasses such as California (47.7% endemic species) (Bond and Goldblatt 1984). Despite this high concentration of endemics, 75.7 % of the original vegetation of the Cape Floral Kingdom has been lost (Myers *et al.* 2000), and it is for this reason that they have classified this region as one of 25 biodiversity hotspots of international significance.

Much of the Cape Floral Kingdom (as well as many other parts of the world) has been and currently still is being transformed by alien plant encroachment. Other land practices such as pastoral and agricultural farming, and urban developments also play a big role in reducing diversity both through habitat destruction and fragmentation (Rebelo 1996, Wood *et al.* 1994, Richardson *et al.* 1996). The city of Cape Town sits on two centres of endemism (the Cape Peninsula and the Cape Flats), and here urban expansion threatens several hundreds of species (Rebelo 1996). This is clearly demonstrated on the Cape Peninsula (49 127 ha in extent) where urbanization and agriculture have transformed 37% of the original natural vegetation (Richardson *et al.* 1996), with at least 39 species having become extinct in the 20th century. The lowland vegetation types on the Peninsula have been worst affected whereas regions of high altitude have been relatively free of urbanization and agriculture, but here alien plants threaten diversity. Of the area not affected by direct habitat destruction, 10.7 % is under dense alien stands (>25 % canopy cover) and another 32.9 % is lightly invaded (Richardson *et al.* 1996). The Cape Flats has also been severely transformed due to urban expansion, agriculture and woody alien infestations. Here only 1.2% and 32% of the original extent of sand plain fynbos and strandveld (the two dominant vegetation types of the area) respectively, remains (Wood *et al.* 1994).

Invasive alien plants not only threaten the diversity of natural systems, but they pose a threat to water supplies (Fenn 1980). Alien trees form dense stands in mountain catchment areas and because of their larger size and biomass use more water than native fynbos (van Wilgen *et al.* 1996). They also create higher intensity fires (Richardson and van Wilgen 1986), increase soil erosion (Scott and van Wyk 1992), degrade riparian regions (Hoffman and Moran 1988) and change habitat suitability for native animal species (Donnelly and Giliomee 1985b). These species therefore pose a serious threat to the structure (measures of species composition) and function (ecosystem processes such as primary productivity, water clarity, soil erosion and the loss or retention of nutrients) of fynbos ecosystems.

1.2. Fynbos Vegetation

1.2.1. Community Structure

Fynbos (meaning fine-leaved bush) is a common term used to describe the dominant vegetation type in the Cape Floristic Kingdom. It is a sclerophyllous, fire-prone shrubland occurring on nutrient-poor (oligotrophic) soils (Moll and Jarman 1984). It may be characterized by three elements (Taylor 1978): restioids (aphyllous, evergreen graminoids of the Restionaceae), ericoids (heath-like shrubs with small, rolled leaves, especially Ericaceae), and proteoids (shrubs with large isobilateral leaves of the Proteaceae), with the presence of restioids being the essential feature defining Cape fynbos (Campbell 1985). Proteoid shrubs usually form a non-sprouting overstorey of approximately 1,5 m – 4 m high. Proteoid Fynbos vegetation resembles the kwongan vegetation of the mediterranean-climate region of southwestern Australia in terms of its structure and function (Milewski 1983).

1.2.2. Diversity and Endemism

Fynbos vegetation forms the largest component of the Cape Floral Kingdom. Its high level of diversity and endemism are well known (Takhtajan 1986, Cowling 1990, Cowling *et al.* 1992). It has over 7 000 plant species distributed amongst its five principal vegetation types (grassy, laterite, limestone, mountain and sand plain fynbos) and has an endemicity of over 80 % of plant species confined to the biome (Rebelo 1996). The high levels of diversity and endemism may be explained in terms of beta and gamma diversity: there is a high turnover in species between different habitats (beta) and between species in similar habitats along geographic gradients (gamma diversity) (Cowling *et al.* 1992, Simmons and Cowling 1996).

The biological profile of a typical fynbos endemic is a dwarf, non-sprouting shrub with soil stored seed banks and ant dispersed seeds (Cowling *et al.* 1992, McDonald and Cowling 1995, Trinder-Smith *et al.* 1996). Endemics also were shown to form symbiotic associations with microbes (Cowling *et al.* 1992).

1.2.3. Why conserve the biodiversity of fynbos?

South Africa contains a wealth of biological diversity, yet the loss of this natural heritage is increasing as in many other parts of the world (D'Antonio and Vitousek 1992, Enserink 1999). Industry, agriculture, population growth and per capita resource use primarily drives the loss of biological diversity (D'Antonio *et al.* 1997). The need for preventing this loss, and increasing financial investment to conserve biodiversity, has to be reconciled with the basic needs of the South African people. These needs include jobs, land, housing, clean water, electricity, a clean and healthy environment, health-care and social welfare (African National Congress 1994). However, for the majority of the South African population, these criteria have not been met, and therefore biodiversity conservation is of low national priority (Holmes and Richardson 1999).

Due to the current socio-economic conditions in the country such as poverty and illiteracy etc. it may be essential that the value of the ecosystem be considered in monetary terms, so as to competitively justify public funding (van Wilgen *et al.* 1996). Direct uses (leading to increased revenue) of biological resources include: gathering, harvesting, or hunting of plant and animal species for food, shelter, medicine, fuel and fibre, to the use of ecosystems and specific habitats for grazing, croplands, mining, recreation and tourism (Laird and Wynberg 1996).

Biodiversity also provides indirect values and benefits, including ecosystem functions like nutrient recycling, waste assimilation, climate regulation, maintenance of fertile soils, prevention of soil erosion etc. Non–consumptive values include aesthetic appeal, spiritual, cultural and moral benefits (Laird and Wynberg 1996). It is difficult to set a value on biodiversity services, although this has been tried (Westman 1977, van Wilgen *et al.* 1996). Figures relating to some of the more direct values of South African biodiversity shows that tourism brings in the greatest amount. In 1993, in protected areas alone, it contributed between R710 million–R6.09 billion to S.A GDP and created some 17 000–148 000 jobs (Wells 1995). In the same year, the wild flower industry, using mostly flora from the fynbos brought in R70 million in export earnings and provided jobs for 20 000–30 000 people (Cowling *et al.* 1995, Cowling and Richardson 1995). Many types of indigenous products are commercially harvested for food, shelter and medicinal purposes. These include certain strongly aromatic buchus e.g. *Agathosma betulina* and *Agathosma crenulata* (buchu oil); *Aspalathus linearis* (rooibos tea); *Aloe ferox* (Aloe bitters); at least 9 out of 20 *Cyclopia* species (honeybush tea); the fruits of *Carpobrotus acinaciformis*, *Carpobrotus deliciosus* and *Carpobrotus edulis* and certain *Thamnochortus* and *Chondropetalum* species that are harvested for thatch. South African plants also make a large contribution to the horticultural industry in other parts of the world (Cowling and Richardson 1995).

Biodiversity also provides the informal sector with the opportunities for self- employment. In the Witwatersrand area alone, the informal medicinal plant or "muti" trade is estimated at R 21 million per annum, and nationally (including the Cape Floral Kingdom) it is believed to vary from R500 million to R1 billion (Williams *et al.* 1996). Within this sector several indigenous plant species are used for roofing and structural material or for firewood (Cowling and Richardson 1995).

Fynbos has thus been identified as a resource of economic, cultural and ecological/functional importance. It is our duty to conserve this heritage, through sustainable use thereof to ensure a heritage for future generations. Biodiversity conservation needs to be prioritized above development needs that only benefit private individuals. Educating people at grass roots level could also lead to an environmental conscience, which could go far to conserving this floral heritage.

1.3. Invasive Alien Trees and Shrubs

1.3.1. Introduction of alien plants to the Cape

After European settlement in 1652, a wide variety of alien plants were introduced from Europe and Asia. These were mainly agricultural and horticultural crops with only one of the introduced species becoming invasive i.e. *Pinus pinaster* (Shaughnessy 1986). This species is currently one of the 12 most successful invaders in the CFK. However after 1830 some of the most important invaders of fynbos vegetation were intentionally introduced and extensively propagated for a range of purposes. These included dune stabilization, wind-breaks, fuel, shelter, construction materials, horticultural uses, commercial forestry and aesthetic appeal, and have spread at increasing rates ever since (Shaughnessy 1986, Richardson *et al.* 1992, Richardson *et al.* 1997). These "transformer" species mainly include woody trees and shrubs such as five *Acacia* species, *Paraserianthes (=Albizia) lophantha,* two species of *Hakea* and *Leptospermum laevigatum.* These species are all of Australian origin. The remaining two species are *Pinus halepensis* and *Pinus pinaster.* These two are both of European origin with the earliest known occurrence of *Pinus halepensis* being around 1830 (Richardson *et al.* 1992). Shaughnessy (1986) described 13 species, with *Acacia pycnantha* and *Hakea gibbosa* included as "pest" plants but not *Pinus radiata*, which has its origins in California.

These alien plants originate from regions of similar climate or ecological environments, and therefore are pre-adapted to local environments and generally were introduced without their natural enemies. This gives them a competitive edge, as they are equipped biologically to overcome barriers that prevent other species from invading the fynbos biome (Cowling and Richardson 1995). The ability of invasive alien species to transform habitats and reduce diversity to dense mono-specific stands can also be attributed to their short juvenile periods and large seed stores. Some species such as *A. saligna* and *A. cyclops* have an annual seed production reaching up to 40 000 and 8 000 seeds respectively (Milton and Hall 1981).

Table 1. The origins of the most important alien plant invaders of the Cape Floristic Kingdom. From Shaughnessy (1986) and Richardson *et al.* 1992.

Species	Common name	Earliest known occurrence
Region of Origin: Australia		
Acacia cyclops	Rooikraans	1857
Acacia longifolia	Long-leaved Wattle	1827
Acacia mearnsii	Black Wattle	1858
Acacia melanoxylon	Blackwood	1848
Acacia pycnantha	Golden Wattle	1865
Acacia saligna	Port Jackson	1848
Paraserianthes lophantha	Stinkbean	1835
Hakea gibbosa	Rock Hakea	1835
Hakea sericea	Silky Hakea	1858
Hakea suaveolens (=drupacea)	Sweet Hakea	1850
Leptospermum laevigatum	Australian Myrtle	1850
Region of Origin: Europe		
Pinus halepensis	Aleppo Pine	ca 1830
Pinus pinaster	Cluster Pine	1685-1693
Region of Origin: California		
Pinus radiata	Radiata pine	1865

1.3.2. The history of alien plant control

In South Africa, most invasive aliens have been the targets of alien plant control. This started as early as 1860 and 1883, for species such as *Xanthium spinosum* and *Opuntia ficus-indica* respectively (Stirton 1978). Woody plant invaders were also targeted by alien plant control methods as early as 1860 (Macdonald and Richardson 1986; chapter 2 this thesis). However, efforts to control invasive plants were without exception uncoordinated and haphazard. They were applied on a trial and error basis without adequate knowledge of the ecology or population biology of the individual species, or of the economic and sociological factors involved (Richardson *et al.* 1997). It is for these reasons that alien plant control remained highly unsuccessful for a long time. Detailed reviews exist for the history of control operations conducted within certain nature reserves (e.g. Macdonald 1983 for Hluhluwe/Umfolozi; Macdonald 1988 for Kruger National Park; and Macdonald, Clark and Taylor 1989 for the Cape of Good Hope Nature Reserve). This last mentioned reserve, now known, as the Cape of Good Hope Nature Reserve section of the Cape Peninsula National Park provides an excellent example of these erratic initial control efforts. Records dating back to 1941 indicate attempts to control invasive aliens, but with little success for at least another 35 years (van Wilgen *et al.* 1997). This largely was due to a lack of follow-up after initial clearing (van Wilgen *et al.* 1997) as well as confusing two operations with incompatible aims: (a) clearing alien plants and (b) supplying firewood for recreational areas (Taylor 1970). Clearing alien plants calls for the plant being completely eliminated whereas cutting trees for firewood requires that the resource be renewable. Hence conservation and demand, two opposing concepts, led to the mismanagement of both funds and labour, with the result that initial control was completely unsuccessful.

The introduction of a coordinated and well-funded eradication plan drawn up in 1974 and implemented in the late 1970's saw the start of a new era of management for the reserve. This plan incorporated a prescribed burning regime, on a ten-year rotation and has proved to be very successful largely as a result of integrated control (van Wilgen *et al.* 1997). Integrated control refers to the application of a number of control options in concert to achieve a favourable outcome, taking economic, sociological and ecological implications into account (Kluge *et al.* 1986).

Much of the early management programmes for invasive alien plants stemmed from the threat that they posed to water supplies through reduced runoff from catchment areas compared to indigenous vegetation (Fenn 1980). Thus adequate government funding was initially (early 1970's- 1980's) available. This was spent clearing aliens in the mountain catchment areas as well as on research. The outcome of this period was an integrated control programme for the removal of alien plants in which mechanical and chemical means were used to ensure that trees were killed (van Wilgen *et al.* 1997). In 1995 a renewed effort to control aliens was undertaken by the "Working for Water" programme when officials once again realized that dense stands of aliens could lead to a substantial reduction in stream flow (Holmes and Gelderblom 1998).

1.3.3. The impacts of alien plant invasions

In a review of the threats posed by invasive alien plants to natural ecosystems worldwide, Cronk and Fuller (1995), identified seven categories of threats:

1) Replacement of diverse systems with single or mixed stands of alien invader species.
2) Alteration of the soil chemistry.
3) Alteration of the geomorphological processes.
4) Changes in the fire regime.
5) Changes to the hydrology of an area.
6) Invasions leading to plant extinction.
7) Invasions posing a threat to soil fauna.

In the Cape Floristic Kingdom a number of studies have documented these threats. Several recent studies have documented the extent of alien invaders in the biome (Macdonald and Richardson 1986, Richardson *et al.* 1992, 1996, 1997, Henderson 1998). Other studies have documented a reduction in species richness after invasion by alien plants mainly due to competitive exclusion (Richardson and van Wilgen 1986, Richardson *et al.* 1989, Richardson *et al.* 1996, Holmes and Cowling 1997a,b). As a result of the suppression of indigenous plant species many taxa face extinction, with 750 currently at risk (Richardson *et al.* 1992). Holmes and Cowling (1997b) also have indicated a decrease in diversity and richness in the seed bank with increasing duration of invasion. In this study, however, the major growth form guild was represented in the seed bank even after twenty years of dense infestation, indicating good restoration potential.

Several studies relate changes in soil chemistry to invasive plants (Witkowski 1991, Musil 1993, Musil and Midgley 1993). Dense stands of alien nitrogen-fixers (family Fabaceae) increased the levels of soil nitrogen and other nutrients, such as phosphorus, due to association with nitrogen-fixing bacteria and increased litterfall. This enriched environment may, in-turn, promote fast growing annuals and grasses (Holmes and Richardson 1999).

Changes to the fire regime brought about by invaders are one of the most serious threats to fynbos vegetation. Van Wilgen and Richardson (1985) showed that alien shrubs such as *Hakea* and *Acacia* change the structure of the fuel bed, by eliminating the understorey and radically altering fuelbed characteristics. These two species increased the fuel loads by 60% and 50 % respectively, and *Hakea* also reduced the moisture content. Versfeld and van Wilgen (1986) found that in self-sown pine forests the biomass might be five times higher than in fynbos shrublands. Thus under extreme weather conditions, fires in alien stands may be much more intense than in fynbos, they burn through the taller, live canopies. These fires are therefore more damaging to the ecosystem than fires in natural fynbos, and can lead to fires becoming less frequent but more intense. Intense fires are also experienced when serotinous (plants with seeds stored in cones or fruits) alien plants such as *Hakea sericea* and *Pinus pinaster* are subjected to weed control. To eliminate these plants they are felled before burning. The seeds are then released and either germinates or is eaten by rodents. Approximately one year after felling, the area is burned and seedlings are killed (van Wilgen *et al.* 1996, Holmes and Marais 2000). These intense fires, especially following a "fell and burn" treatment (where the biomass is closer to the soil), increases water repellency in the soil which aids soil erosion following the first heavy rains (Richardson and van Wilgen 1986, Scott and Van Wyk 1992).

Van Wilgen *et al.* 1996 and Le Maitre *et al.* 1996 have documented the increased water use of invasive alien species. Dense stands of aliens can reduce runoff by between 30% and 70%, depending on such factors as the annual rainfall, age and density of the stand (van Wyk 1987).

Owing to the large impacts of invasive alien trees and shrubs on fynbos biodiversity and ecosystem processes, it is important that the alien species are controlled. Various techniques have been used in clearance operations, yet the side effects of these operations have yet to be adequately assessed (Macdonald *et al.* 1989). This applies especially to the Cape of Good Hope Nature Reserve which now forms part of the Cape Peninsula National Park. This reserve forms one of the areas in the Cape Floral Kingdom for which good data (dating back to 1941) is available on the history of attempts at managing invasive alien plants.

These earlier studies provided a basis from which to compare the effectiveness of the different clearance treatments on the recovery of fynbos vegetation. In addition, it is important to determine whether additional restoration actions (such as seed addition) may be necessary after initial control or follow-up work.

1.4. A detailed description of the study sites

1.4.1. Study region

This study was conducted in the Cape of Good Hope Section of the Cape Peninsula National Park, which is situated at the south–western corner of the Cape Floristic Kingdom. It covers 7750 hectares and is roughly triangular in shape (Figure 1). This reserve (34°15'S; 18°25'E) contains many famous features such as the views from the rocky promontories of Cape Point and the Cape of Good Hope, which alone attracts many local and foreign tourists. This large influx of visitors to the reserve poses serious management problems. However the challenge is to utilize this large number of people to benefit conservation, and promote the region as an eco-tourism destination.

Topography and geology

The highest points of the reserve occur along the False-bay coast. It takes the form of steep hills with the highest point reaching up to 360 m at Paulsberg. The western slopes of these mountains fall to a level central plateau that stretches for 5-6 km westward. This area contains scattered low sandstone outcrops with a low rock cover. Here seepage zones are also prevalent. These zones occur where a band of tilted sandstone rises above the surface and dams soil water behind it. Drainage is poor on the plateau (altitude ca. 150m) and a few streams contribute to the marshy conditions here. Another interesting feature is Cirkelsvlei. This permanent water body lies between sandstone ridges on high ground west of the plateau (Taylor 1984), and has no rivers draining into or out of it, but is fed by springs from beneath. The main drainage system is the Krom River. This river joins the Klaasjagers River that flows past the ruins of the homestead at Theefontein, to form a lagoon on the west coast. Here waters are deflected southwards as a result of "beach drift" (Mabbutt 1952).

The geology of the Cape of Good Hope Nature Reserve is composed of quartzitic sandstone beds of the Table Mountain Group of the Cape Supergroup, resting on old Cape Granite. Both the sandstone and granite are very hard and erosion resistant substrates and has prevented the land from being consumed by tempestuous wind and waves very prevalent in the area. The Cape Supergroup is represented by the Graafwater and Peninsula Formations within the reserve (Fraser and Macmahon 1994). The former was laid down 450 million years ago and is composed of a narrow bed (up to 65 m deep) of medium grained sandstone and mudstone. The Peninsular Formation is composed of a large bed (up to 1200 m deep) of almost pure quartzite sandstones (Deacon *et al.* 1992). At the base of Cape Maclear and at isolated outcrops on Dias Beach sediments derived from the Graafwater Formation occur as a result of erosion. The Peninsula Formation forms the dominant surface rock over the entire reserve (Privett 1998).

At two places in the reserve outcrops of brown, red, or yellow colored rocks occur. These are made of ferricrete that is sandstone whose grains have been cemented together by iron to form hard pans (Opie 1967). This can be seen at "Rooi Hoogte".

The quaternary deposits in the reserve is represented by the older (siliceous) and younger (calcareous) sands. The younger dunes are still actively forming and can be seen along both coastlines e.g. Olifantsbos Point (Privett 1998). On the east coast the older calcrete hills and cliffs are found. These are projections of the seabed and consist of sand and shellgrit cemented together with calcium carbonate (Opie 1967).

Soil

Table Mountain Sandstone is a course rock and in the reserve the softer faces easily weather into shallow, grey, quartzitic, sandy soils. These soils also contain little organic material i.e. they are low in nutrients and tend to be well drained. These shallow, infertile soils occur mostly on the plateau, on the upper slopes and on the summits of mountains and hills. These areas are relatively acidic with a pH ranging from 4 to 5. Soils of the lower lying central plateau is poorly drained thus more organic material accumulates and gives it a dark brown to black color. This soil is derived from sandstone bedrock and has a high acidity (pH 3-4).

Soils of the raised beaches, limestone outcrops and ancient dunes have different chemical properties from the sandstone substrates. Coastal soils are of marine origin. They are derived from marine sand and often contain large shell fragments. These younger sands are deep and have a neutral to alkaline pH. The major fossil dune of the reserve bisects it, reaching from Buffels Bay to Papkuilskloof. Other dunes also occur at Olifantsbos and at "The Meadows".

Climate

As a result of the latitudinal position of the reserve a mediterranean-type climate with warm, dry summers and cold wet winters are experienced. This resembles other mediterranean-climate regions on similar latitudes like Chile, California and Southwestern Australia.

Wind plays an important role in the climate of the reserve. Cape Point is the most exposed spot on the coast of South Africa and receives strong winds for most of the year. Summer is characterized by high-pressure cells and southeasterly winds which carry little moisture. These winds of 15-40 km/h can last for longer than a week at a time and often reaches gale force of up to 102 km/h. In winter the reserve is dominated by a succession of cold fronts which strike as northwesterly winds, with regular rains which follow. These winds are of shorter duration than the southeasterly winds and are not known to form dunes. They do however have a great effect on the coastal vegetation. This vegetation, which is often subjected to severe wind and storm, show browning of the growing tips which is most likely the result of wind borne salt spray (Taylor 1984b).

The southwestern Cape experiences a true Mediterranean-type climate receiving over 70% rain during winter (April-September) and less than 10% of the annual rain between November and January (summer) (Fuggle 1981). The annual rainfall averages 700 mm per annum near the mountains, but at Cape Point the rain is relatively low (402 mm per year) but higher at Klaasjagersberg (565 mm), which is further north in the Cape of Good Hope Nature Reserve.

Vegetation

Acocks (1975), in his map of the Veld Types of South Africa, depicted the greater part of the vegetation of the reserve as Macchia with a narrow strip of strandveld running down the west coast. Numerous others have described the vegetation of the Cape of Good Hope Nature Reserve since then. Besides the documentation of alien invasive species in the reserve (Taylor *et al*. 1985), the natural vegetation here has also been well documented (Fraser and MacMahon 1995, Privett 1998 and 2001) and a comprehensive species list of

1073 vascular plant species has been compiled (Taylor 1969, Taylor 1983, Taylor 1984a 1984b). Eleven species are reported to be endemic to the region (Privett 1998).

Taylor (1984b) divides the vegetation of the reserve into two structural formations i.e. Fynbos and broad-leaved thicket, with fynbos occupying 97 % of the area of the reserve. Fynbos occurs on sites exposed to fire whereas thicket occurs where fire is excluded. This latter vegetation type is a floristically impoverished form of the coast forests and thickets of the Knysna region.

Fynbos have been further broken down into 12 vegetation types by Cowling *et al.* 1996. Here communities were categorized on the basis of structural features and dominant species, following Campbell's (1986) system of classification. Privett (1998) used a combination of maps published by Taylor 1984(b) and Cowling *et al.* 1996 as well as further reconnaissance mapping in the field to produce a map with 11 vegetation categories for the Cape of Good Hope Nature Reserve.

1.4.2. The history of environmental impacts at the Cape of Good Hope Nature Reserve

Initial settlement by humans and the resulting impacts on the environment.

Prior to the establishment and subsequent management of alien plant invaders in the Cape of Good Hope Nature Reserve, this area was not pristine. Humans have been influencing the natural vegetation of the fynbos biome from about half a million years ago (Deacon 1992). Accounts of human occupation and their resultant impacts on the vegetation of the reserve has been well documented (Opie 1967, Taylor 1969, Hallinan 1992). The following review emphasizes activities that may have influenced the vegetation in the reserve.

Prehistory

Evidence shows that the southern Cape Peninsula was home to the stone age people from at least 200 000 years ago (Table 2). The stone-age people were habitat generalists and able to adapt and exploit a range of habitats. Taylor (1969) details some of the shelters within the Cape of Good Hope Nature Reserve that housed primitive people. He also, discusses the variety of foods such as carbohydrates (such as geophytes) and proteins that formed the major components of the diet of man.

Fire was used regularly from the middle stone age onwards, although it may have been used earlier (Deacon 1986). In fact (Deacon 1992) implied that fire was used as a management tool to harvest geophytes, with fire increasing the productivity of these plants. Other resources that were consumed include seafood such as fish, shellfish, marine birds and marine mammals. Evidence for this was found at middle stone-age sites in the southern Cape (Deacon 1986) as well as below Bonteberg in the Cape of Good Hope Nature Reserve (Maggs and Speed 1967).

Hunter-gatherers occupied the fynbos biome from 21000 years ago. They were part of a pan-bushman lifestyle shared by others in southern Africa (Deacon 1992). Archaeological sites from 10 000 years ago, suggests that hunter-gatherers were territorial patch foragers, hunting zebra, wildebeest and eland. It has also been suggested that they used fire-stick farming to encourage geophytes as well as using this technique to capture small animals such as antelope (Deacon 1992).

Early evidence for a pastoral lifestyle by the Khoi-khoi in the southern Cape dates back to 2000-1600BP (Sealy and Yates 1994, Smith 1992a, Schweitzer and Scott 1973, Schweitzer 1979). During this period sheep bones as well as pottery remains were found at the site Die Kelders near Gansbaai. Cattle remains only appear around 1300BP in the records for the Western Cape (Smith 1992b). The major impact of this mode of life on the fynbos biome was decreased intervals between fires relative to the natural fire regime and an increase in soil erosion due to an alteration in the vegetation cover (Deacon 1992).

Colonial Settlement and the resultant impact on vegetation inside the Cape of Good Hope Nature Reserve

In 1652 the Dutch East India Company established a permanent settlement in Cape Town. The main aim of the company was to service travellers on the East Indian trade route. Fresh produce was grown, livestock bartered from the Khoi-khoi, and supplies of water, wood and game were available for trade (Deacon 1992).

A century later Simons Bay was developed as an alternative to Table Bay due to the fact that the latter port was very exposed to northwest gales that caused a number of shipping disasters. The establishment of Simons Town thus encouraged settlement of free burgers in the southern Cape Peninsula and started the colonization of the Cape of Good Hope Nature Reserve by Europeans. The first man to be granted land was Jurgen Schuster in 1738. It is believed that his homestead was located just outside of the Cape of Good Hope Nature Reserve, while his grazing rights stretched from below Red Hill all the way to Schusterskraal on the West Coast. By 1786 three other families were granted grazing rights within the Cape of Good Hope Nature Reserve. These areas include Schusters Kraal, Olifantsbosch Bay and at the top of the Buffels River Valley in the area of the farm "Buffelsfontein" (Hallinan 1992).

Following the second British occupation of the Cape in 1806 there was an influx of settlers into the area of the reserve (Hallinan 1992). From about 1810 land was granted to farmers on a quitrent basis. This meant that subject to certain conditions, ownership of land was complete and could be passed on from generation to generation (Hallinan 1992, Taylor 1969).

Most landholders appeared to derive merely a subsistence livelihood from their land. Vegetable farming was one of these subsistence activities, which took place mainly on the alluvial soils along the Klaasjagers River and at Olifantsbos due to the permanent water supply (Taylor 1969, Hallinan 1992). The main crops that were grown include potatoes, tomatoes, and green vegetables. Barley was also grown as feed for stock (Taylor 1969). While vegetables were grown on the arable land adjacent to the river, the surrounding fynbos hills were utilized for grazing. Other regions used for grazing include the Schusters and Krom River valleys and the plateau south of Smith's farm. These regions bear evidence of considerable disturbance (Taylor 1969). From 1855-1885 the southern portion of the reserve up to Cape Point and including the farm Buffelsfontein was used to breed cattle, horses, pigs and ostriches (Hallinan 1992).

In the early 1900's many southern farmers abandoned their land or used it sporadically for seasonal grazing, as they were unable to compete with large-scale commercial agriculture that was developing in the surrounds. In the 1930's a few hundred cattle still grazed the abandoned farms (Taylor 1969).

Table 2. General chronology of hominids in South Africa with particular reference to the Cape Floral Kingdom biome before European colonisation. Data comes from Taylor (1969), Deacon (1992), and Hallinan (1992).

Date	Period	Main Impacts on Vegetation
1.5 myr-150 000 BP	Earlier stone age	The beginning of stone tool making. Tools include stone axes and cleavers. People were habitat generalists and opportunists. Thus, were able to expand their habitat range. Minimal impact due to low population numbers.
125 000 – 60 000 BP	Middle stone age	There was a change in the range of stone tools (flake-blades instead of stone axes). Klaasies River site on the Tsitsikamma coast provided evidence for the evolution of modern man (*Homo sapiens*). Clear evidence for the use of fire. Geophytes were also harvested for their carbohydrate content and fire was used to increase the productivity of the bulbs. Clear evidence that people exploited seafood such as fish, shellfish, marine birds and marine mammals as well as terrestrial resources which were found at coastal sites in the Southern Cape. Low impact on the vegetation as a result of small population sizes and unsophisticated technology and hunting techniques.
40 000 – 2000 BP	Later stone age	Change in the stone tools. Artifacts include parts of bows and arrows, bored stones, ostrich eggshell water containers with decorations and painted beads and engraved stones. Hunter-gatherers appeared from 21000 BP, and their impacts on vegetation were mainly due to the use of fire to encourage geophytes
2000 BP – historic times	Early pastoral	The introduction of domestic animals to South Africa by the stone age people. Evidence for a pastoral lifestyle by the Khoi-khoi (known to Europeans as "Hottentots"). Burning of vegetation may have lead to improved grazing for livestock.
1652 AD –present	colonization	Widespread landscape transformation

Agriculture and grazing were but two factors that had an impact on the natural vegetation. After the route to the Cape was pioneered, many more ships than ever before rounded the Cape. Shipwrecks were a common feature at the time. Hallinan (1992) details the location of these various ships, with the greatest concentration being around the Olifantsbosch Point. These stranded ships often carried valuable merchandise that had to be salvaged. These operations were time consuming often lasting for many months. Temporary dwellings or huts were erected and draught animals concentrated in certain areas, thus destroying the local vegetation through overgrazing and trampling (Taylor 1969). The Cape of Good Hope Nature Reserve was also a site for military activity. Between the world wars regular military training exercises such as artillery practices were held in the area of the reserve. Initially tented camps for personnel were established between Theeberg and the Klaasjagers River and later a permanent barracks was established at the present Klaasjagersberg administrative offices (Hallinan 1992). These military operations would have lead to an increase in fires, as well as other disturbances such as trampling and grazing by horses.

1.5. The alien plant invaders within the reserve

Alien plant invaders has been a long-standing problem within the Cape of Good Hope Nature Reserve (Taylor and Macdonald 1985, Taylor *et al.* 1985, Macdonald *et al.* 1989). These woody trees and shrubs were both intentionally and unintentionally introduced (Macdonald *et al.* 1987) with the most important invaders being (in order of importance): *Acacia cyclops*, *Acacia saligna, Acacia longifolia*, *Pinus pinaster* and *Eucalyptus lehmannii*.

Many of the early farmers in the reserve introduced these aliens deliberately. Acacia species were planted as windbreaks around farmlands, as shade trees near homesteads and also as sand stabilizers of coastal dunes (Taylor 1984b, Macdonald *et al.* 1989). At the end of the 1800's and the early 1900's small hold farming was no longer viable and farmers abandoned their land. These woody species then proliferated and formed dense impenetrable thickets, aided by fire and spread unchecked into the adjoining veld (Taylor 1984b).

Currently the only invader posing a problem to indigenous vegetation in the Cape of Good Hope Nature Reserve is *Acacia cyclops*. This species which has reached a dense cover, occur on steep slopes in the extreme south west of the reserve and is included in future alien eradication plans for the reserve (Privett 2001). Other alien invaders such as pines, eucalypts, *Acacia saligna*, *Acacia longifolia*, *Leptospermum laevigatum* have all been brought under control with only occasional juveniles now present.

1.6. **Site Selection**

Three localities were selected (Figure 1) to encompass areas supporting dense stands (> 70% cover) of invasive alien species, that had been exposed to different alien clearance methods. In order to determine which treatments to study or where to set up plots, a thorough review of the three localities was conducted. Information was obtained from publications relating to the area, scrutiny of the relevant rangers reports (dating back to 30 years), consultations with managers and field visits. Thereafter a database was drawn up of the various alien plant control methods employed and this information was then used to develop maps using a Geographical Information System (Arcview 3.1). These maps (Figure 1.2 through 1.13) were used to identify various treatments to study within each locality, with each cell in the maps representing an area of 250m x 250m. Thus, ten study sites were identified with final site selection being on the basis of the frequency of control methods, as well as on a suitable combination of methods. Table 3 details the control methods employed in the last thirty years in the sites itself. If it was not certain whether alien invasive control methods occurred in the study site, it was not included in the table. Three control sites were also chosen (Table 4), after consultations with managers, in areas that had no history of dense alien plant infestations.

It is important to mention that the maps have limitations (See chapter 4) due to the nature of record keeping by the reserve management. Initial alien clearing methods in the reserve was sporadic and not well documented. From 1974-1989 records simply indicate in which areas invasive alien clearing operations took place without an indication of size. There was no accurate grid system indicating exactly where control operations took place until mid 1989. Therefore, pre 1990 information (pertaining to alien clearing operations) that was used in the mapping process was an estimate of where alien clearing may have taken place. In some instances, the entire locality was indicated to have been subjected to alien plant clearance. Thus many cells in Figures 1.2 through Figure 1.13 are an overestimate of the number of

Table 3. Localities, sites and treatment details for 10 study sites in the Cape of Good Hope Nature Reserve. At each site 5 plots of $50m^2$ were established.

	Localities									
	Klaasjagersberg				Meadows		Platboom			
Sites	K1a	K1b	K2	K3: Control	M5	M4: Control	P6a	P6b	P7	P8: Control
1963	Fire	Fire	Fire	Fire	~	~	~	~	~	~
1986	Fire	Fire	Fire	Fire	~	~	~	~	~	~
1986 (February-March)	Bulldozing	Bulldozing	~	~	~	~	~	~	~	~
1986 (October)	Aerial-spraying	Aerial-spraying	Aerial-spraying	~	~	~	~	~	~	~
1988	Foliar-spraying	Foliar-spraying	Foliar-spraying							
1990	Weeding + herbicide	Weeding + herbicide	~	~	~	~	~	~	~	~
1991 (March)	~	~	~	~	~	~	Fire	Fire	Fire	Fire
1991	Weeding	Weeding	Weeding	~	Weeding	Weeding				
1991	Foliar spraying	Foliar spraying	Weeding + herbicide	~	~	~	~	~	~	~
1992 (April)	~	~	~	~	Fire	Fire	~	~	~	~
1992 (September)	~	~	~	~	Burnt heaps	~	~	~	~	~
1993 (February)	~	~	~	~	~	~	Boom-spraying	Boom-spraying	Foliar-spraying	~
1993	Weeding	Weeding	Weeding	~	~	~	~	~	~	Weeding
1993	Weeding + herbicide	Weeding + herbicide	Weeding + herbicide	~	~	~	~	~	~	Weeding + herbicide
1993 (September)	~	~	~	~	Foliar-spraying	~	~	~	~	~
1994	~	~	~	~	~	~	~	~	~	Weeding
1995	Weeding + herbicide	Weeding + herbicide	Weeding + herbicide	~	~	~	Foliar-spraying	Foliar-spraying	~	~
1995	~	~	Weeding	~	~	~	~	~	~	~
1996	~	~	~	~	~	~	Weeding	Weeding	Weeding	Weeding
1997	Weeding	Weeding	Weeding	Weeding	Weeding	Weeding	~	~	~	~

Most records prior to 1990 (which were questionable, in terms of exact location) were not reflected in the table, although they were included in the maps. This was largely due to the fact that they occurred in the broader locality, but may not have occurred in the immediate vicinity of the study sites.

times an area has been subjected to alien clearing. From 1990 however, reserve management records for invasive alien control methods were much more efficiently documented. Rangers used a 500m x 500m grid system for the Cape of Good Hope Nature Reserve, to indicate where control operations took place and these could then be accurately reflected in the maps.

Table 4. Biophysical Information of three control sites in the Cape of Good Hope Nature Reserve

Sites	Vegetation post-fire age	Altitude (m)	Dominant indigenous plant species	Slope	Aspect
Klaasjagersberg (K3)	14	31	*Erica subdivaricata*	0°	Flat
Meadows (M4)	8	85	*Leucadendron coniferum*	5°	SE
Platboom (P8)	9	100	*Metalasia muricata*	0°	Flat

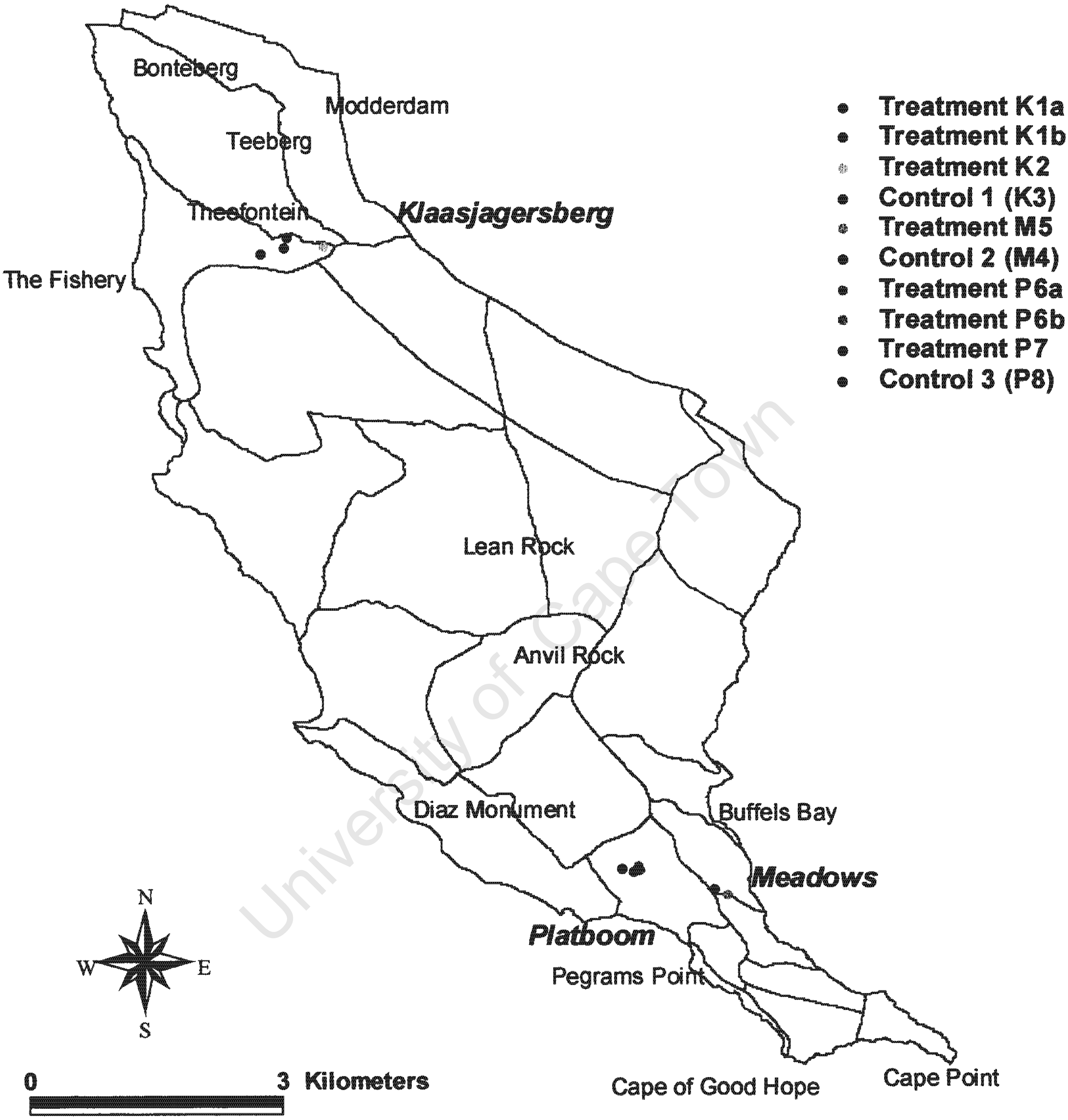


Figure 1.1. Map of the Cape of Good Hope Nature Reserve Section of the Cape Peninsula National Park. The coloured dots represent the exact position of study plots.

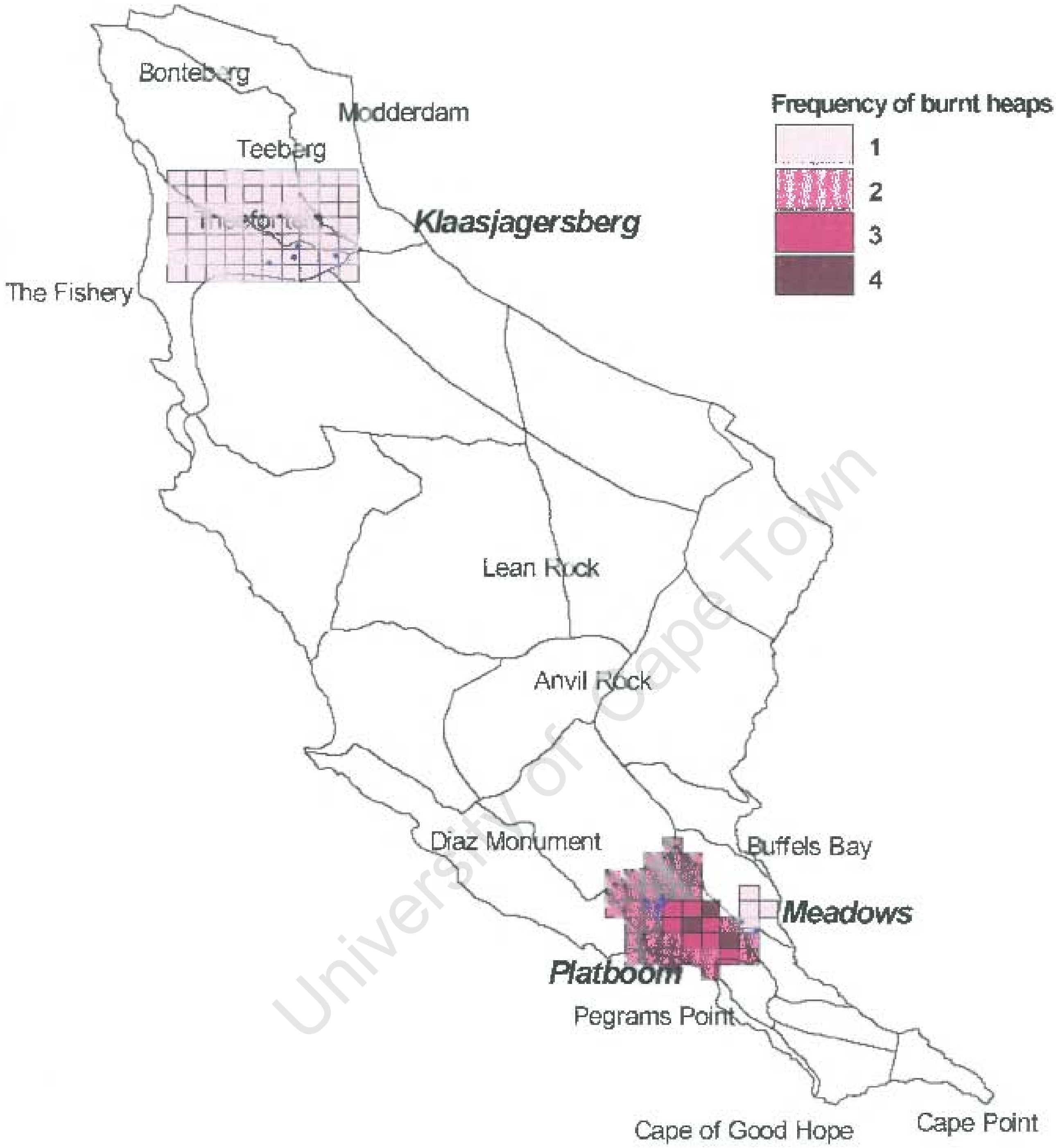


Figure 1.2. Frequency of burnt heaps in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

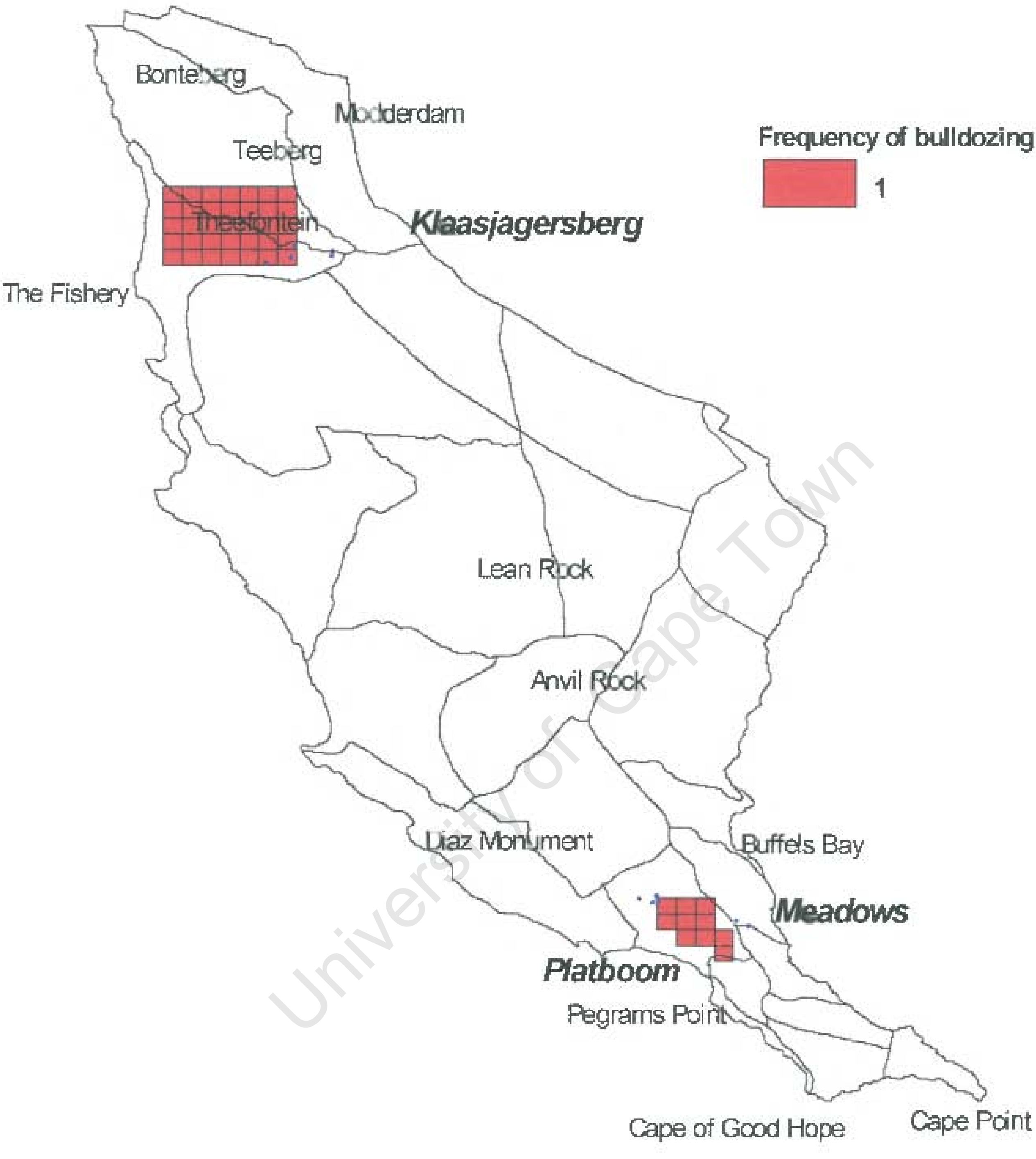


Figure 1.3. Frequency of bulldozing in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m. No cell experienced bulldozing more than once in the past thirty years, and although it may seem that bulldozing occurred in treatment P6a, plots did not overlap with this treatment.

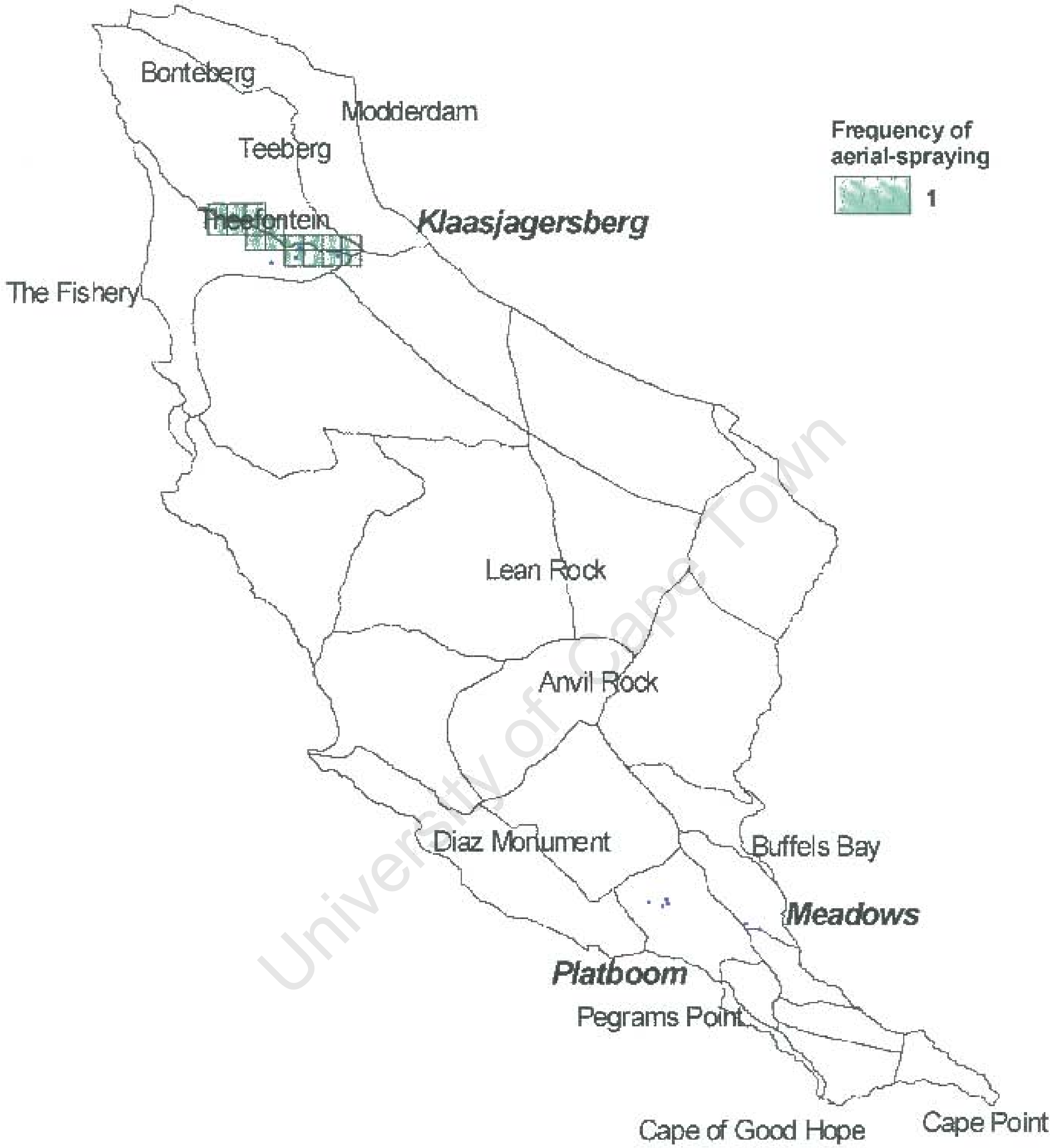


Figure 1.4. Frequency of "aerial spraying" invasive alien plants in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. "Aerial-spraying" was done using aircraft crop dusters and the herbicide Triclopyr was used. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m. No cell experienced "aerial-spraying" more than once in the past thirty years.

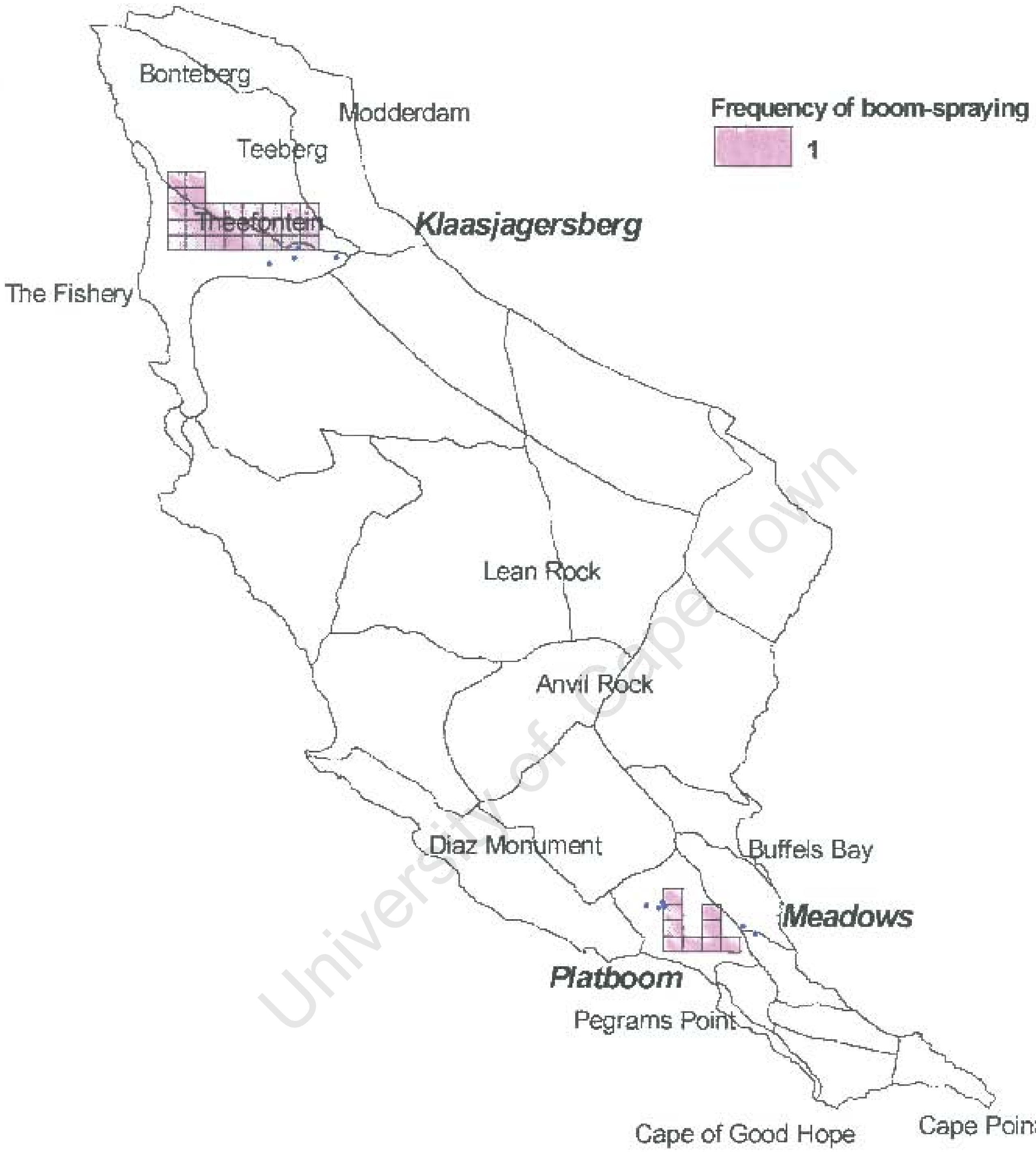


Figure 1.5. Frequency of "boom-spraying" in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m. No cell experienced "boom-spraying" more than once in the past thirty years

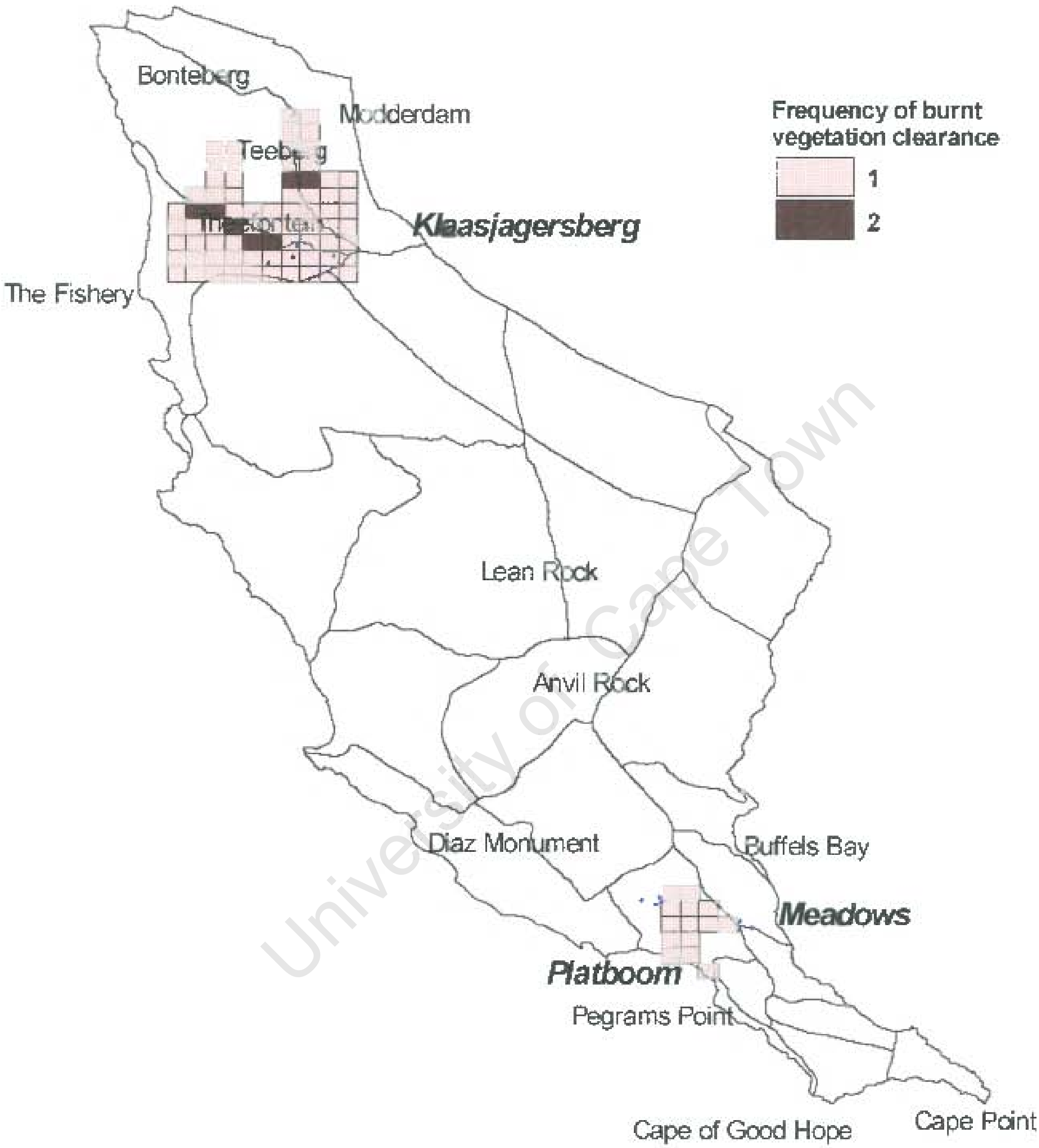


Figure 1.6. Frequency of the occurrence of clearing alien vegetation after fire in 250m x 250m grid cells over the past thirty years, in the three localities at Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading.

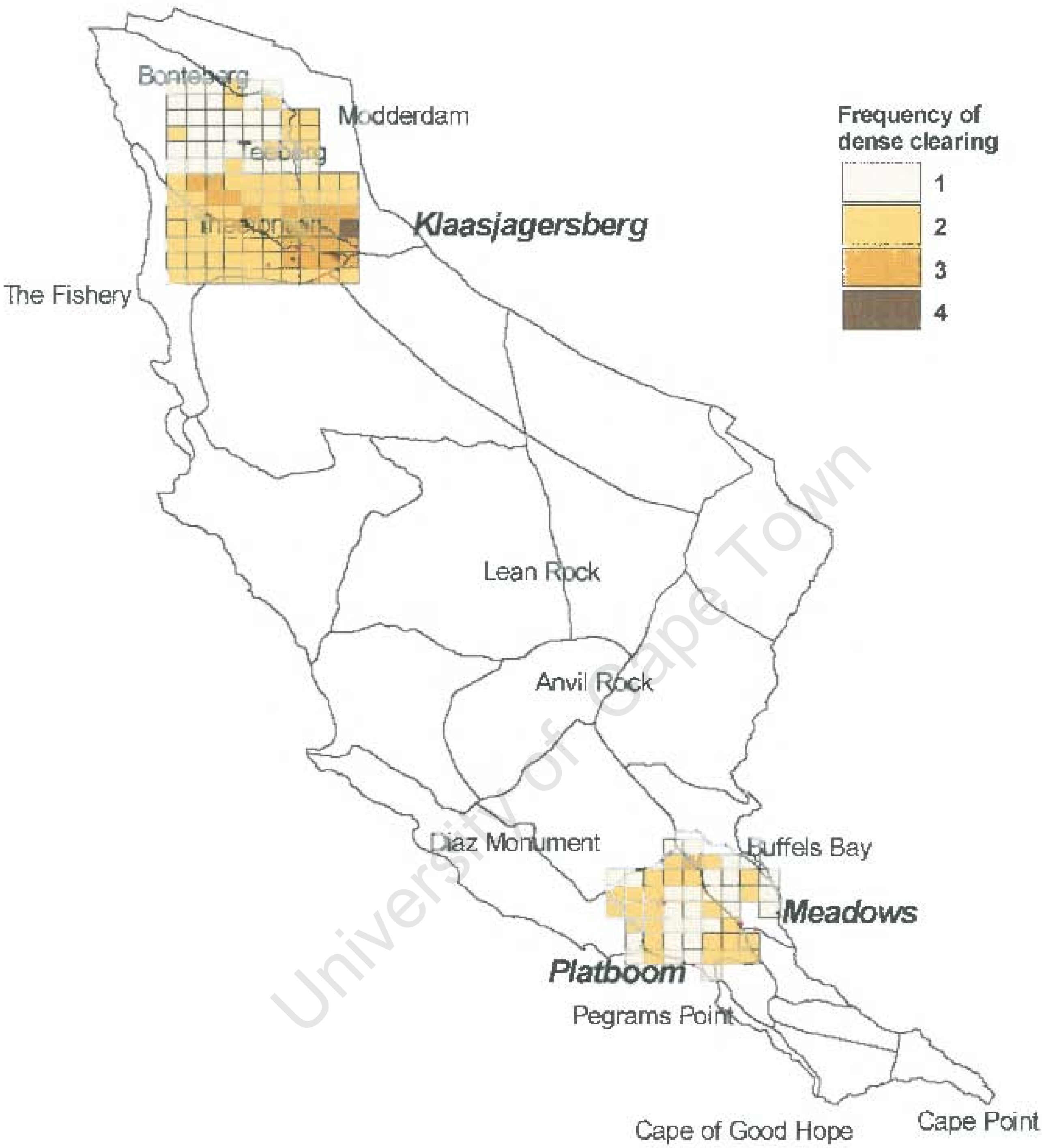


Figure 1.7. Frequency of clearing dense alien infestations in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. Dense clearing entailed the use of chainsaws to reduce infestations. The red dots represent the study plots shown in figure 1.1. Red shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

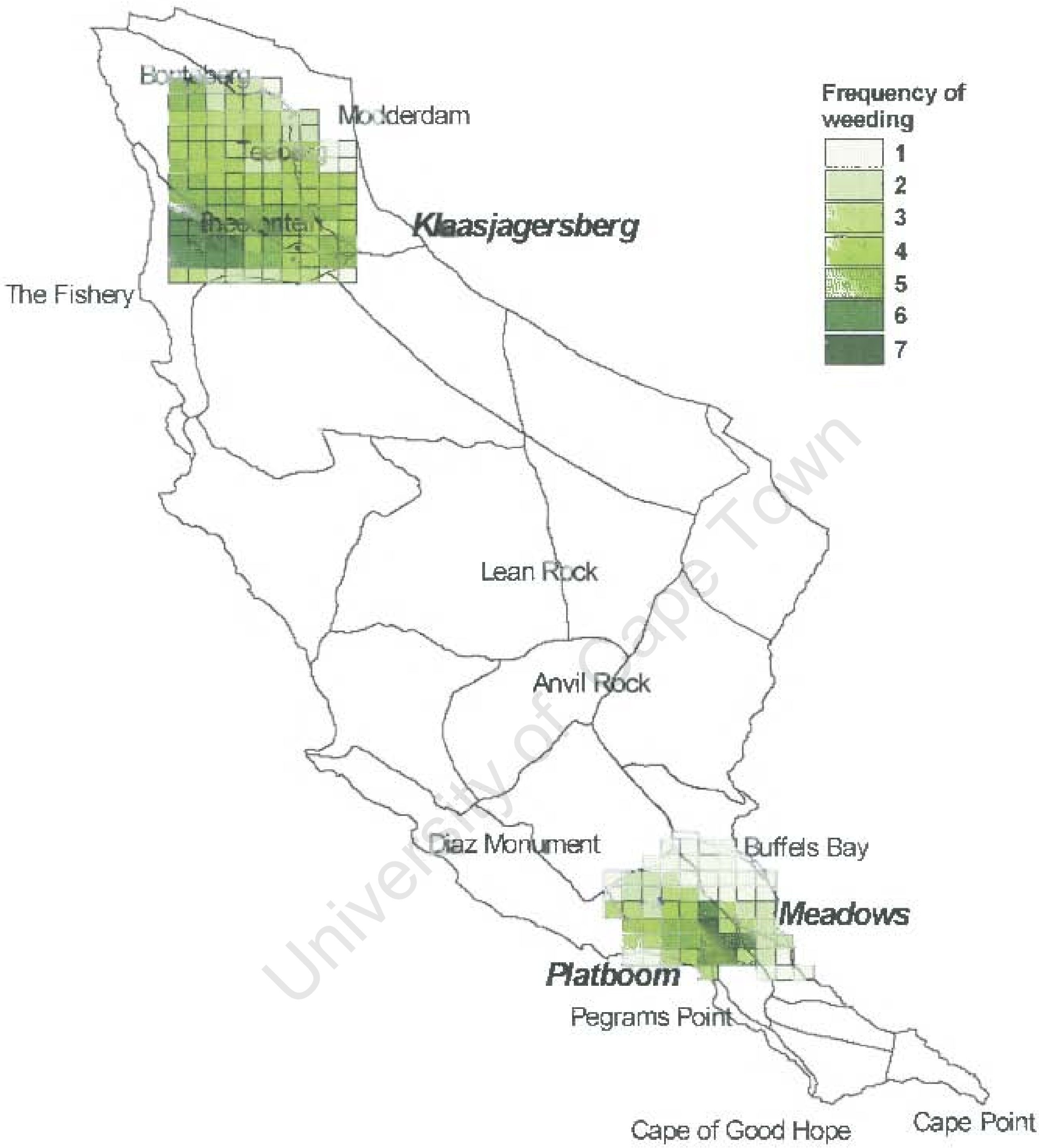


Figure 1.8. Frequency of weeding in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. Weeding mostly entailed hand-pulling and slashing of alien vegetation. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

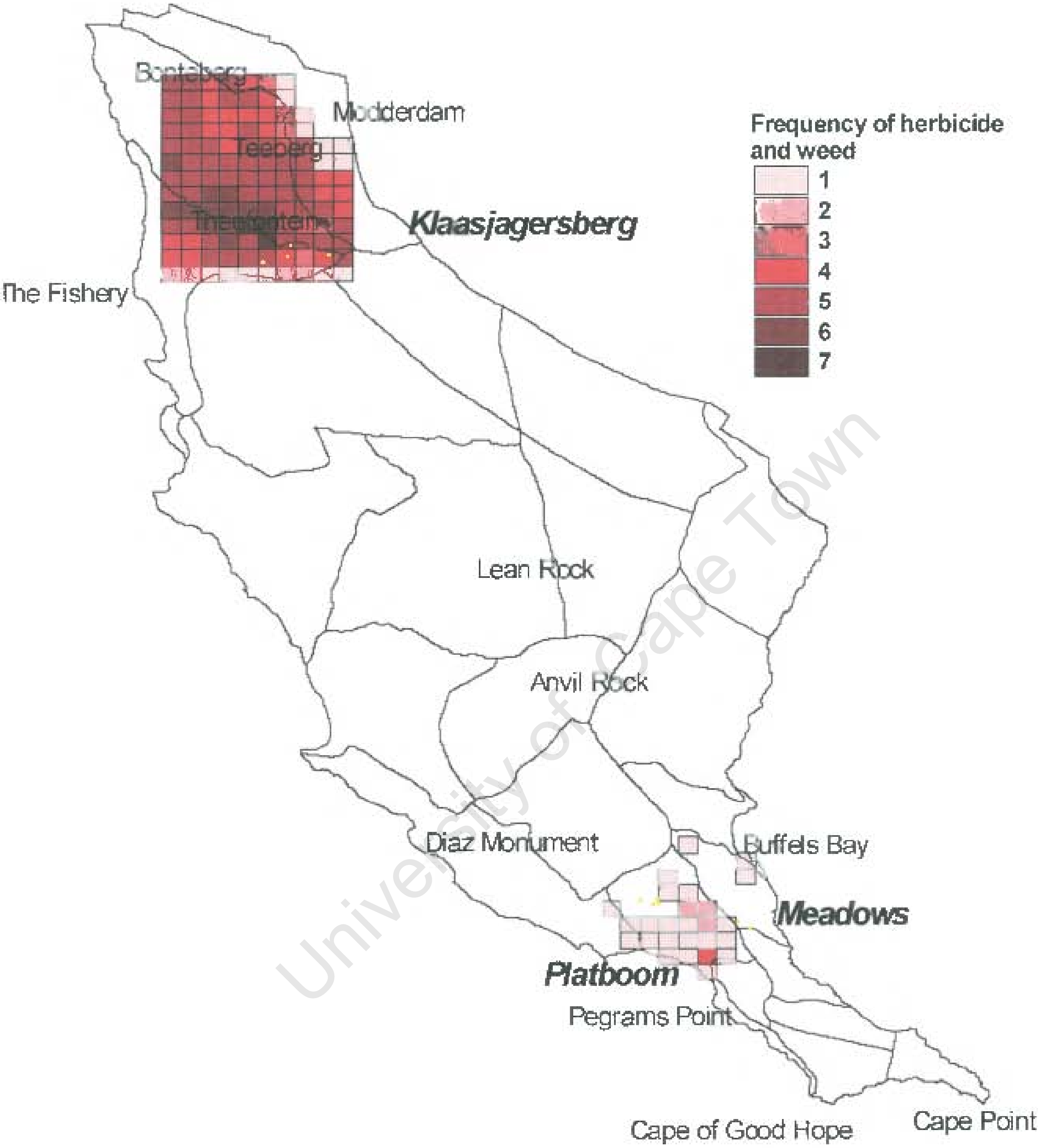


Figure 1.9. Frequency of herbicide and weeding in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. This method entailed the alien vegetation being slashed and treated with herbicide like Triclopyr. The yellow dots represent the study plots shown in figure 1.1. Yellow shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

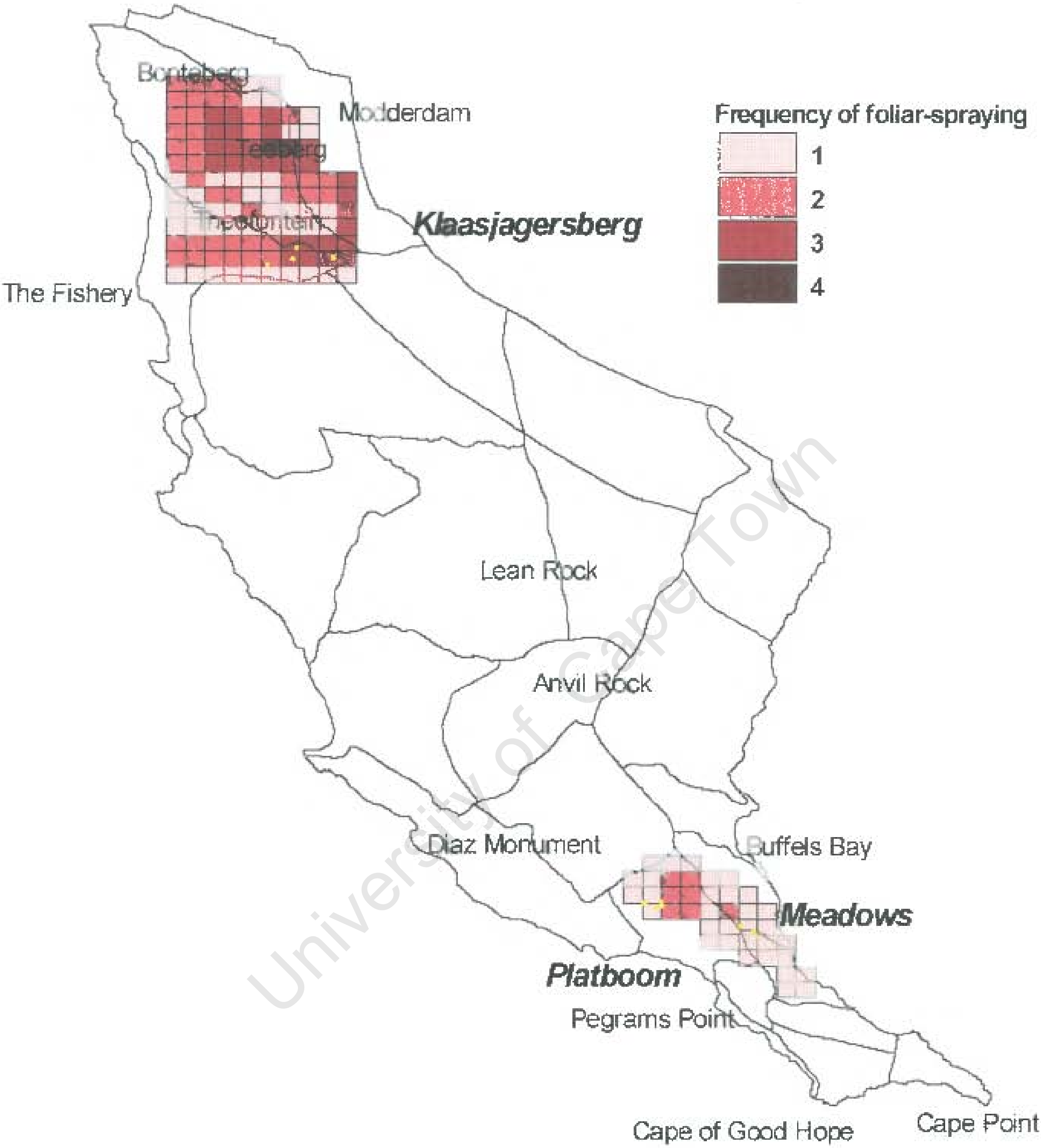


Figure 1.10. Frequency of foliar-spraying in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. Foliar spraying basically involved knapsack spraying by labourers, directly on the alien vegetation. The yellow dots represent the study plots shown in figure 1.1. Yellow shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

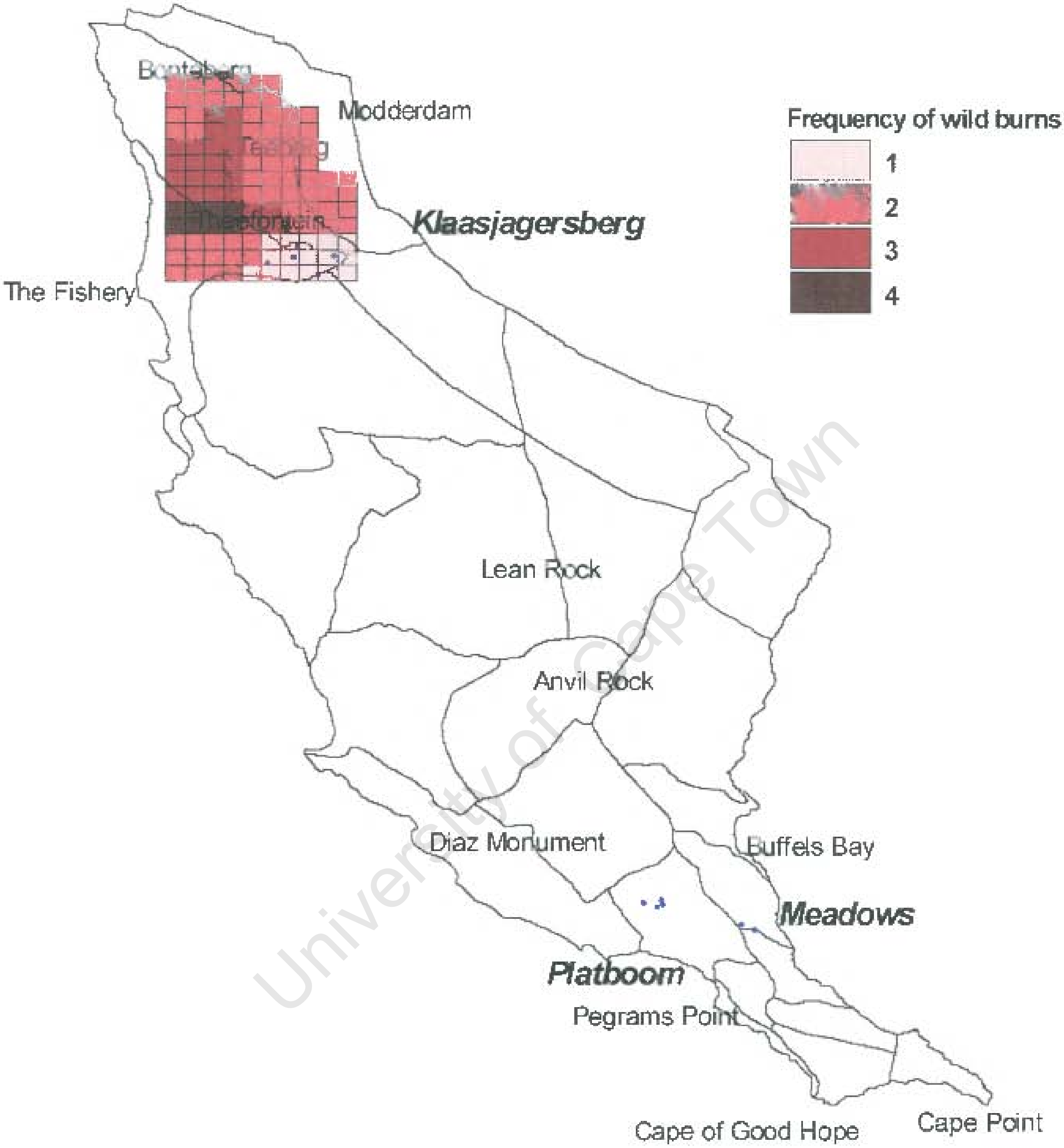


Figure 1.11. Frequency of wild burns in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

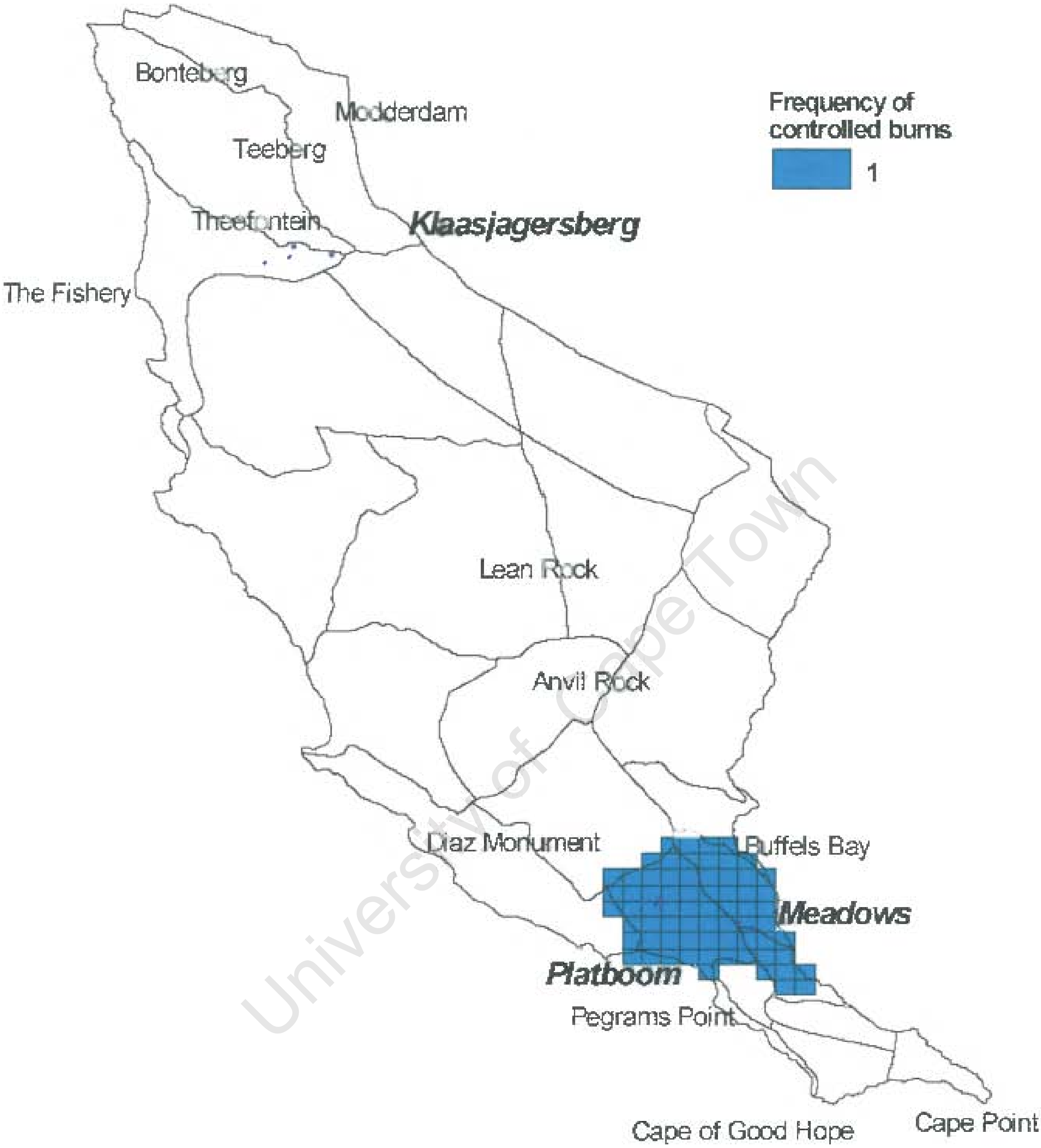


Figure 1.12. Frequency of controlled burns in the past thirty years, in the three localities at the Cape of Good Hope Nature Reserve. The blue dots represent the study plots shown in figure 1.1. Blue shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m. No cell experienced a control burn more than once in the past thirty years

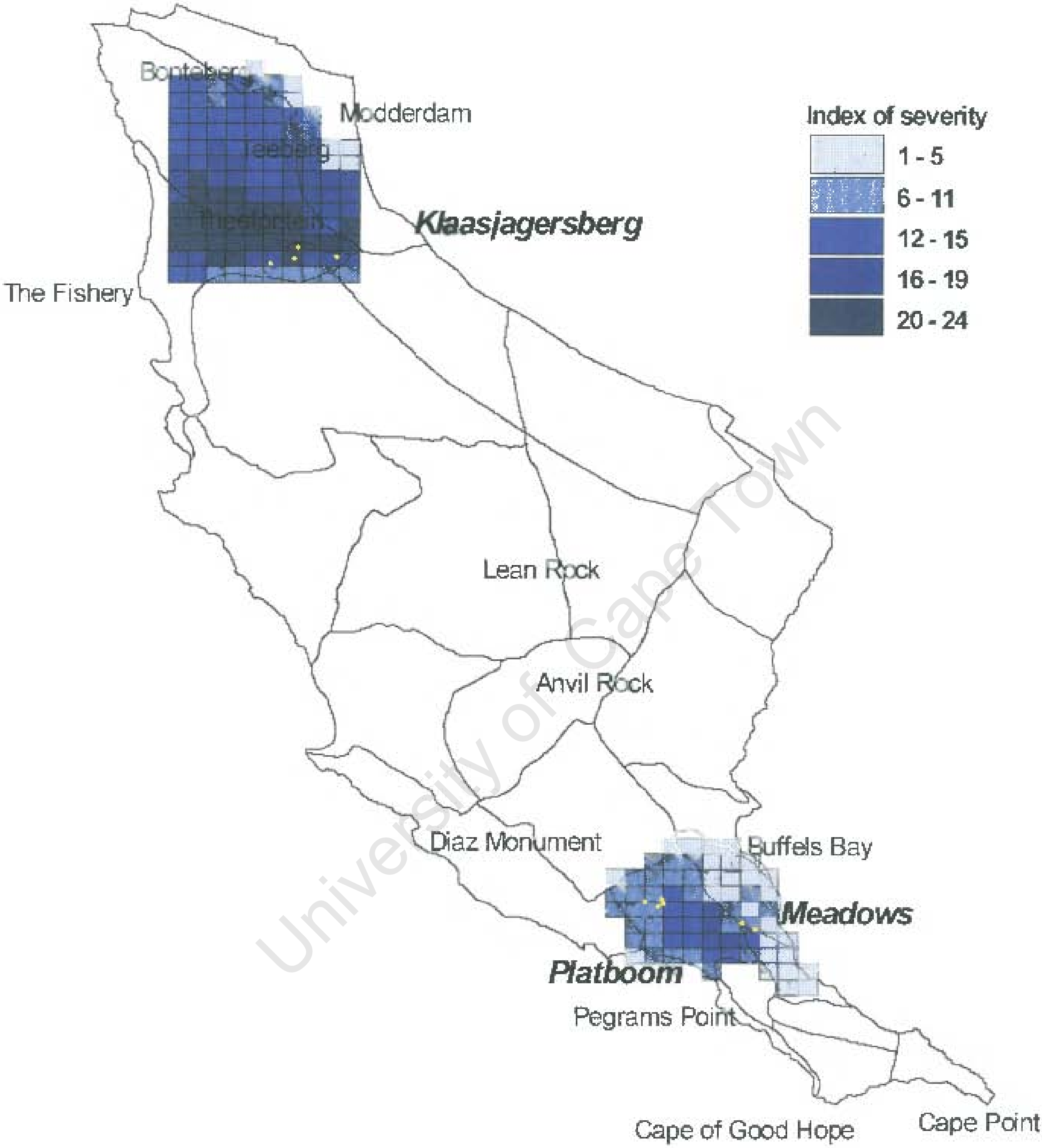


Figure 1.13. The frequency of all control methods applied, for the past thirty years in the three localities, at the Cape of Good Hope Nature Reserve. The yellow dots represent the study plots shown in figure 1.1. Yellow shading has been used for clarity, as opposed to using individual colour shading. Each grid cell has an area of 250m x 250m.

1.7. Locality details

Locality 1: Klaasjagersberg

An intense wild fire passed through the densely alien infested Klaasjagersberg region in February 1986. The dominant alien tree/shrub species was *Acacia saligna, and Acacia longifolia*, *Acacia cyclops, Pinus pinaster* and *Eucalyptus lehmanni* were also present. This vegetation had formed an impenetrable thicket, and was burnt prior to 1986, in 1963. The vegetation following the fire was burnt standing in all sites in this locality.

In the "bulldozed and aerial-sprayed" treatment site (Figure 1.14) the standing burnt vegetation was bulldozed (February-March 1986). This was done owing to the high density of the stumps, to make follow-up treatment simpler and more cost-effective. Officials also decided that in order to minimize the impact of the bulldozing treatment, the blade of the bulldozer should be kept approximately half a metre above the ground so that the soil would not be disturbed more than necessary.

In the "aerial-sprayed" treatment site (Figure 1.15), bulldozers were not used due to the nature of the terrain. Here the soil was too moist for such heavy machinery, but both Klaasjagersberg treatment sites had aerial application of the herbicide Triclopyr (Garlon 4) onto alien regrowth which was carried out using aircraft (crop dusters). This aerial application took place in October 1986.

Figure 1.14. Vegetation structure in some of the quadrats in the **"bulldozed and aerial-sprayed" treatment** (site K1a and K1b in Table 3). These quadrats contain predominantly grasses such as *Cynodon dactylon* and *Stenotaphrum secundatum* (Photograph taken in March 1999).

Figure 1.15. Vegetation structure of the **"aerial-sprayed"** quadrats (site K2 in Table 3). There is a higher cover of restioids and grasses in this site than in control quadrats. The restioid component of the vegetation can be seen in the foreground. (Photograph taken in March 1999).

The control site had no history of dense alien invasion, but burnt in the same fire. It was an area of natural vegetation and had similar soils to the treatment sites. The history of this area indicates that the bank of the Klaasjagersriver was utilized for farming practices, with the hills having been used for grazing. Hence, this site would most likely fall into this latter category.

The earlier history of this area has already been detailed in this chapter. After colonial settlement in the Cape, the Klaasjagersberg area on the banks of the river was used for farming. The ruin of the old homestead is still visible. As this land was arable, it implies that ploughing was also carried out here. Thus farming activities was carried out in both the "bulldozed and aerial-sprayed" treatment sites and the "aerial-sprayed" treatment. Lastly, the bulldozed area was also previously an old rubbish dump until about 1975 (R. Erntzen, personal communication).

Figure 1.16. Another weedy species of grass *Pentaschistus macrourum* occurs in abundance in the **"aerial-sprayed" treatment site** (site K2). Here it is seen in the foreground, with *Berzelia* sp. in the background. (Photograph taken in November 2000).

Figure 1.17. In the mature **Klaasjagersberg control** (site K3 in Table 3), the overstorey is primarily composed of *Leucadendron coniferum* and the understorey is made up of numerous ericoids and some graminoids. The photograph was taken in March 1999.

Locality 2: Meadows

This area was given a prescribed burn in April 1992. The area supported a forty-one year old dense (>70% projected canopy cover) *Acacia cyclops* infestation. Prior to the management burn, extensive brush piles had accumulated from past clearing operations. This site (Figure 1.18 and 1.19) was chosen to determine the impact of a burnt-standing treatment together with "foliar-spraying" as the follow-up (September 1993). In September 1992, stack burning was also conducted in this site (M5). This treatment was compared to an area of natural vegetation, on similar soils that burnt in the same fire, but with no history of dense alien invasion (Figure 1.20 and 1.21).

Figure 1.18. Vegetation structure of the **"foliar-sprayed"** region at the **Meadows** site (site M5 in Table 3). The vegetation is predominantly grass, and the slope (15°) of this site can be observed in this photograph. (Photograph taken in March 1999).

Figure 1.19. Some quadrats were exceptionally bare after the 1992 fire with **"foliar-spraying"** as follow-up (site M5), despite seven years of recovery time (Photograph taken in March 1999).

Figure 1.20. Vegetation structure of the **control site at the Meadows** (site M5 in Table 3), which had no alien infestation but had burnt in the same fire. This site has *Leucadendron coniferum* as the dominant species (Photograph taken in November 2000).

Figure 1.21. Vegetation at the **Meadows control site** (site M4), with *Leucadendron coniferum* in the background and *Passerina* in the foreground. (Photograph taken in November 2000)

Locality 3: Platboom

The dense forty-five-year-old *Acacia cyclops* stand was given a prescribed burn on 27th March 1991 (i.e. a "burnt standing" treatment). The alien plants recruiting after the fire was "boom-sprayed" in the one treatment site (P6a+P6b), and in the other treatment site (P7) the alien plants were "foliar-sprayed". In the former treatment (conducted in February 1993), the boom was a horizontal pipe containing numerous holes through which the herbicide Triclopyr was sprayed onto aliens from a truck (which drove through the vegetation) whereas foliar-spraying was more direct. Here labourers carrying knap-sacks containing the herbicide sprayed this onto aliens via hand-held mist-blowers (January to February 1993). Both these treatments were compared to an area of natural vegetation, on similar soils that burnt in the same fire but with no history of dense alien invasion.

Figure 1.22. The vegetation structure of the **"boom-sprayed" treatment site** (site P6a and 6b in Table 3). The structure looks very similar to that of the "foliar-sprayed" treatment at the Meadows.

Figure 1.23. The vegetation structure of the **"foliar-sprayed" treatment site at Platboom** (site P7 in Table 3). The "skeletons" of the alien *Acacia's* can be seen in the background. The dominant shrubs in this area are *Metalasia muricata* and *Erica gilva*. The photograph (taken in March 1999) clearly shows the higher diversity and richness of this treatment compared to the "boom-sprayed" treatment within the same locality.

Figure 1.24. Another example of the **high plant species richness and diversity** at the Platboom **"foliar-sprayed" treatment site.** The photograph (taken in March 1999) shows *Leucospermum conocarpodendron* in the background, with a very good representation of *Metalasia muricata* throughout.

Figure 1.25. Vegetation structure of the **Platboom control** site (site P8 in table 3). This vegetation comprises an array of restioids and ericoids. **It is the most species rich study site** ($50m^2$ plot scale).

The effects of different alien plant control methods on the recovery of fynbos vegetation at the Cape of Good Hope Nature Reserve

Introduction

Flora and vegetation

The Cape Floral Kingdom, well known for its exceptionally high levels of species diversity and endemism (Cowling and Holmes 1992, and Cowling *et al.* 1992, McDonald and Cowling 1995, Goldblatt and Manning 2000), is one of the world's 25 "priority biodiversity hotspots for conservation" (Myers *et al.* 2000). This area comprises vegetation types from the Fynbos, Succulent Karoo and Forest biomes (Low and Rebelo 1998), but fynbos shrublands contain about 70% of the Cape Floral kingdom's vascular plant species (Rebelo and Siegfried 1990). Approximately 2700 Red Data Book plant species occur in the Cape Floral Kingdom (Hilton-Taylor 1996), out of a total of approximately 9000 species (Goldblatt and Manning 2000), thus emphasizing the need to conserve this distinctive flora.

Invasive alien plants in fynbos

One of the main threats to biodiversity in the Cape flora is the spread of invasive alien plants, especially trees and shrubs (Richardson *et al.* 1992, 1996). The most widespread and damaging invasive alien plant species are trees and shrubs of the genera *Acacia*, *Hakea* and *Pinus*. These species are pre-adapted to local conditions, and have invaded large areas in both the mountains and the lowlands of the fynbos biome (Macdonald and Richardson 1986, Richardson *et al.* 1992). Once these species have colonized an area they grow faster and soon overtop the natural vegetation. All these invasive species recruit dense populations after fire, and a series of fires in a particular area often results in dense thickets of the alien species that transforms the vegetation from shrubland to woodland/forest (Richardson *et al.* 1989). Such invasions bring about many changes to ecosystems, including a marked change to the amount, properties and distribution of combustible biomass (van Wilgen and Richardson 1985), a change to the nutrient cycling patterns (Witkowski 1991, Yelenik 2000) and increased litterfall (Milton 1981). Thus, invasive alien species threaten both ecosystem processes and biodiversity over large parts of the Cape Floral Kingdom.

Invasive plant control

Managing fynbos for goals such as nature conservation, optimal water yield and reduced fire hazard centres on managing fire and controlling invasive alien plants. The need for management of these plants was recognized as early as the end of the 19th century (Stirton 1978), yet it is only in the last few decades that substantial progress has been made in research relevant to management (van Wilgen *et al.* 1992). Previously, the war against invaders was largely mechanical, involving the chopping out of vegetation. This was sporadic, with little follow-up taking place because of a lack of funding or the lack of knowledge of the target species (van Wilgen *et al.* 1997).

Recent innovations in invasive alien plant control have led to the integrated use of three types of control: biological, chemical and mechanical. In fynbos, the last-mentioned involves mechanical tools and/or fire i.e. cut and burn, cut and leave, and burn-standing (van Wilgen *et al.* 1992). Computer technology also has been used to simulate the growth, spread and water use of alien plants in fire-prone environments (Le Maitre *et al.* 1996) and to explore the implications of different management strategies (Higgins *et al.* 2000). These advances have resulted in better integration of the different available control measures to combat invasive alien trees in fynbos (van Wilgen *et al.* 2000).

The effectiveness of any control operation depends on the life-history characteristics of the target species. Certain options are more suitable or practical for a particular species than other options. For instance, in addition to producing copious amounts of seed, *Acacia saligna* is a vigorous resprouter and requires herbicide application in addition to mechanical measures to achieve control. More recently, however, biological control (using species-specific insects and diseases from the region of origin of the alien plant) has been used to control *A. saligna* (Morris 1997).

Fire is an essential part of controlling alien plants in fynbos (van Wilgen *et al.* 1992), but needs to be used cautiously because the felling of dense stands of alien trees results in a large accumulation of dead fuel near the soil surface. If fires occur in hot, dry conditions, fires can be extremely intense, which can cause further damage to the remaining native plants, seed banks and soil and cause severe erosion (Richardson and van Wilgen 1986, Breytenbach 1989, Holmes *et al.* 2000). Fire also stimulates the release of seeds by serotinous invaders (Lamont *et al.* 1991) and the germination of alien seeds present in the soil-stored seed bank. This results in mass recruitment of juveniles forming denser stands than the original parent thickets. This means that an additional intensive step in the control program is required (e.g. hand-pulling of seedlings or the application of herbicides).

Assessing the effectiveness of invasive plant control

Early efforts at alien control in the Western Cape were generally uncoordinated. This was largely due to poor knowledge of the ecology of invasive plant species, and because follow-up operations often were neglected (van Wilgen *et al.* 1992, 1997). Many parts of the fynbos have a long history of control efforts. Many attempts at control have exacerbated the problem and caused further damage to native communities. Unfortunately, few areas have adequate records of the control methods that were used, or the levels of success that were achieved in controlling the alien plants. Furthermore, there are no records that detail the type of impacts of different treatments on elements of the native communities. Consequently, managers in different areas tend to base plans for control on their personal knowledge and experience. To improve management of alien plants throughout the fynbos biome, managers need to learn from past experiences, compile general guidelines and effectively monitor their goals.

Since a key element of current alien management programs is the restoration of sites after removal of the invaders (Holmes and Richardson 1999), good information is needed on the extent to which the different control options may exacerbate site transformation. This is because the management itself has the capacity to alter the rate at which the site potentially may return to a condition approximating the pre-invasion state. Such insights are needed in order to know, among other things, the level of resources to allocate to proactive restoration measures, such as the introduction of keystone species to speed up succession. Thus, it is important to know what types of restoration practices should be applied with each type of control method in order to promote vegetation recovery.

Fynbos plant species are not distributed in a uniform or random fashion but are concentrated in small nodes (Kruger and Taylor 1979, Simmons and Cowling 1996). Such localized distribution patterns, and the high turnover of species, even over short distances, and also the large number of plant species within identifiable "functional groups" (Richardson *et al.* 1995) means that comparing the species composition of different sites is not very helpful when seeking indicators of the impact of a particular management treatment. The problem lies in determining whether any differences in species composition are "natural" (i.e. due to the normal pattern of species turnover), or due to the elimination or addition of species as a result of management. The approach taken in this study was to investigate changes to measures of biodiversity (i.e. abundance, species richness, diversity, evenness) and vegetation structure, rather than comparing species composition across sites.

This study reports on the recovery of fynbos vegetation following four alien clearance treatments applied to dense stands of invasive alien plants. All alien stands were burnt standing as the initial treatment, with first follow-up treatment as: 1) "bulldozing and aerial-spraying" 2) "aerial spraying" 3) "boom-spraying" and 4) "foliar-spraying". No scientific/published work has been conducted on the relative susceptibility of fynbos plant guilds to any of the treatments applied. Hence it was not feasible to draw detailed hypothesis relating the various guilds to the above treatments. This study was therefore largely exploratory in nature with the main objectives:

1) To determine whether the management practices employed have affected species richness, diversity, evenness, density and total cover.
2) To determine whether treatments have changed the guild structure of the vegetation.
3) To determine whether any guilds should be re-introduced into the community following alien clearance in order to promote the recovery of a structurally representative fynbos stand.

Materials and Methods

Information pertaining to the history of the sites can be found in chapter 1.

The vegetation survey was conducted during March 1999 and encompassed a total of 5 plots on each site (See Table 3 on page 26). The species lists of Taylor (1985) and Privett (1998) were used to aid with species identifications, which were confirmed with reference to herbarium specimens in the Guthrie Herbarium collection housed at the Bolus Herbarium, University of Cape Town. Only perennially identifiable species were included in the survey.

The following variables were measured within each plot:

a) *Ranked abundance of each species.* Instead of counting the number of each species, plants were assigned into one of five classes based on a simplified form of Acocks (1975) system of abundance symbols (Table 1). For some later analyses, each class was assigned its median value to give an estimate of plant density.

b) *Species richness.* This value was derived by assessing the number of perennial species occurring at 2 scales: $1m^2$ (quadrat placed at the centre of each plot) and $50m^2$ (the entire plot).

c) *Total estimated projected canopy cover* of the plot ($50m^2$).

d) *The percentage projected canopy cover of each species.* All species covering less than 1% of the plot were recorded as present and in later analyses were accorded the value of 0.1 % canopy cover.

Table. 1. Abundance categories used in the study.

Category	Condensed Acocks category	Number of individuals in plot
1	occasional	1 to 4
2	fairly frequent	5 to 10
3	frequent	11 to 50
4	common	51 to 100
5	abundant	>100

The diversity (H) and evenness (J) of plant communities were calculated using Shannon-Wienner formulae:

$$H = - \Sigma P_i \log P_i \qquad (1)$$

where P_i is the proportion of individuals in the community represented by species P, (because the number of individuals were not counted, the median value of the Condensed Acocks category was used),

$$J = H/H_{(max)} \qquad (2)$$

Where $H_{(max)}$ is the maximum diversity if all species in the community had equal numbers of individuals (Begon *et al.* 1995).

Both the diversity and evenness indices were used to explain community patterns. The diversity index was used, as it is more informative than species richness alone. This index also takes into account the abundance of individual species, as opposed to just count data as species richness does (Magurran 1991). The evenness index was used to express quantitatively whether certain species were more dominant, as this could then be related to the treatments.

e) *Soil pH and depth*. Three soil samples were collected at random within each plot, for the determination of soil pH (measured in 0.01 $mol.l^{-1}$ $CaCl_2$, using a WTW "Germany" pH meter). Means of the three values for each plot were used in analyses. Soil depth (cm) was determined by knocking a steel rod into the ground until it could penetrate the soil no further. This was replicated five times within each plot and the mean value was then derived.

<u>Data Analysis</u>

The effects of different clearance treatments on vegetation variables (see page 53 for variables) were compared using a one-way analysis of variance (ANOVA). In the analysis each treatment was compared to its respective control site. The ANOVA also tested whether the control sites differed to each other, and within-site treatment comparisons were also determined using this type of analysis. The richness and density (abundance) data were log transformed before the ANOVA's were conducted and the total canopy cover values were arc-sine transformed to meet the required assumptions of normality.

To determine whether environmental variables such as total bare ground, soil pH and soil depth differed significantly between treatments and respective controls, the non-parametric Mann Whitney U-test was used (Table 2).

To investigate whether guild structure remained constant after alien plant clearance, each indigenous species was assigned to a growth form, dispersal mode and regeneration mode category (See figure 2 for categorization information). Categories were based on information in Goldblatt and Manning (2000), van Wilgen and Forsyth (1992) and Trinder-Smith (1995). The null hypothesis that cover (a biomass surrogate) in each biological attribute class remained constant after different clearance treatments was investigated. Initially thirteen different variables were derived when species were assigned to biological categories. However, the "geophyte guild" was combined with the forb guild in this analysis, as there was only one geophyte species with the very low cover of 0.1 %. The data was arc-sine transformed before conducting an Analysis of Variance. This was done to determine whether treatments were in fact affecting the percentage cover of each variable. Before the analysis of variance of the twelve functional guilds, a normality test was conducted, and a resulting six variables had non-normal distributions. The categories with the skewed distribution patterns are shown in table 5.

Detrended correspondence analysis (DCA) (Gauch 1982) was used to assess patterns in functional group composition among the different treatments and controls. For this analysis the sum of the cover values for all species within the 13 functional groups were used. The data was detrended by segments, as the unimodal method of detrending was selected. This multivariate analysis was conducted using the program CANOCO for Windows (ter Braak and Smilauer 1998).

Results

Environmental variables

There was no significant difference in the percentage of bare ground between the treatment and control plots (Table 2). The "foliar-sprayed" treatment at the Meadows site had the highest percentage bare ground, but here the standard deviation was also quite high, which could largely have been attributed to one plot that was extremely bare.

When soil pH was analysed using the Mann-Whitney U-Test, two observations were made: 1) the pH differed among sites, with Klaasjagersberg and Platboom occurring on acid soils (pH < 5), and the Meadows on alkaline soil (pH > 7). 2) The soil pH was significantly higher for the "foliar-sprayed" treatment at the Meadows compared to its control, and was significantly higher between one of the "boom-sprayed" treatments and the Platboom control (Table 2).

There was no consistent pattern or trend in soil depth between the treatments and controls. Although the depth of the soil at the bulldozed site was not as deep as the control, it was not significantly different.

Plant variables

Plant density

Plant densities were high at the Meadows and Platboom control sites (Figure 1a), as well as in the "bulldozed and aerial-sprayed" treatment and the "aerial-sprayed" treatment (Table 3). The density of plants was significantly lower in the "foliar-sprayed" treatment at the Meadows (P=0.05), but the same did not apply for the "foliar-sprayed" treatment at Platboom. No other control versus treatment comparisons, or within-site treatment comparisons differed significantly for this variable (Table 4a).

Total canopy cover

There was no significant difference in the total canopy cover between the treatments and the control plots (Table 4a). Within the "bulldozed and aerial-sprayed" treatment, as well as the "aerial-sprayed" treatment the high total cover (Table 3) can be attributed to two species of grass (Appendix 1) rather than a range of species or growth forms. The same applies to the "foliar- sprayed" treatment at the Meadows and the "boom-sprayed" treatment. The high total percentage cover values at these treatment sites largely reflect the dominance of one species (Appendix 1).

Species richness

The highest species richness of 23.6 species occurred in the Platboom control site at the $50m^2$ scale, and at the $1m^2$ scale the highest richness of 8.2 was recorded at the Meadows control (Table 3). Species richness at the $1m^2$ scale was significantly lower for the "foliar-sprayed" treatment and the "boom-sprayed" treatment compared to its respective control sites (Table 4b). At the $50m^2$ scale the richness was also significantly lower for both the "foliar-sprayed" treatments (at the Meadows and Platboom) as well as the "boom-sprayed" treatment sites. The "bulldozed and aerial-sprayed" treatment site and the "aerial-sprayed" treatment were similar in richness to the control.

When the plant species richness values were compared amongst the control sites ($50m^2$ and $1m^2$), the lowest species richness occurred at Klaasjagersberg, which comprised the oldest vegetation (Figure 1e and 1f). This control differed significantly from both the Meadows and Platboom controls (Table 4b).

Table 2. Environmental variables for the 10 study sites in the Cape of Hope Nature Reserve (means, SD in parentheses, n=5).

The values in superscript are the probabilities of significant differences between treatments and controls. (Mann-Whitney U-test).

Figures in bold represent values which are significantly different between treatments and controls

Plot number	K1a	K1b	K2:	K3:	M5:	M4:	P6a:	P6b:	P7:	P8:
Treatment	burnt, bulldozed & aerial sprayed	burnt, bulldozed & aerial sprayed	burnt & aerial sprayed	Control	burnt & foliar sprayed	Control	burnt & boomsprayed	burnt & boomsprayed	burnt & foliar sprayed	Control
% Bare Ground	11 (9.62) $^{0.92}$	5.4 (6.31) $^{0.35}$	13 (9.75) $^{0.60}$	9.8 (7.82)	27.4 (39.45) $^{0.75}$	7.8 (4.17)	22.4 (21.18) $^{0.92}$	22 (21.97) $^{0.60}$	20.6 (17.08) $^{0.83}$	14.6 (5.77)
Soil ph	4.65 (0.82) $^{0.35}$	4.64 (1.31) $^{0.92}$	4.06 (041) $^{0.53}$	4.248 (.38)	**7.49 (0.04)** $^{0.02}$	7.33 (0.15)	**6.77 (0.96)** $^{0.02}$	5.28 (.89) $^{0.25}$	4.56 (0.43) $^{0.35}$	4.742 (.26)
Soil depth (cm)	57.36 (24.32) $^{0.17}$	52.92 (25.69) $^{0.17}$	78.12 (9.70) $^{0.75}$	74.76 (25.58)	91.64 (18.23) $^{0.17}$	79.56 (11.89)	99.36 (3.67) $^{0.35}$	101 (0.00) $^{0.12}$	79.24 (21.17) $^{0.08}$	94.6 (13.42)

Table 3. Plant variables for the 10 study sites in the Cape of Good Hope Nature Reserve (means, SD in parentheses, n=5).

Refer to ANOVA (Table 4a, 4b and 4c) for significant differences between treatment and controls, for within site treatment comparisons and control site comparisons

Plot number	K1a:	K1b:	K2:	K3:	M5:	M4:	P6a:	P6b:	P7:	P8:
Plant density (abundance)	120 (68.98)	234.4 (77.11)	209.3 (59.08)	131 (61.80)	144.4 (75.54)	247.2 (113.78)	133 (80.73)	133.8 (63.57)	129.6 (76.18)	202.9 (70.18)
Total Canopy Cover (%)	89 (9.62)	94.6 (6.31)	87 (9.75)	90.2 (7.82)	72.6 (39.45)	92.2 (4.17)	77.6 (21.18)	78 (21.97)	79.4 (17.08)	85.4 (5.77)
Plant species richness ($50m^2$)	10.4 (3.05)	13.2 (3.42)	15.8 (3,12)	15 (4)	9 (5.57)	23.2 (5.15)	8 (1.87)	9.6 (4.16)	13 (4.06)	23.6 (11.74)
Plant species richness ($1m^2$)	3.2 (1.48)	3.4 (0.89)	3.77(2.63)	4.4 (1.34)	2.6 (1.82)	8.2 (2.59)	3.4 (0.89)	2.8 (1.10)	4.8 (1.48)	8 (3.16)
Diversity (H)	0.64 (0.11)	0.75 (0.11)	0.83 (0.18)	0.86 (0.23)	0.39 (0.24)	1.08 (0.15)	0.51 (0.13)	0.53 (0.29)	0.83 (0.09)	1.04 (0.20)
Evenness (J)	0.64 (0.08)	0.68 (0.09)	0.69 (0.11)	0.74 (0.16)	0.44 (0.33)	0.79(0.06)	0.58 (0.14)	0.54 (0.25)	0.76 (0.11)	0.78(0.04)

Within-site treatment comparisons ($50m^2$) showed that species richness was significantly lower in the "bulldozed & aerial-sprayed" versus the "aerial-sprayed" treatment. Species richness in the "boom-sprayed" treatment was significantly lower than the "foliar-sprayed" treatment. The species richness values were an underestimate however, owing to the season of the survey. Ephemeral species such as some geophytes and annual forbs are not represented in the vegetation during the hot dry summer months, and especially not in vegetation older than 2 years post-fire.

Diversity indices

The lowest diversity amongst the controls occurred at Klaasjagersberg (Table 3). Overall, the lowest plant diversity occurred at the "foliar-sprayed" treatment at the Meadows (Figure 1c), which was significantly less diverse compared to its control site (Table 4c). Treatment versus control comparisons showed that the diversity was also highly significantly lower in the "boom-sprayed" treatment. Within-site treatment comparisons showed that the "boom-sprayed" treatment had significantly lower diversity than the "foliar-sprayed" treatment at Platboom.

Evenness

Evenness indices were highest in the control sites, as well as at the "foliar-sprayed" site at Platboom (Figure 1d). The area with the most uniform distribution of plant species was the Meadows control (with an evenness of 0.79), followed very closely by the Platboom control which had an evenness of 0.78. The plant community that had the lowest evenness index was that of the "foliar-sprayed" treatment site, at the Meadows (Table 3).

Treatment versus control comparisons indicated that the evenness of the "foliar-sprayed" treatment site at the Meadows was significantly lower than its control (Table 4c). The "boom-sprayed" treatment also had a significantly lower evenness index compared to the control. Within-site treatment comparisons also indicated that the "boom-sprayed" treatment differed significantly from the "foliar-sprayed" treatment at Platboom, which had an evenness index similar to the Platboom control.

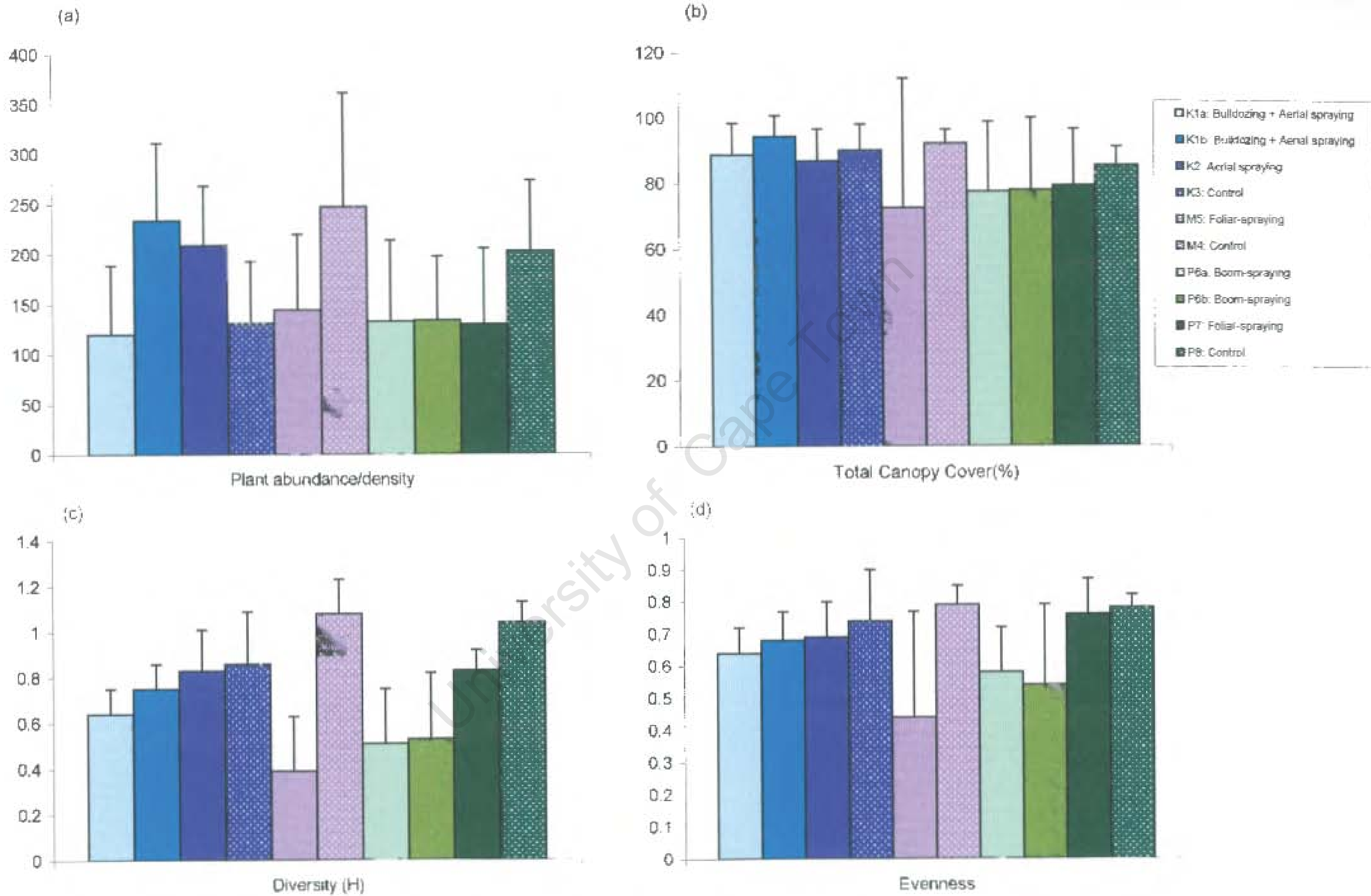
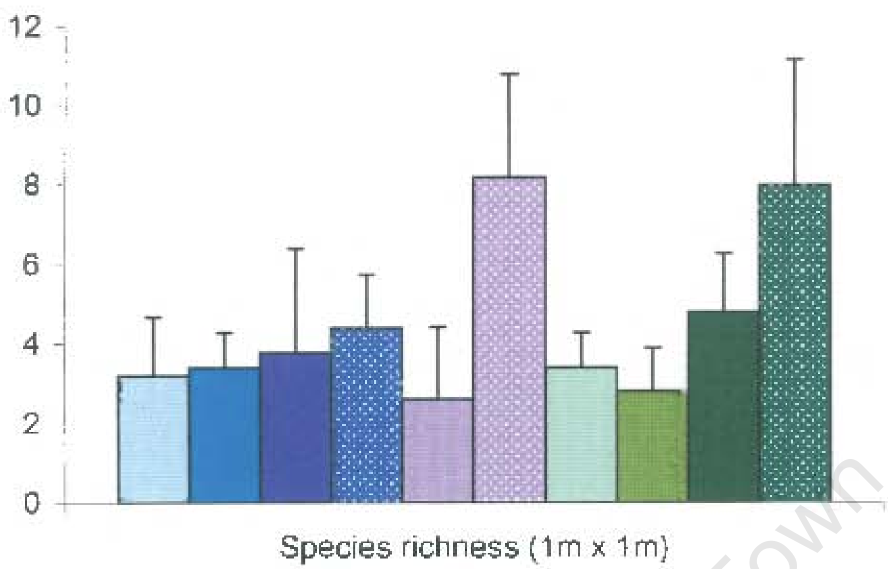


Figure 1 (a-d). Mean abundance, total canopy cover, diversity and evenness for the 10 study sites in the Cape of Good Hope Nature Reserve. Standard deviations are indicated as bars, and all graphs follow the legend in figure 1b.

(e)



(f)

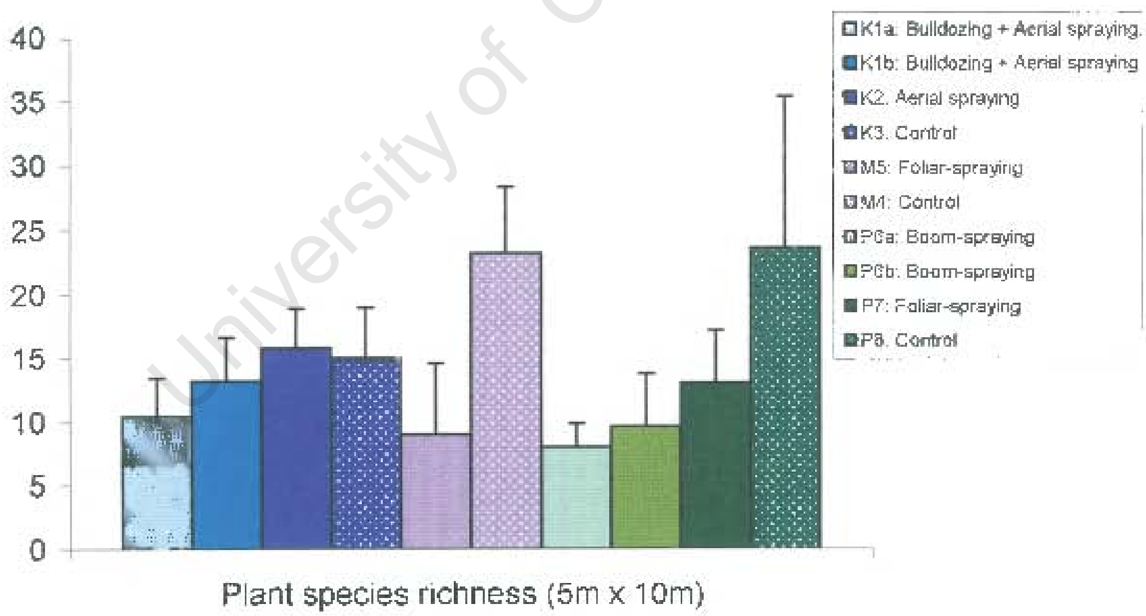


Figure 1 (e-f). Mean plant species richness at 1m^2 and 50 m^2 respectively. The standard deviations are indicated as bars. Both graphs follow the legend in figure 2f.

Guild structure

<u>Biological attributes</u>

Six of the twelve biological attribute classes had data with skewed distributions (Tables 5a, 5b and 5c). Most of the plant species used passive and wind dispersal strategies compared to ant and bird dispersal (Figure. 2). The analysis of variance for passively dispersed plants showed that there was no overall difference in this mode of dispersal in a comparison between all treatments and their controls (P=0.335). When the percentage cover of wind-dispersed plant species was analysed, there was a significant difference between the "foliar-sprayed" treatment at the Meadows and its control site (Table 5a). The cover of this guild was very low in the "foliar-sprayed" treatment (M5). When the controls were compared to each other, the cover of the wind dispersed plant species was significantly higher at the Meadows compared to the Klaasjagersberg control (P=0.03). Within-site treatment comparisons showed the "foliar-sprayed" treatment had a significantly higher percentage cover of wind dispersed species than the "boom-sprayed" treatment (P=0.03). Control comparisons for ant-dispersed plants showed that the projected canopy cover of plants, using ants as their dispersal agent, was significantly higher at the Platboom control compared to the Klaasjagersberg control.

Treatment versus control comparisons however, showed that it was only the "foliar-sprayed" treatment at Platboom that contained a significantly lower cover of ant-dispersed plants compared to its respective control.

Most of the plant cover was concentrated in the low shrub and graminoid growth form categories (Figure. 2). There was a decrease in the "low-shrub" component of the vegetation and an increase in the graminoids following all except the "aerial-sprayed" treatment. The graminoid component increased significantly following the "bulldozing and aerial-spraying", "boom-spraying" and the "foliar-sprayed" treatment at the Meadows (Table 5b). The same was not observed in the Platboom "foliar-sprayed" treatment. In the latter treatment, the percentage cover of graminoids was similar to that of its control. Within-site treatment comparisons showed that the "boom-sprayed" treatment had a significantly higher cover of sprouters, which were mainly graminoids compared to "foliar-sprayed" treatment at Platboom.

Table 4a. The effect of alien clearing treatments on plant density and projected canopy cover using ANOVA. Significant differences between comparisons are indicated in bold.

	Plant density					Total projected canopy cover (%)				
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.	d.f.	s.s.	m.s.	v.r.	F pr.
All treatments & controls	9	0.98	0.11	1.48	0.189	9	1479.5	164.4	0.79	0.626
Within site treatment versus control comparisons										
K1a+K1b vs K3	1	0.04	0.04	0.61	0.441	1	21.2	21.2	0.1	0.751
K2 vs K3	1	0.10	0.10	1.36	0.25	1	132.3	132.3	0.64	0.43
M5 vs M4	**1**	**0.29**	**0.29**	**3.97**	**0.053**	1	431.6	431.6	2.08	0.157
P6a+P6b vs P8	1	0.16	0.16	2.16	0.149	1	13.3	13.3	0.06	0.801
P7 vs P8	1	0.16	0.16	2.13	0.152	1	36.1	36.1	0.17	0.679
Between site control comparisons										
K3 vs M4	1	0.22	0.22	2.92	0.095	1	0.2	0.2	0	0.974
M4 vs P8	1	0.01	0.01	0.18	0.671	1	100.5	100.5	0.48	0.491
K3 vs P8	1	0.22	0.22	2.98	0.092	1	27.5	27.5	0.13	0.718
Within site treatment comparisons										
K1a+K1b vs K2	1	0.00	0.00	0.02	0.883	1	153.2	153.2	0.74	0.396
P6a+P6b vs P7	1	0.00	0.00	0.05	0.831	1	10.8	10.8	0.05	0.821

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)

M5 = burnt & foliar sprayed; M4 = control (Meadows)

6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom).

Table 4b. The effect of alien clearing treatments on plant species richness using ANOVA. Significant differences between comparisons are indicated in bold.

	Plant species richness ($50m^2$)					Plant species richness ($1m^2$)				
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.	d.f.	s.s.	m.s.	v.r.	F pr.
All treatments & controls	**9**	**1.26**	**0.14**	**5.8**	**<0.001**	**9**	**1.56**	**0.17**	**4.7**	**<0.001**
Within site treatment versus control comparisons										
K1a+K1b vs K3	1	0.04	0.04	1.59	0.215	1	0.07	0.07	1.83	0.184
K2 vs K3	1	0.04	0.04	1.6	0.213	1	0.01	0.01	0.32	0.575
M5 vs M4	**1**	**0.53**	**0.53**	**22.16**	**<0.001**	**1**	**0.81**	**0.81**	**22.1**	**<0.001**
P6a+P6b vs P8	**1**	**0.14**	**0.14**	**23.62**	**<0.001**	**1**	**0.56**	**0.58**	**15.13**	**<0.001**
P7 vs P8	**1**	**0.14**	**0.14**	**5.69**	**0.022**	1	0.11	0.11	3.1	0.086
Between site control comparisons										
K3 vs M4	1	0.09	0.09	3.83	0.057	**1**	**0.18**	**0.18**	**4.76**	**0.035**
M4 vs P8	1	0.00	0.00	0.05	0.824	1	0.00	0.00	0.01	0.916
K3 vs P8	**1**	**0.11**	**0.11**	**4.54**	**0.039**	**1**	**0.22**	**0.22**	**6.05**	**0.018**
Within site treatment comparisons										
K1a+K1b vs K2	**1**	**1.26**	**0.14**	**5.8**	**<0.001**	1	0.01	0.01	0.18	0.669
P6a+P6b vs P7	**1**	**0.11**	**0.11**	**4.44**	**0.042**	1	0.13	0.13	3.45	0.071

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)

M5 = burnt & foliar sprayed; M4 = control (Meadows)

6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

Table 4c. The effect of alien clearing treatments on plant diversity and evenness using ANOVA. Significant differences between comparisons are indicated in bold.

	Plant diversity					Plant Evenness				
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.	d.f.	s.s.	m.s.	v.r.	F pr.
All treatments & controls	**9**	**2.33**	**0.26**	**7.66**	**<0.001**	**9**	**0.58**	**0.06**	**2.47**	**0.024**
Within site treatment versus control comparisons										
K1a+K1b vs K3	1	0.09	0.09	2.75	0.105	1	0.02	0.02	0.69	0.413
K2 vs K3	1	0.02	0.02	0.63	0.432	1	0.00	0.00	0	0.956
M5 vs M4	**1**	**1.20**	**1.20**	**35.42**	**<0.001**	**1**	**0.30**	**0.30**	**11.68**	**0.001**
P6a+P6b vs P8	**1**	**0.89**	**0.89**	**26.46**	**<0.001**	**1**	**0.16**	**0.16**	**6**	**0.019**
P7 vs P8	1	0.11	0.11	3.32	0.076	1	0.00	0.00	0.03	0.861
Between site control comparisons										
K3 vs M4	1	0.11	0.11	3.39	0.073	1	0.01	0.01	0.3	0.585
M4 vs P8	1	0.00	0.00	0.12	0.733	1	0.00	0.00	0.03	0.861
K3 vs P8	1	0.13	0.13	3.71	0.061	1	0.01	0.01	0.28	0.597
Within site treatment comparisons										
K1a+K1b vs K2	1	0.00	0.00	0.11	0.745	1	0.00	0.00	0.14	0.712
P6a+P6b vs P7	**1**	**0.31**	**0.31**	**9.23**	**0.004**	**1**	**0.13**	**0.13**	**5.04**	**0.03**

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)

M5 = burnt & foliar sprayed; M4 = control (Meadows)

6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

When regenerative mode was analysed, all treatments except "aerial-spraying" caused a significant decrease in the long-lived obligate seeding plants. The "boom-sprayed" treatment had a significantly higher percentage cover of sprouting plant species compared to its control.

The effect of clearance on guild structure

Changes in the percentage cover of the various biological attributes differed among treatments (Figure 2). Growth form structure changed least following the "aerial-spraying" and Platboom "foliar-spraying" treatments. Detrended correspondence analysis (DCA), which explains 58% variance of species data on two axes, showed all the controls as well as the "foliar-sprayed" treatment at Platboom to be more similar to each other (Figure 3a) in terms of guild structure than to the other treatments (Figure 3b). The Meadows "foliar-sprayed" treatment formed a cluster to the far left of the DCA plot, overlapping with the "boom-sprayed" treatment. The "aerial-sprayed" treatment was intermediate between the latter treatments and the controls. A biplot showing the various guilds (Figure 4), indicates that high percentage cover of sprouters and graminoids are more closely associated with the treatments than the controls, whereas the low, medium and tall shrubs have a higher representation in the controls and the "foliar-sprayed" treatment at Platboom. Some of the "boom-sprayed" plots occur towards the top left of the diagram, and are closely associated with a high cover of short-lived seeders (e.g. *Vellereophyton dealbatum* and *Plecostachys serpyllifolia*).

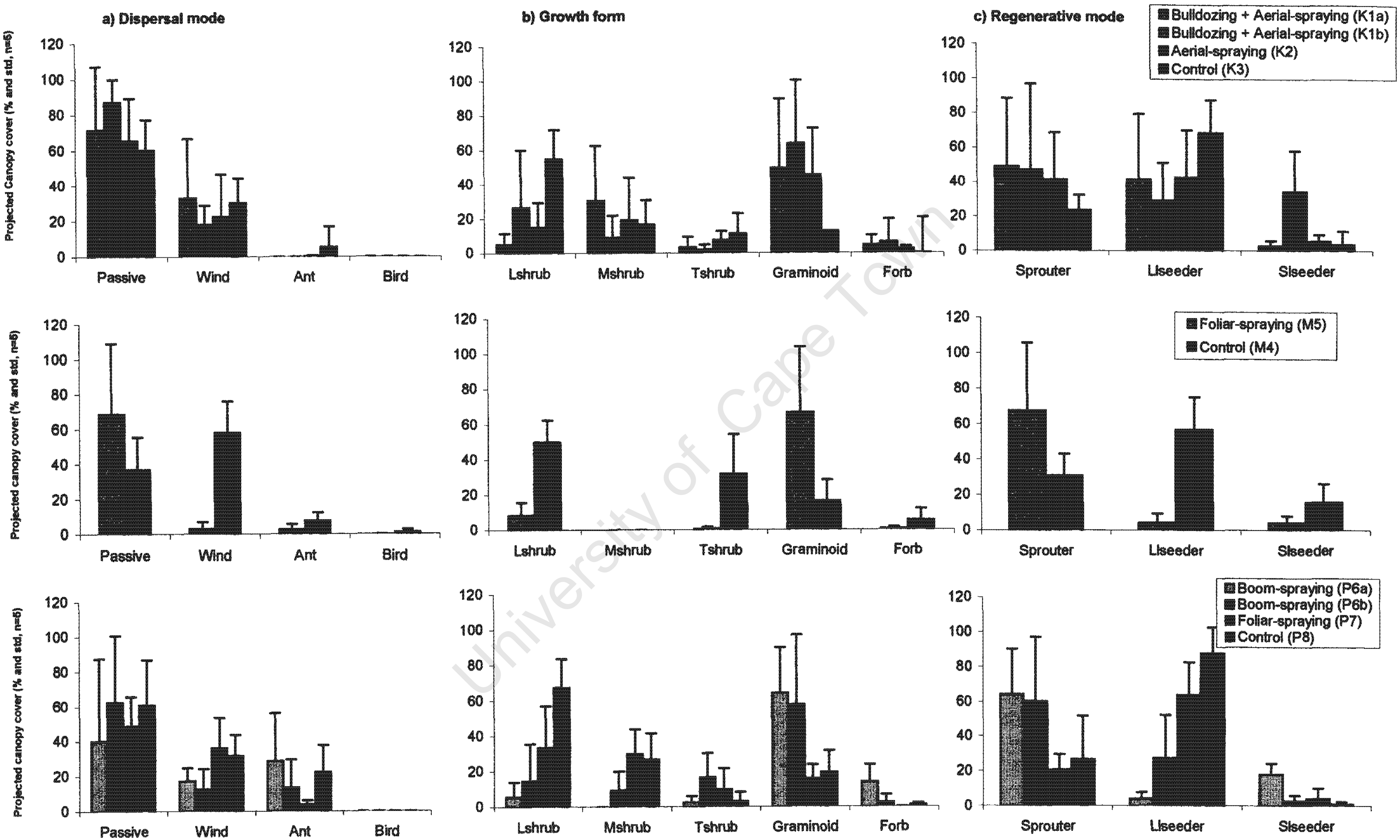


Figure 2. Mean percentage canopy cover (with standard deviations indicated as bars) of plant species classified into guilds by a) mode of dispersal, b) growth form and c) regenerative mode, at three localities in the Cape of Good Hope Nature Reserve. (Lshrub = low shrub; Mshrub= medium shrub; Tshrub = tall shrub; Slseeder = short lived seeder and Llseeder = long lived seeder).

Table 5a. ANOVA results for four plant dispersal guilds between alien clearing treatments and respective controls, among controls and among treatments within the same localities. Ant and bird dispersal modes had skewed distribution patterns
Figures in bold represent significant differences between the various comparisons as indicated below.

	Passive					Wind					Ant					Bird				
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.	d.f.	s.s	m.s	v.r.	F pr.	d.f	s.s	m.s.	v.r.	F pr.	d.f.	s.s	m.s	v.r.	F pr.
All treatments & controls	9	5939	659.9	1.18	0.335	**9**	**6020**	**668.9**	**4.31**	**<.001**	**9**	**4605**	**511.6**	**4.23**	**<.001**	**9**	**64.79**	**7.199**	**2.91**	**0.01**
Within site treatment versus control comparisons																				
K1a+K1b vs K3	1	912.4	912.4	1.63	0.209	1	93	93	0.6	0.444	1	133.4	133.4	1.1	0.3	1	0	0	0	1
K2 vs K3	1	110.4	110.4	0.2	0.66	1	37	37	0.24	0.628	1	2.3	2.3	0.02	0.892	1	0	0	0	1
M5 vs M4	1	1066	1066	1.9	0.176	**1**	**4664**	**4664**	**30.04**	**<.001**	1	159.8	159.8	1.32	0.257	**1**	**36.44**	**36.44**	**14.71**	**<.001**
P6a+P6b vs P8	1	171.4	171.4	0.31	0.583	1	539.4	539.4	3.47	0.07	1	77.4	77.4	0.64	0.429	1	0.438	0.438	0.18	0.676
P7 vs P8	1	280.8	280.8	0.5	0.483	1	16.3	16.3	0.1	0.748	1	700.7	700.7	5.79	**0.021**	1	0.328	0.328	0.13	0.718
Between site control comparisons																				
K3 vs M4	1	538.4	538.4	0.96	0.333	**1**	**731.6**	**731.6**	**4.71**	**0.036**	1	191.5	191.5	1.58	0.216	**1**	**36.44**	**36.44**	**14.71**	**<.001**
M4 vs P8	1	132.3	132.3	0.24	0.63	1	185.3	185.3	1.19	0.281	1	281.7	281.7	2.33	0.135	**1**	**14.92**	**14.92**	**6.02**	**0.019**
K3 vs P8	1	24.1	24.1	0.04	0.837	1	3.5	3.5	0.02	0.881	**1**	**1025**	**1025**	**8.47**	**0.006**	1	0.328	0.328	0.13	0.718
Within site treatment comparisons																				
K1a+K1b vs K2	1	399.1	399.1	0.71	0.404	1	6.3	6.3	0.04	0.841	1	5.9	5.9	0.05	0.827	1	0	0	0	1
P6a+P6b vs P7	1	39.1	39.1	0.07	0.793	**1**	**777.5**	**777.5**	**5.01**	**0.031**	1	473.9	473.9	3.91	0.055	1	0	0	0	1

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)
M5 = burnt & foliar sprayed; M4 = control (Meadows)
6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

Table 5b. ANOVA results for five biological attributes of growth form between alien clearing treatments and respective controls, among controls and among treatments within the same localities. Mid and tall shrubs as well as forbs had skewed distributions
Figures in bold represent significant differences between the various comparisons as indicated below.

	Low shrub					Mid Shrub					Tall shrub				
Source of variation	d.f	s.s.	m.s.	v.r	F pr.	d.f	s.s.	m.s	v.r.	F pr.	d.f	s.s.	m.s	v.r.	F pr.
All treatments & controls	**9**	**12299.5**	**1366.6**	**7.79**	**<.001**	**9**	**7517.7**	**835.3**	**5.19**	**<.001**	**9**	**3758.9**	**417.7**	**4.04**	**<.001**
Within site treatment versus control comparisons															
K1a+K1b vs K3	**1**	**2883.8**	**2883.8**	**16.43**	**<.001**	1	21.9	21.9	0.14	0.714	1	175.1	175.1	1.69	0.201
K2 vs K3	1	213.9	213.9	1.22	0.276	1	0.9	0.9	0.01	0.94	1	153.2	153.2	1.48	0.231
M5 vs M4	**1**	**2254.7**	**2254.7**	**12.85**	**<.001**	1	0.3	0.3	0	0.964	**1**	**2174.7**	**2174.7**	**21.03**	**<.001**
P6a+P6b vs P8	**1**	**5627.8**	**5627.8**	**32.07**	**<.001**	**1**	**1866**	**1866**	**11.6**	**0.002**	1	149.7	149.7	1.45	0.236
P7 vs P8	**1**	**1143.6**	**1143.6**	**6.52**	**0.015**	1	12	12	0.07	0.786	1	174.5	174.5	1.69	0.201
Between site control comparisons															
K3 vs M4	1	26.9	26.9	0.15	0.697	**1**	**1363.1**	**1363.1**	**8.47**	**0.006**	**1**	**934.4**	**934.4**	**9.04**	**0.005**
M4 vs P8	1	9.7	9.7	0.06	0.815	**1**	**1956**	**1956**	**12.16**	**0.001**	**1**	**655.7**	**655.7**	**6.34**	**0.016**
K3 vs P8	1	146.7	146.7	0.84	0.366	1	122.7	122.7	0.76	0.388	1	88.9	88.9	0.86	0.359
Within site treatment comparisons															
K1a+K1b vs K2	1	16.9	16.9	0.1	0.758	1	0.4	0.4	0	0.959	1	258.7	258.7	2.5	0.122
P6a+P6b vs P7	**1**	**1293.9**	**1293.9**	**7.37**	**0.01**	**1**	**2228.3**	**2228.3**	**13.85**	**<.001**	1	9.1	9.1	0.09	0.768

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)
M5 = burnt & foliar sprayed; M4 = control (Meadows)
6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

Table 5b. Continued.

	Graminoid					Forb				
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.	d.f.	s.s.	m.s.	v.r.	F pr.
All treatments & controls	**9**	**11658.9**	**1295.4**	**2.9**	**0.01**	**9**	**1446.16**	**160.68**	**2.71**	**0.014**
Within site treatment versus control comparisons										
K1a+K1b vs K3	**1**	**3654.3**	**3654.3**	**8.18**	**0.007**	1	174.57	174.57	2.95	0.094
K2 vs K3	1	0.5	0.5	0	0.974	1	16.91	16.91	0.29	0.596
M5 vs M4	**1**	**2532.9**	**2532.9**	**5.67**	**0.022**	1	154.32	154.32	2.6	0.114
P6a+P6b vs P8	**1**	**3068.8**	**3068.8**	**6.87**	**0.012**	1	215.87	215.87	3.64	0.064
P7 vs P8	1	16.5	16.5	0.04	0.848	1	43.18	43.18	0.73	0.398
Between site control comparisons										
K3 vs M4	1	20.6	20.6	0.05	0.831	1	202.03	202.03	3.41	0.072
M4 vs P8	1	31	31	0.07	0.794	1	69	69	1.16	0.287
K3 vs P8	1	67.9	67.9	0.15	0.699	1	19.87	19.87	0.34	0.566
Within site treatment comparisons										
K1a+K1b vs K2	1	380.5	380.5	0.85	0.362	1	0.28	0.28	0	0.946
P6a+P6b vs P7	**1**	**3611**	**3611**	**8.08**	**0.007**	**1**	**496.41**	**496.41**	**8.38**	**0.006**

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)

M5 = burnt & foliar sprayed; M4 = control (Meadows)

6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

Table 5c. ANOVA results for three biological attributes of regenerative mode between alien clearing treatments and respective controls, among controls and among treatments within the same localities. Short lived seeders has skewed distributions
Figures in bold represent significant differences between the various comparisons as indicated below.

	Sprouter					Long lived seeder					Short lived seeder				
Source of variation	d.f	s.s.	m.s.	v.r.	F pr.	d.f	s.s.	m.s.	v.r.	F pr.	d.f.	s.s.	m.s.	v.r.	F pr.
All treatments & controls	9	6591.1	732.3	1.35	0.242	**9**	**18770.1**	**2085.6**	**8.96**	**<.001**	**9**	**4475.97**	**497.33**	**6.86**	**<.001**
Within site treatment versus control comparisons															
K1a+K1b vs K3	1	1126.1	1126.1	2.08	0.157	**1**	**1689.6**	**1689.6**	**7.25**	**0.01**	**1**	**445.84**	**445.84**	**6.15**	**0.017**
K2 vs K3	1	4.7	4.7	0.01	0.926	1	10	10	0.04	0.837	1	77.44	77.44	1.07	0.307
M5 vs M4	1	1246.5	1246.5	2.3	0.137	**1**	**3921.9**	**3921.9**	**16.84**	**<.001**	**1**	**388.03**	**388.03**	**5.35**	**0.026**
P6a+P6b vs P8	**1**	**2410.4**	**2410.4**	**4.45**	**0.041**	**1**	**9968**	**9968**	**42.8**	**<.001**	**1**	**524.34**	**524.34**	**7.24**	**0.01**
P7 vs P8	1	24.1	24.1	0.04	0.834	**1**	**1098.3**	**1098.3**	**4.72**	**0.036**	1	41.43	41.43	0.57	0.454
Between site control comparisons															
K3 vs M4	1	70.3	70.3	0.13	0.721	1	157.6	157.6	0.68	0.416	**1**	**444.25**	**444.25**	**6.13**	**0.018**
M4 vs P8	1	38.8	38.8	0.07	0.79	1	587.3	587.3	2.52	0.12	**1**	**552.67**	**552.67**	**7.63**	**0.009**
K3 vs P8	1	5	5	0.01	0.924	1	752.4	752.4	3.23	0.08	1	86.55	86.55	1.19	0.281
Within site treatment comparisons															
K1a+K1b vs K2	1	174.9	174.9	0.32	0.573	1	115	115	0.49	0.486	1	235.16	235.16	3.24	0.079
P6a+P6b vs P7	**1**	**2998.5**	**2998.5**	**5.53**	**0.024**	**1**	**3791.2**	**3791.2**	**16.28**	**<.001**	1	239.19	239.19	3.3	0.077

K1a + K1b = burnt, bulldozed & aerial sprayed; K2 = burnt & aerial sprayed; K3 = control (Klaasjagersberg)
M5 = burnt & foliar sprayed; M4 = control (Meadows)
6a + 6b = burnt & boomsprayed; P7= burnt & foliar sprayed; P8 control (Platboom)

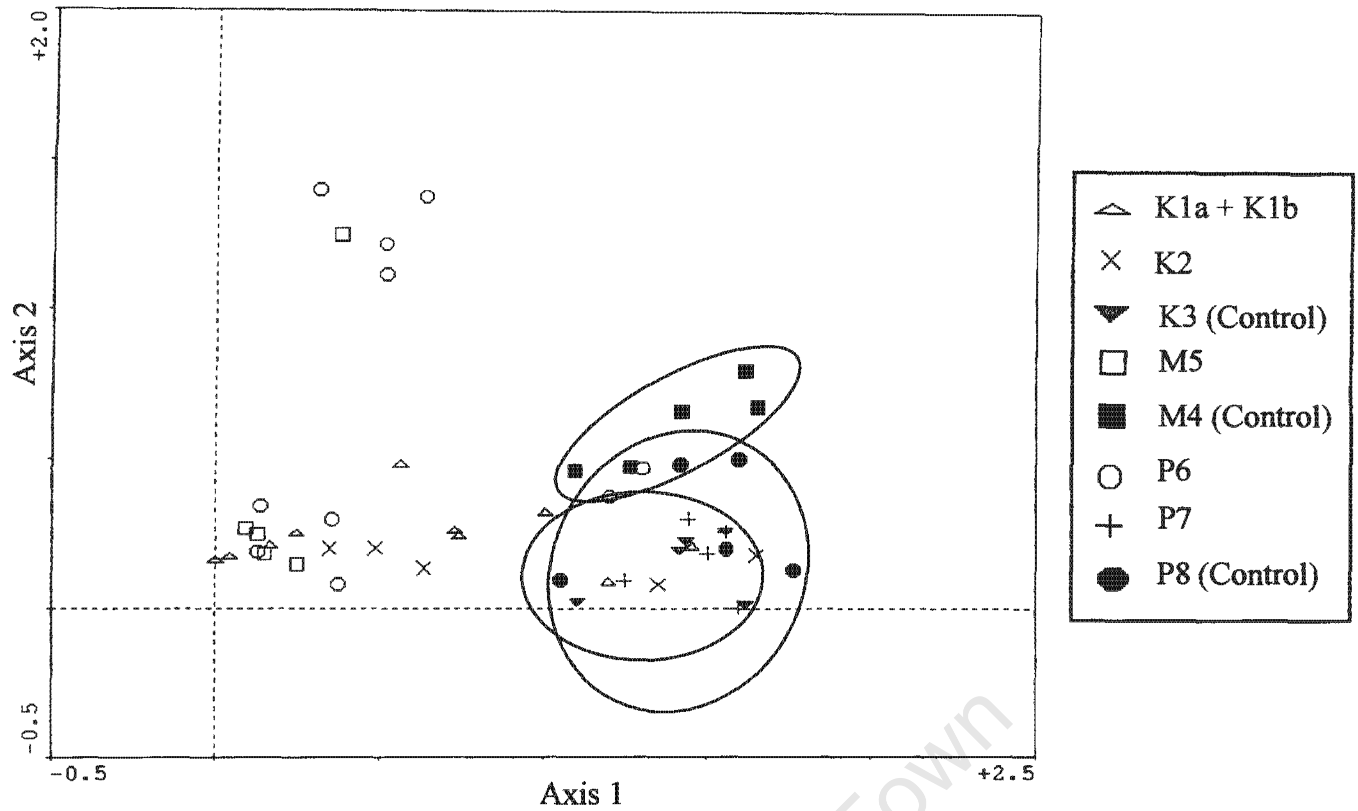


Figure 3a. Detrended correspondence analysis of the various guilds into which vegetation was classified. Encircled symbols represents the controls only.

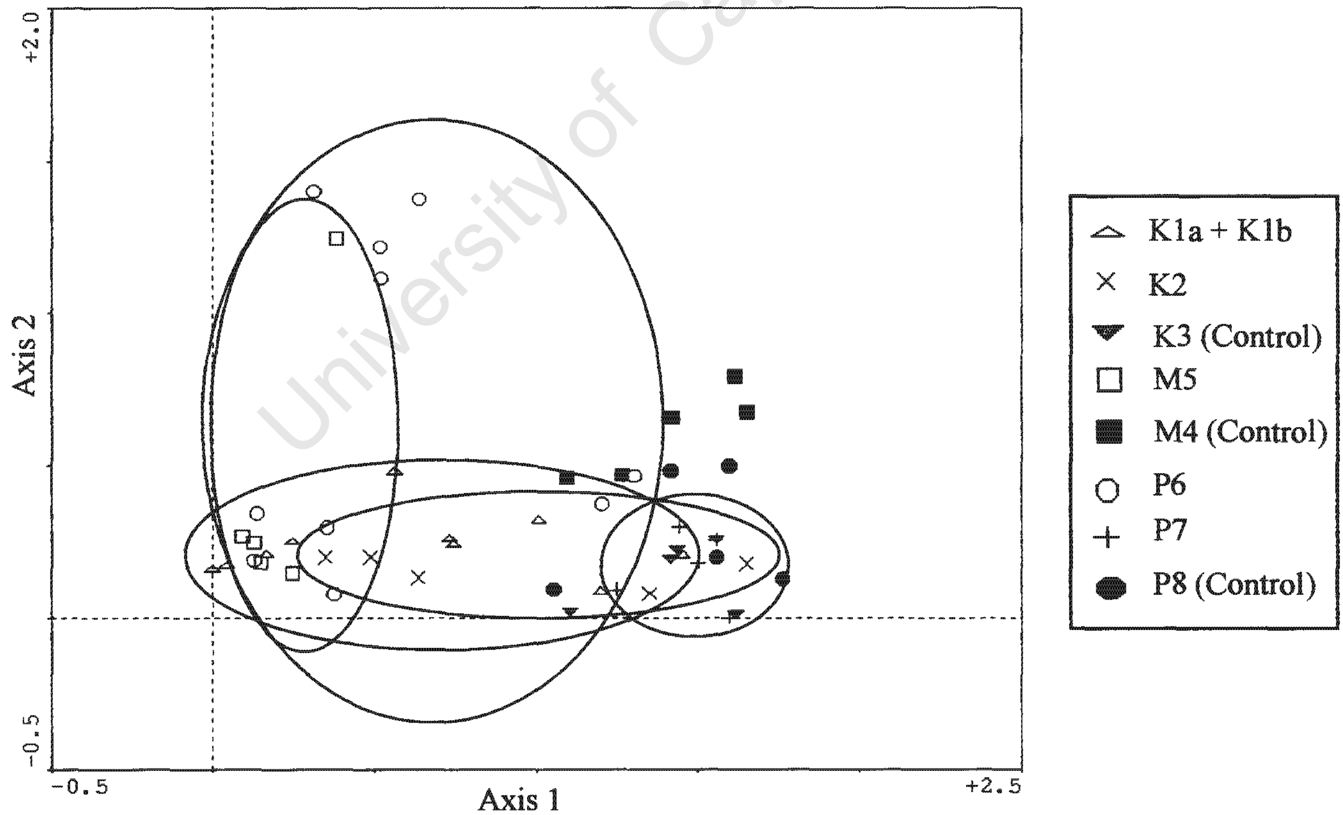


Figure 3b. Detrended correspondence analysis of the various guilds into which vegetation was classified. Encircled symbols represents the treatments only.

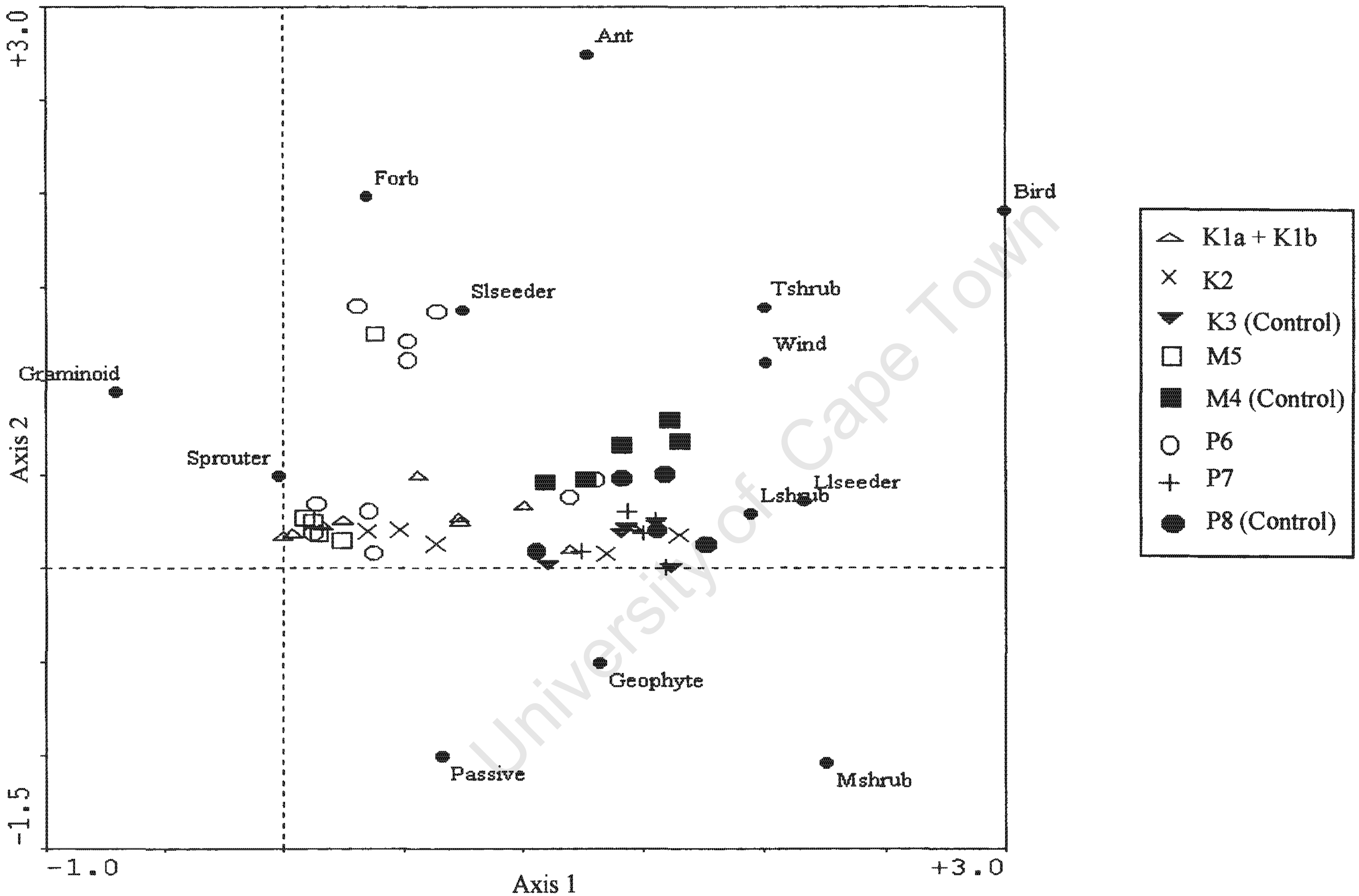


Figure 4. Detrended correspondence analysis of plant percentage cover data, with a biplot of the thirteen different guilds. (Lshrub = low shrub; Mshrub= medium shrub; Tshrub = tall shrub; Slseeder = short lived seeder and Llseeder = long lived seedeer). The eigenvalues for axis 1 and axis 2 are 0.321 and 0.119 respectively.

Discussion:

Effect of alien control methods on diversity and abundance measures

According to the analysis of variance plant species richness at both the quadrat ($1m^2$) and plot scales were significantly lower for most of the previously invaded areas, except for the "bulldozed and aerial-sprayed" treatment and the "aerial-sprayed" treatment sites (Table 4b). The exceptionally low richness in the Klaasjagersberg region may however be attributed to historical factors that are unrelated to alien plant management (see chapter one), since other studies conducted by van Wilgen (1981) and Holmes and Cowling (1997a) showed higher richness values for similarly aged vegetation. In the former study, twenty-one year old vegetation contained 24 species (5 m x 10 m plot) and the latter study contained a mean of 9.7 species ($1m^2$) in eleven-year-old vegetation.

The density of fynbos species was also lower in the previously invaded areas, except for one of the "bulldozed and aerial-sprayed" treatments and the "aerial-sprayed" treatment (Table 4a). Here two widespread and troublesome grass weeds (*Stenotaphrum secundatum* and *Cynodon dactylon)* were dominant, indicating a disturbed habitat (Bromilow 1995). Holmes and Cowling (1997a,b) also observed decreases in species richness, diversity and abundance in previously invaded sites which resulted from a reduction of seed input under alien trees and the gradual attrition of the soil-stored seed banks.

In this study, it was shown that herbicidal application impacted fynbos communities to varying degrees. The foliar-spraying treatment at Platboom caused a significant reduction in richness at the plot scale, although no other variables were significantly affected for this treatment (Table 4b). The "aerial-sprayed" treatment had a similar species richness to the natural community. The taller (eight-month old) *Acacia* canopies intercepted the blanket application of herbicide (Triclopyr), with little consequence (i.e. mortality) to the natural indigenous community. A small-scale study conducted within The Cape of Good Hope Nature Reserve, looking at the effect of spot spraying on the fynbos vegetation, showed proteoids, ericoids and to a lesser degree, graminoids to be sensitive to the chemicals (Triclopyr) used for alien plant control (Ellis *et al.* undated).

Boom-spraying alien vegetation was found to have a negative effect on the recovery of fynbos vegetation as it significantly reduced the species richness at both the quadrat and plot scale. Boom-spraying (with the boom approximately 1m above ground) involved very direct contact of herbicide with both alien and indigenous vegetation as the vehicle traversed the two-year old patch of aliens. A ranger report by Alex Wood for February 1993 indicated that boom-spraying was hampered to a certain extent by strong south-easterly winds. Wind would thus have resulted in a large amount of chemical drift onto indigenous vegetation, thus resulting in significantly lower plant diversity and created a plant community that was very unevenly distributed (Table 4c).

Plant variables such as species richness, density, diversity and evenness were all significantly lower at the "foliar-sprayed" treatment site at the Meadows compared to the control site (Figures 1a-1f). The only variable not significantly impacted in this site was the total canopy cover, but the standard deviation was high (Table 3). Since this treatment affected most variables, it would imply that this was one of the most detrimental clearance treatments in relation to fynbos vegetation recovery. Here high fuel loads, due to the accumulation of extensive brush piles from past clearing operations (Wood 1993) as well as stack burns (see chapter one) may have exacerbated the effect of post-fire foliar spraying. Studies done in the fynbos biome have shown that invasion by alien trees and shrubs leads to a substantial increase in the above ground biomass (Versfeld and van Wilgen 1986), with *Acacia saligna* increasing the biomass threefold (van Wilgen and Richardson 1985). This increased fuel load would then lead to a more intense fire than is experienced by native fynbos communities (van Wilgen and Richardson 1985), causing hydrophobic organic substances to coat the soil particles (Scott and van Wyk, 1992). When subsequent rainfall occurs, sediment movement in the form of soil and gully erosion may result from an inability of water to be absorbed by the soil and pass down the soil profile (Scott and van Wyk 1992). This topsoil, which erodes away, is usually the source of important indigenous plant propagules. This site was running at a slope, which would greatly aid erosion, and lead to poor recruitment and ultimately a less functional system. It is possible that erosion may have played a role in poor recruitment but this was not empirically tested. Richardson and van Wilgen (1986) reported that serious erosion occurred because of a failure in the re-establishment of fynbos vegetation following a "fell and burn" treatment in the Wemmershoek catchment in Stellenbosch, and Holmes *et al.* (2000) also showed a reduction in density and cover after the same treatment.

Guild structure

Results of the DCA (Figure 4) indicate that the vegetation structure in all the controls as well as the "foliar-spraying" treatment in Platboom is more similar to each other (in terms of percentage cover) than to the other treatments. The range of growth forms, dispersal syndromes and regenerative modes were well represented in these communities as opposed to the other treatment sites. The implication is that most of the clearance methods may have impacted the guild structure in the treatment areas, and exacerbated the impact of the alien invasion itself.

This study showed that most species utilised both passive and wind dispersal (species in the: Ericaceae, Restionaceae and Poaceae families) more than ant and bird dispersal (Figure 2). The plant species that use wind and passive dispersal mechanisms (See appendix 2), produce small seeds that are distributed closer to the soil surface. In the "foliar-sprayed" treatment site at the Meadows, failure of the various guilds (such as the low shrubs, tall shrubs and the long-lived seeders) to re-establish may have been as a consequence of higher intensity fire that killed off the small seeded plant species. Holmes *et al.* 2000 found that following a "fell and burn" treatment through *Hakea*, which resulted in a high intensity fire, guild structure was severely negatively affected. Only myrmecochores persisted relatively well, since these were buried (by ants) to a greater depth in the soil.

In most treatments, there was a decrease in the low shrub and an increase in the graminoid (predominantly grasses) component of the vegetation (Tables 5b). This was largely due to the fact that the herbicide used (Triclopyr) does not kill monocots. Growth form structure changed the least following "aerial-spraying" and "foliar-spraying" at Platboom, and in the latter case the growth form structure remained constant, despite changes in species richness, thus buffering the ecosystem from a loss of function.

"Boom-spraying" and "foliar-spraying" caused a significant reduction in the long-lived obligate reseeding species, which form a dominant component of uninvaded fynbos (Holmes and Cowling 1997b). Many of these species, such as the serotinous proteas, form part of the overstorey, which is thought to maintain understorey diversity in the long term (Vlok 1996). Thus loss of overstorey species can threaten diversity, which may in turn affect ecosystem function.

Implications for Management

Recent research by Holmes and Cowling (1997a) into the restoration potential of Mountain fynbos has shown that this vegetation type is relatively resilient to disturbance such as alien plant invasion. Although 70% of the flora disappeared from the standing vegetation following two fire cycles in areas invaded by alien *Acacia saligna*, representatives of all the major growth forms survived in the seed bank as soil stored propagules. Thus, following clearance of alien stands, indigenous vegetation was able to re-establish, primarily from the soil seed bank.

Although fynbos is resilient in that it has good potential to recover following dense alien invasion (Holmes and Cowling 1997a, Holmes and Marais 2000) species re-introductions may be needed where plant guilds have been eliminated or are under-represented, to improve the long-term resilience of the restored community (Holmes and Richardson 1999). In treatments such as the "bulldozed and aerial-sprayed" and the "boom-sprayed" treatment, as well as the "foliar-sprayed" treatment at the Meadows, where guilds such as long-lived seeders, tall shrubs, medium shrubs and low shrubs have been significantly reduced, re-introduction of those guilds would accelerate recovery.

Since the majority of obligately reseeding species have soil-stored seeds it may be necessary to augment recruitment from the seed bank by sowing seed harvested from another site in the area, first giving seeds a smoke treatment to stimulate germination (Brown 1993). Species such as serotinous Proteaceae that store seeds in the canopy and which have been eliminated by invasion may be re-introduced if local seed sources are far away (> 1 km). Such introductions into the ecosystem would accelerate recovery and lead to a greater probability of a structurally diverse and more functional ecosystem re-establishing. It is important to realise however that dormant soil seed banks are cued to germinate after a fire. Thus, re-introduction by seed will only be successful after fire or on bare ground since seed germination cues are tied to the post-fire environment. Cues such as a direct heat pulse (Cocks and Stock 1997), smoke from a fire (Brown 1993) and an increase in diurnal temperature fluctuation due to vegetation removal (Pierce and Moll 1994) all play a role in stimulating germination in the post-fire environment.

In the bulldozed area, sowing could be conducted after the treatment itself, but in the case of "boom-spraying", it may be necessary to wait for another fire-cycle, after applying this treatment, to meet the ideal "post-fire" fynbos recruitment environment.

At the Meadows, high fuel loads coupled with possible soil erosion may have exacerbated the effect of post-fire "foliar-spraying". The management implications following this is to burn under cooler weather conditions or if possible to remove some of the fuel following felling to reduce the intensity and severity of the fire. Also, surface erosion could be reduced using logs pegged across steeper slopes and sowing with fast growing fynbos pioneers to provide early plant cover to protect the soil (Holmes and Richardson 1999, Holmes *et al.* 2000). At this site where the soil seed bank has been lost, the restoration of a structurally representative community would require the sowing of a comprehensive seed mix containing the major guilds such as graminoids, forbs, low shrubs and overstorey shrubs.

The "aerial-spraying" treatment and the "foliar-spraying" at Platboom caused the least change to the vegetation. Thus, natural recovery is adequate and no seed additions are required. If the former treatment is to be used in future management plans, practitioners need to ensure that herbicidal applications (using crop dusting aircraft) are delayed until the faster growing alien species have created a dense canopy above the regenerating indigenous species. This would reduce the possibility of herbicidal drift killing the fynbos under-storey. When "foliar-spraying" of alien plant species is conducted using handheld mistblowers, chemical drift plays a role in the mortality of indigenous vegetation. Thus, there is a need to explore alternative application methods. Currently, a new method of application in which the herbicide is wiped directly onto the foliage of the target plants is being tested. This method should minimise mortality of non-target species since there is no dripping or spray-drift. But for this method to be practicable in dense stands, the alien species would need to be taller than the indigenous non-target species.

"Boom-spraying" reduced the species richness and altered the structure of the vegetation. Triclopyr (Garlon) selectively kills dicotyledonous plants. The manner in which the herbicide was applied was not species-specific and vegetation recovery was thus negatively affected. Hence, although this herbicide may control alien vegetation "boom-spraying" is a less successful method of application in terms of preserving the natural environment, one of the essential reasons for control.

Priorities for future research

Herbicidal application has been used for many years in the control of invasive alien plants. Spot spraying as well as blanket herbicidal applications have been conducted in parts of the CFR, but very little research has been done to determine the side-effects of this treatment. It is essential that more studies be conducted concerning the impact of herbicides on the indigenous fynbos vegetation, especially if one considers the small-scale differentiation of communities as well as the highly localised endemics in the CFR. Trials pertaining to different methods of herbicidal application and timing of application on alien vegetation after a fire (6 months, twelve months or 18 months) would be particularly useful. There is also a need for determining the best time of year to apply herbicides. This will enable managers to improve the efficiency of their follow-up methods and potentially will aid the natural recovery of the ecosystem.

The bulldozing treatment should be re-assessed in a site where farming is not a contributing factor. Managers need to know beforehand whether this is a viable method for eradicating burnt alien vegetation to enable adequate recovery of a site.

From the alternate perspective it is essential that biological control agents are researched and released for alien species not currently under efficient biological control. Successful agents generally reduce the alien's fecundity and follow-up clearance costs. In addition, mechanical control could be improved by the development of efficient tools (e.g. the "tree popper" which is a mechanical device which cuts stems below the ground). Although mechanical control may reduce the need for herbicides, this type of device is still time consuming when clearing dense stands and may lead to huge disturbance in the form of vegetation trampling.

Are ants useful as indicators of recovery success in South African fynbos cleared of invasive alien plants by different methods?

Introduction

The use of plants as indicators of recovery or restoration success is problematic in fynbos. Besides the high beta and gamma diversity of fynbos plant communities, and the localised distribution patterns of many taxa (Kruger and Taylor 1979, Simmons and Cowling 1996), practitioners also have to consider that certain plant groups, such as geophytes are seasonal, while groups such as annuals, are present or not equally abundant at all stages of post-fire succession. These factors make it difficult to compile a complete species list for an area in a once-off survey. Thus, plants give one perspective on rehabilitation success, but the rehabilitation success of other components of the ecosystem might be different.

<u>The use of ants as indicators</u>

A potential alternative to using botanical surveys to assess the impact of land-use practices on the ecosystem is the use of invertebrates, which play a major functional role in the soil system (Majer 1996). Invertebrates have a long and successful history of use as biological indicators in aquatic ecosystems (Cairns and Pratt 1993) and have only recently received more attention in terrestrial systems (Kremen 1992, 1994, Peck *et al.* 1998). Ants, however, have been used extensively as bioindicators of restoration success (for the past twenty years) by the mining industry in Australia (Majer *et al.* 1984; and Majer 1985, Majer and Nichols 1998, Bisevac and Majer 1999). These protocols and guidelines have then been applied to Brazil (Majer 1992) and South Africa (Majer and de Kock 1992).

Traditionally, bioindicators have been used to assess ecosystem responses to environmental perturbation, often associated with human land-use (Noss 1990). In mine sites undergoing restoration, it was found that the pattern of ant species richness and composition was reflected by other invertebrate groups (Majer 1983), as well as changes in soil microbial biomass (Anderson and Sparling 1997).

To assess the effects of restoration practices on invertebrate species diversity, we chose to study ants for the following reasons: ants are abundant amongst the invertebrate fauna, making up 70% of individual insects in some areas (Wilson 1992). Ants contain many specialist species, show high species richness and occur higher up the trophic level (Anderson 1995, Majer 1996). An additional good reason for using ants is that the worker stage is present all year round, i.e. they are not highly seasonal like most other insect groups. They also meet other criteria essential for good indicator taxa (Noss 1990, Caro and O'Doherty 1999), in that ants are easy to sample, monitor and identify and are responsive to changing environmental conditions. These invertebrates have also already successfully been used in comparing alien invaded areas, with natural communities in fynbos vegetation in South Africa (Donnelly and Giliomee 1985b, French and Major 2001).

Ants play an important role in the success of rehabilitation. They facilitate soil aeration and drainage when creating channels and chambers for their nests (Abbott 1989). This then leads to an increase in porosity and water-holding capacity of the soil as well as the movement of a substantial amount of subterranean soil to the surface (Petal 1978). Ants play a vital role in nutrient cycling and energy flow (Folgarait 1998). Anderson and Sparling (1997) also found a correlation between microbial biomass and ant activity, providing useful information on the soil microbial populations and the quality of the soil organic matter. In symbiotic associations with plants, ants derive food and shelter from their hosts; they protect foliage, buds, and reproductive structures; they disperse and store seeds, and enrich the soil with essential nutrients (Beattie 1985, Holldobler and Wilson 1990).

In fynbos, as in the mediterranean–climate regions of Australia, ants are important in the dispersal of seeds (myrmecochory) (Berg 1975, Slingsby and Bond 1981, Milewski and Bond 1982, Bond and Slingsby 1983). Based on seed morphology, it is estimated that approximately 2,500 species of fynbos plants (30 % of species) are ant-dispersed (Bond and Slingsby 1983, Bond *et al.* 1991, Cowling *et al.* 1994). This includes 175 species (53%) of Proteaceae, 60% of the Restionaceae, 95% of the Rutaceae and 22% of threatened coastal fynbos (Slingsby 1982). Fynbos is fire-adapted, with different plants having evolved various reproductive strategies to deal with this disturbance. Some plants store seeds in the canopy of the parent plant within woody cones, while others resprout from underground storage organs. Other plants, occurring in many different families and a range of growth forms (Le Maitre and Midgley 1992), use ants to bury their seeds which are then protected from predators and the harsh post-dispersal environment. Indigenous ants favour seeds with a fleshy food body (elaiosome). The seed is picked up and carried underground to nutrient enriched nests. Here the food body is consumed and the seed discarded in middens (the rubbish dump of the nest). When a fire burns through the area, the nest may be damaged, but the seed will be able to germinate and grow (Bond and Slingsby 1983).

In disturbed areas of fynbos, such as in areas that have been invaded by pines and hakeas, ant diversity decreases (Donnelly and Giliomee 1985b, Koen and Breytenbach 1988). Here elaiosome-gathering ants are eliminated due to changes in the vegetation structure, microclimate and food content (Donnelly and Giliomee 1985b, Koen and Breytenbach 1988). Ant diversity also decreases under stands of Australian acacia species despite an abundance of food in the form of elaiosomes on acacia seeds (French and Major 2001). Hence, in a disturbed situation like this, it is evident that an alteration of the vegetation composition driven by one or a few invasive plant species can have a major effect on the survival of ant–dependent plant species (Richardson *et al.* 1995).

The aim of this study was to investigate how ant species diversity and composition was affected by the type of alien clearing methods employed at the Cape of Good Hope Nature Reserve. Are there differences in ant species composition between treatment and matched control areas and can ant diversity be used as an indicator of rehabilitation success following alien plant clearance?

Materials and Methods

Information pertaining to the history of the sites can be found in chapter 1.

Ant sampling

Pitfall traps were placed 2.5 meters from the corners of each quadrat and one meter towards the inside of the plot (Figure 1). The pitfall traps were plastic cups,

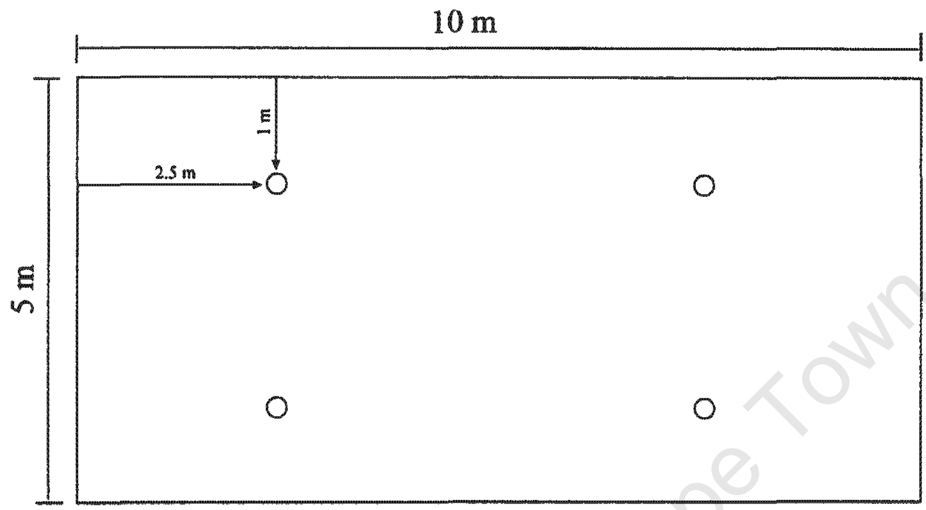


Figure 1. Placement of four pitfall traps within each 5 x 10 m quadrat.

9.5 cm deep with a diameter of 6.5 cm at the top, containing c. 75 ml of ethylene glycol. The pitfall trapping survey was conducted from late February to March for a period of one month. Four pitfalls were placed in each 5 x 10 m quadrat. For each plot, two samples were processed at random, yielding a total of 100 pitfall traps. Twice the number of pitfall traps was installed to counteract losses and breakage of pitfalls.

After 31 days, samples were collected and on return to the laboratory, saltwater extractions (Fisher, 1999) were used to separate organic matter from inorganic matter and the resultant cleaned. Samples were stored in 80 % ethanol. Ants were removed from the rest of the invertebrates and sorted to morphospecies based on external morphological characters. A reference collection for the reserve, housed at the South African Museum, was used for identifying species, and voucher specimens from each site were mounted and added to this collection. In cases where large numbers of ants of one species were found, the contents of the pitfalls were sub-sampled (method follows) for that species, from which the estimated total number was calculated. For sub-sampling all ant species were placed into a flat rectangular dish (area = 6600mm^2), and were evenly spread therein. A circular plastic tube, 706.9mm^2 in area, was then placed in the dish and the ants that occurred within the tube were removed and placed into a petri dish that was divided into 4 equal quarters. As it was only one ant species that was being estimated, that species was then counted in any quarter. The number of ants in that quarter was then multiplied by 4 and then multiplied by a conversion factor of 9.34 (6600 mm^2 / 706.9 mm^2) to give a total estimate of the number of specimens of that species.

Data Analysis

Only worker ants were used in the data analysis as queens are difficult to identify and their presence in pitfalls may simply be the result of them flying into the vicinity of the pitfalls rather than being resident in the area. Two measures of species abundance (frequency of occurrence and total individuals) were used to compare ant assemblages in the treatments and controls. Ants occur in colonies, and are thus patchily distributed. Therefore, frequency of occurrence often is a more accurate estimate of abundance than the total numbers of individuals (Andersen 1991). Species richness as well as diversity and evenness indices were calculated from these data. The diversity index takes both abundance and species richness into account.

Indices for diversity (H) and evenness (J) of ant communities for various treatments and controls were calculated using the Shannon-Wienner formulae:

$$H = - \Sigma P_i \log P_i$$

where P_i is the proportion of individuals in the *i*th pitfall represented by species P;

$$J = H/H_{(max)}$$

where $H_{(max)}$ is the maximum diversity if all species in the pitfall had equal numbers of individuals (Begon *et al.*,1995).

Similarities in community species composition between treatments and controls were explored by deriving a Bray-Curtis similarity matrix on transformed presence/absence data using the program PRIMER (Plymouth Routines in Multivariate Ecological Research). Data was transformation to reduce the contribution of superabundant species.

The program EstimateS was used to compute the incidence-based coverage estimator (ICE) which is a non-parametric species richness estimator. Computations were based on 50 randomised sample orders for each treatment/control (Colwell and Coddington 1994).

Estimates of species richness were analysed graphically, by plotting the estimated and observed species richness as a function of the cumulative number of pitfalls sampled.

To assess whether different alien clearing treatments have had any significant impacts on ant community composition at the three localities in the Cape of Good Hope Nature Reserve, a two-step process was conducted. The first step was to reduce the multivariate species and site matrix into a single variate using Principal Components Analysis (PCA) (Ludwig and Reynolds 1988). Thereafter orthogonal contrasts (Steel and Torrie 1980) were used to make comparisons between treatments and controls and between the controls themselves. PCA has the advantage of simultaneously producing species and sample scores in one integrated analysis (Gauch 1982). The sample scores (in this case from the pitfall traps), was then used for the analysis of variance for the ant abundance data. A second multivariate technique, Detrended Correspondence Analysis (DCA) was also performed. This technique is usually superior to other ordination techniques and is computationally very efficient (Gauch 1982). Data were log transformed to reduce the contribution of superabundant species.

Most of the species found in this study were generalist myrmecines (such as *Crematogaster*, *Tetramorium*, *Pheidole*, *Monomorium*, *Cardiocondyla*) and generalist formicines (*Camponotus*) and scavengers (e.g. *Ocymyrmex, Plagiolepis*), rather than encompassing a range of functional groups. It was for this reason and a general lack of knowledge of the ecology of individual species that it was not feasible to divide the species into functional groups, such as was done by Andersen (1997). The only exception was those taxa known to disperse the seeds of fynbos plants and these are considered in this study.

Results

Composition of the ant fauna

A total of 42817 individual ants were identified, comprising 44 species and 18 genera (Table 1). The most species rich genera were *Tetramorium* (12 species), *Camponotus* (7 species) and *Monomorium* (5 species) respectively. Overall, the most numerous species caught was *Crematogaster peringueyi* which occurred in all treatments and controls. This predominantly arboreal species was by far the most abundant ant at the Klaasjagersberg site (94 % of all ants counted at this site belonged to this species). The second and third most common species caught, *Pheidole capensis* and *Anoplolepis steingroeveri* were also found at all the sites. At the Meadows treatment site *A. steingroeveri* was the most abundant, but in both the Meadows control and all the Platboom study sites *P. capensis* was the dominant ant in terms of abundance. Two invasive alien ant species, the Argentine Ant, *Linepithema humile,* and *Technomyrmex albipes* were amongst the ten most abundant species. The last-mentioned species was found in all study sites except one of the control sites (M4), whereas *L. humile* was trapped at only two study sites, both in the "bulldozed and aerial-sprayed" treatment. As many as nine, out of a total of 44 ant species only occurred once.

In the "boom-sprayed" treatment the mean abundance of *Crematogaster peringueyi* was significantly lower compared to the control site, whereas *Tetramorium quadrispinosum, Lepisiota capensis* and *Pheidole capensis* were significantly more abundant in the same treatment compared to the control (Table 3). The latter two species (both ground nesting) were also found to be significantly less abundant in the "bulldozed & aerial-sprayed" treatment and *T. quadrispinosum,* very nearly approached a significant value (P = 0.06).

Effect of clearing treatments on ant variables

The Klaasjagersberg locality and its treatment sites

The highest abundance of ants amongst the ten study sites was attained in the "bulldozed & aerial-sprayed" treatment with a mean of 2194 ants (Table 4). Although ant abundance was high in this treatment, it did not differ significantly to the Klaasjagersberg control site (Table 5). The ant diversity was significantly lower however, when compared to the undisturbed control site with the species richness and evenness also very nearly approaching a significantly lower value (P=0.06). Within site treatment comparisons

Table 1. Abundance and species richness of ants at the 10 study sites within the Cape of Good Hope Nature Reserve. For each entry, the first number indicates the total number of individuals and in parentheses the number of pitfalls that trapped the particular species.
At each of the three localities, at least one treatment site was chosen and one control site which had natural vegetation and which had never been cleared of alien vegetation.

Species	Localities and treatments										
	Klaasjagersberg				Meadows		Platboom				
	K1a: burn bulldozed & aerial sprayed (n=10)	K1b: burn bulldozed & aerial sprayed (n=10)	K2: burn & aerial sprayed (n=10)	K3: Control Klaasjagersberg (n=10)	M5: burnt & foliar sprayed (n=10)	M4: Control Meadows (n=10)	P6a: burnt & boomsprayed (n=10)	P6b: burnt & boomsprayed (n=10)	P7: burnt & foliar sprayed (n=10)	P8: Control Platboom (n=10)	Total individuals (n=100)
Subfamily Cerapachyinae											
Cerapachys sp. 1 (cribinodis -group)	0(0)	0(0)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Cerapachys sp. 2	0(0)	0(0)	1(1)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	1(1)	3
Subfamily Dolichoderinae											
Linepithema humile	220(4)	72(4)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	292
Technomyrmex albipes	92(2)	1(1)	7(2)	5(2)	7(3)	0(0)	48(5)	16(3)	6(2)	17(4)	199
Subfamily Formicinae											
Anoplolepis steingroeveri	6(3)	2(2)	13(3)	1(1)	1012(6)	41(3)	19(7)	19(5)	13(3)	191(9)	1317
Camponotus angusticeps	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	2(2)	2
Camponotus baynei	0(0)	0(0)	1(1)	6(2)	0(0)	3(3)	0(0)	0(0)	0(0)	0(0)	10
Camponotus bertolonii	0(0)	0(0)	0(0)	0(0)	0(0)	1(1)	0(0)	1(1)	1(1)	0(0)	3
Camponotus cognatus	0(0)	5(2)	2(2)	16(5)	0(0)	2(2)	19(4)	2(2)	15(7)	31(8)	92
Camponotus klugii	0(0)	0(0)	0(0)	0(0)	0(0)	2(2)	0(0)	0(0)	0(0)	0(0)	2
Camponotus niveosetosus	9(3)	3(2)	1(1)	28(7)	1(1)	17(6)	7(6)	7(4)	6(4)	1(1)	80
Camponotus sp. 4 (emarginatus -group)	0(0)	0(0)	0(0)	0(0)	0(0)	3(2)	0(0)	0(0)	5(2)	3(2)	11
Lepisiota capensis	35(3)	18(5)	143(10)	131(10)	101(6)	60(8)	126(10)	254(10)	298(10)	116(9)	1282
Lepisiota laevis	0(0)	0(0)	1(1)	1(1)	0(0)	21(1)	11(2)	150(3)	0(0)	193(2)	377
Lepisiota sp. 1	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1(1)	1
Plagiolepis sp. 2	0(0)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	1(1)	0(0)	2
Subfamily Myrmicinae											
Cardiocondyla shuckardi	0(0)	0(0)	0(0)	0(0)	0(0)	8(3)	1(1)	0(0)	0(0)	6(2)	15
Crematogaster peringueyi	4430(5)	21632(6)	2406(10)	1490(10)	27(2)	214(5)	6(2)	77(8)	410(8)	270(7)	30962
Leptothorax denticulatus	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Monomorium fastidium	0(0)	1(1)	0(0)	0(0)	3(2)	73(10)	0(0)	2(1)	4(1)	0(0)	83
Monomorium rhopalocerum	1(1)	2(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	3

Monomorium sp. 4	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Monomorium tablense	10(4)	0(0)	11(4)	1(1)	0(0)	3(2)	2(2)	84(3)	1(1)	15(4)	127
Monomorium xanthognathum	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Ocymyrmex barbiger	1(1)	14(4)	0(0)	4(3)	8(3)	8(5)	23(6)	35(7)	31(9)	21(8)	145
Pheidole capensis	73(3)	0(0)	198(7)	80(7)	376(10)	2195(9)	1276(10)	1262(10)	663(10)	406(8)	6529
Pheidole sp.2	137(1)	4(2)	0(0)	2(1)	0(0)	0(0)	0(0)	49(1)	0(0)	0(0)	192
Solenopsis punctaticeps	24(2)	1(1)	22(2)	2(1)	11(5)	18(5)	11(5)	38(6)	55(9)	19(6)	201
Tetramorium akermani	0(0)	0(0)	0(0)	0(0)	40(3)	0(0)	0(0)	0(0)	0(0)	0(0)	40
Tetramorium erectum	2(1)	0(0)	17(1)	1(1)	1(1)	0(0)	4(2)	0(0)	5(4)	3(3)	33
Tetramorium frigidum	12(7)	34(5)	4(2)	2(1)	0(0)	2(2)	19(3)	6(2)	9(4)	7(7)	95
Tetramorium grassii	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1(1)	0(0)	0(0)	1
Tetramorium pusillum	0(0)	12(3)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	17(1)	29
Tetramorium quadrispinosum	81(9)	130(3)	85(8)	46(9)	0(0)	0(0)	102(10)	87(10)	10(3)	0(0)	541
Tetramorium regulare	8(5)	12(2)	6(3)	4(1)	0(0)	4(2)	3(2)	3(1)	12(6)	5(3)	57
Tetramorium sp. 1 (oculatum complex)	10(4)	1(1)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	2(2)	7(3)	21
Tetramorium sp. 10	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1(1)	1
Tetramorium sp. 4 (squaminode - group)	0(0)	0(0)	4(3)	0(0)	0(0)	6(2)	6(1)	11(4)	5(2)	5(1)	37
Tetramorium sp. 8 (squaminode - group)	0(0)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	2(1)	0(0)	0(0)	3
Tetramorium sp.9	0(0)	0(0)	0(0)	0(0)	2(2)	0(0)	3(3)	1(1)	0(0)	0(0)	6
Subfamily Ponerinae											
Hypoponera sp. 2	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Leptogenys castanea	0(0)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	1
Leptogenys peringueyi	0(0)	0(0)	1(1)	0(0)	0(0)	4(3)	0(0)	0(0)	1(1)	0(0)	6
Subfamily Pseudomyrmecinae											
Tetraponera clypeata	0(0)	1(1)	10(2)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)	11
Total no. of individuals :	5153	21947	2936	1822	1590	2685	1686	2107	1553	1338	42817
Species Richness:	19	20	22	19	13	20	18	21	21	23	44

Table 2. Percentage similarity (Bray-Curtis) of ant species between the 10 study sites in the CGHNR.

Study sites	Localities and treatments									
	Klaasjagersberg				Meadows		Platboom			
	K1a: burn bulldozed & aerial sprayed	K1b: burn bulldozed & aerial sprayed	K2: burn & aerial sprayed	K3: Control Klaasjagersberg	M5: burnt & foliar sprayed	M4: Control Meadows	P6a: burnt & boomsprayed	P6b: burnt & boomsprayed	P7: burnt & foliar sprayed	P8: Control Platboom
K1a										
K1b	71.79									
K2	58.54	52.38								
K3	78.95	66.67	73.17							
M5	56.25	48.48	51.43	56.25						
M4	51.28	50.00	66.67	66.67	48.48					
P6a	70.27	57.89	75.00	81.08	64.52	73.68				
P6b	65.00	63.41	69.77	75.00	58.82	73.17	82.05			
P7	70.00	63.41	74.42	75.00	58.82	78.05	76.92	76.19		
P8	61.90	55.81	66.67	71.43	55.56	69.77	78.05	63.64	72.73	

Table 3. Abundance of the 10 most common species found within the 10 study sites in the CGHNR (means, standard deviation in parentheses, n=10).

The values in superscript are the probabilities of significant differences between treatment and control (the Mann-Whitney U-test was used).

Significant treatments (P<=0.05) are indicated in bold.

Species	Localities and treatments									
	Klaasjagersberg				Meadows		Platboom			
	K1a: burn bulldozed & aerial sprayed	K1b: burn bulldozed & aerial sprayed	K2: burn & aerial sprayed	K3: Control Klaasjagersberg	M5: burnt & foliar sprayed	M4: Control Meadows	P6a: burnt & boomsprayed	P6b: burnt & boomsprayed	P7: burnt & foliar sprayed	P8: Control Platboom
Crematogaster peringueyi	443 (933.83) $^{0.07}$	2163.2 (2046.59) $^{0.45}$	240.60 (351.14) $^{0.97}$	149.00 (213.97)	2.70 (7.86) $^{0.15}$	21.40 (31.95)	**0.60 (1.58)** $^{\mathbf{0.02}}$	7.70 (10.83) $^{0.33}$	41.00 (41.05) $^{0.31}$	27.00 (41.84)
Pheidole capensis	7.3 (14.55) $^{0.34}$	**0.00 (0.00)** $^{\mathbf{0.01}}$	19.80 (23.85) $^{0.41}$	8.00 (11.41)	37.60 (31.92) $^{0.11}$	219.50 (368.94)	**127.60 (129.31)** $^{\mathbf{0.01}}$	**126.20 (111.46)** $^{\mathbf{0.01}}$	66.30 (101.02) $^{0.27}$	40.60 (63.75)
Anoplolepis steingroeveri	0.6 (1.07) $^{0.41}$	0.2 (0.42) $^{0.71}$	1.30 (3.43) $^{0.43}$	0.10 (0.32)	101.20 (134.23) $^{0.11}$	4.10 (12.29)	1.90 (2.12) $^{0.29}$	1.90 (3.03) $^{0.10}$	**1.30 (2.37)** $^{\mathbf{0.04}}$	19.10 (39.27)
Lepisiota capensis	**3.5 (7.65)** $^{\mathbf{0.01}}$	**1.8 (2.62)** $^{\mathbf{0.00}}$	14.30 (13.74) $^{0.82}$	13.10 (10.02)	10.10 (25.75) $^{0.17}$	6.00 (4.55)	12.60 (11.21) $^{0.85}$	**25.40 (10.04)** $^{\mathbf{0.01}}$	29.80 (37.58) $^{0.31}$	11.60 (9.32)
Tetramorium quadrispinosum	8.1 (9.98) $^{0.60}$	13 (27.15) $^{0.06}$	8.50 (8.85) $^{0.65}$	4.60 (3.72)	0.00 (0.00) $^{1.00}$	0.00 (0.00)	**10.20 (4.92)** $^{\mathbf{0.00}}$	**8.70 (7.36)** $^{\mathbf{0.00}}$	1.00 (2.49) $^{0.26}$	0.00 (0.00)
Lepisiota laevis	0.00 (0.00) $^{0.71}$	0.00 (0.00) $^{0.71}$	0.10 (0.32) $^{1.00}$	0.10 (0.32)	0.00 (0.00) $^{0.71}$	2.10 (6.64)	1.10 (2.43) $^{0.88}$	15.00 (43.98) $^{0.76}$	0.00 (0.00) $^{0.45}$	19.30 (43.48)
Linepithema humile	22(46.82) $^{0.13}$	7.2 (15.94) $^{0.13}$	0.00 (0.00) $^{1.00}$	0.00 (0.00)	0.00 (0.00) $^{1.00}$	0.00 (0.00)	0.00 (0.00) $^{1.00}$	0.00 (0.00) $^{1.00}$	0.00 (0.00) $^{1.00}$	0.00 (0.00)
Solenopsis punctaticeps	2.4 (6.04) $^{0.65}$	0.10 (0.32) $^{0.97}$	2.20 (6.29) $^{0.68}$	0.20 (0.63)	1.10 (1.45) $^{0.73}$	1.80 (2.53)	1.10 (1.37) $^{0.55}$	3.80 (4.34) $^{0.41}$	5.50 (7.37) $^{0.15}$	1.90 (2.28)
Technomyrmex albipes	9.2 (28.06) $^{0.94}$	0.10 (0.32) $^{0.88}$	0.70 (1.89) $^{0.97}$	0.50 (1.26)	0.70 (1.25) $^{0.26}$	0.00 (0.00)	4.80 (9.40) $^{0.57}$	1.60 (2.68) $^{0.85}$	0.60 (1.35) $^{0.43}$	1.70 (2.95)
Pheidole sp. 2	13.7 (43.32) $^{0.97}$	0.40 (0.97) $^{0.71}$	0.00 (0.00) $^{0.71}$	0.20(0.63)	0.00 (0.00) $^{1.00}$	0.00 (0.00)	0.00 (0.00) $^{1.00}$	4.90 (15.50) $^{0.71}$	0.00 (0.00) $^{1.00}$	0.00 (0.00)

Table 4. Total ant abundance, species richness, diversity and evenness per pitfall at 10 study sites in the CGHNR (means, SD in parentheses, n=10).

Ant variables	Localities and treatments									
	Klaasjagersberg				Meadows		Platboom			
	K1a: burn bulldozed & aerial sprayed	K1b: burn bulldozed & aerial sprayed	K2: burn & aerial sprayed	K3: Control Klaasjagersberg	M5: burnt & foliar sprayed	M4: Control Meadows	P6a: burnt & boomsprayed	P6b: burnt & boomsprayed	P7: burnt & foliar sprayed	P8: Control Platboom
Total number of individuals	515.3 (912.00)	2194.7 (2055.88)	293.6 (340.48)	182.2 (226.597)	159 (116.95)	268.5 (354.71)	168.6 (150.4)	210.7 (139.83)	155.3 (117.39)	133.8 (100.15)
Species richness	6 (2.71)	4.8 (2.09)	6.6 (2.55)	6.5 (6.05)	4.5 (2.01)	7.6 (1.95)	8.1 (1.85)	8.4 (2.27)	9 (1.76)	9.3 (2.7)
Diversity (h)	0.33 (0.22)	0.14 (0.21)	0.41 (.26)	0.40 (0.38)	0.24 (0.18)	0.35 (0.19)	0.45 (0.12)	0.55 (0.18)	0.54 (.22)	0.59 (.19)
Evenness (j)	0.44 (0.26)	0.23 (0.34)	0.48 (.25)	0.50 (0.491)	0.35 (0.20)	0.39 (0.18)	0.51 (0.16)	0.61 (0.16)	0.57 (.21)	0.62 (.15)

indicate that when the "bulldozed and aerial-sprayed" treatment was compared to the "aerial-sprayed" treatment the ant diversity and evenness was also significantly lower (P=0.018).

A comparison of the abundance, richness, diversity and evenness (Table 4) between the "aerial-sprayed" treatment and the control indicates that the ant variables were very similar. In-fact, none of the variables differed significantly to the control (Table 5).

<u>The Platboom locality and its treatment sites</u>

The Platboom control supported the most rich and diverse ant community leading to a high evenness of 0.62. Overall, ant species richness, diversity and evenness were high at Platboom, with treatments only slightly less than the control for these three variables (Table 4). Results of the ANOVA, based on the first principal components score (explaining 40% of the variation in the ant abundance data), indicate that it was only the "boom-spraying" treatment that differed significantly to its control site in terms of ant abundance (Table 5). Within site treatment comparisons indicated that the "boom-sprayed" treatment differed significantly to the "foliar-sprayed" treatment at Platboom.

The ant variables were similar between the "foliar-sprayed" treatment (P7) and the Platboom control (Table 4). None of the variables differed significantly (Table 5).

<u>The Meadows locality and its treatment sites</u>

The abundance, species richness, diversity and evenness were all lower in the "foliar-sprayed" treatment site at the Meadows compared to its control (M4)(Table 4), although only the species richness was significantly lower (P<0.001) (Table 5).

Ant species richness was plotted against plant variables such as: 1) plant species richness at the plot scale (Figure 2a) and the point scale (Figure 2b) 2) abundance 3) diversity 4) evenness 5) percentage canopy cover and 6) percentage bare ground and 7) vegetation age (Figure 2c). All variables (except bare ground and vegetation age) were positively correlated, although only the species richness at the point scale was significantly correlated with ant species richness (where r = 0.38, P=0.0065). These figures indicate that at a high plant species richness ant species richness is also high, but at a low plant species richness ant richness can be either high or low.

Table 5. Results of the ANOVA's for abundance, species richness, diversity and evenness of ants. Principal components analysis reduced the abundance matrix into a score for each pitfall. Thereafter the ANOVA (of which the results are displayed below) was conducted for ant abundance. Significant treatments (P<=0.05) are indicated in bold.

	Abundance / Species Composition					**Species richness**					**Diversity**					**Evenness**				
Source of Variation	d.f.	s.s	m.s	v.r	F.pr	df	s.s	m.s	v.r	F.pr.	df	s.s	m.s	v.r	F.pr.	df	s.s	m.s	v.r	F.pr.
All treatments and controls	**9**	**405.95**	**45.11**	**9.26**	**< .001**	**9**	**1.33**	**0.15**	**6.04**	**<.001**	**9**	**1.83**	**0.20**	**5.16**	**<0.001**	**9**	**1.35**	**0.15**	**3.03**	**0.003**
Within site treatment versus control																				
K1a+K1b vs K3	1	0.11	0.11	0.02	0.883	1	0.09	0.88	3.62	0.06	1	0.19	0.19	4.75	**0.032**	1	0.19	0.19	3.75	0.056
K2 vs K3	1	2.64	2.64	0.54	0.463	1	0.33	0.33	1.37	0.245	1	0.10	0.10	2.63	0.108	1	0.06	0.06	1.14	0.289
M5 vs M4	1	1.90	1.90	0.39	0.534	1	0.30	0.30	12.34	**<0.001**	1	0.06	0.06	1.44	0.234	1	0.01	0.01	0.12	0.734
P6a+P6b vs P8	**1**	**26.55**	**26.55**	**5.45**	**0.022**	1	0.01	0.01	0.51	0.477	1	0.06	0.06	1.40	0.241	1	0.03	0.03	0.55	0.461
P7 vs P8	1	0.27	0.27	0.06	0.814	1	0.00	0.00	0.00	0.983	1	0.01	0.01	0.37	0.546	1	0.01	0.01	0.29	0.591
Between site control comparisons																				
K3 vs M4	**1**	**67.02**	**67.02**	**13.76**	**< .001**	1	0.02	0.02	0.94	0.336	1	0.01	0.01	0.34	0.561	1	0.07	0.07	1.43	0.235
M4 vs P8	1	5.44	5.44	1.12	0.293	1	0.03	0.03	1.34	0.25	1	0.30	0.30	7.51	**0.007**	1	0.28	0.28	5.62	**0.02**
K3 vs P8	**1**	**65.72**	**65.72**	**13.49**	**<0.001**	1	0.08	0.08	3.19	0.078	1	0.03	0.03	0.82	0.367	1	0.00	0.00	0.00	0.991
Within site treatment comparisons																				
K1a+K1b vs K2	1	0.03	0.03	0.01	0.936	1	0.04	0.04	1.48	0.226	1	0.23	0.23	5.82	**0.018**	1	0.20	0.20	3.99	**0.049**
P6a+P6b vs P7	**1**	**33.12**	**33.12**	**6.80**	**0.011**	1	0.01	0.01	0.48	0.492	1	0.01	0.01	0.23	0.631	1	0.00	0.00	0.01	0.907

The observed species accumulation curves levelled off after eight samples for the treatments and controls at the Meadows and Platboom, but not so for Klaasjagersberg (Figure 3). The ICE accumulation curves for Klaasjagersberg showed the sites containing the Argentine Ant levelling off at a low diversity; Treatment 2 levelled off at a higher diversity and the control was still increasing after 10 samples.

<u>Effect of clearing treatments on ant composition</u>

The ordination (Figure 4) shows three distinct groups. Pitfalls from Treatment 1 ("bulldozed & aerial-sprayed") and some samples from Treatment 5 ("foliar-sprayed") separated in different directions from the main cluster. In the case of the former treatment, the eight pitfalls, which occurred as a separate entity to the main cluster, contained the Argentine ant, whereas the four pitfalls in "foliar-sprayed" treatment of the Meadows had large patches of bare ground, which were largely associated with high abundances of *Anoplolepis steingroeveri*.

When controls were compared to each other (Figure 5), Klaasjagersberg formed a clearly separate group while there was overlap between the Meadows and Platboom controls. Environmental variables such as soil pH and depth were also included in the analysis, and most of the variation between the species and environment data was explained by the second axis (44.8%). Arrows in the diagram indicate the direction of increase of the variables involved.

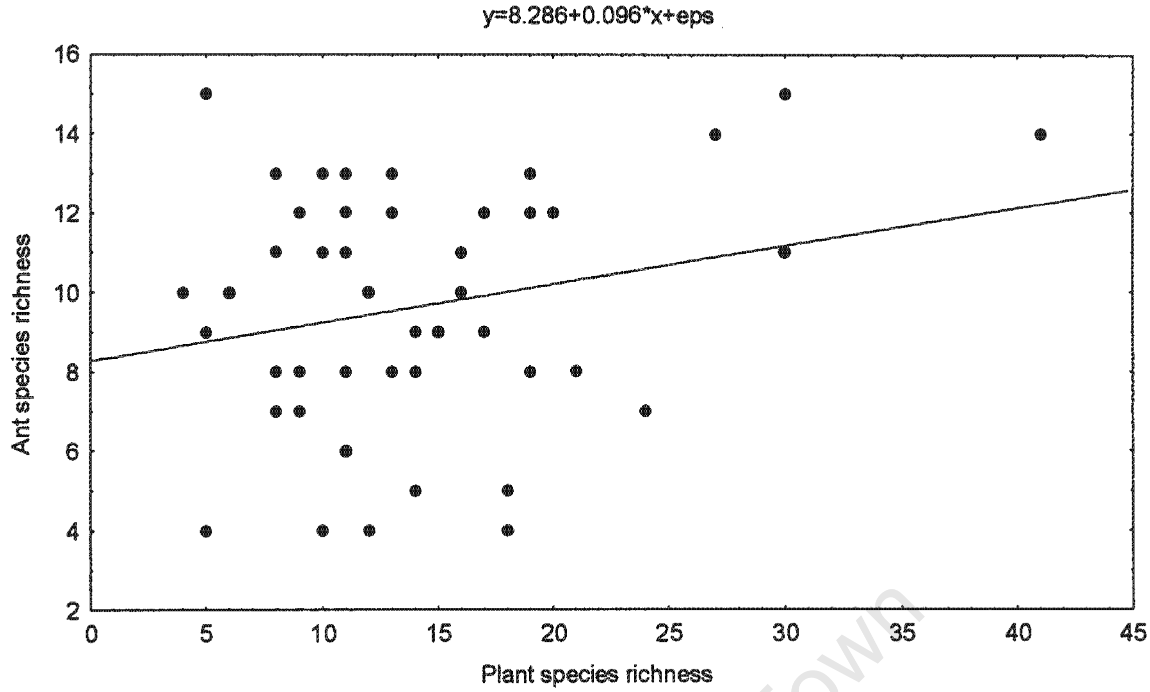


Figure 2a. Correlation between plant species richness and ant species richness in 50 m^2 quadrats ($r^2 = 0.048$, $P = 0.123$).

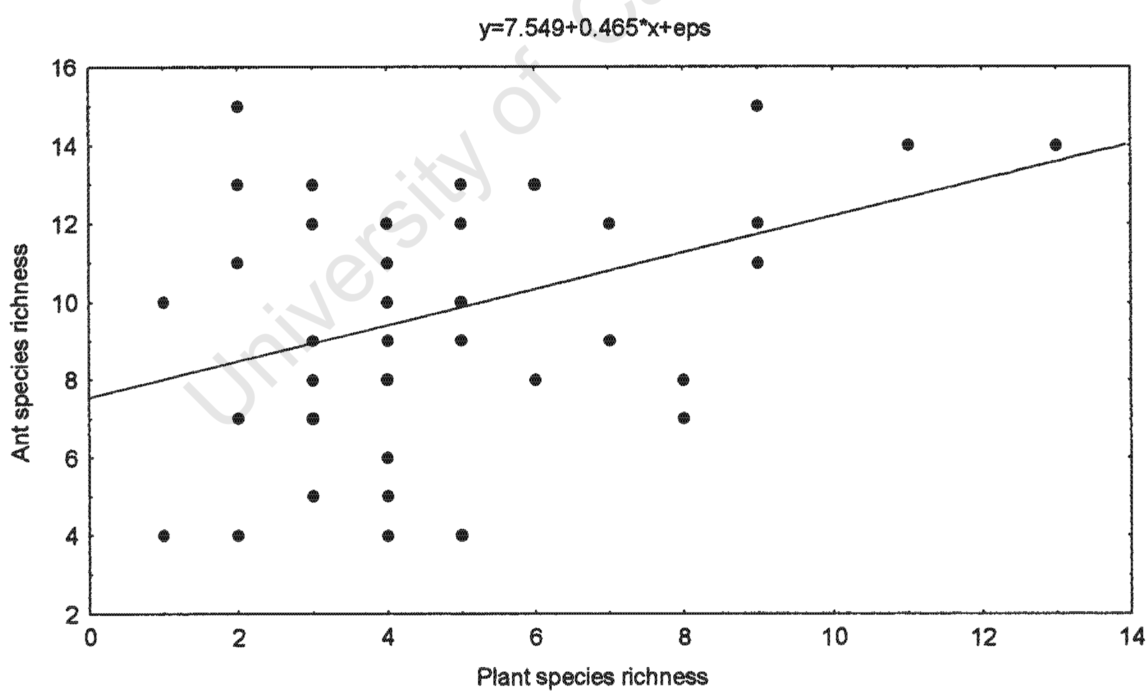


Figure 2b. Correlation between plant species richness and ant species richness where plant species were calculated within a 1 x 1 m quadrat ($r^2 = 0.144$, $P = 0.0065$).

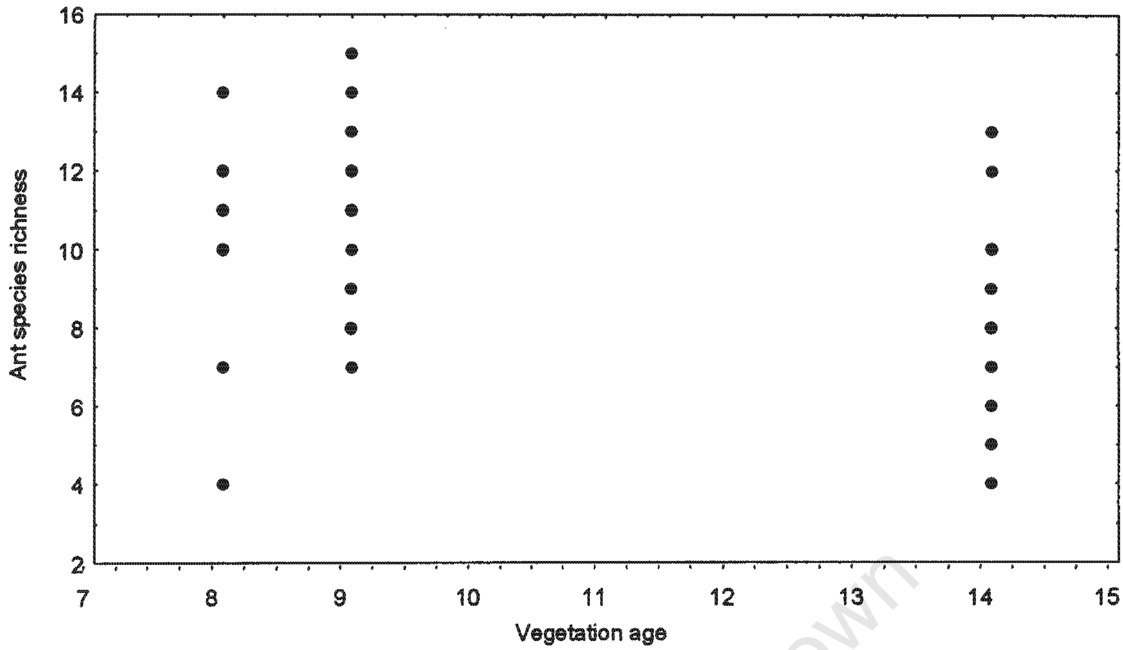


Figure 2c. Correlation between vegetation post-fire age and ant species richness

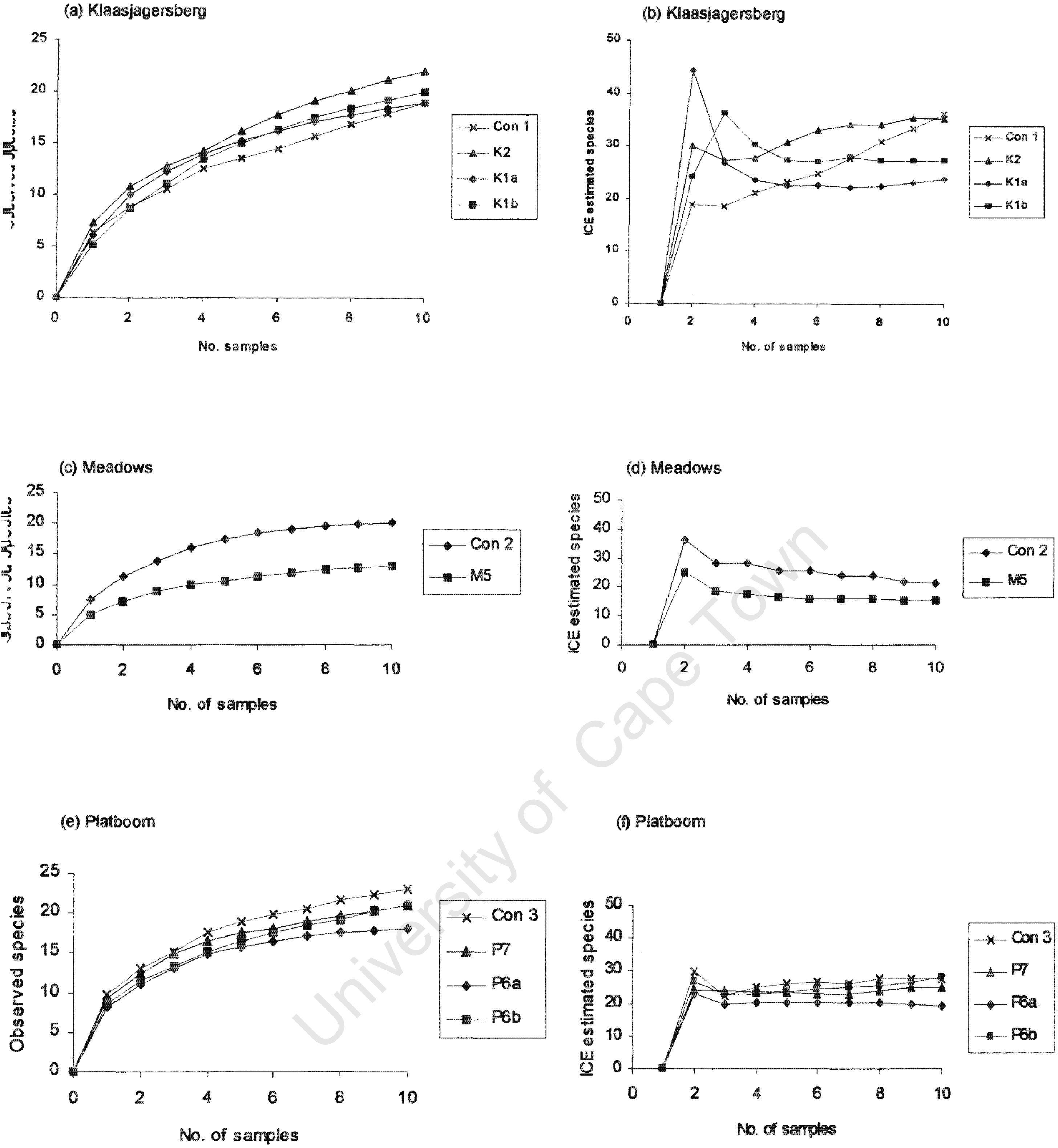


Figure 3. Assessment of pitfall trapping sampling techniques for three sites within the reserve. Graphs on the left shows species accumulation curves plotted for the observed number of species as a function of the number of pitfalls. The graphs on the right is the estimated species richness after 50 randomizations of sample order using the program EstimateS.

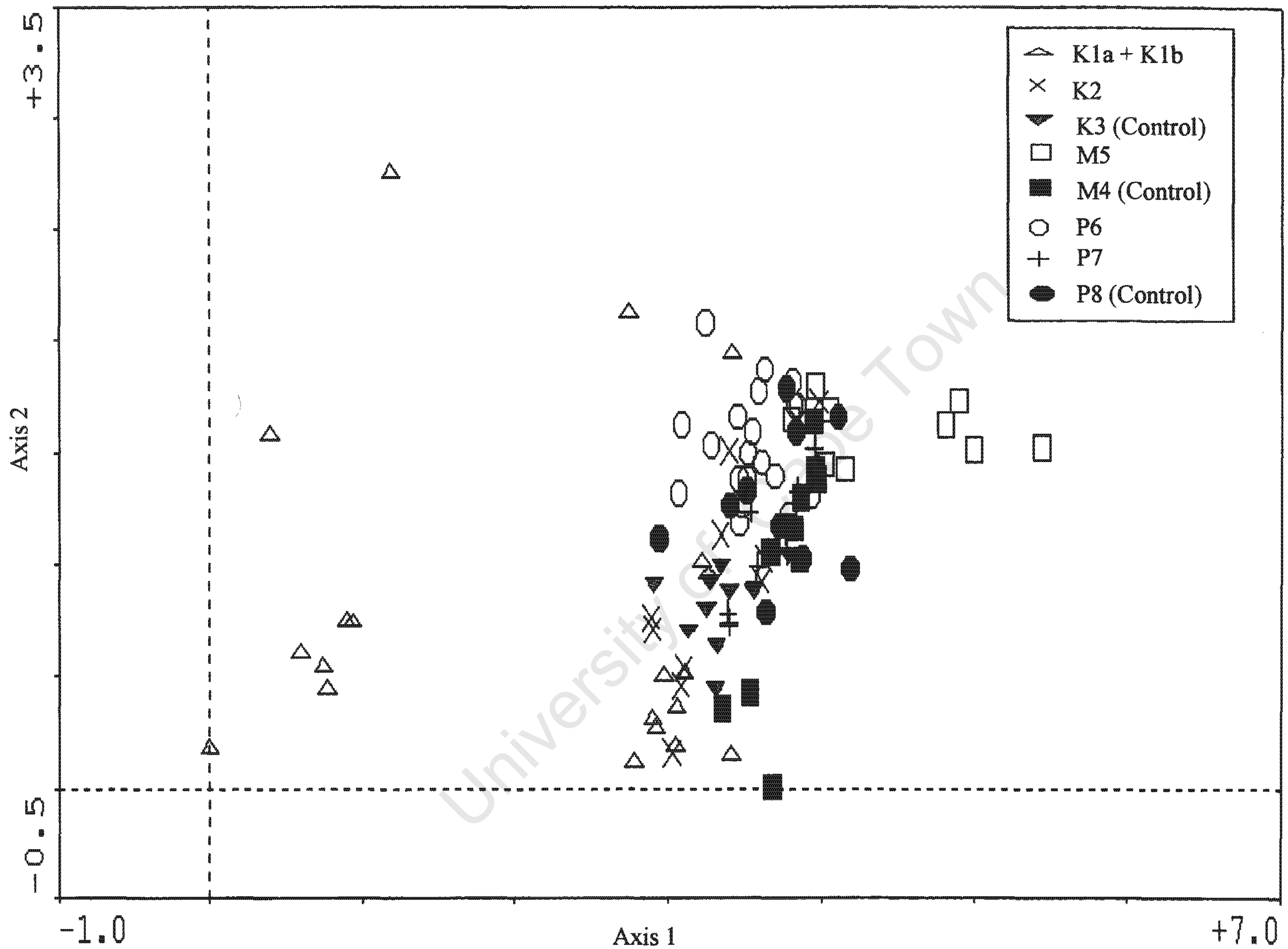


Figure 4. Detrended correspondence analysis of log transformed ant abundance data for all treatments and controls. Eigenvalues for axis one and two are 0.631 and 0.241 respectively.

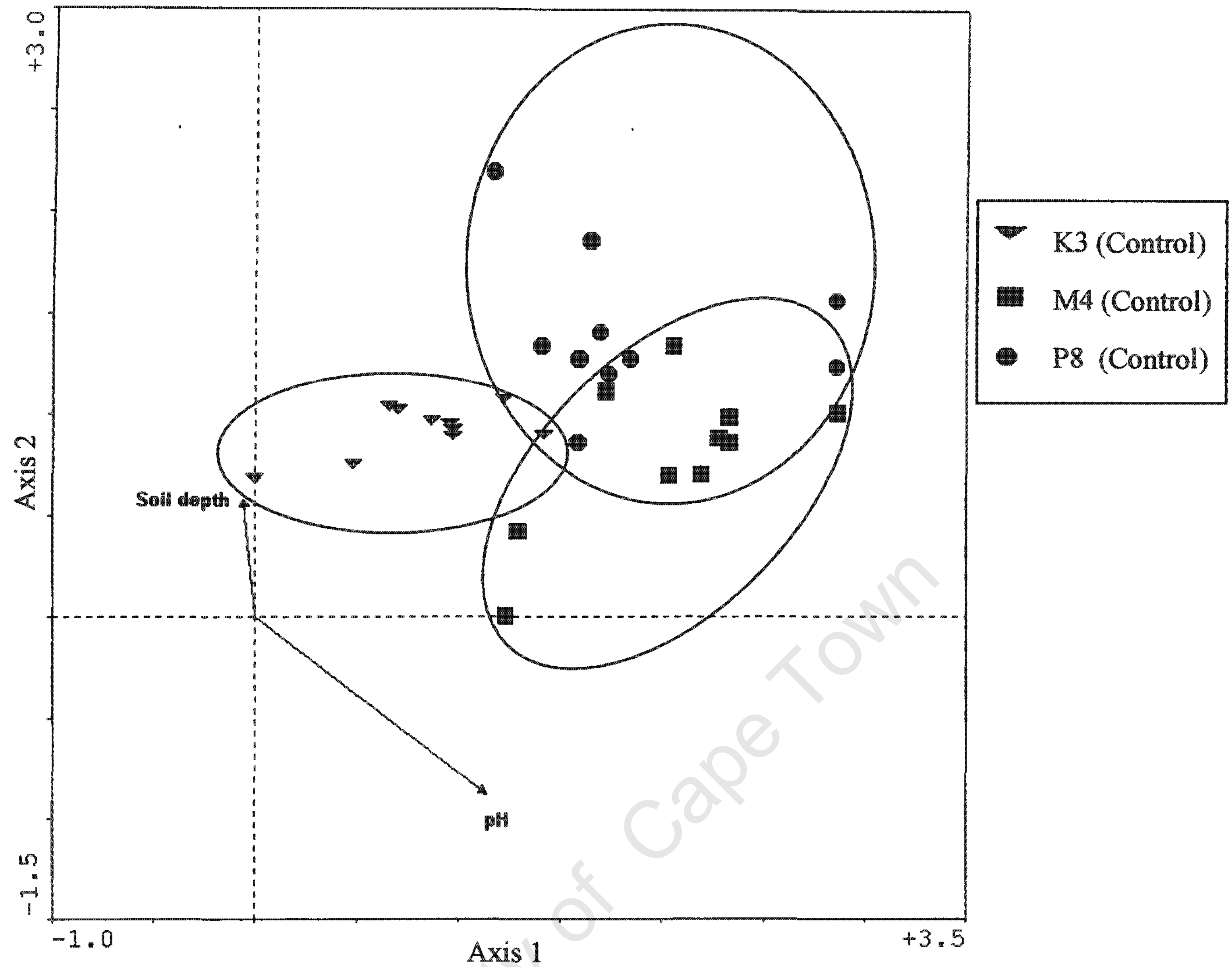


Figure 5. Detrended Correspondence analysis of log transformed ant abundance data for three control sites within the reserve. Eigenvalues for axis 1 and 2 are .391 and .208 respectively.

Discussion:

The number of ant species found in this study (44) is similar to that found in studies conducted by Donnelly and Giliomee (1985a) and Cody *et al.* (1979). In the former study, ants were collected by pitfall trapping in fynbos vegetation of the Jonkershoek valley near Stellenbosch; 45 species were found. Cody *et al.* (1979) recorded a total of 45 ant species in Californian chaparral, which is physiognomically similar to fynbos.

<u>The impact of the "bulldozed and aerial-sprayed" treatment on ants</u>

The bulldozed and aerial-sprayed site had a significantly lower ant species diversity compared to the Klaasjagersberg control site. The species richness was also quite low in this region and very nearly approached a significantly lower value when compared with the undisturbed control site. Despite this reduction in ant species richness, the abundance of ants increased in this site due to the dominance of *Crematogaster peringueyi* and *Linepithema humile*. Folgarait (1998) reported that this was the case in many events where different types and extents of land-use practices caused a reduction in richness and an increase in abundance due to dominance of aggressive or exotic ants. The evenness at this site was also low as a result of the dominance of the above two species.

The reduction in diversity at this site could be attributed to its long history of disturbance (see Chapter One), with bulldozing being but one of these disturbances. Studies in the forestry industry have shown that soil damage takes place as a result of vehicle movement (Greacen and Sands 1980), and Warkotsch *et al.* (1994) stated that compaction (which could be brought about by heavy machinery) leads to a reduction in pore volume which in-turn affects life within in the sandy soil.

<u>The impact of bulldozing on ant species composition with special reference to the Argentine Ant (*Linepithema humile*)</u>

Ordination by detrended correspondence analysis of the "bulldozed and aerial sprayed" treatment with the control (Figure. 4) indicates: two separate clusters of points for the "bulldozed and aerial sprayed" treatment. The eight pitfalls that contained the Argentine Ant, *L. humile* formed a separate cluster to the left while the 12 pitfalls without *L. humile* clustered closely with the control pitfalls. In fact one pitfall trap contained only this species. Hence, the grouping of the pitfalls to the left is most likely the result of changes in ant species composition brought about by the Argentine Ant. The diversity of the ant species was significantly reduced in the "bulldozed and aerial-sprayed site (Table. 5), and the pitfalls containing *L. humile* generally contained fewer species. *L. humile* has been reported to replace dominant indigenous ants in many systems, radically altering the diversity and composition of native communities (Hattingh, 1945; Skaife 1961; Crowell 1968; Bond and Slingsby 1984; De Kock *et al.* 1992; Ward 1987).

The ability of *L. humile* to reduce ant species richness is related to the fact that *L. humile* is omnivorous and has generalist feeding and nesting needs. Studies conducted in Northern California, show that *L. humile* workers are very efficient foragers, recruiting to food sources in high numbers and more quickly than native ants (Human and Gordon 1999). The native species may then suffer decreased foraging success in invaded areas, as a result of retreating from aggressive or passive encounters initiated by the Argentine Ant at food sources. If reduced foraging ability by native species leads to mortality or emigration, this invasive ant would successfully have displaced native species. Thus they manage to out compete native species by interference and exploitation of resources in the system (Human and Gordon 1996).

It is not all native species however, that *L. humile* can oust from the ant community. For instance *T. quadrispinosum* manages an uneasy co-existence with the Argentine Ant (Bond and Slingsby 1984). To occur together, species need to differ from each other in their use of available resources like food and nesting sites (Begon *et al.* 1995), with some species managing this by temporal separation of foraging times. In fact, the greater the difference in niches between species, the more likely they are to co-exist (Holldobler and Wilson 1990). In the case of *Tetramorium quadrispinosum*, it manages to co-exist with *L. humile* by closing its nest entrances after foraging periods (Witt 1993), thereby reducing the possibility of *L. humile* attacking workers as has been observed in excavated nests (Witt and Giliomee 1999).

Other factors besides its feeding requirements, which makes *L. humile* a successful invader includes it's nesting needs. They nest in the ground, and have many queens in a nest, which means that expansion of the colony can take place rapidly, and new colonies can be formed very easily (Majer 1994).

L. humile is often found in areas disturbed by humans (De Kock and Giliomee 1989; Ward 1987), and in this study the same applies. The area containing this species is close to an old homestead, which would have acted as the focus for the introduction of the Argentine Ant. The plots were situated close to a river and had a very high grass cover offering sufficient shade and reduced soil temperatures for this ant species to thrive. Witt and Giliomee (1999) showed that *L. humile* was active at low soil temperatures (15-19^{0}C). Studies looking at the habitat preference of *L. humile* in fynbos showed this ant to prefer old moist vegetation (Donnelly and Giliomee 1985, Kemp 1995), and Ward (1987) also found the Argentine Ant to be more closely associated with a permanent water source than native ants. In contrast to these cases, a study conducted by Human *et al.* (1998) showed that although *L. humile* tended to be firmly established in areas close to water, there were many areas far away from water in which these ants were also well established.

The presence of the Argentine Ant in this study site has very serious implications for the persistence of ant-dependent plant species due to the mutualistic relationship shared between many fynbos plant and ant species. Bond and Slingsby (1984) reported that in areas infested by the Argentine ant, seedling recruitment in *M. cucullatus* was significantly reduced because these ants do not bury seeds like native species do. Other myrmechocorous Proteaceae, with large seeds, share a dispersal process similar to the *Mimetes cucullatus* and therefore the effects of the Argentine invasion would be applicable to >170 species of ant dispersed Cape Proteaceae (Bond and Slingsby 1984).

Herbicidal control measures

Both the "foliar-sprayed" treatment (P7) at Platboom and the "aerial-sprayed" treatment (K2) had ant species richness (and diversity and evenness) indices that were more similar to their respective controls than the other treatments. Infact ant species were 73% similar between treatments and controls for the two respective sites (Table 2). Majer *et al.* (1984) demonstrated that the restoration of a rich and diverse ant fauna was assisted by rehabilitation procedures that encouraged a rich and diverse plant community. Though "foliar-spraying" affected the species richness of the plants in this study, the structural integrity of the system was maintained and was more similar to the controls (See chapter 2). Therefore, the "ideal" habitat for ants, described by Greenslade (1979) as, "a mosaic of bare soil and areas of herbs, small shrubs and grasses, under an open canopy of trees or taller shrubs", were well met in both treatments. These sites provided a variety of microhabitats in which ants have proliferated, as well as sufficient biomass providing a food base for consumers (Majer, *et al.* 1982).

Although the above two methods of alien plant control did not impact the ant composition and diversity in the respective sites, the use of herbicides did however impact the ant community in the Platboom "boom-sprayed" treatment. Principal components analysis indicates that ant species composition was significantly different in the "boom-sprayed" treatment site compared to its control (Table 5). This blanket method of herbicidal application involved direct transmission of Garlon 4 (triclopyr) onto vegetation (see chapter one for detail). This herbicide selectively knocked out the dicotyledenous plants, with the resulting community being composed of a high graminoid cover (see chapter 2), which had a lower mortality level than other components of the fynbos (A. Ellis *et al.* undated). This kind of vegetation does not support as rich a faunal component as does the control as there are essentially less microhabitats to support the more rare or cryptic ant species. For instance, the tree nesting *C. peringueyi*, though neither rare or cryptic, and ubiquitous in this study, was found in significantly lower numbers and less frequently in this study site. This was most likely the result of the altered habitat (significantly lower shrub cover) brought about by the boom-spraying treatment. French and Major (2001) also found that in areas with a reduction in plant species richness, such as weed infested areas (*Acacia saligna*), although the ant species richness remained the same, the composition of the ant communities changed. They suggested that although there was an abundance of food (in the form of *A. saligna* elaiosomes) ants were responding to the diversity of food available.

The two "foliar-spraying" treatment sites did not produce similar results or trends in the respective sites (Table 4). At Platboom (P7) the composition and diversity of ants was similar to the control whereas in the Meadows (M5), the ant species richness was significantly lower for this treatment site compared to its respective control (Table 5). This reduction in richness could be as a consequence of damage caused by the severity of fire, to the soil properties of the area. This region had a forty-one year-old dense *Acacia* infestation. In addition to this, certain areas within this site also had fuel stacked and these heaps burnt. This increased fuel biomass, when compared to natural fynbos vegetation, would have resulted in a very intense fire, with certain areas which had these stack burns, being worst affected. This heat release at the surface would ultimately encourage soil erosion (see chapter 2). The topsoil which erodes away is usually the source of important plant growth nutrients, a rich microflora and fauna which are essential for nutrient cycling (Brady 1974) and is also an important source of indigenous plant propagules. Due to a poor establishment of plant species and growth forms in this treatment (chapter 2), the variation of microclimates, and therefore the variation in habitats available to ants are significantly reduced. Greenslade and Greenslade (1977) proposed that vegetation had a threefold influence on ant communities through its effects on microclimate, carrying capacity and structural complexity of the habitat. Plants provide ants with nesting sites and food sources, and other fauna which are associated with the vegetation act as potential predators, prey and competitors (Buckley 1982). Within the *Camponotus* genus itself, there was a marked reduction in species number in the Meadows treatment (M5) which was most likely the result of habitat modification and / or the dominance of *Anoplolepis steingroeveri*, *Camponotus* was excluded due to competition for nectar food sources.

Ant versus plant diversity

There was a significantly positive relationship between ant species richness and plant species richness at the plot scale (Figure 2b), with a rich plant community perhaps encouraging a diverse ant community through increased provision of microclimates, a diverse array of food-sources and buffering against temperature extremes. This was demonstrated by numerous other studies dealing with ant species that recolonize the rehabilitation site (Majer *et al.* 1984, 1999). Here it was shown that ant recolonization rates are associated with a structurally and floristically diverse plant community. Results also indicate that when plant richness is low, ant richness may be high or low. A rich assemblage of ants, in a site with a low plant richness could be as a result of ant colonization from surrounding species rich areas. Roth *et al.* (1994) suggested that ant species recovery might be due to the close proximity of primary forest that serves as a source from which ants could colonize a disturbed site.

Plant diversity in fynbos vegetation is highest in the first few years following a fire (le Maitre and Midgley, 1992), with more mature vegetation being less diverse (van Wilgen, 1980). Studies conducted by Donnelly and Giliomee (1985a,b), in the fynbos vegetation of the Cape, recorded the lowest diversity and richness values for ants in twenty-one year old, and thirty-nine year old fynbos vegetation. Both these sites had the lowest plant diversity in comparison to the other fynbos areas in the study. Thus vegetation age may influence the results (as shown by regression analysis- Figure 2c). In this study, comparisons among controls indicate that the maturest (oldest post-fire age) site (Klaasjagersberg) had the lowest plant and ant species richness.

The role of ants in seed dispersal

Seeds are the only stage in the life cycle of most fynbos plants that survives fire, and therefore the role of the seed dispersal agent (seed burial) is of paramount importance in maintaining fynbos community structure and diversity as well as providing a buffer against local extinction. In the boomsprayed treatment as well as the "foliar sprayed" treatment at the Meadows (M5) known seed dispersers such as *Anoplolepis steingroeveri*, *Pheidole capensis, Camponotus niveoseosus* and *T. quadrispinosum* were present. *A. steingroeveri* and *P. capensis* have been found to be very efficient dispersers, recruiting very rapidly to elaiosome bearing seeds such as *Leucospermum conocarpodendron* and *Mimetes cucullatus* and then transporting them into their nests (Slingsby and Bond 1984, Bond and Slingsby 1984). Christian (2001) demonstrated the importance of seed dispersers in the fynbos community by investigating the disruption of the mutualistic relationship between plants and ants by the Argentine Ant. This study proved two abundant ant species that are very easily displaced by the Argentine Ant, *A. custodiens* and *P. capensis,* to be very effective dispersers of large seeded Proteaceae. The latter two species dispersed up to 70 % of large seeds placed outside its nests, compared to 15% dispersed by *T. quadrispinosum and Meranoplus peringueyi* (latter species not found in this study). These four ants dispersed small seeded species equally well. This study therefore highlighted that elimination of dispersers leads to greater vulnerability to seed predation of large seeded species (relying on fewer more specialized dispersers) but not small seeded species.

Ordination results indicate the separation of certain pitfalls in the "foliar-sprayed" treatment at the Meadows. These pitfalls contained high numbers of *Anoplolepis steingroeveri,* which coincided with bare ground. Bond *et al.* 1991 found that these ants prefer to make their nests in the open, unshaded by large shrubs, which then has important ecological consequences for plants.

Conclusion

The ant communities at the three control sites were different, especially Klaasjagersberg versus The Meadows and Platboom (Figure 5). There was a greater difference between these sites than between treatment and controls within the sites, making comparisons more complicated. This study set out to determine whether the diversity and composition of ants was significantly impacted by the alien clearing treatments applied and the results indicated that certain treatments did have a marked effect on ant communities (Table 5.). The exact causes of these impacts were not studied, but some likely causes can be suggested. Ant community composition was significantly different compared to the control area following heavy "boom-spraying" of the herbicide Triclopyr. This change in ant composition was most likely the result of an alteration to the vegetation structure that consequently lead to habitat modification. In the "foliar-sprayed" treatment (M5) ant species richness was significantly less compared to its control. Here, high fuel loads in the "burn-scarred" regions, resulted in patches of bare ground. Although richness was less in this treatment, these bare patches favoured the major seed disperser *Anoplolepis steingroeveri*.

This investigation has shown that ant communities have changed in the "foliar sprayed" treatment site (M5) and the "boom-sprayed" treatment where composition and richness were negatively affected. In the former treatment the ant species were only 49% similar to the control (Table 2). Despite this, results in these sites indicate that there was a good representation of seed dispersing ants in these treatments. It is possible that ants could colonise from adjacent areas. The implication for recovery with the presence of a variety of seed-dispersers is a positive one, with the important keystone process of myrmecochory still intact. Fynbos plant recolonisation would therefore be well buffered against poor alien clearing techniques because important ant seed dispersers tend to occur in high densities where there is bare ground.

The "bulldozed and aerial-sprayed" treatment showed a significant reduction in ant diversity. This reduced diversity was especially evident in the pitfalls that contained the *L. humile*. Severe disturbance as that caused by bulldozers can encourage the presence of the Argentine Ant, although in this case the presence of this invasive species is mostly likely due to historical factors (see chapter one) unconnected to restoration. *L. humile* may be considered a keystone invader with negative system impacts.

This study has shown that ants are good indicators of ecosystem recovery (but see chapter 4 for the limitations), indicating the decrease in diversity in the "bulldozed and aerial-sprayed" treatment, the decrease in species richness in the "foliar-sprayed" treatment at the Meadows, as well as the change in composition in the "boom-sprayed" treatment. No other significant differences were indicated, for any of the treatment versus control comparisons, for any of the four ant variables tested. Thus, ant community composition does, to some extent, reflect the changes in the composition of plant communities due to the types of manipulations that were studied. Given the rather rudimentary understanding of ant habitat requirements in fynbos, the inferences made in this study must be seen as preliminary. A priority for future research would therefore be to derive a better understanding of fynbos ant species habitat requirements or feeding guilds, which could then be used in a functional group analysis as was done with the plants. This would then allow one to determine whether any guild such as predators or ground nesting species has been eliminated as a result of the treatments employed.

This trial highlights that despite the changes in composition and diversity of ants in the different treatment sites, the presence of seed dispersing ants deems well for fynbos plants and for ecosystem recovery. Ants could therefore be considered a useful tool for conservation, and can provide useful insight into the relationship between the management practice and biodiversity.

Chapter 4. General conclusions

This study set out to determine the impacts of different alien plant control methods on the recovery of fynbos ecosystems. To determine whether the control methods affected the ecosystem, I chose to survey the vegetation, and did an assessment of the ant communities in the treatment areas and compared them to areas of natural vegetation. The rationale behind the comparative monitoring of the vegetation and identification of ants was to provide baseline data from which one could derive an understanding of the behaviour and dynamics of the fynbos ecosystem in the treatment areas. This information, I felt, would be very valuable for making informed management decisions for future alien clearing operations.

The impacts of invasive alien control operations *per se* have not been well studied anywhere in the world. Little formal attention has been given to defining "impacts" and it is important that we distinguish invasive species with minor effects from those with major effects in order to prioritise management efforts. Parker *et al.* 1999 set out a framework for understanding the ecological effects of invaders. In this study I simply looked at community level impacts of alien invaders on the natural community measures such as species richness, evenness, diversity, abundance and projected canopy cover (for plants). Since these indices might miss some essential information, multivariate methods were also used.

The main aims of this study were:

1. To determine the impacts of the different clearance methods on vegetation structure and diversity.
2. To determine whether any guilds need to be re-introduced into the community to restore vegetation structure and function.
3. To determine whether the vegetation rehabilitation requirements differ among the different treatments?
4. To determine the impacts of various clearing treatments on ant species composition and diversity.
5. To determine whether ants are useful indicators of ecosystem recovery following different methods of alien plant clearance.

The main findings of this study with regards to the aims above were as follows:

The total projected canopy cover did not vary significantly between the treatments and controls, although the density of the "foliar-sprayed" treatment at the Meadows was significantly less (Table 1). The vegetation structure and diversity were significantly different in the treatments compared to the controls. Certain treatments appeared to affect the indigenous vegetation more severely than others, leading me to conclude that restoration action may be needed.

There were significantly fewer low shrubs in the treatment sites (except for the "aerial-spraying" treatment), and an increase in the graminoids (predominantly grasses). Graminoids were significantly more abundant in the "bulldozed and aerial-sprayed" treatment, the "boom-sprayed" treatment as well as the "foliar-sprayed" treatment at the Meadows. Thus, the growth forms in the Platboom "foliar-sprayed" treatment (P7) and the "aerial-sprayed" treatment seem to have been least affected by management actions.

When regenerative mode was investigated, there was a significant decrease in the long-lived obligate reseeding species for all treatments except the "aerial-sprayed" treatment. This is a well-known impact of invasive woody plants in fynbos (Richardson and van Wilgen 1986a,b, Holmes and Cowling 1997a,b). Again, the vegetation structure was least affected in the "aerial sprayed" treatment compared to the controls. There was also a significant increase in the sprouter component of the vegetation in the "boom-sprayed" area.

The species richness at both the quadrat and plot scales showed significant reduction in the treatments (Table 1), although not so for the "bulldozed and aerial-sprayed" treatment and the "aerial-sprayed" treatment. The control site, however, had an unusually low richness, and therefore significant differences in richness values were not derived. Diversity and evenness were also significantly lower for the "boom-sprayed" treatment, as well as the "foliar-sprayed" treatment site at the Meadows.

Multivariate analysis indicates that the structure of the controls and the "foliar-sprayed" treatment at Platboom were much more similar to each other than to the other treatments, with the "aerial-sprayed" treatment forming an intermediate group. Here, most of the growth forms, dispersal syndromes and regenerative modes were well represented as opposed to the other treatments. Since the most desirable condition for successful recovery is a spread of guilds which approximates the controls, it leads me too conclude that the treatment sites that were the most severely impacted were the "boom-sprayed" site as well as the "foliar-sprayed" treatment site at the Meadows. In the latter treatment, accumulated fuel loads may have exacerbated the effect of post-fire foliar spraying. In severe cases like this restoration actions in the form of guild re-introductions may be necessary. The current policy in the Cape Peninsula National Park is to control alien species and allow indigenous species to recolonize unassisted. This protects the genetic integrity of the ecosystem by conserving local gene pools. However, in a situation such as the above where guilds are missing and indigenous seedlings have been killed without being given the opportunity to set seed the site may take too long to recover on its own. Thus reintroducing important guilds such as graminoids, low shrubs, overstorey shrubs and forbs are essential for preserving biological diversity but also for maintaining important ecological processes. This type of restoration measures would allow a more rapid return of a structurally representative fynbos stand (Holmes and Richardson 1999). Which in turn would lead to a more functional ecosystem.

Results of the ant study complement the vegetation study. In Chapter 3, I found that the composition of ants was significantly different following "boom-spraying" in the Platboom area. Ant species richness and diversity were also significantly lower in the "foliar-sprayed" treatment site at the Meadows and the "bulldozed and aerial-sprayed" treatment site respectively. Both these sites had a distinct ant faunal composition as shown in the multivariate analysis, with the pitfalls containing the Argentine Ant separating to the left of the diagram for the "bulldozed and aerial-sprayed" treatment. In the "foliar-sprayed" treatment at the Meadows ant species composition was also different, separating from the main cluster whereas other treatments overlapped in composition with their respective control sites. The species richness of ants was also significantly reduced in the "foliar-sprayed" treatment, as in the vegetation study. Although the richness was lower for this treatment at the Meadows, the important seed dispersers such as *Anoplolepis steingroeveri* and *Pheidole capensis* were present in this treatment. *P. capensis* was also present in significantly higher amounts in the "boom-sprayed" treatment.

The "bulldozed and aerial-sprayed" treatment also contained the invasive Argentine Ant, which has been reported to transform many other ecosystems into which it has been introduced. There is no conclusive evidence however that its presence here can be attributed to the treatment applied. The presence of this species is more likely the result of disturbance in general and the proximity of human-impacted sites, which serve as foci for invasion, although the species may have been encouraged by this disturbance.

These results also showed that ants respond to changes in vegetation structure and composition that has also been observed in mine site rehabilitation. This suggests that floristically diverse fynbos communities support the most diverse ant assemblages. This leads me to conclude that ants can be useful indicators of recovery following alien clearance, since it tracks changes in the vegetation.

Limitations or drawbacks experienced with this study.

The initial stages of this project dealt with the selection of study sites, in which vegetation surveys would be conducted. During the last three decades, various treatments had been employed in an effort to eradicate alien infestations in the Cape of Good Hope Nature Reserve. These control methods were then mapped in the three localities, and GIS coverages were derived. This method had its limitations, due to the nature of the literature that was available and consulted.

From 1968 onwards management of alien species was rather haphazard, and there wasn't much to report. From 1974-1976 there was a slightly better indication of where exactly clearing operations took place in the reserve, but this still wasn't mapped using a proper grid system. After 1976 to mid 1989 the alien invasive clearing program was documented by simply stating in which areas treatments occurred. In many instances there was no indication of size, or the amount of labourers. It would then have been ideal to have a degree of significance, this being then an idea of the amount of days worked by the labourers or perhaps the amount of labourers to give an indication of size but this kind of additional information wasn't always available in the reports. Thus, pre-1990 rangers reports of alien clearing were not very informative. There was no accurate reflection of the size of the areas that were subjected to a specific treatment. Since 1990, however, a grid system was used by rangers to indicate where control operations took place as well as the extent of the area that was worked in. Although the coverages have their limitations at a very fine scale (250m x 250m) they generally provide very good information on the history of the sites. These coverages and the consultations with rangers were then used to determine which treatments to investigate/survey.

In chapter two, the vegetation survey clearly illustrated the difficulties facing the field ecologist studying fynbos plant communities. The task of compiling a species list was challenging, considering the high diversity of fynbos in a given area, as well as the possibility of omitting or misnaming a species because of this. This problem is compounded by the fact that species are easily confused in their vegetative state. Despite these difficulties, species lists of Taylor (1985) and Privett (1998) made accurate species identifications easier at the Cape of Good Hope site than would be possible in most other parts of the fynbos biome. Also, for species unidentifiable in its vegetative state, flowering material was collected the next flowering season to enable identifications. It was also useful to work in pairs, searching jointly thereby reducing the likelihood of overlooking or misidentifying species.

It was generally easier to identify ants because of the excellent reference collection for the Cape of Good Hope Nature Reserve housed at the South African Museum. One constraint of this method was the considerable time required to clean and sort samples prior to identifications. It took approximately one day to clean and sort three samples before identification, although this may vary depending on the density of the sample. In this study the abundance of *Crematogaster peringueyi* also slowed down this process. The pitfall traps were also left out for a month, which in retrospect was more time than actually needed to sample the common ant species in the vicinity. However, sampling for a month increased the chances of catching rare species. In recent ant studies pitfall-trapping surveys have been conducted typically for a week (H. Robertson, personal communication). Initially, ant identifications also proved to be time consuming. After the initial learning curve however, identifications became more rapid. Another drawback of using ants, or any invertebrate group, as indicators is that their wider distribution within the fynbos biome is poorly known, so it is difficult to assess levels of rarity and endemism, compared with plants (H. Robertson, personal communication).

I felt that using both plants and ants as indicators of ecosystem recovery, to address whether the treatments were negatively impacting the ecosystem was very useful, since they provided two perspectives of the ecosystem. Plants, however, proved to be the more sensitive indicator as significant differences could be seen between treatments and controls using composite community measures like diversity and abundance, as well as at the multivariate level where plants were classified into functional groups. The latter method very successfully showed the difference in functional guild structure of the treatments compared to the controls, providing very insightful information about the ecosystem. The effects of disturbance on ants however, are not as persistent a legacy as in communities of long-lived plants, which takes years to re-establish (Anderson 1995). This is largely due to the rapid colony growth rates and high dispersal abilities displayed by ants (Holldobler and Wilson 1990). For example, trees in a clear-felled forest may take decades to re-establish no matter how favourable the conditions. Ant species however can recover in a few years provided the conditions are suitable. The major effects of site disturbance on ants are therefore secondary and stress related, through modification to habitat structure and microclimate (Anderson1995). It is possibly for this reason that ant community measures were more similar between treatments and controls than between controls themselves.

For the reasons discussed above, I feel that using ants could be useful as indicators if used in conjunction with plants but if used on its own might miss essential information pertaining to the negative impacts of the treatments.

In conclusion this study has shown that management practices such as "foliar-spraying" and "aerial-spraying" has lead to the successful recovery of the indigenous vegetation, with ecosystem function still intact. Treatments such as "boom-spraying" however should be avoided since indigenous vegetation structure was severely impacted by this treatment. The vegetation structure in the "bulldozed and aerial-sprayed" treatment was also severely impacted, but here historic factors such as farming may have contributed to the change in vegetation structure. Lastly, high fuel loads should also be avoided for subsequent indigenous vegetation recovery.

Table 1. Main findings and management implications of the treatments employed at the Cape of Good Hope Nature Reserve. Abreviations of guilds are as follows: Lshrub = low shrub; Mshrub= medium shrub; Tshrub = tall shrub; Slseeder = short lived seeder and Llseeder = long lived seeder.

	Treatments		
	Bulldozing and aerial-spraying (K1a + K1b)	Aerial-spraying(K2)	Foliar-spraying (M5)
Plant density	No effect	No effect	Reduction in density from 247.2 to 144.4
Total canopy cover	No effect	No effect	No effect
Plant species richness (50 m^2)	No effect	No effect	Reduction in richness from 23.2 to 9
Plant species richness (1 m^2)	No effect	No effect	Reduction in richness from 8.2 to 2.6
Plant diversity	No effect	No effect	Reduction in diversity from 1.08 to 0.39
Plant evenness	No effect	No effect	Reduction in evenness from 0.79 to 0.44
Functional guilds	Decrease in Lshrubs but increase in graminoids, also a decrease in Llseeders	No effect	Decrease in wind and bird dispersed plants. Decrease in Lshrubs and Tshrubs but an increase in graminoids. Increase in sprouters but a decrease in Llseeders and Slseeders.
Ant density	No effect	No effect	No effect
Ant richness	No effect	No effect	Reduction in richness from 7.6 to 4.5
Ant diversity	Reduction in diversity from 0.40 to 0.33 and 0.14 for K1a and K1b respectively	No effect	No effect
Ant evenness	No effect	No effect	No effect
Overall assessment	Plants and ants are affected but suggest further study due to historical factors.	This method has been successful with no difference between treatment C22and control.	One of the worst treatments, but mostly due to high fuel loads.
Management Implications	Re-introduction of guilds such as low shrubs.	No restoration action required	Sowing with a comprehensive seed mix containing the major guilds. This should encourage the return of the invertebrate fauna as well.

Table 1 continued. Main findings and management implications of the treatments employed at the Cape of Good Hope Nature Reserve.

	Treatments		
	Foliar-spraying (P7)	Boom-spraying (P6a + P6b)	All treatments
Plant density	No effect	No effect	No effect
Total canopy cover	No effect	No effect	No effect
Plant species richness (50 m^2)	Reduction in richness from 23.6 to 13	Reduction in richness from 23.6 to 8 and 9.6 for P6a and P6b respectively	Reduction in richness from 20.6 to 11.3 species per 50 m^2
Plant species richness (1 m^2)	No effect	Reduction in richness from 8 to 3.4 and 2.8 and for P6a and P6b respectively	Reduction in richness from 6.9 to 3.42 species per 1m^2
Plant diversity	No effect	Reduction in diversity from 1.04 to 0.51 and 0.53 for P6a and P6b respectively	Reduction in diversity from 0.99 to 0.32 species per 50 m^2
Plant evenness	No effect	Reduction in evenness from 0.78 to 0.58 and 0.54 for P6a and P6b respectively	Reduction in evenness from 0.77 to 0.62 species per 50 m^2
Functional guilds	Decrease in ant dispersed plants. Decrease in Lshrubs and Llseeders.	Decrease in low and medium shrubs and an increase in graminoids. Increaser in sprouters and Slseeders but a decrease in Llseeders.	
Ant density	No effect	Increase in density from 133.8 to 168.6 and 210.7 for P6a and P6b respectively	Increase in density from 83.5 to 528.17 species per 50 m^2
Ant richness	No effect	No effect	Reduction in richness from 7.8 to 6.77 species per 50 m^2
Ant diversity	No effect	No effect	Reduction in diversity from 0.45 to 0.38 species per 50 m^2
Ant evenness	No effect	No effect	Reduction in evenness from 0.50 to 0.46 species per 50 m^2
Overall assessment	Although richness is affected, structure and function has been maintained. Hence, knapsack application has been successful.	"Boom-spraying" has negatively effected the structure and therefore has been unsuccessful.	Most treatments have negatively impacted indigenous recruitment.
Management Implications	No restoration action required.	Sowing with a comprehensive seed mix containing the major guilds.	

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Appendix 1. The mean percentage cover per species (PC) and frequency of occurrence in ten study sites in the Cape of Good Hope Nature Reserve.

	FAMILY	SPECIES	K1a (n=5)		K1b (n=5)		K2 (n=5)		K3 (n=5)		M4 (n=5)		M5 (n=5)		P6a (n=5)		P6b (n=5)		P7 (n=5)		P8 (n=5)	
			PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.	PC	FREQ.
1	RUTACEAE	*Adenandra uniflora*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
2	RUTACEAE	*Agathosma hookeri*	0	0	0	0	0	0	0	0	0.2	1	0.04	2	0	0	0	0	0	0	0	0
3	RUTACEAE	*Agathosma imbricata*	0	0	0	0	0	0	0	0	2.02	4	0	0	0	0	0	0	0	0	0	0
4	RUTACEAE	*Agathosma serpyllacea*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2
5	ASTERACEAE	*Anaxeaton laeve cf.*	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0
6	RUBIACEAE	*Anthospermum aethiopicum*	0	0	0.02	1	0.02	1	0	0	0.6	2	0	0	0	0	0.02	1	0	0	0.6	2
7	IRIDACEAE	*Aristia sp.*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
8	ASPARAGACEAE	*Asparagus stipulaceus*	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0
9	ASTERACEAE	*Athanasia crithmifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1	1.2	1	1.4	2	0	0
10	BRUNIACEAE	*Berzelia abrotanoides*	0	0	0.8	1	0	0	6.8	3	0	0	0	0	0	0	0	0	0	0	0	0
11	BRUNIACEAE	*Berzelia lanuginosa*	0	0	0	0	0	0	1.22	2	0	0	0	0	0	0	0	0	0.8	1	0	0
12	RESTIONACEAE	*Calopsis fruticosa*	0	0	0	0	0	0	0	0	0.8	3	0	0	0	0	0	0	0	0	0	0
13	MESEMBRYANTHEMACEAE	*Carpobrotus acinaciformis*	0	0	0	0	0	0	0	1	0.2	0	0	0	0.4	2	0.82	2	0.02	1	0.6	2
14	MESEMBRYANTHEMACEAE	*Carpobrotus edulus*	1.82	4	0.24	3	1.4	3	0.02	0	0	1	0	0	0	0	0	0	0	0	0	0
15	LAURACEAE	*Cassytha ciliolata*	0	0	0	0	0.02	1	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
16	GENTIANACEAE	*Chironia baccifera*	0	0	0	0	0	0	0	0	1.02	4	0.2	1	0	0	0.02	1	0.62	2	0	0
17	RESTIONACEAE	*Chondropetalum microcarpum*	0	0	0	0	0	0	0	0	0.62	3	0	0	0	0	0	0	0.02	1	0.2	1
18	RESTIONACEAE	*Chondropetalum nudum*	0.02	1	1.62	4	2.62	3	1.02	5	0	0	0	0	0	0	0	0	0	0	0.8	2
19	ROSACEAE	*Cliffortia filifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.2	1
20	ROSACEAE	*Cliffortia fulcata*	0	0	0	0	0	0	0	0	0.6	2	0.02	1	0	0	0.02	1	0	0	0.26	4
21	ROSACEAE	*Cliffortia obcordata*	0	0	0	0	0	0	0	0	5	2	0	0	0	0	0	0	0	0	0	0
22	ROSACEAE	*Cliffortia subsetacea*	0	0	0.6	1	4.4	2	2.02	4	0	0	0	0	0	0	0	0	0	0	0	0
23	ASTERACEAE	*Cullumia setosa*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1	0	0	0.02	1
24	POACEAE	*Cynodon dactylon*	20	2	22.02	3	23.4	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
25	CYPERACEAE	*Cyperus odoratus*	0.02	1	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
26	CYPERACEAE	*Cyperus sphaerospermus*	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
27	PROTEACEAE	*Diastella divaricata*	0	0	0	0	0	0	5	1	0	0	0	0	0	0	0	0	0	0	12.2	5
28	RUTACEAE	*Diosma hirsuta*	0	0	0	0	0	0	0	0	1.22	4	0	0	0	0	0	0	0.04	2	0	0
29	RUTACEAE	*Diosma oppositifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.6	1
30	ASTERACEAE	*Disparago anomala*	0	0	0	0	0	0	0	0	0.8	1	0	0	0	0	0	0	0	0	0	0
31	ASTERACEAE	*Disparago lasciocarpa*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
32	POACEAE	*Ehrharta villosa*	0	0	0	0	0	0	0	0	0	0	64	4	34.8	2	41.2	3	0.02	1	0	0
33	RESTIONACEAE	*Elegia cuspidata*	0	0	0.4	1	0	0	1	3	0	0	0	0	0	0	0	0	0	0	0	0
34	RESTIONACEAE	*Elegia filacea*	0	0	5	1	0	0	2.8	4	0	0	0	0	0	0	0	0	1.22	2	1.2	1
35	ASTERACEAE	*Elytropappus scaber*	0	0	0	0	0	0	0	0	0	0	0	0	0.4	1	0	0	0	0	0.2	1
36	MESEMBRYANTHEMACEAE	*Erepsia anceps cf.*	0	0	0	0	0.02	1	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0
37	MESEMBRYANTHEMACEAE	*Erepsia bracteata cf.*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
38	ERICACEAE	*Erica clavisepala*	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	ERICACEAE	*Erica corifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
40	ERICACEAE	*Erica gilva*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4.82	3	10	3
41	ERICACEAE	*Erica gnaphaloides*	0	0	0	0	1.2	1	1.2	1	0	0	0	0	0	0	0	0	0	0	0	0
42	ERICACEAE	*Erica hirtiflora*	0	0	0	0	0.04	2	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0

43	ERICACEAE	*Erica imbricata*	0.4	1	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0.84	3
44	ERICACEAE	*Erica laeta*	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
45	ERICACEAE	*Erica multumbellifera*	0	0	0	0	0.04	2	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
46	ERICACEAE	*Erica phylicifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0.4	1	1.2	1	1.42	3	0	0
47	ERICACEAE	*Erica plukenetti*	0	0	0	0	0	0	1.2	2	0	0	0	0	0	0	0	0	0	0	0	0
48	ERICACEAE	*Erica pulchella*	0	0	0	0	1	1	0.42	2	0	0	0	0	0	0	0	0	0	0	6.2	5
49	ERICACEAE	*Erica subcapitata*	0	0	0	0	0	0	0	0	2.4	2	0	0	0	0	0	0	0	0	0	0
50	ERICACEAE	*Erica subdivaricata*	0	0	15.02	3	0	0	37.6	5	0	0	0	0	0	0	0	0	18	2	0.02	1
51	EBENACEAE	*Euclea racemosa*	0	0	0	0	0	0	0	0	2.42	4	0.6	2	0	0	0	0	0	0	0	0
52	ASTERACEAE	*Felicia tenela*	0	0	0	0	0	0	0	0	0.04	2	0	0	0.4	1	0	0	0	0	0	0
53	POACEAE	*Festuca scabra*	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
54	CYPERACEAE	*Ficinia deusta*	0	0	0	0	0	0	0	0	0	0	0.4	1	26.6	4	10.8	2	0	0	0	0
55	CYPERACEAE	*Ficinia filiformis*	0	0	0.04	2	0.02	1	0.44	3	3.8	3	0.02	1	0	0	0	0	0.02	1	0.02	1
56	CYPERACEAE	*Ficinia secunda cf.*	0.02	1	0	0	0.02	1	0	0	0	0	0	0	0.04	2	2.4	1	2	2	0	0
57	CYPERACEAE	*Ficinia sp. 1*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2
58	CYPERACEAE	*Ficinia sp. 2*	0	0	0	0	0	0	0.06	3	0	0	0	0	0	0	0	0	0	0	0	0
59	CYPERACEAE	*Ficinia trichoides*	0	0	0	0	0	0	0	0	4.04	5	0	0	0	0	0	0	0	0	0	0
60	GERANIACEAE	*Geranium incanum*	0.04	2	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
61	THYMELAEACEAE	*Gnidia juniperifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
62	THYMELAEACEAE	*Ginidia oppositifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.2	1
63	ASTERACEAE	*Helicrysum anomilum*	0	0	0	0	0	0	0	0	0.02	1	2.8	3	0	0	0	0	0	0	0	0
64	ASTERACEAE	*Helicrysum crispum*	0	0	0	0	0	0	0	0	3.42	4	0.02	1	0	0	0	0	0	0	0	0
65	ASTERACEAE	*Helicrysum dasyanthum*	0	0	0	0	0	0	0	0	5.22	5	0.2	1	0	0	0	0	0	0	0	0
66	STERCULIACEAE	*Hermania procumbens*	0	0	0	0	0	0	0	0	0	0	0	0	0.4	1	0.2	1	0	0	0	0
67	STERCULIACEAE	*Hermania ternifolia*	0	0	0	0	0	0	0	0	1.02	3	0.4	1	0	0	0	0	0	0	0	0
68	FABACEAE	*Indegofera brachystachya*	0	0	0	0	0	0	0	0	4.4	5	0.02	1	0	0	0	0	0	0	0.02	1
69	RESTIONACEAE	*Ischyrolepis capensis*	1.22	2	0	0	0.02	1	0.4	1	0	0	0	0	0	0	2.6	3	7.22	4	10	3
70	RESTIONACEAE	*Ischyrolepis cincinnata*	0.4	1	0.02	1	1.4	2	0	0	0	0	0	0	0	0	0	0	0.02	1	0	0
71	JUNCACEAE	*Juncus capensis*	0.4	1	3.6	5	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
72	JUNCACEAE	*Juncus krausii*	0.6	2	14.8	4	3.4	4	2	1	0	0	0	0	0	0	0	0	0	0	0	0
73	THYMELAEACEAE	*Lachnea densiflora*	0	0	0	0	0.2	1	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
74	MESEMBRYANTHEMACEAE	*Lempranthus tenuifolius cf.*	1.2	1	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2	0	0	0	0
75	PROTEACEAE	*Leucadendron coniferum*	0	0	0	0	0	0	11	3	22.4	4	0	0	0	0	0	0	0.02	1	2.22	3
76	PROTEACEAE	*Leucadendron salignum*	0	0	0	0	0	0	0.8	1	0	0	0	0	0	0	0	0	0.4	1	1.8	2
77	PROTEACEAE	*Leucospermum conocarpodendron*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.8	2	0	0
78	CAMPANULACEAE	*Lightfootia longifolia*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
79	CAMPANULACEAE	*Lightfootia tenela*	0	0	0	0	0	0	0	0	0.04	2	0.2	1	0	0	0	0	0	0	0	0
80	LOBELIACEAE	*Lobelia setacea*	0.02	1	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
81	POACEAE	*Merxmuellera stricta cf.*	0	0	0	0	0	0	0	0	2.4	2	0	0	0.02	1	0	0	0	0	0	0
82	ASTERACEAE	*Metalasia brevifolia*	0	0	0	0	0.4	1	0	0	0.02	1	0.02	1	0.02	1	0.84	3	0	0	0.02	1
83	ASTERACEAE	*Metalasia cephalotes*	0	0	0	0	0	0	0	0	16.8	5	0.8	1	0.6	1	0.02	1	0	0	0	0
84	ASTERACEAE	*Metalasia muricata*	30.42	5	8	2	19	4	8.6	4	0	0	0.02	1	0	0	9.2	3	28	5	26.6	5
85	LOBELIACEAE	*Monopsis lutea*	0.02	1	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
86	MYRICACEAE	*Myrica quercifolia*	0	0	0	0	0	0	3.6	1	0	0	0	0	0	0	0.6	1	0.04	2	0.42	2
87	ASTERACEAE	*Oedera prolifera*	0	0	0	0	0	0	0	0	5.2	4	0.4	1	0	0	0	0	0	0	0	0
86	SCROPHULARIACEAE	*Oftia africana*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	1	0.84	3	0	0
89	OLEACEAE	*Olea capensis*	0	0	0	0	0	0	0	0	0.8	1	0	0	0	0	0	0	0	0	0.02	1
90	OLEACEAE	*Olea exasperata*	0	0	0	0	0	0	0	0	0.42	2	0.02	1	0	0	0	0	0	0	0	0
91	GENTIANACEAE	*Orphium frutescens*	0	0	0.42	2	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0

92	POACEAE	*Paspalum dilatatum*	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
93	THYMELAEACEAE	*Passerina paleacea*	0	0	0	0	1	1	0	0	2.6	5	1.2	1	0	0	0	0	0.8	1	0	0
94	THYMELAEACEAE	*Passerina vulgaris*	3.22	3	0.42	3	7.4	5	0.2	1	0	0	0.02	1	2.42	3	14.4	5	7	3	0.66	5
95	GERANIACEAE	*Palargonium betulinum*	0	0	0	0	0	0	0	0	0.04	2	0	0	0	0	0	0	0	0	0	0
96	GERANIACEAE	*Pelargonium capitatum*	0.02	1	0	0	0.02	1	0.02	1	0	0	0.02	1	0.02	1	0.02	1	0	0	0	0
97	GERANIACEAE	*Pelargonium chamaedrypholium*	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2	0	0	0	0	0	0
98	GERANIACEAE	*Pelargonium cuculatum*	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.06	3
99	GERANIACEAE	*Palargonium rapaceum cf.*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
100	POACEAE	*Pannisetum macrourum cf.*	0	0	0	0	0.4	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
101	POACEAE	*Pentashustis acinosa*	0.02	1	0.02	1	0	0	0	0	0.04	2	0	0	0	0	0.06	3	0	0	0.02	1
102	POACEAE	*Pentashustis aurea*	0	0	0.2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
103	POACEAE	*Pentashustis curvifolia*	0	0	0.02	1	2.4	1	0	0	0	0	0	0	0	0	0	0	0	0	0.82	2
104	AIZOACEAE	*Phameceum cordifolium*	0	0	0	0	0	0	0	0	0	0	0.06	3	0	0	0	0	0	0	0	0
105	RHAMNACEAE	*Phylica dodii*	0	0	0	0	0	0	0	0	0.04	2	0.02	1	0.02	1	0	0	0	0	0	0
106	RHAMNACEAE	*Phylica imberbis*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.64	3
107	RHAMNACEAE	*Phylica ericoides*	0	0	0	0	0	0	0	0	0.8	1	1.82	4	0	0	0	0	0	0	0	0
108	RHAMNACEAE	*Phylica parviflora*	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0.4	1	2.06	5	0.4	2
109	RHAMNACEAE	*Phylica stipularis*	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	4	1
110	RESTIONACEAE	*Platycaulos compressus*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
111	ASTERACEAE	*Plecostachys serpyllifolia*	0.2	1	9	3	1.04	4	1.4	3	0.02	1	0.42	2	4	2	0	0	2	2	0	0
112	POACEAE	*Pseudopentamaris macrantha*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2
113	FABACEAE	*Psoralea aphylla*	0	0	0.2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
114	FABACEAE	*Psoralea laxa*	0.02	1	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
115	FABACEAE	*Psoralea pinnata*	0.02	1	1.2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
116	CELASTRACEAE	*Pterocelastrus tricuspidatus*	0	0	0	0	0	0	0	0	0.22	2	0	0	0	0	0	0	0	0	0	0
117	RESTIONACEAE	*Restio bifidus*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.6	2
118	RESTIONACEAE	*Restio laptostachys*	0	0	0	0	4	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
119	RESTIONACEAE	*Restio praeacutus*	0.6	2	0	0	0.4	1	2.2	3	0	0	0	0	0	0	0	0	0	0	0	0
120	RESTIONACEAE	*Restio quinquefarius*	11	2	0.84	4	9.4	4	2.6	4	0	0	0	0	0	0	0	0	0	0	2	1
121	RESTIONACEAE	*Restio tetragonus*	0	0	0.02	1	0.8	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
122	RESTIONACEAE	*Restio triticeus*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2.4	2	3.02	4
123	ANACARDIACEAE	*Rhus glauca*	0	0	0	0	0	0	0	0	2.8	3	0.06	3	0	0	0	0	0	0	0	0
124	ANACARDIACEAE	*Rhus laevigata*	0	0	0	0	0	0	0.4	1	0.2	1	0	0	0	0	0.02	1	0	0	0.2	1
125	ANACARDIACEAE	*Rhus lucida*	0.22	2	0.22	2	0.02	1	0	0	2.8	4	0	0	0.02	1	2.04	3	1.82	4	0.06	3
126	MESEMBRYANTHEMACEAE	*Rushia sarmentosa*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
127	ERICACEAE	*Salaxis axilaris*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8.02	2
128	SCHIZAEACEAE	*Schizea pectinata*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
129	ERICACEAE	*Scyphogyne muscosa*	2.4	2	1.2	2	5.6	3	1.02	3	0	0	0	0	0	0	0.2	1	0	0	9.4	3
130	SELAGINACEAE	*Salago cylindrica cf.*	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0.2	1
131	SELAGINACEAE	*Salago dregei cf.*	0	0	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0
132	ASTERACEAE	*Senecio pubigerus*	0	0	0	0	0.02	1	0.2	1	0	0	0	0	0	0	0	0	0	0	0	0
133	ASTERACEAE	*Senecio rigidis*	0	0	0.02	1	0.02	1	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
134	PROTEACEAE	*Serruria glomerata*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.6	1	9.4	4
135	PROTEACEAE	*Sarruria villosa*	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0
136	POACEAE	*Sporobolus capensis*	0.42	2	2.4	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
137	BRUNIACEAE	*Staavia radiata*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.2	1	0	0
138	POACEAE	*Stanotaphrum secundatum*	15	2	13	2	0	0	0	0	0	0	0	0	0.8	1	0	0	0	0	0	0
139	ASTERACEAE	*Stoebe cuspidata*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.8	1	0	0
140	ASTERACEAE	*Stoeba incana*	0.4	1	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1

141	THYMELAEACEAE	*Struthiola ciliata*	0	0	0.2	1	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0.22	2
142	THYMELAEACEAE	*Struthiola dodecandra*	0	0	0	0	0	0	0	0	0.82	3	0	0	0	0	0	0	0	0	0	0
143	SCROPHULARIACEAE	*Sutera hispida*	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2	0	0	0	0	0	0
144	ASTERACEAE	*Syncarpha vestita*	1.6	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
145	CYPERACEAE	*Tetraria compacta*	0	0	0	0	0	0	0.8	1	0	0	0	0	0	0	0	0	0	0	0	0
146	CYPERACEAE	*Tetraria sp. 1*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04	2
147	RESTIONACEAE	*Thamnochortus fraternus*	0	0	0	0	0	0	0	0	4.22	3	0	0	0	0	1.2	1	0	0	0.02	1
148	RESTIONACEAE	*Thamnochortus fruticosus*	0	0	0	0	1	2	0	0	0.4	1	0	0	0	0	0.02	1	2.4	2	0	0
149	RESTIONACEAE	*Thamnochortus gracilis*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.2	1	0.22	2
150	POACEAE	*Tribolium uniolea*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1
151	ASTERACEAE	*Troglophyton parvulum*	0	0	0	0	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
152	ASTERACEAE	*Ursinia crithmifolia*	0	0	0	0	0	0	0	0	0	0	0.02	1	0.82	2	1.2	1	0	0	0	0
153	ASTERACEAE	*Ursinia tenuifolia*	1.2	1	8	1	0.04	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
154	ASTERACEAE	*Vellereophyton dealbatum*	0.2	1	0	0	0.82	3	0	0	0	0	0	0	11.8	4	0	0	0	0	0.4	1
155	CAMPANULACEAE	*Wahlenbergia procumbens*	0.02	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
156	RESTIONACEAE	*Willdenowia glomerata*	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0
157	ZYGOPHYLLACEAE	*Zygophyllum spinosum*	0	0	0	0	0	0	0	0	0	0	0	0	0.02	1	0	0	0	0	0	0

Appendix 2. The biological attributes of species . The catergories are abbreviated as follows:
DM = Dispersal mode, RM = Regenerative mode and GF = Growth form

	FAMILY	SPECIES	DM	RM	GF
1	RUTACEAE	*Adenandra uniflora*	Ant	L. lived seeder	Low shrub
2	RUTACEAE	*Agathosma hookeri*	Ant	Sprouter	Low shrub
3	RUTACEAE	*Agathosma imbricata*	Ant	Sprouter	Low shrub
4	RUTACEAE	*Agathosma serpyllacea*	Ant	L. lived seeder	Low shrub
5	ASTERACEAE	*Anaxeaton laeve cf.*	Wind	L. lived seeder	Low shrub
6	RUBIACEAE	*Anthospermum aethiopicum*	Passive	L. lived seeder	Low shrub
7	IRIDACEAE	*Aristia sp.*	Passive	Sprouter	Geophyte
8	ASPARAGACEAE	*Asparagus stipulaceus*	Passive	Sprouter	Low shrub
9	ASTERACEAE	*Athanasia crithmifolia*	Wind	Sprouter	Low shrub
10	BRUNIACEAE	*Berzelia abrotanoides*	Wind	Sprouter	Medium shrub
11	BRUNIACEAE	*Berzelia lanuginosa*	Wind	Sprouter	Medium shrub
12	RESTIONACEAE	*Calopsis fruticosa*	Passive	Sprouter	Graminoid
13	MESEMBRYANTHEMACEAE	*Carpobrotus acinaciformis*	Passive	L. lived seeder	Forb
14	MESEMBRYANTHEMACEAE	*Carpobrotus edulus*	Passive	L. lived seeder	Forb
15	LAURACEAE	*Cassytha ciliolata*	Wind	L. lived seeder	Low shrub
16	GENTIANACEAE	*Chironia baccifera*	Passive	S. lived seeder	Low shrub
17	RESTIONACEAE	*Chondropetalum microcarpum*	Passive	L. lived seeder	Graminoid
18	RESTIONACEAE	*Chondropetalum nudum*	Passive	Sprouter	Graminoid
19	ROSACEAE	*Cliffortia filifolia*	Passive	L. lived seeder	Low shrub
20	ROSACEAE	*Cliffortia fulcata*	Passive	L. lived seeder	Low shrub
21	ROSACEAE	*Cliffortia obcordata*	Passive	L. lived seeder	Low shrub
22	ROSACEAE	*Cliffortia subsetacea*	Passive	L. lived seeder	Low shrub
23	ASTERACEAE	*Cullumia setosa*	Ant	L. lived seeder	Low shrub
24	POACEAE	*Cynodon dactylon*	Passive	Sprouter	Graminoid
25	CYPERACEAE	*Cyperus odoratus*	Passive	L. lived seeder	Graminoid
26	CYPERACEAE	*Cyperus sphaerospermus*	Passive	L. lived seeder	Graminoid
27	PROTEACEAE	*Diastella divaricata*	Ant	L. lived seeder	Low shrub
28	RUTACEAE	*Diosma hirsuta*	Ant	Sprouter	Low shrub
29	RUTACEAE	*Diosma oppositifolia*	Ant	Sprouter	Low shrub
30	ASTERACEAE	*Disparago anomala*	Passive	S. lived seeder	Low shrub
31	ASTERACEAE	*Disparago lasciocarpa*	Wind	S. lived seeder	Low shrub
32	POACEAE	*Ehrharta villosa*	Passive	Sprouter	Graminoid
33	RESTIONACEAE	*Elegia cuspidata*	Passive	Sprouter	Graminoid
34	RESTIONACEAE	*Elegia filacea*	Passive	Sprouter	Graminoid
35	ASTERACEAE	*Elytropappus scaber*	Wind	S. lived seeder	Low shrub
36	MESEMBRYANTHEMACEAE	*Erepsia anceps cf.*	Passive	S. lived seeder	Forb
37	MESEMBRYANTHEMACEAE	*Erepsia bracteata cf.*	Passive	S. lived seeder	Forb
38	ERICACEAE	*Erica clavisepala*	Passive	L. lived seeder	Low shrub
39	ERICACEAE	*Erica corifolia*	Passive	L. lived seeder	Low shrub
40	ERICACEAE	*Erica gilva*	Passive	L. lived seeder	Low shrub
41	ERICACEAE	*Erica gnaphaloides*	Passive	L. lived seeder	Low shrub
42	ERICACEAE	*Erica hirtiflora*	Passive	L. lived seeder	Low shrub
43	ERICACEAE	*Erica imbricata*	Passive	L. lived seeder	Low shrub
44	ERICACEAE	*Erica laeta*	Passive	L. lived seeder	Low shrub
45	ERICACEAE	*Erica multumbellifera*	Passive	L. lived seeder	Medium shrub
46	ERICACEAE	*Erica phylicifolia*	Passive	L. lived seeder	Low shrub
47	ERICACEAE	*Erica plukenetti*	Passive	L. lived seeder	Low shrub
48	ERICACEAE	*Erica pulchella*	Passive	L. lived seeder	Low shrub
49	ERICACEAE	*Erica subcapitata*	Passive	L. lived seeder	Low shrub
50	ERICACEAE	*Erica subdivaricata*	Passive	L. lived seeder	Low shrub
51	EBENACEAE	*Euclea racemosa*	Passive	Sprouter	Tall shrub
52	ASTERACEAE	*Felicia tenela*	Wind	S. lived seeder	Forb
53	POACEAE	*Festuca scabra*	Passive	Sprouter	Graminoid
54	CYPERACEAE	*Ficinia deusta*	Ant	Sprouter	Graminoid
55	CYPERACEAE	*Ficinia filiformis*	Passive	Sprouter	Graminoid
56	CYPERACEAE	*Ficinia secunda cf.*	Ant	Sprouter	Graminoid
57	CYPERACEAE	*Ficinia sp. 1*	?	Sprouter	Graminoid

58	CYPERACEAE	*Ficinia sp. 2*	?	Sprouter	Graminoid
59	CYPERACEAE	*Ficinia trichoides*	Ant	Sprouter	Graminoid
60	GERANIACEAE	*Geranium incanum*	Wind	S. lived seeder	Forb
61	THYMELAEACEAE	*Gnidia juniperifolia*	Passive	L. lived seeder	Low shrub
62	THYMELAEACEAE	*Ginidia oppositifolia*	Passive	Sprouter	Low shrub
63	ASTERACEAE	*Helicrysum anomilum*	Wind	S. lived seeder	Low shrub
64	ASTERACEAE	*Helicrysum crispum*	Wind	S. lived seeder	Low shrub
65	ASTERACEAE	*Helicrysum dasyanthum*	Wind	S. lived seeder	Low shrub
66	STERCULIACEAE	*Hermania procumbens*	Passive	S. lived seeder	Forb
67	STERCULIACEAE	*Hermania ternifolia*	Passive	Sprouter	Low shrub
68	FABACEAE	*Indegofera brachystachya*	Passive	L. lived seeder	Low shrub
69	RESTIONACEAE	*Ischyrolepis capensis*	Passive	Sprouter	Graminoid
70	RESTIONACEAE	*Ischyrolepis cincinnata*	Passive	Sprouter	Graminoid
71	JUNCACEAE	*Juncus capensis*	Passive	S. lived seeder	Graminoid
72	JUNCACEAE	*Juncus krausii*	Passive	S. lived seeder	Graminoid
73	THYMELAEACEAE	*Lachnea densiflora*	Ant	L. lived seeder	Low shrub
74	MESEMBRYANTHEMACEAE	*Lampranthus tenuifolius cf.*	Passive	L. lived seeder	Forb
75	PROTEACEAE	*Leucadendron coniferum*	Wind	L. lived seeder	Tall shrub
76	PROTEACEAE	*Leucadendron salignum*	Wind	Sprouter	Low shrub
77	PROTEACEAE	*Leucospermum conocarpodendron*	Ant	Sprouter	Tall shrub
78	CAMPANULACEAE	*Lightfootia longifolia*	Passive	L. lived seeder	Low shrub
79	CAMPANULACEAE	*Lightfootia tenela*	Passive	L. lived seeder	Low shrub
80	LOBELIACEAE	*Lobelia setacea*	Passive	S. lived seeder	Forb
81	POACEAE	*Merxmuellera stricta cf.*	Passive	Sprouter	Graminoid
82	ASTERACEAE	*Metalasia brevifolia*	Wind	L. lived seeder	Low shrub
83	ASTERACEAE	*Metalasia cephalotes*	Wind	L. lived seeder	Low shrub
84	ASTERACEAE	*Metalasia muricata*	Wind	L. lived seeder	Medium shrub
85	LOBELIACEAE	*Monopsis lutea*	Passive	S. lived seeder	Forb
86	MYRICACEAE	*Myrica quercifolia*	Passive	Sprouter	Low shrub
87	ASTERACEAE	*Oedera prolifera*	Wind	S. lived seeder	Forb
88	SCROPHULARIACEAE	*Oftia africana*	Passive	Sprouter	Low shrub
89	OLEACEAE	*Olea capensis*	Bird	Sprouter	Tall shrub
90	OLEACEAE	*Olea exasperata*	Passive	Sprouter	Low shrub
91	GENTIANACEAE	*Orphium frutescens*	Passive	S. lived seeder	Low shrub
92	POACEAE	*Paspalum dilatatum*	Passive	Sprouter	Graminoid
93	THYMELAEACEAE	*Passerina paleacea*	Passive	L. lived seeder	Low shrub
94	THYMELAEACEAE	*Passerina vulgaris*	Passive	L. lived seeder	Tall shrub
95	GERANIACEAE	*Pelargonium betulinum*	Wind	Sprouter	Low shrub
96	GERANIACEAE	*Pelargonium capitatum*	Wind	Sprouter	Forb
97	GERANIACEAE	*Pelargonium chamaedrypholium*	Wind	Sprouter	Forb
98	GERANIACEAE	*Pelargonium cuculatum*	Wind	Sprouter	Low shrub
99	GERANIACEAE	*Pelargonium rapaceum cf.*	Wind	Sprouter	Low shrub
100	POACEAE	*Pennisetum macrourum cf.*	Passive	Sprouter	Graminoid
101	POACEAE	*Pentashustis acinosa*	Passive	Sprouter	Graminoid
102	POACEAE	*Pentashustis aurea*	Passive	Sprouter	Graminoid
103	POACEAE	*Pentashustis curvifolia*	Passive	Sprouter	Graminoid
104	AIZOACEAE	*Phamaceum cordifolium*	Passive	S. lived seeder	Forb
105	RHAMNACEAE	*Phylica dodii*	Passive	L. lived seeder	Low shrub
106	RHAMNACEAE	*Phylica imberbis*	Passive	Sprouter	Low shrub
107	RHAMNACEAE	*Phylica ericoides*	Passive	L. lived seeder	Low shrub
108	RHAMNACEAE	*Phylica parviflora*	Passive	L. lived seeder	Low shrub
109	RHAMNACEAE	*Phylica stipularis*	Passive	Sprouter	Low shrub
110	RESTIONACEAE	*Platycaulos compressus*	Passive	Sprouter	Graminoid
111	ASTERACEAE	*Plecostachys serpyllifolia*	Wind	S. lived seeder	Low shrub
112	POACEAE	*Pseudopentamaris macrantha*	Passive	Sprouter	Graminoid
113	FABACEAE	*Psoralea aphylla*	Passive	L. lived seeder	Medium shrub
114	FABACEAE	*Psoralea laxa*	Passive	Sprouter	Low shrub
115	FABACEAE	*Psoralea pinnata*	Passive	L. lived seeder	Tall shrub
116	CELASTRACEAE	*Pterocelastrus tricuspidatus*	Bird	Sprouter	Tall shrub
117	RESTIONACEAE	*Restio bifidus*	Passive	L. lived seeder	Graminoid
118	RESTIONACEAE	*Restio leptostachys*	Passive	Sprouter	Graminoid

119	RESTIONACEAE	*Restio praeacutus*	Passive	Sprouter	Graminoid
120	RESTIONACEAE	*Restio quinquefarius*	Passive	Sprouter	Graminoid
121	RESTIONACEAE	*Restio tetragonus*	Passive	L. lived seeder	Graminoid
122	RESTIONACEAE	*Restio triticeus*	Passive	Sprouter	Graminoid
123	ANACARDIACEAE	*Rhus glauca*	Passive	Sprouter	Tall shrub
124	ANACARDIACEAE	*Rhus laevigata*	Passive	Sprouter	Low shrub
125	ANACARDIACEAE	*Rhus lucida*	Passive	Sprouter	Tall shrub
126	MESEMBRYANTHEMACEAE	*Rushia sarmentosa*	Passive	S. lived seeder	Forb
127	ERICACEAE	*Salaxis axilaris*	Passive	L. lived seeder	Low shrub
128	SCHIZAEACEAE	*Schizea pectinata*	Wind	Sprouter	Graminoid
129	ERICACEAE	*Scyphogyne muscosa*	Passive	L. lived seeder	Low shrub
130	SELAGINACEAE	*Selago cylindrica cf.*	Passive	S. lived seeder	Low shrub
131	SELAGINACEAE	*Selago dregei cf.*	Passive	S. lived seeder	Low shrub
132	ASTERACEAE	*Senecio pubigerus*	Wind	S. lived seeder	Forb
133	ASTERACEAE	*Senecio rigidis*	Wind	S. lived seeder	Medium shrub
134	PROTEACEAE	*Serruria glomerata*	Ant	L. lived seeder	Low shrub
135	PROTEACEAE	*Serruria villosa*	Ant	L. lived seeder	Low shrub
136	POACEAE	*Sporobolus capensis*	Passive	Sprouter	Graminoid
137	BRUNIACEAE	*Staavia radiata*	Ant	Sprouter	Low shrub
138	POACEAE	*Stenotaphrum secundatum*	Passive	Sprouter	Graminoid
139	ASTERACEAE	*Stoebe cuspidata*	Wind	L. lived seeder	Medium shrub
140	ASTERACEAE	*Stoebe incana*	Wind	L. lived seeder	Low shrub
141	THYMELAEACEAE	*Struthiola ciliata*	Passive	Sprouter	Low shrub
142	THYMELAEACEAE	*Struthiola dodecandra*	Passive	Sprouter	Low shrub
143	SCROPHULARIACEAE	*Sutera hispida*	Passive	S. lived seeder	Low shrub
144	ASTERACEAE	*Syncarpha vestita*	Wind	L. lived seeder	Low shrub
145	CYPERACEAE	*Tetraria compacta*	Passive	Sprouter	Graminoid
146	CYPERACEAE	*Tetraria sp. 1*	?	Sprouter	Graminoid
147	RESTIONACEAE	*Thamnochortus fraternus*	Wind	Sprouter	Graminoid
148	RESTIONACEAE	*Thamnochortus fruticosus*	Wind	Sprouter	Graminoid
149	RESTIONACEAE	*Thamnochortus gracilis*	Wind	Sprouter	Graminoid
150	POACEAE	*Tribolium uniolea*	Passive	Sprouter	Graminoid
151	ASTERACEAE	*Troglophyton parvulum*	Wind	S. lived seeder	Forb
152	ASTERACEAE	*Ursinia crithmifolia*	Passive	S. lived seeder	Forb
153	ASTERACEAE	*Ursinia tenuifolia*	Passive	S. lived seeder	Forb
154	ASTERACEAE	*Vellereophyton dealbatum*	Wind	S. lived seeder	Forb
155	CAMPANULACEAE	*Wahlenbergia procumbens*	Passive	S. lived seeder	Forb
156	RESTIONACEAE	*Willdenowia glomerata*	Ant	Sprouter	Graminoid
157	ZYGOPHYLLACEAE	*Zygophyllum spinosum*	Ant	L. lived seeder	Low shrub