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Impact of fire on geophyte abundance, diversity and composition in the Jarrah Forests of South-West Western Australia

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Impact of Fire on Geophyte Abundance, Diversity and Composition in the Jarrah Forests of south-west Western Australia.

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Photo: *Drosera pallida*, Perup Treatment Site (Source: J. Cargill 20/10/04).

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ABSTRACT

Within fire-prone environments, such as the dry sclerophyll forests of south-west Western Australia, species morphology and distributions have established and maintained long-term dynamic relationships with fire regimes. These fire regimes can alter ecosystems, creating spatial and temporal changes in species composition, structure and fuel accumulation at both local and landscape scales.

The cessation of Indigenous fire regimes and the establishment and expansion of a fire vulnerable European society produced the need for fire management strategies such as prescribed burning to be implemented to ensure protection of life, property and other values. Concurrent with these human-focused objectives has been the ever-increasing goal of utilizing these methods for maintaining biodiversity in the long-term.

For prescribed burning to meet the goal of maintaining biodiversity at varying landscape scales, efforts are needed to gather information on taxa where little to no knowledge on fire ecology exists. One group of organisms that is poorly studied in relation to fire ecology are the geophytes, which contribute a significant portion of the south-west flora.

The aim of this research was to address five key objectives via a space-for-time survey, monitoring the impacts of fire on geophyte populations in the southern jarrah forests, at two experimental sites (Perup and McCorkhill) established and maintained by the Department of Conservation and Land Management. The first three interrelated objectives sought to determine the impacts that time since last fire, frequency of fire and season of fire have on geophyte abundance, diversity and composition. The fourth objective was to determine whether ecosystem structure and microclimate are as important to geophyte populations and species assemblages as the direct effects of fire itself. The fifth objective was to ascertain whether microhabitats such as large, fallen burnt logs are used as refuge areas by geophyte species.

The results of this study indicated that geophyte species assemblages were decidedly different between plots experiencing the same fire treatment, and spatially distributed in a very patchy nature within treatment plots. As such, geophytes could be responding to fine-

scale microclimate variations in such things as fire intensity, burn duration, fire effects on the availability of nutrients and fire impaction through changes in community structure. Overall geophyte species richness was shown to peak 5-10 years post-fire within the two treatment sites, which could be a reflection of the time taken for mychorrizal associations to establish. Total geophyte assemblages were found not to be significantly affected by variations in fire treatment, although individual geophyte species did display a significant preference for different fire ages and fire frequencies. Species exhibiting fire treatment preferences were found to be very similar across the two treatment sites, at both a treatment scale and at a micro-scale. Microhabitats such as large, fallen burnt logs were identified as vital refuge areas within a more dry and open jarrah complex.

It is hoped that the knowledge gained from this study will enable fire management plans to continue to adapt and improve by including new research on areas where scientific data has been lacking. New generalisation that can be obtained from this research will hopefully guide the conservation and management of geophytes and hopefully protect their long-term abundance, diversity and composition.

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CHAPTER 1: INTRODUCTION

Fire is a significant environmental factor across terrestrial landscapes and its short and long-term effects on major vegetation types are reasonably well known (Kemp, 1981; Hopper, 2003). This is particularly true in fire prone environments, such as the fynbos shrublands of South Africa, the chaparral shrublands of California and the dry sclerophyll forests of Australia (Gill et al., 1981; Wardell-Johnson, 2003), in which species morphology and distributions have maintained long-term dynamic relationships with fire. Individual plant species within these environments have also adapted to respond to particular fire regimes or to avoid the impact of fire itself (Gill et al., 1981; Keeley, 2002; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003).

Long-term relationships with fire have shaped ecosystems and landscapes, and influenced the life-history strategies of many plant species. This has lead to the evolution of many mechanisms and plant morphologies in response to frequent fire, including obligate seeders, resprouters, fire ephemerals, regular ephemerals and geophytes (Wardell-Johnson, 2003). Strategies such as those mentioned above enable plants to survive fire disturbance and to rapidly recolonise burnt areas (Gill et al., 1981).

Fire regimes are generally considered to have three interrelated components: fire frequency (how often fire occurs), season of fire (what time of year fire occurs) and fire intensity (the heat output from fire) (Bond & van Wilgen, 1996; Morrison, 2002). Fires can also vary in their type: subterranean fires which burn in organic layers of the soil; surface fires that begin their burn just above the ground, such as the litter layer; and crown fires that burn in the canopies of trees (Bond & van Wilgen, 1996). These fire regime factors can alter ecosystems, creating spatial and temporal changes in species composition, structure and fuel accumulation. The last mentioned factor, being the amount of organic matter present, is strongly associated with the availability of moisture and nutrients (Gill et al., 1981; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003).

1.1 Fire in South-west Western Australian Jarrah Forests

The fire-prone environments of the south-west Western Australian jarrah forests are controlled by factors such as climate and weather, topography, vegetation type, fuel amount and ignition sources (lightning and human caused). These factors influence the rates of ignition and the behaviour and extent of fire across the south-west landscape (Gill et al., 1981; McCaw & Hanstrum, 2003). As a result, both the plants and the animals in this region display a wide variety of physical and behavioural adaptive traits that enable them to survive and reproduce in a fire-prone environment (Christensen & Kimber, 1975; Gill et al., 1981; Dixon & Barrett, 2003). The flora of south-west Western Australia has evolved in isolation from other major land masses and the occurrence of fire both natural and human induced has contributed (mostly due to climate and geological processes) to a highly endemic biota in the region (Gill et al., 1981; Dixon & Barrett, 2003; Hopper, 2003; McCaw & Hanstrum, 2003).

Forests in the south-west of Western Australia occur in a Mediterranean type climate characterised by warm, dry summers and mild, wet winters. Climatic variations within the region influence the distribution of major forest types dominated by, jarrah (*Eucalyptus marginata*), marri (*Corymbia calophylla*) and karri (*Eucalyptus diversicolor*) or a mix of these species (McCaw & Hanstrum, 2003). These forests have different understorey components and the amount of organic matter (fuel) generated is therefore variable. Within karri forests between 25 and 30 t/ha of fuel can be accumulated fifteen years post-fire, compared to 10-15 t/ha fifteen years post fire within a jarrah complex. The density of understorey vegetation, nature (size and distribution) and depth of fuel will dictate the intensity of fire (Sneeuwjagt, 1973).

Climate not only affects the type and amount of vegetation (fuel) occurring in a region but it also influences the seasonal patterns of fuel moisture through variations in rainfall, temperature and humidity. Fuel moisture determines the flammability and rate of combustion of woody fuels, thus influencing rate of spread and fuel consumption (Gill et al., 1981). Of major influence to fuel moisture content in forests of the south-west is the reliability and persistence of summer and early autumn drought periods (McCaw & Hanstrum, 2003). During these hot, dry conditions fuels that have remained damp over the

winter months are able to dry. They eventually reach a point at which they are able to burn and this is known as the 'fire season'. A long period of drought lengthens the fire season and increases the likelihood of fire events across the landscape (Gill et al., 1981). Compounding this issue are the dry continental easterly winds that occur during the summer months, further increasing the rate of spread of forest fires by increasing oxygenation to the combustion process (McCaw & Hanstrum, 2003).

Indigenous burning regimes have been a factor instrumental in shaping the landscape of south-west Western Australia. Fire was a common tool used in indigenous society and was utilised in rituals, domestic life and in the process of obtaining food from the land, the latter being termed 'fire-stick farming' (Nicholson, 1981). The fire regimes utilised by indigenous peoples evolved through centuries of experience in burning the landscape, although the specific regimes utilised are still a matter of conjecture (Nicholson, 1981; Abbott, 2003; Hassell & Dodson, 2003). It is speculated that indigenous fire regimes were carried out at a frequency of every 1 to 10 years in highly populated areas, more accurate data suggesting that burns took place every 3-5 years. This is compared to intervals between major fires in uninhabited areas of every 30 to 100+ years (Hassell & Dodson, 2003). Indigenous Australians are thought to have had a controlling effect on vegetation composition and structure through frequent, low intensity burning, which thus involved lower build up of organic matter (Abbott, 2003; Hassell & Dodson, 2003). European settlement and the subsequent cessation of indigenous fire regimes have seen a decrease in fire frequency and a corresponding increase in the intensity of fires (Gill et al., 1981; Hassell & Dodson 2003). The broad scale implications of these changes to fire regimes are still yet to be fully understood.

European settlement in the south-west of Western Australia saw the reduction of indigenous influence on fire regimes and establishment and expansion of a fire-vulnerable society into a fire prone environment. This produced the need for fire-management strategies to be implemented that protect human life and property and protect and enhance forest values (McCaw & Burrows, 1989; Abbott, 2003).

1.2 Prescribed Burning in South-west Western Australian Jarrah Forests

Prescribed burning for the purpose of hazard reduction and regeneration has been used extensively in south-west Western Australia to manage the spatial distribution of fuel ages since the mid 1960's (McCaw & Hanstrum, 2003). Hazard-reduction burning is a standard method utilised to limit the occurrence of heavy fuel loads in forests and reduce the occurrence of large intense wildfires. Regeneration burning following logging utilises high-intensity fire to create favourable seedbeds through the removal of competition and logging trash (Shea et al, 1981).

Prescribed burning objectives have been achieved through rotational prescription burning of small patches creating a mosaic of different age classes. The landscape mosaic created is capable of acting as a barrier to the spread of large wildfires, limiting the scale of high-intensity fires, moderating physical damage to the forest and reducing the possibility of impacts to human interests (Keeley, 2002).

The most common methods used for lighting prescribed fires are incendiaries dropped from aircraft or fires lit by ground crews using torches (Shea et al, 1981). The intervals at which these prescribed burns take place are dependent on the ability of the ecosystem to cope with fire and the fuel loads within a forest. The frequency of fire within an area could therefore range anywhere from 3-4 years in fire tolerant systems to 10-15 years where species require longer periods for phytomass production and for the juvenile phase (Burrows & Abbott, 2003; Hassell & Dodson, 2003). The timing of the burn is also crucial, with fuel moisture and wind speed readings used to determine the Fire Danger Index and the Rate of Spread Index. Coupled with weather predictions, these indices assist in limiting the possibility of uncontrolled fires, thus reducing the risk of spotting and minimizing potential smoke and fire impacts on residential areas (Gill et al., 1981; Sneeuwjagt & Peet, 1985).

The key to the implementation of prescribed burning plans is an understanding of the effects of fire on the biographical properties of the landscape. The complexity of fire regimes within heterogeneous landscapes means that no single fire regime will prove beneficial to all species or to all aspects of the environment. Nevertheless, fire will continue to influence the nature of landscapes and their biodiversity and hence

management plans should be open to change. Flexibility and the capacity to incorporate relevant research findings is important (Burrows & Abbott, 2003).

1.3 Maintaining Biodiversity Through Prescribed Fire Regimes

Prescribed burning of native vegetation for hazard reduction and regeneration has been established as an integral management tool for ensuring protection of life and property. Concurrent with these human-focused objectives, has been the ever-increasing goal of utilizing these methods for maintaining biodiversity in the long-term (Burrows & Wardell-Johnson, 2003). This broadening of the role of prescribed burning practices raises many questions. One concern to ecologists is whether the current human-focused methods of fuel reduction burning can be employed or whether they need to be modified to ensure the maintenance of biodiversity in south-west Western Australian jarrah forests. To answer this question it is essential for research to be undertaken to determine the impacts of different fire treatments on all facets of the ecosystem.

Impacts of fire on larger forest species such as jarrah, banksias and legumes are well documented, and as a result knowledge of their fire ecology is well known (Burrows, 1985; Burrows, Ward & Robinson, 1995; Bell & Pate, 1996). For prescribed burning to meet the goal of maintaining biodiversity at varying landscape scales, efforts are needed to gather information on taxa where little to no knowledge on fire ecology exists. Findings from research of this type may help in devising burning practices that will not adversely affect any of the species or processes within that ecosystem. Research aimed at expanding the knowledge behind burning practices is particularly relevant in Western Australia, given the highly endemic and patchy distribution of many organisms (Dixon & Barrett, 2003). Such taxa are prone to local extinctions following prolonged or intense periods of disturbance such as fire (Huston, 2003).

1.4 Geophytes: Definitions and Known Impacts by Fire

One group of organisms that is poorly studied in relation to fire ecology are the geophytes, which contribute a significant portion of the south-west flora (Burrows & Wardell-Johnson, 2003). Geophytes, as described by Pate and Dixon (1982), are land-dwelling species that perreniate by means of underground storage organs such as bulbs, corms and

tubers. By diverting energy to underground organs rather than to above ground biomass, the plants can remain dormant during adverse periods such as those experienced during drought and natural fire events. The term fire response species has been synonymous with geophytes, with many species from different regions of the world such as *Drosera erythrorhiza*, *Erharta capensis* and *Prasophyllum regium* known to be fire-dependant or fire-induced (flowering, germination) organisms at various stages in their life-cycles (Dixon & Pate, 1978; Hoffman & Brown, 1992; Verboom, Stock & Linder, 2002). Ethylene gas produced by bushfires has been shown as the likely agent for floral stimulation in many geophytes, while heat, smoke and increased nitrogen content due to ash-bed production during fire events is thought to stimulate germination (Dixon and Barrett, 2003).

Vegetative growth of the geophyte storage organ is totally dependent on the photosynthetic gains being made by new foliage (Pate & Dixon, 1982). As a result the seasonality and frequency of fire could hinder the plant's ability to refill or replace the storage organ, by damaging the storage organ or killing above ground biomass. This could impact on the individual plants ability to survive the subsequent growing season, as well as reducing its reproductive output.

Broad speculations about the response of geophytes to fire events have meant that their ecological requirements have often been overlooked in fire management decisions. Although many geophyte species respond favourably to fire, no definitive research has been undertaken within southern jarrah forests to determine how prescribed burning practices are impacting on geophyte diversity and abundance. In particular, the consequences of changes made to the frequency and season of fire treatments on geophyte life cycles have been poorly studied.

Given the lack of knowledge on the impacts of fire treatments on geophytes and the increased use of prescribed burning as a tool for maintaining biodiversity, it is appropriate to research the fire ecology of geophytes in order to ensure that fire management plans continue to adapt and improve by including new research on areas where scientific data have been lacking. New generalisations that can guide conservation and management of geophytes will hopefully protect their long-term abundance, diversity and composition.

1.5 Aims

This project is a preliminary study, in that it will collect and synthesise information on the abundance, diversity and composition of specific groups of geophytes (members of the Orchidaceae, Droseraceae and Stylidiaceae families) within areas of varying fire treatments. These treatment variations include season of fire, frequency of fire and time since last fire. Long-term experimental sites established and maintained by the Department of Conservation and Land Management (DCLM) will be used to determine which treatments reduce and which treatments enhance geophyte abundance and diversity. Limitations through the use of current fire effects sites namely Perup and McCorkhill are discussed within the methodology. By gathering this information, and with the aid of further studies, recommendations may be made to the relevant management bodies about the suitability of various fire-management options based on the response of geophyte species. This should ensure that management is able to preserve the unique species diversity of our southern jarrah forests.

The specific objectives of my project are as follows:

1. *To determine population size, diversity and composition of geophytes in relation to time since last fire.*
2. *To determine population size, diversity and composition of geophytes in relation to frequency of fire treatments.*
3. *To determine population size, diversity and composition of geophytes in relation to the season of fire treatments.*
4. *To determine whether ecosystem structure and its effect on microclimate is more important in determining population sizes of geophytes than the direct effect of fire itself.*
5. *To determine whether large, fallen burnt logs are being utilised by geophytes as a refuge area.*

1.6 Thesis Structure

The aim of this thesis was firstly to examine the impact of season, frequency and time since last fire on abundance, diversity and composition of geophyte populations. This research took place with two sites located in the jarrah forests of south-west Western Australia. Secondly, the aim was to investigate whether ecosystem structure and microclimate are important in controlling geophyte abundance, diversity and composition. Finally, the aim was to determine whether microhabitats such as large, fallen burnt logs are utilised by geophytes as refuge areas.

This thesis consists of four chapters:

Chapter 1 provides an overview of the justification and use of prescribed burning within ecosystems. Further it gives details of the roles natural fire and prescribed fire regimes have on shaping the unique flora of south-west jarrah forests. Lastly the chapter outlines the need for ongoing collection of data about, for example, geophytes in the interest of developing knowledge and understanding of the ecosystem, which is essential to the maintenance of biodiversity in the long term.

Chapter 2 describes the methodology used in this project. This involves a description of the study area within the jarrah forests of the south-west of Western Australia. Detailed descriptions of the Perup and McCorkhill treatment sites are given together with indicates of limitations the study encountered. Sampling design, field and laboratory methods and data analysis are also described.

Chapter 3 presents the results of the project, addressing the five key objectives of this study. The first three intertwined objectives seek to assess whether time since last fire, frequency of fire and season of fire have an impact on geophyte abundance, diversity and composition. Concurrent with these three objectives I attempt to identify species (if any) showing specific fire disturbance preferences, i.e. are 'unique', or are significantly impacted by different treatments. The chapter then addresses the fourth project objective by analysing the impact of microclimate on individual geophyte species. The last section of the chapter deals with the fifth project objective and identifies geophyte species that used microhabitats such as large, fallen burnt logs as refuge areas.

Chapter 4 discusses the findings of the study in the context of other research studies dealing with the impact of fire treatment on the flora of south-west Western Australia. The chapter also outlines recommendations for future studies and raises management implications associated with the findings of this project.

CHAPTER 2: METHODS

2.1 Introduction

This chapter describes the approaches used in the project to address five key objectives given in the introduction through a space-for-time survey, monitoring the impacts of fire on geophyte populations at two sites in the southern jarrah forests. The first three interrelated objectives sought to determine the impacts that Time Since Last Fire (yrs), Frequency of Fire and Season of Fire have on geophyte abundance, diversity and composition. In conjunction with these three objectives, I determined which species (if any) have specific preferences, i.e. are 'unique', or are significantly impacted by different treatments.

The fourth objective was to determine whether ecosystem structure and micro-climate are important in controlling geophyte abundance, diversity and composition. This objective seeks to ascertain whether the secondary impacts of fire treatment (physical alterations in micro-climate) are as important to geophyte populations and species assemblages as the direct effects of fire itself.

Fifthly I also set out to determine whether microhabitats such as large, fallen burnt logs are used as refuge areas by geophyte species, and to assess which species (if any) show significant preferences for these habitats.

2.2 Description of Study Area: South-west Western Australian Jarrah Forests

2.2.1 Location

The south-western section of the Darling Plateau (location of Perup Site, Figure 2.2) extends from the Blackwood River in the west to Mount Barker in the east and stretches from Bridgetown in the north to Walpole (Scott Coastal Plain) in the south (Department of Conservation and Land Management, 1987a). The Blackwood Plateau region (location of McCorkhill Site, Figure 2.2) extends from Margaret River (Dunsborough Fault) in the west to the Blackwood River (Darling Fault) in east and stretches from Swan Coastal Plain (10km south of Busselton) in the north to the Scott Coastal Plain (Brockman Highway) in the south (DCLM, 1987b)

2.2.2 Geology, Topography and Soils

The Darling Plateau is an ancient erosion surface. Composed of Precambrian crystalline rocks that have been extensively laterised (Churchward and McArthur, 1980; DCLM, 1987a). The southern forest region of the plateau has an average elevation of approximately 200m above sea level, ranging from 100m to 435m at Mount Lindsey. Rivers such as the Donnelly, Frankland and Warren dissect this area causing the formation of many valley environments (DCLM, 1987a).

Soils in the southern forest region of the Darling Plateau are diverse with variations occurring at different landscape levels. Higher in the landscape soils consist of laterites with lateritic associated podsol. Leached grey sand with deep pallid zones occurs on or near drainage divides (DCLM, 1987a; Strelein, 1988). Lower in the landscape laterites become less common with many minor rivers dissecting the area. This leads to the formation of brown sands and orange earths, with slopes below laterite consisting of gravely red earths or red podsol (DCLM, 1987b). The eastern section of the area contains gravely red earth and podsolic soils emerging from sandy humus podsol in drainage lines (DCLM, 1987a; Strelein, 1988).

The Blackwood Plateau is underlain by Mesozoic sediments and Bunbury Basalt (Churchward and McArthur, 1980; DCLM, 1987b). The Plateau has a gently undulating surface with elevation ranging from 80m to 180m above sea level. Most of this area is poorly drained with the surface covered by lateritic gravel and sand (DCLM, 1987b).

2.2.3 Climate

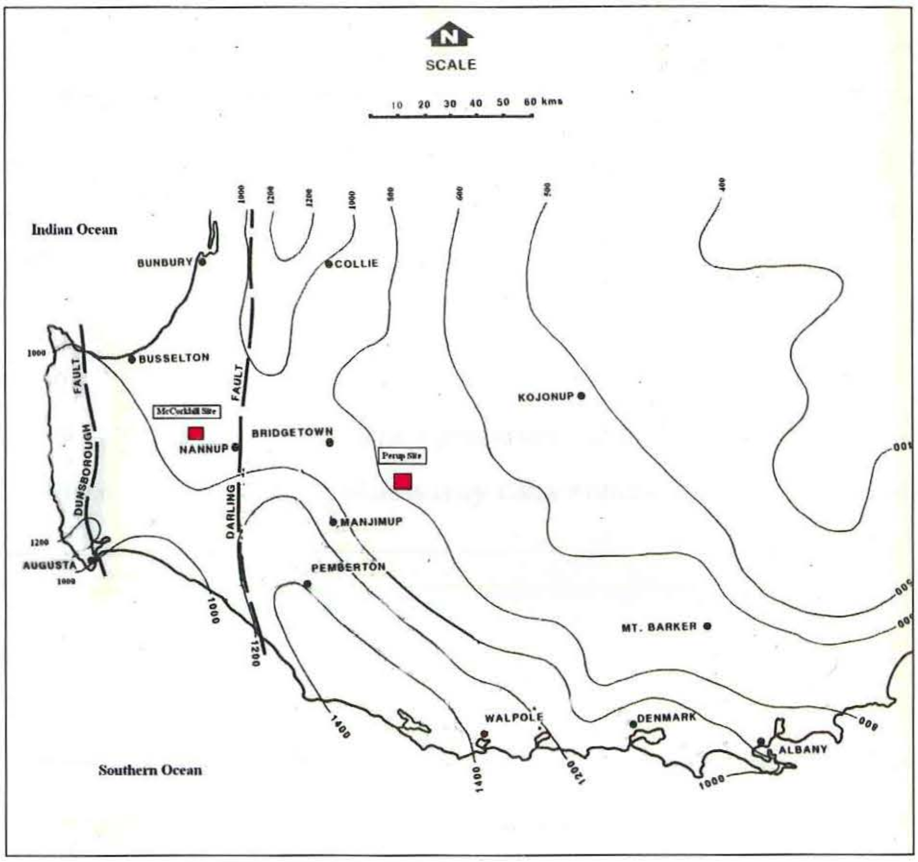

Figure 2.1 Rainfall map for the south-west Western Australian jarrah forests, indicating a decline in annual rainfall from the McCorkhill Site in the west to the Perup Site in the east (Adapted from Map 4B: DCLM, 1987 A).

The climatic conditions of the south-west jarrah forests are generally described as Mediterranean with cold wet winters and hot dry summers. Oceanic and topographical features strongly influence the duration and degree of rainfall in these areas (DCLM, 1987a+b; Strelein, 1988). Annual rainfall ranges from 1400mm in the south to around 600mm at the north-eastern boundary. This reflects the north-east to south-west pattern of rainfall within the southern forests (Figure 2.1) (DCLM, 1987a+b). Temperature variations in minimum and maximum values are seen to be lowest in the northern regions (Bridgetown) of the southern forests, with summer temperatures being the lowest closer to the coast. These variations across the region have a dramatic impact on the growing season of most plants, with peak growth season in the far south occurring mid February, end of March within the central and mid April in the dryer north-west areas of the region (DCLM, 1987a+b; Strelein, 1988).

2.2.4 Vegetation

The southern forests region of the Darling Plateau consists predominantly of Jarrah/Marri in the north and eastern areas with Karri/Marri/Jarrah forests in the south an south west of the region (Strelein, 1988). This region also consists of small pockets of Marri/Wandoo/Jarrah woodland in the north-east. Swamp/sand ridge areas, Sheoak/Banksia/Jarrah and other low woodlands are dotted across the landscape (DCLM, 1987a; Strelein, 1988).

The Blackwood Plateau region primarily consists of Jarrah/Marri forests covering the landscape. Low woodlands of *Melaleuca preissiana* and Banksia species occupy damp areas within the complex. Shallow uplands may carry stunted jarrah and grass trees (*Kingia australis*) (DCLM, 1987b).

2.3 Site Selection

Two sites (Perup and McCorkhill) established and maintained by the Department of Conservation and Land Management to investigate long-term effects of different fire regimes on woody plants chosen given their available fire records and broad ranging age, frequency and seasonal treatments. The location of the two sites was illustrated on Figure 2.2.

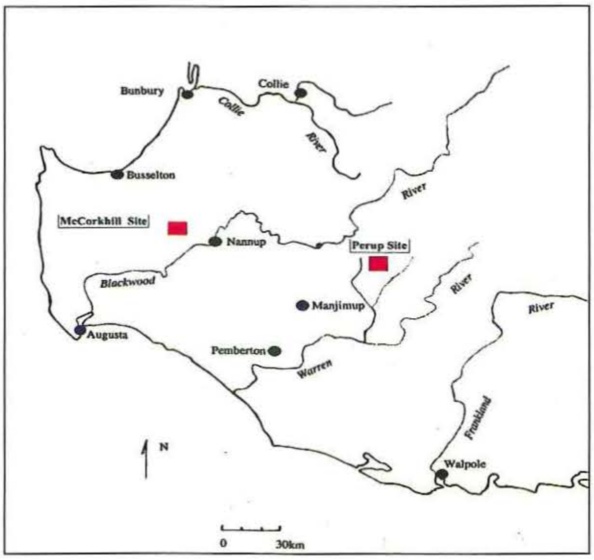

Figure 2.2 Location of Perup and McCorkhill study sites relative to major regional centres of south-west Western Australia. (Source: Ward, 2004).

2.3.1 Perup Treatment Site

Plate 1. Jarrah complex of the Perup Treatment Site. (Source J. Cargill 18/10/04)

Site description

The Perup Treatment Site is situated 350m east of the Northern and Boyup Brook-Cranbrook Road intersection. It consists of 12, 200m x 200m, plots set up in a 'grid-like' formation. This site was established in the 1981/82 fire season and as such long unburnt reference areas (controls) are 22 years old.

The Perup site is a relatively open and homogenous jarrah/marri (*E. marginata, C. calophylla*) forest located in close proximity to marri/wandoo jarrah woodland forests (*E. calophylla, E. wandoo*) (Figure 2.3). This site has a low annual rainfall of approximately 600-800mm (Figure 2.1). The terrain is very flat with shallow sand on laterite (Figure 2.1). A distinct feature of the Perup Site is the abundance of *Macrozamia riedlei* amongst the understorey within the majority of plots. Large quantity of woody debris are scattered across the treatment from past logging activities. Intensive herbivory was also observed.

2.3.2 McCorkhill Treatment Site

Plate 2 Jarrah complex of the McCorkhill Treatment Site. (Source J. Cargill 20/10/04)

Site description

The McCorkhill site is situated off Junction Road approximately 20km west of Nannup town centre. It consists of 12, 200m x 200m plots set up in 'grid-like' formation. This site was established in 1983, with long unburnt reference areas (control) dating back to 1976. Burning of the vegetation at the northern end of the site proved beneficial giving an additional age structure of 1 year post-fire.

The McCorkhill Site is a fairly dense jarrah/marri (*E. marginata, C. calophylla*) forest with a prominent understorey consisting mainly of Casuarina and Banksia species (Figure 2.4). The annual rainfall of this site is approximately 1000mm given the orographic effect of its location 20km from the foot of the Darling Scarp and close distance to the ocean (60km) (Figure 2.1 & 2.2). The terrain is slightly undulating with deep sands over laterite. Laterite nodules protrude along small ridges in the centre of the site.

2.4 Sampling Design

2.4.1 Taxa Selection

Geophyte taxa selection was based on issues such as time available for field work, presence of flowering specimens during the sampling period (September to November) and indications from the literature that some species are more prone to fire impacts. Preliminary surveys and information gathered from DCLM Fire Effects Vegetation Lists conducted on the treatment sites recorded geophyte species present within the Anthericaceae, Amaranthaceae, Asteraceae, Colchidaceae, Droseraceae, Orchidaceae and Stylidiaceae families. After careful consideration it was decided that data collection would revolve around species from the Droseraceae, Orchidaceae and Stylidiaceae families. Species from these families were relatively simple to identify both at early life cycle (rosette) and later sample periods (15th September to the 15th of November). Many of the species are at peak flowering from September through November which makes identification and recognition in the field considerably easier during the survey sweeps.

2.4.2 Plot and Treatment Selection

Plot and treatment selection was based on the existing experimental design within the Perup and McCorkhill Sites. Unavoidable limitations such as limited replication, restricted plot size and the choice of treatments were faced when attempting to match treatments across sites. The most significant problems encountered were the validity of comparing variations in frequency and season across plots with different age structures and being able to determine whether frequency, season or age was the determining factor for plots with contrasting species abundance, diversity and composition (Gill et al, 1981; Morrison, 2002; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003). Despite the limitations plots were selected to cover a range of treatments and thus enhance the chances of answering the project's objectives.

Further concerns with the selection of plots included the lack of knowledge about the historical fire regimes of the two sites and the lack of information pertaining to geophyte abundance, diversity and composition pre the onset of the current fire treatments (1981/82 for the Perup Treatment Site and 1976 for the McCorkhill Treatment Site). The lack of

records of past occurrences of wildfires in and around the research areas and the impact the number and intensities of these events could have had (if any) on the floristic composition of the treatment sites prior the application of treatments is unknown. Although the occurrence of wildfires could indeed have impacted geophyte populations the site size and site homogeneity may in fact negate this issue in terms of 'within site' data analyses, since major fires would have affected the entire site not specific plots within the site. The degree to which fire intensity and other fine scale variations differed across the site also remains unknown.

2.4.2.1 Perup Treatment Site

Due to the limited range of available treatments within the Perup Treatment Site, plots were only selected along a gradient of time since last fire (Table 2.1).

Table 2.1 Details of the plots selected in relation to time since last fire at the Perup Treatment Site.

Time Since Last Fire (yrs)	Plot Number	Replicate Plot Number
2	2	10
9	4	5
10	1	12
12	7	9
22	3	6

Within the Perup Treatment Site 5 age ranges were used with each age having two replicate plots per fire treatment: 2 year old (plot 2 and 10), 9 year old (plot 4 and 5), 10 year old (plot 1 and 12), 12 year old (plots 7 and 9) and 22 year old control (plots 3 and 6). By selecting these plots data were gathered from recently burnt treatments (2 years old), intermediate aged treatments (9, 10 and 12 years old) and treatments that have remained unburned for a long period (22 years old) (Table 2.1).

2.4.2.2 McCorkhill Treatment Site

In comparison to the Perup Treatment Site the McCorkhill Treatment Site displayed a wide treatment variety between plots and as such data were gathered to examine the time since last fire, frequency and season of burn effects on geophyte abundance, diversity and composition (details of the plots are given in Table 2).

Table 2.2 Details of the plots selected in relation to time since last fire, frequency and season of fire within the McCorkhill Treatment Site.

Plot Number	Time Since Last Fire (yrs)	Number of Fires Since 1976	Season of Fire
12 and 13	1	~	~
4 and 9	4	4	Autumn/April
3 and 5	5	4	Summer/December
6 and 10	5	5	Summer/February
8 and 11	10	2	Spring/November
7 and 14	21	2	Summer/February
1 and 2	28	1	~

Time Since Last Fire

Within the McCorkhill Treatment Site a range of 6 age treatments were utilized: 1 year old (plots 12 and 13), 4 year old (plots 4 and 9), 5 year old (plots 6 and 10), 10 year old (plots 8 and 11), 21 year old (plots 7 and 14) and 28 years old control (plots 1 and 2) (Table 2). It must be noted that the 21 year old treatment was only contained within one plot (plot 7) as a result plot 7 was divided in half with those two halves being treated as the individual plots of 7 and 14.

Frequency of Fire

Within the McCorkhill Site 4 different fire frequencies were utilized: 1 treatment (plots 1 and 2), 2 treatments (plots 8 and 11), 4 treatments (plots 3 and 5) and 5 treatments (plots 6 and 10) (Table 2.2). Frequency records for the site began in 1976 and as such data on treatments and/or wildfires before that time could not be included, i.e. Plots 6 and 10 have had 5 known treatments since 1976.

Season of Fire

Within the McCorkhill Treatment Site 4 different treatment seasons were utilised: Autumn/April treatments (plots 4 and 9), Summer/February treatments (plots 6 and 10), Summer/ December treatments (plots 3 and 5) and Spring/November treatments (plots 8 and 11) (Table 2). The Spring/November treatments within plots 8 and 11 were substantially older than the other season of fire plots, 10 years old in comparison to plots 4 and 9, 6 and 10 and 3 and 5 that were 4, 5 and 5 years old respectively.

2.5 Sampling Methods

Sweep Transects

Many factors needed to be taken into account when devising a sampling method for obtaining baseline species data within treatment plots. First and foremost was the problem of gaining an accurate assessment of species richness and abundance given the patchy distribution of geophytes and the inherent large size of the plots (200m x 200m). As such any sampling methods utilized needed to not only cover substantial ground in order to gather sufficient numbers for analysis but spatially allow for the patchy distribution of geophytes.

The second issue was the small size of the majority of geophyte species in question, especially those individuals not in flower. As a result many species were only identifiable by small rosettes of leaves close to the ground as with orchids in general and species such as *Drosera glandulegera* that at full size only measures approx. 5mm in diameter

(Wildflower Society of Western Australia, 2001). Therefore a sampling method would not only have to account for the small size of species and the fact that individuals could easily be missed or overlooked but also take into consideration the amount of time it would take to effectively and efficiently sample twenty-six 200m x 200m plots.

Sweep transects were selected as the principal technique for the collection of baseline species data within both the Perup and McCorkhill Treatment Sites. Within each plot five 100m transects were set out (Figure 2.5) with individuals recorded as per a 1m ground-survey sweep either side of the main transect line. In addition, each individual transect was also divided into four 25m sections (Figure 2.5). Note: To negate possible edge effects from the boundary fire-breaks transects started 5-10m inside the plots.

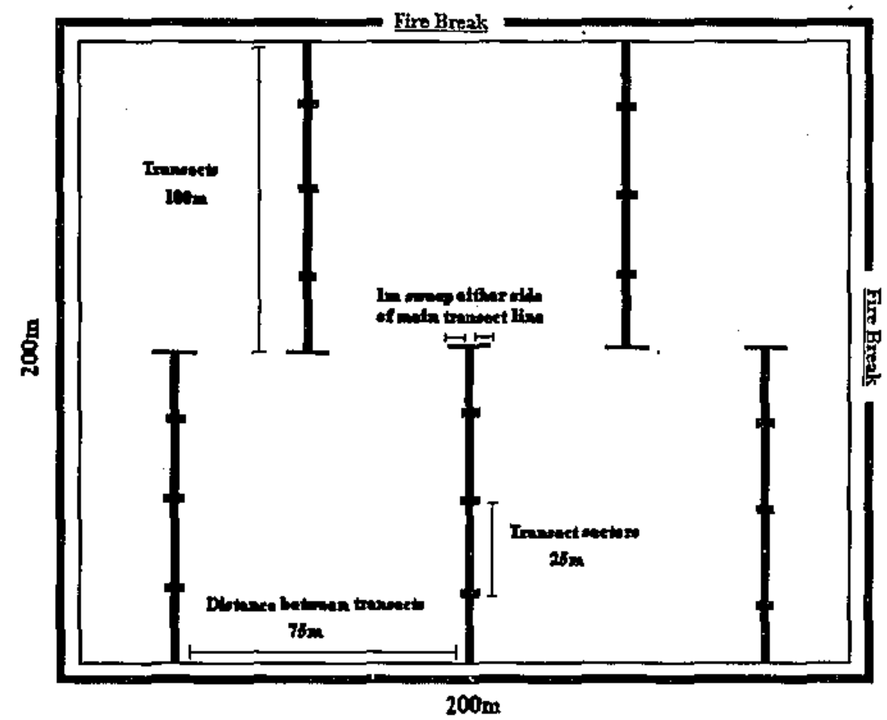

Figure 2.3 Layout of the sweep transect sample design for plots within the Perup and McCorkhill Treatment Sites. Plots were approximately 200m x 200m with five 100m long transects marked out alternatively across the area. Each transect was further divided into four 25m sectors for micro-scale analysis. Ground-level surveys or 'sweeps' were carried out 1m either side of the main transect line.

Sweep transects allowed for the intensive surveying of a large area sufficient to ensure representative data was collected from each plot and associated treatment, in the most accurate and time efficient way possible. By dividing transects into four 25m sections the

spatial distribution of individuals and possible micro-scale correlations between indirect factors could be gauged and subsequently these markers served as a sample point for soil collection and litter depth measurements. The length of transects also gave the most effective means of picking up fine scale variations in species abundance and richness that may occur due to fire intensity or burn rate changes across the plot i.e. measurements were taken from the outer boundary of the plot right into the centre of the plot (Figure 2.5).

The 100m transects lines and 25m sections within, were marked out using 1.5m jarrah stakes with painted tips enabling them to be easily seen amongst the dense shrubs of some of the long unburnt plots. Given that one-hundred and thirty 100m transects were set out across 2 sites each with 5 stakes (650 stakes), time constraints and the fact that only one person was putting out the stakes, a precise measurement of distances utilizing a 100m measuring tape proved almost impossible. As such the transect distances were measured by pacing out the 100m, although this seems like a crude method, 1 pace approximately equalled 1m in any event, It was then possible to set up the transects very quickly whilst still maintaining an acceptable degree of accuracy in terms of distance.

2.6 Field Methods

2.6.1 Baseline Data

Taxa collection and identification

Individuals observed along each sweep transect were identified in the field where possible and samples taken (vegetative structures, flowers) and placed in a 'field sample folder'. These samples were subsequently assigned a number for future reference in order to avoid duplication in identification (folder was also pressed in order to preserve flowers vital in the identification phase). As a result every new species was sampled and numbered within this folder allowing for fast and accurate field identification as a taxa database with species samples was on hand to compare and contrast individuals. Where individuals could not be identified they were still numbered and added to the sample folder but identified at a later date utilizing flora key guides that dealt specifically with the three focus families.

Many of the orchid species collected were still in their early vegetative growth phase and were only visible as small rosette leaves. Without the orchids characteristic floral structures and with only small rosette leaves to work with, identification proved extremely difficult. Assistance was therefore sought from DCLM botanists and DCLM orchid expert Andrew Brown.

Soil Sampling

Soil samples were taken in the middle of every 25m section (refer to Figure 2.5) as such 4 samples were collected for each transect, 20 samples per plot (5 transects) and in addition with the replicate plot 40 soil samples were taken for every treatment. Approximately, the first 5cm of topsoil was collected from each sample point as this is where the vast majority of the storage organs and root systems from sampled species were located (Pate & Dixon, 1981). Utilizing a small trowel the soil was collected and placed in an airtight snap-lock bag with the site, plot, transect and section number labelled for future analysis.

Canopy Cover

Canopy cover was measured at 5m intervals along each transect utilizing a 'periscope' that consisted of a 2m long pole with a mirror inserted at 45 degrees within a piece of PVC pipe, a cross was then made using wire at the top of the PVC pipe. By holding the pole upright and looking straight into the mirror a cross could be seen, if vegetation (canopy) happened to be lined up by the centre of the cross it was marked as yes or indeed if blue sky was in line with the cross it marked as no.

By walking along the transect and looking onto the periscope at 5m intervals a yes/no ratio was gathered and from that an average canopy cover for that section, transect, plot and finally treatment could be determined. At a fine scale, 5 readings were taken for each transect section and 20 readings taken at a whole transect scale, for example readings that resulted in ratio of 8 'no' readings and 12 'yes' readings would have revealed an average canopy cover of 60% for that transect.

Litter Depth

Litter depth was measured in the middle of every 25m section within the sweep transects (Figure 2.5). Consequently 4 readings were taken per transect, 20 per plot and 40 per treatment. At each sample point along the transect line a ruler was carefully inserted into the leaf litter until it reached the topsoil. Readings were then taken for the distance in centimetres from the topsoil to the top of the litter layer. A plastic ruler was utilised that had been filed down to avoid having to compensate for the 1cm of extra plastic after the 0 cm mark.

Shrub Cover

Shrub cover data were gathered by measuring the amount of cover for each 25m section as compared to the amount of open area. Using Figure 2.6 as an example, the distance along each 25m section covered by shrubs was measured and subsequently collated i.e. shrubs cover 16m of the 25m distance giving a shrub cover of 64%. This data was then averaged for the entire transect line, plot then treatment.

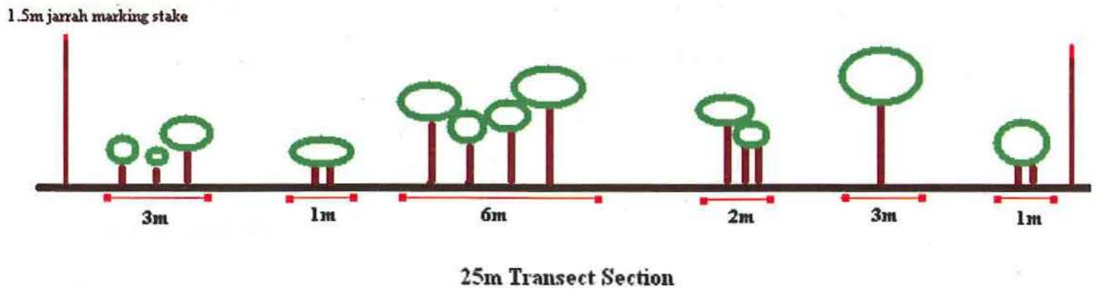

Figure 2.4 Illustrates the method utilised in gathering percentage shrub cover data within the Perup and McCorkhill Treatment Sites.

2.6.2 Microhabitat data

Log Selection

Log data was gathered within both the Perup and McCorkhill Treatment Sites
in order to assess species density and richness in geophyte species, (orchids in particular) in and around large burnt logs and their associated ash-beds. Along each sweep transect 1

large log (minimum size 5m x 0.5m) was selected, giving 5 logs per plot and 10 logs per treatment.

Data was gathered by identifying and counting species that existed either on the log, under the log or within the ash-bed (identified visually through soil colour and presence of burnt woody debris). This enabled a measure of the species density and richness within log micro-habitats to be gained, the size of the selected logs and the distance from the 0m transect point was also measured.

2.7 Laboratory Methods

Soil Analysis

Soils samples were placed in aluminium trays and dried at 100 degrees C for 24 hours and sieved to remove large particulate matter (>2mm). Utilizing two replicates for each section sample (refer Figure 2.5) soil pH, electrical conductivity (EC) and salinity were measured. The pH of the soil was measured through a calibrated glass electrode pH meter. Calcium Chloride ($CaCl_2$) was added to the replicate sample and a mean value gathered between the two samples. Electrical conductivity and subsequent salinity readings were gathered using a calibrated conductivity meter. A mean value between the two replicates was then obtained.

2.8 Data Analysis

Species Evenness

Species Evenness calculations were undertaken in order to show the relative distribution of individuals among species, with a low evenness value signifying that an individual species was dominating the treatment plot (Fowler, Cohen & Jarvis, 1998).

The following formula was used to calculate Species Evenness (E):

$E = H / \ln (R)$ Where: E = Species Evenness value
H = Shannon-Weaver Diversity Index ($-\sum^{s} p_i \ln p_i$)
R = Species Richness

Species evenness was calculated utilizing the Perup and McCorkhill species matrix at a transect level and subsequently a Mean Species Evenness value was obtained for each treatment. In doing so identifying whether individual species were present in a higher relative abundance than other species.

Analysis of Variance (ANOVA)

One-Way Analysis of Variance was conducted within Microsoft Excel to test the variation between mean values of species density, species richness and species evenness between treatments within the Perup and McCorkhill Treatment Sites. As such the ANOVA tests were utilized to gauge the significance of variation between plots with different treatments (P-value < 0.05) (Tabachnick & Fidell, 1996; Fowler et al, 1998). In the case of species density data were log transformed in order to eliminate the skewed values, mainly caused by the large populations of *Drosera menziesii* throughout both treatment sites.

Although the ANOVA test was able to indicate whether a difference between groups existed it was not able to identify which group(s) were indeed different i.e. whether the intermediate values were significantly different from the extreme values (Tabachnick & Fidell, 1996). Given the low statistical power of the base line data (low replication) multi-comparison tests were not conducted. To combat this problem line and bar graphs with error bars were constructed to illustrate the trends across different treatments as to species density, richness and evenness.

Multi- Dimensional Scaling (MDS Ordinations)

Multi-Dimensional Scaling through PRIMER (Plymouth Routines in Multivariate Ecological Research, version 5.2.1) was utilised in order to group plots within and between the Perup and McCorkhill Treatments Sites according to their similarity in species composition. Using the base species data at transect and site level a similarity matrix was developed using a Bray-Curtis similarity measure. Then by performing random permutations on this data, a 2-D ordination diagram was produced via MDS. This ordination illustrates the spatial representation between plots as to the relative difference in their species composition (Tabachnick & Fidell 1996; Anderson & Robinson, 2003).

ANOSIM (Analysis of Similarity)

ANOSIM (Analysis of Similarity) was utilised to test the similarity of species composition between the Perup and McCorkhill Treatment Sites, in order to identify whether the treatment plots within the two sites could be combined to increase replication and thus statistical power. ANOSIM tests were also used to test the difference in species composition between treatments within the same site. These results are expressed as a Global R value and a significance level of sample statistic. A Global R value less than 1 and significance level of sample statistic less than 0.5, the species composition between the sites is significantly different and therefore cannot be treated as similar sites.

Canonical Analysis of Principle coordinates (CAP)

CAP analysis was utilised as a tool for identifying 'unique species. That is species showing a significant preference for a certain treatment such as only occurring in only within that treatment or identified in a high or low abundance in comparison to other treatments (Anderson, 2004). CAP utilises a symmetric matrix (in this case mean species data at a transect scale) and calculated a canonical analysis on the principle coordinates or in other words the effect of 'X' on 'Y'. Firstly the program produced a metric multi-dimensional scaling plot of the Bray-Curtis dissimilarities (unconstrained). Secondly providing a constrained ordination plot of the first two canonical axis' by the output of x and y coordinates that have tested the degree to which species do not conform with the main species assemblages across the treatment groups (Tabachnick & Fidell, 1996; Anderson & Willis, 2003; Anderson, 2004).

The CAP ordination diagram illustrates the separation between the variables (treatment plots) as to the presence and/or absence of unique species (Anderson & Willis, 2003; Anderson, 2004). By focussing on treatment variables that show clear separation across the axis (Axis 1 in particular) individual species causing the discrepancy in distance could be found by looking at the Canonical Axis in relation to Samples (species). Treatment plots at opposing ends of an axis are positioned in that way due to the occurrence of species that have only been sampled within that fire treatment.

CAP also provided a Permutation Test to gauge the significance (<0.05) of variance between unique species within the treatment plots this was measured by a trace statistic. In which the program obtained a P-value by recalculating the squared canonical values through random re-orderings of observations, while keeping X (total number of observations x number of samples) and the initial projection matrix constant (Tabachnick & Fidell, 1996; Anderson, 2004).

Canonical Correspondence Analysis (CCA)

Canonical Correspondence Analysis (CCA) was employed using CANOCO to gauge correlations between environmental factors (soil pH, soil conductivity, soil salinity, canopy cover, litter depth and shrub cover) and individual species at a microhabitat scale. Utilizing the species x site matrices from the Perup and McCorkhill transect section data, a constrained ordination was produced. By ordering the transect sections along gradients of environmental factors with species placed as to their correlation with the gradients on a 2 dimensional bi-plot (Leps & Smilauer, 2003). The length of gradient on the CCA bi-plot is relative to the strength of the correlation between that factor and the species that follow it. Species that reside beside long gradient lines have a higher correlation with that factor than species residing near short gradient lines. As a result of this CCA ordination floristic composition (through individual species) can be explained by environmental data, for example species that only occur in conjunction with high percentage canopy cover or low soil pH (Tabachnick & Fidell, 1996; Leps & Smilauer, 2003). Environmental data used in the CCA analysis was confined to those measured at the transect section scale (Figure 2.5). These were litter depth, canopy cover, shrub cover, soil pH, soil conductivity and soil salinity).

CHAPTER 3 RESULTS

3.1 Perup and McCorkhill Treatment Sites Combined

The MDS ordination (Figure 3.1) illustrates a distinct difference in geophyte species composition between the Perup and McCorkhill sites with a low stress value (<0.2) indicating strong intra-site relationships well represented in the 2-dimensional ordination.

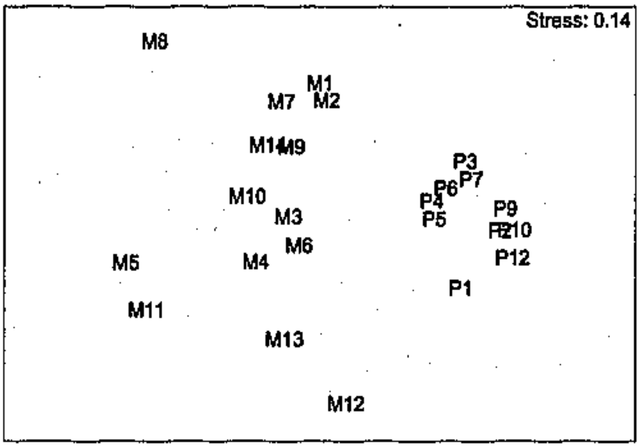

Figure 3.1 MDS ordination based of the comparison between species abundance, diversity and composition of the plots within the Perup Treatment Site and the McCorkhill Treatment Site. Ordination was based on 99 permutations and indicates a stress value of 0.14. Associated with plot numbers, M represents the McCorkhill treatment site and P represents the Perup treatment site.

The ordination shows that plots within the Perup Treatment Site cluster together and are distinct from the McCorkhill site plots. The Perup site grouping was more apparent between plots than that of the McCorkhill plots. A similar pattern is observed irrespective of data transformations used to down weigh the most abundant species. An ANOSIM test of the variance in species abundance, diversity and composition between the two sites was significant given the values of the Global R value (0.659) and Significance Level of Sample Statistic (0.1%). Because of the differences in geophytes between the two sites the fire treatment effects were analysed separately for each.

3.2 Perup Treatment Site: Time Since Last Fire

3.2.1 Species Abundance and Diversity

Table 3.1 Results of a one-way ANOVA testing differences in mean density, species richness and species evenness between sites with different age structures (2, 9, 10, 12 and 22 years old). (*) Indicates values that show significant difference (P<0.05) between treatments.

	df	Ms	F	P - value	F critical
Species Density	4	0.0139	0.1942	0.9312	5.1923
Species Richness	4	9.85	49.25*	0.0003*	5.1932
Species Evenness	4	0.0063	0.5899	0.68591	5.1932

Mean density shows a gradual decline as the time since last fire increases (Figure 3.2). The maximum mean density was recorded at 2 years post-fire, with a gradual decline to 10 years post-fire with the minimum mean density being recorded 22 years post-fire. Although there is a gradual decline in mean density, a one-way ANOVA shows no significant (P<0.05) difference between mean species density and time since last fire within the Perup Site (Table 3.1).

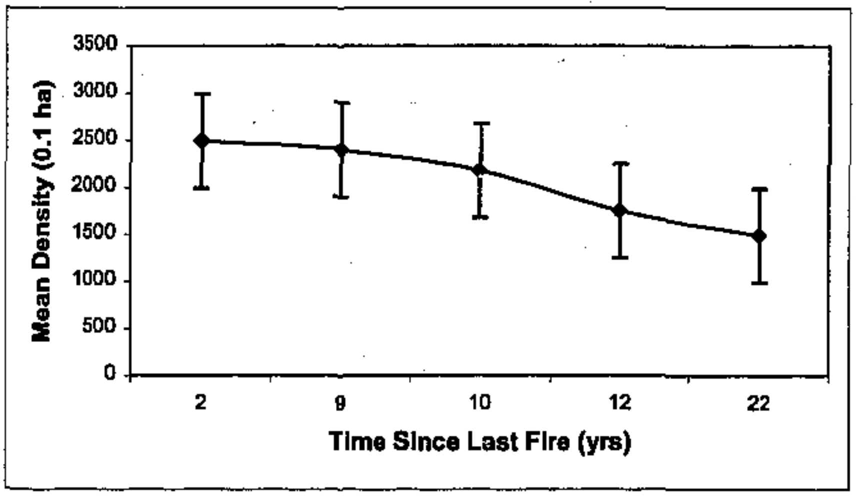

Figure 3.2 Trend between mean density per 0.1 ha (+/- Std- error) and time since last fire within the Perup Treatment Site.

Species richness displays a definite peak 10 to 12 years post-fire (Figure 3.3), with richness at its lowest value within 2 year old treatments. One-way ANOVA test results shows that species richness is significantly impacted by time since last fire within the Perup Site (Table 3.1). Although no formal post-hoc statistical tests were conducted due to limited replication it is apparent, based on comparison of error bars, that 10 year old treatments contained a greater richness than 2 and 9 year old treatments (Figure 3.3).

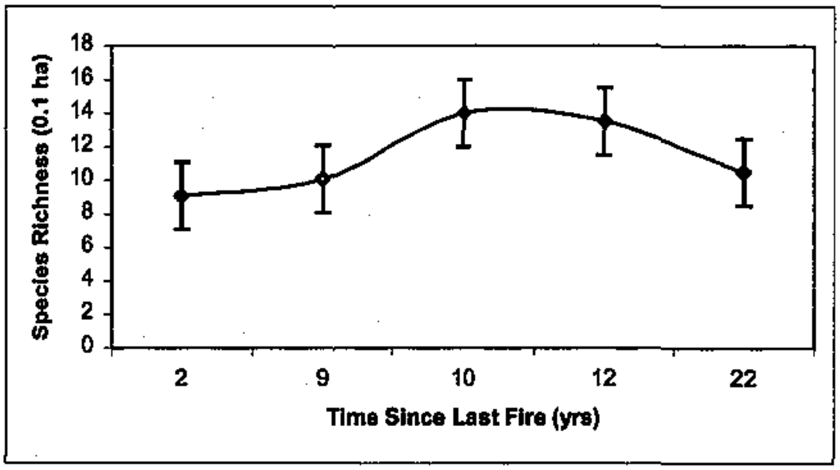

Figure 3.3 Trend between species richness per 0.1 ha (+/- Std- error) and time since last fire within the Perup Treatment Site.

Species Evenness (E=H/ln(R)) declines from 2 years post-fire to 10 years post-fire then increases from 10 to 22 years post-fire (Figure 3.4). As was the case with mean species abundance, although a trend was shown to exist as species evenness declined from 2 years

to 10 years post-fire and subsequently increased 10 to 22 years post-fire a One-way ANOVA tests reveal that time since last fire does not significantly impact species evenness within the Perup Site (Table 3.1).

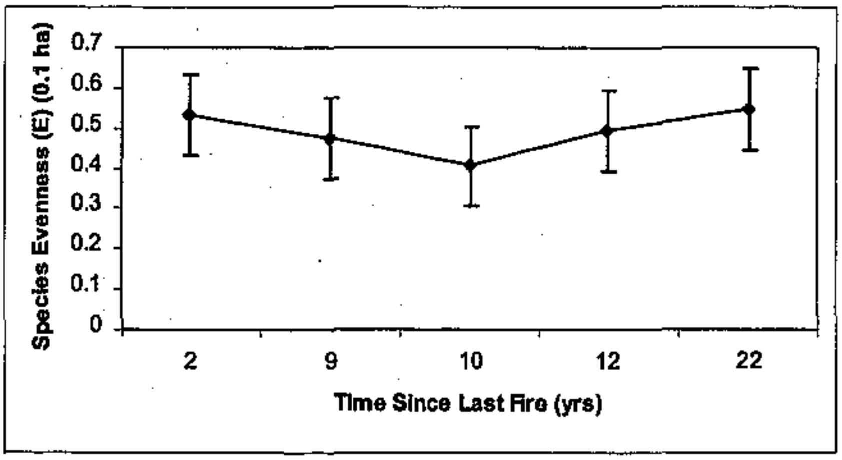

Figure 3.4 Trend between species evenness (E=H/In(R)) (+/- Std-error) and the age since last treatment within the Perup Treatment Site.

3.2.2 Species Composition

The relationship between plot and replicate plots in terms of species composition within the Perup Site is illustrated through an MDS ordination (Figure 3.5). The associated low stress value (<0.20) indicates a strong representation of plot relationships. The ordination shows no distinct groupings, either between different treatments or indeed between plots exhibiting the same treatment. An ANOSIM test of the variance in composition of species between plots and within plots is not significant (Global R = 0.387). Plots 1 and 12 (10 year old) and plots 2 and 10 (2year old) show the largest discrepancies between plots with the same treatment and plots 3 and 6 (22 year old) and plots 4 and 5 (9 year old) show the greatest similarity between plots of the same treatment i.e. similar species composition.

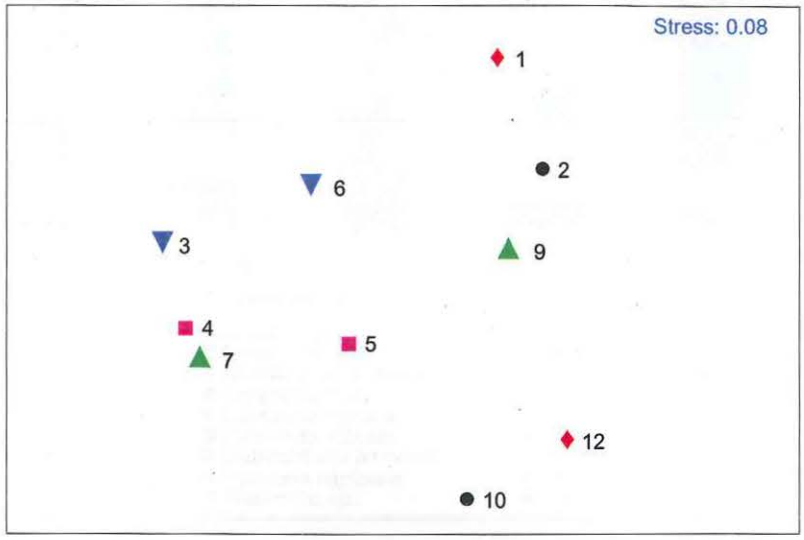

Figure 3.5 MDS ordination illustrating the pattern of different aged plots in relation to species abundance, diversity and composition. Ordination was based on 99 permutations with a stress value of 0.08. Key: ● 2 year old (plot 2 and 10), ■ 9 year old (plot 4 and 5), ♦ 10 year old (plot 1 and 12), ▲12 year old (plot 7 and 9), ▼ 22 year old (plot 3 and 6).

The composition of individual species within plots are shown in a percentage composition graph (Figure 3.2.2.1). It must be noted that the graph bars begin at 55% since *Drosera menziesii* proved to be extremely abundant and as such it comprises a minimum of 55% of the total number of individuals found within all the treatments (Figure 3.6). *Stylidium calcaratum* is abundant within 2 year old treatments, while in the 9 year old treatment *Stylidium sp., Drosera pallida, Stylidium calcaratum* are recorded in high abundances.

Within 10 year old treatments *Stylidium calcaratum, Drosera pallida, Caladenia flava* and *Pyrorchis nigricans* are found in high abundance and within 12 year old treatments *Stylidium sp.* is recorded in very high abundance, with *Drosera pallida, Stylidium calcaratum* and *Cyrtostylis huegelii* also found to be abundant. Of the individuals identified within 22 year old treatments *Stylidium sp, Drosera pallida, Caladenia flava, Cyrtostylis huegelii* and *Thelymitra crinita* are most abundant within the treatment.

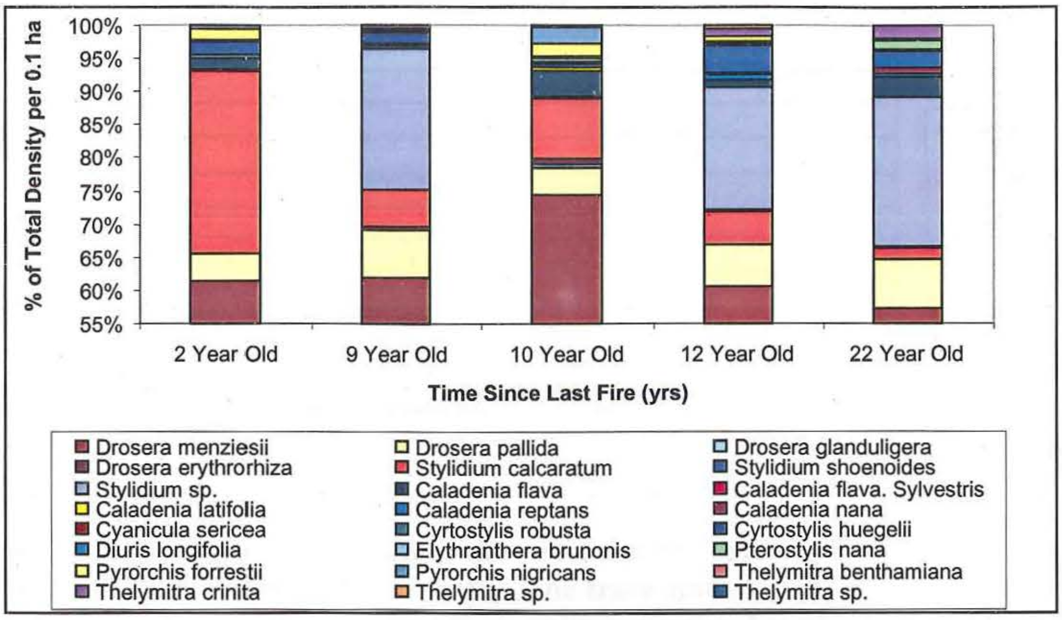

Figure 3.6 Percentage composition of species mean densities between 2, 9, 10, 12 and 22 year old treatments. Note: Bars begin at 55% to decrease the dominance of Drosera menziesii and thus increase the resolution of other less abundant species across all treatments.

3.2.3. Individual Species Analysis

CAP analysis shows that 10 year old treatments (plots 1 and 12) within the Perup Site contain a significantly higher proportion of 'unique species' or species in higher abundance compared to that of other treatments within the Perup Site. The 9 year old treatments (plots 9 and 4) also have species unique to that treatment (Figure 3.7). Although these two treatments were found to contain 'unique species', the trace statistic (<0.05) value indicated that the variance between treatments is not significant (Figure 3.7), given that only the 10 year old plots were grouped distinctively separate from the other treatments.

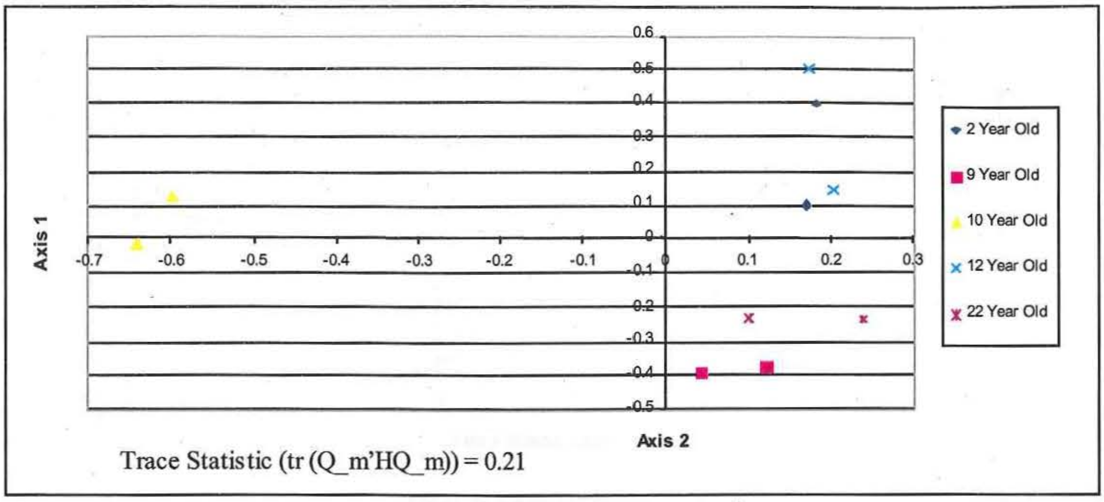

Figure 3.7 CAP analysis illustrating the presence of unique species within different aged plots (2, 9, 10, 12 and 22 years old). The trace statistic indicates the significance of variation between plots in terms of unique species (<0.05) with axis 1 and 2 highlighting the degree to which individual plots contain species unique to that treatment.

Through the analysis of CAP data the individual species accounting for the difference in species composition between treatments (Figure 3.7) were extracted and categorised into Early Response Species, Intermediate Response Species and Late Response Species. Figures 3.8- 3.9 illustrate the trend displayed by species within these three categories and an example species that correlates to the trend: *S. calcaratum* is placed within the 'Early Response Species' category since maximum abundance of the species is found 0-3 years post-fire (Figure 3.8); *C. flava* is placed within the 'Intermediate Response Species' category since maximum mean abundance is reached 9-12 years post-fire (Figure 3.9); and *T. crinita* is placed within the 'Late Response Species' category since the maximum mean abundance of the species is found to occur 18+ years post-fire (Figure 3.10).

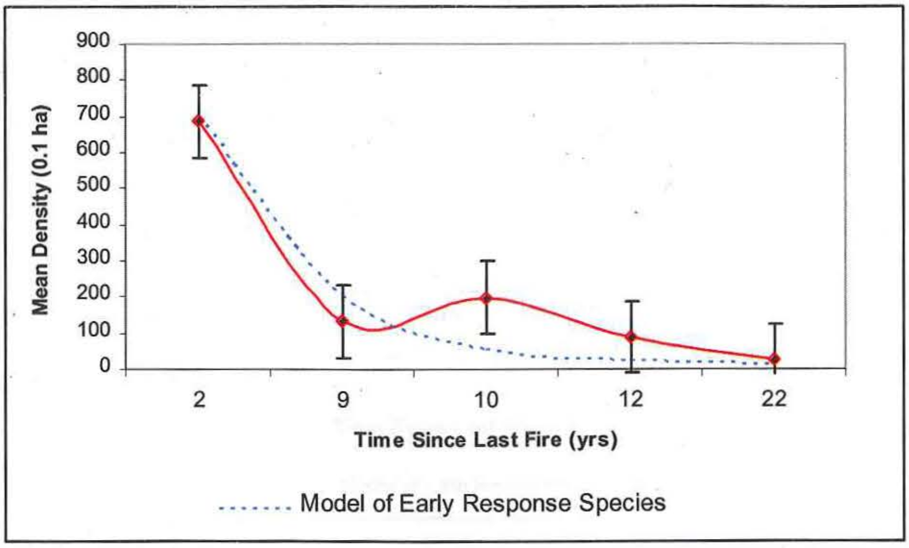

Figure 3.8 Trend in mean density (+/-Std-error) of *Stylidium calcaratum* 2, 9, 10, 12 and 22 years post-treatment and the trend displayed by Early Response Species that reach maximum abundance 0-3 years post-treatment.

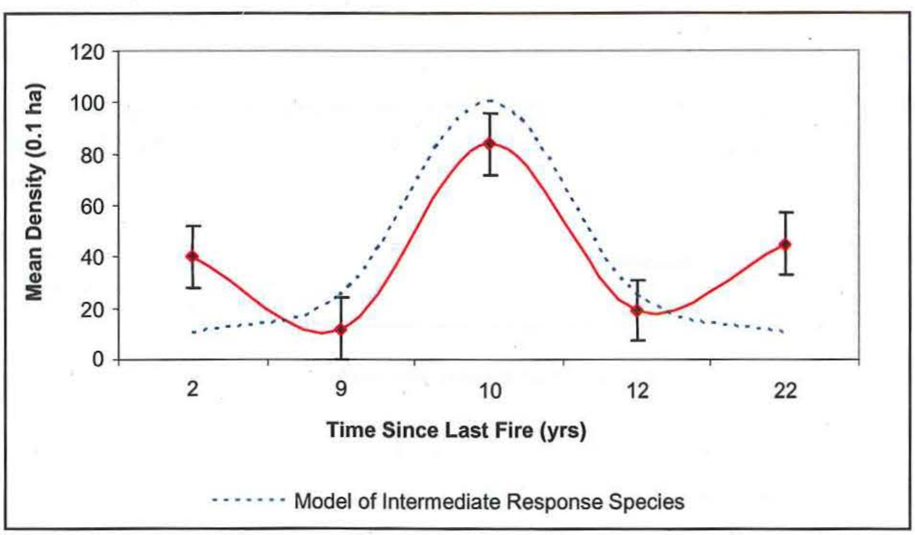

Figure 3.9 Trend in mean abundance (+/- Std-error) of *Caladenia flava* 2, 9, 10, 12 and 22 years post-treatment and the trend displayed by Intermediate Response Species that reach maximum mean abundance 9-12 years post-treatment.

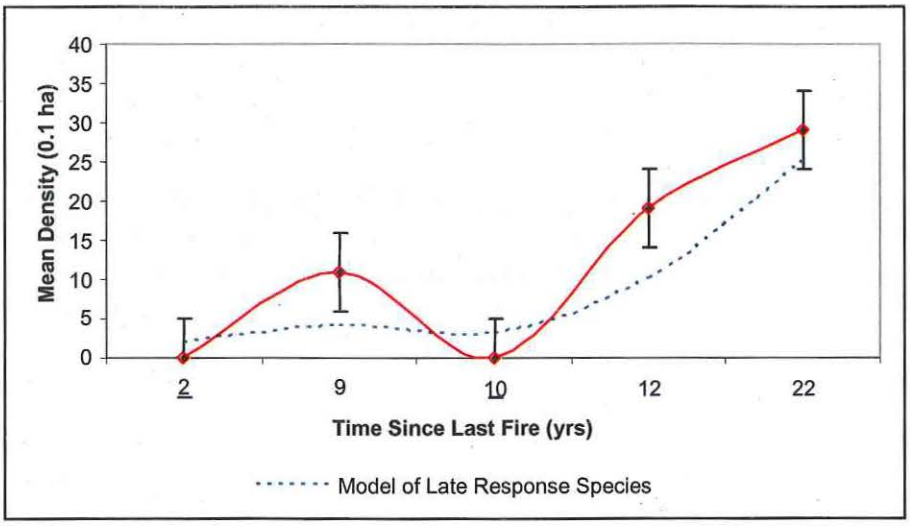

Figure 3.10 Trend in mean abundance (+/- Std-error) of *Thelymitra crinita* 2, 9, 10, 12 and 22 years post-treatment and the trend displayed by Late Response Species that reach maximum abundance 18+ years post-treatment.

The Age Response Category table (Table 3.2) shows species found via CAP analysis to correlate with the general trends illustrated within Figures 3.8- 3.10, i.e. that indicated a significant preference to a particular treatment in terms of time since last fire.

Table 3.2 Age Response Categories for the Perup Treatment Site: Early (0-3 years), Intermediate (9-12 years) and Late (18+ years) and the species that show a significant age response.

Age Response Category (Post-Fire)	Species
Early (0 - 3 years)	*Stylidium calcaratum, Cyrtostylis robusta, Pyrorchis forrestii*
Intermediate (9 - 12 years)	*Drosera glanduligera, Drosera erythrorhiza, Caladenia flava, Caladenia flava. Sylvestris, Caladenia reptans, Elythranthera brunonis, Pyrorchis nigricans, Thelymitra benthamiana*
Late (18+ years)	*Caladenia nana, Pterostylis nana, Thelymitra crinita*

3.3 McCorkhill Treatment Site: Time Since Last Fire

3.3.1 Species Abundance and Diversity

Table 3.3 Results of one-way ANOVA testing differences in species density, species richness and species evenness between sites with different age structures (1, 4, 5, 10, 21 and 28 years old). (*) Indicates values that show significant difference (P<0.05) between treatments.

	df	Ms	F	P - value	F critical
Species Density	5	0.1028	2.6481	0.1338	4.3874
Species Richness	5	7.1333	3.0571	0.1031	4.3874
Species Evenness	5	0.0495	8.9627*	0.0094*	4.3874

Mean species density remains fairly constant throughout 1, 4 and 5 year old treatments (Figure 3.11), with density falling in the 10 and 21 years old treatments. Mean density returns to similar values within 1, 4 and 5 year old treatments at 28 years post-fire. Despite the large decrease in mean density probably due to the variability between replicate plots of the same treatment (Figure 3.11), ANOVA tests indicated that time since last fire has no significant impact on mean species density within the McCorkhill Site (Table 3.3).

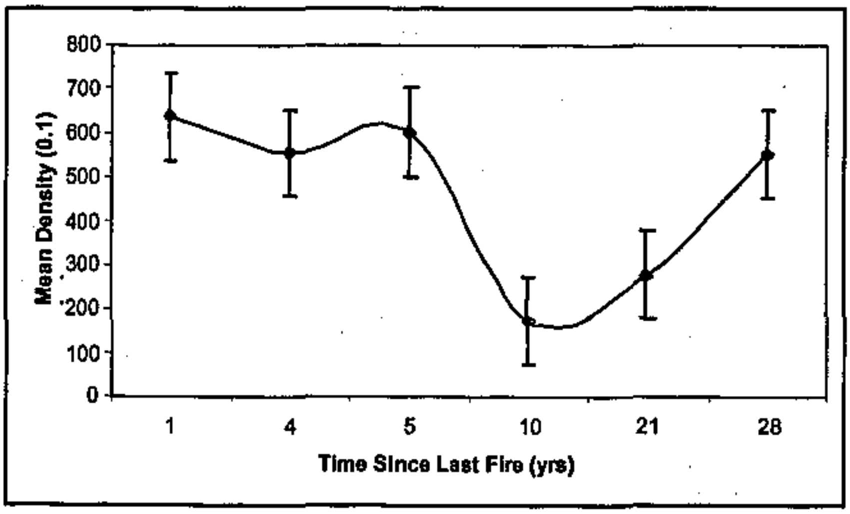

Figure 3.11 Trend between mean species density (+/- Std-error) and time since last fire within the McCorkhill Treatment Site.

The species richness of geophytes in relation to time since fire within the McCorkhill Treatment Site is lowest at 1 year post-fire (Figure 3.12), with a marked increase in species richness from 1 to 5 years post-fire with the maximum species richness found within plots containing 5 year old treatments. After a decline in species richness from 5 to 10 year old treatments, species richness was found to level off from 21 to 28 years post-fire. Although trend in species richness are observed ANOVA tests revealed that time since last fire has no significant impact ($P<0.05$) on species richness within the McCorkhill Site (Table 3.3).

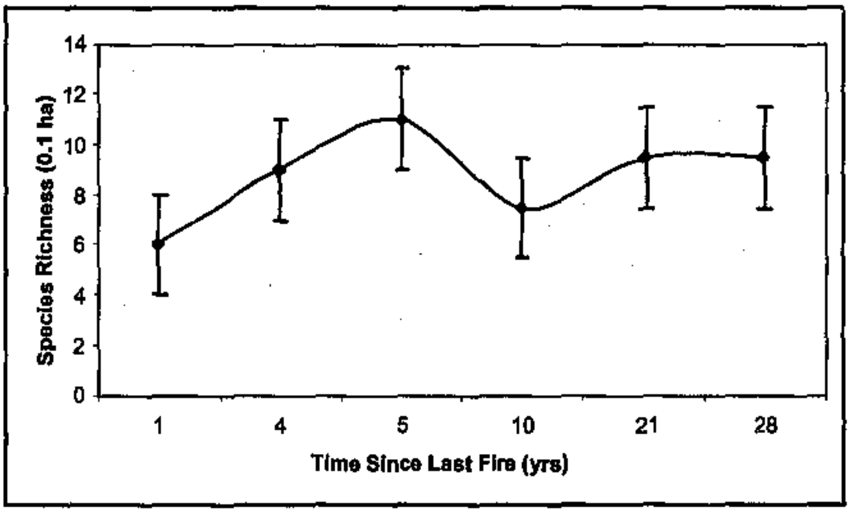

Figure 3.12 Trend between species richness (+/- Std-error) and time since last fire within the McCorkhill Treatment Site.

Species evenness is lowest within 1 and 5 year old treatments, with a plateau in high evenness values reached between 10, 21 and 28 years post-fire (Figure 3.13). ANOVA tests results show that time since last fire significantly impacts species evenness within the McCorkhill Treatment Site (Table 3.3).

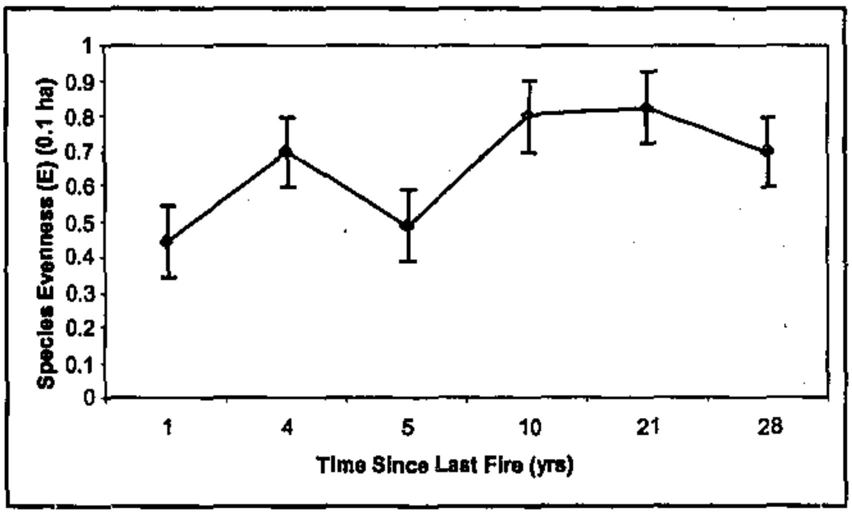

Figure 3.13 Trend between species evenness (E=H/In(R)) (+/- Std-error) and the age since last treatment within the McCorkhill Treatment Site.

3.3.2 Species Composition

The relationship between geophyte populations within different treatment ages was analysed with a MDS ordination (Figure 3.14). The low stress value (<0.20) indicates a strong representation of the relationships between plots. The 28 (plots 1 and 2) and 21 year old treatments (plots 7 and 14) show distinct similarities in species composition between replicate plots, with plots within 4 (plots 4 and 9) and 5 year old treatments (plots 6 and 10) also clustering relatively close together. Far removed from the central clustering of treatments are the 10 year (plot 8 and 11) and 1 year old treatments (plots 12 and 13) that did not display any correlation in geophyte composition either between replicate plots or between other treatments. An ANOSIM test shows no significant differences in species composition between the fire age treatments (Global R = 0.415).

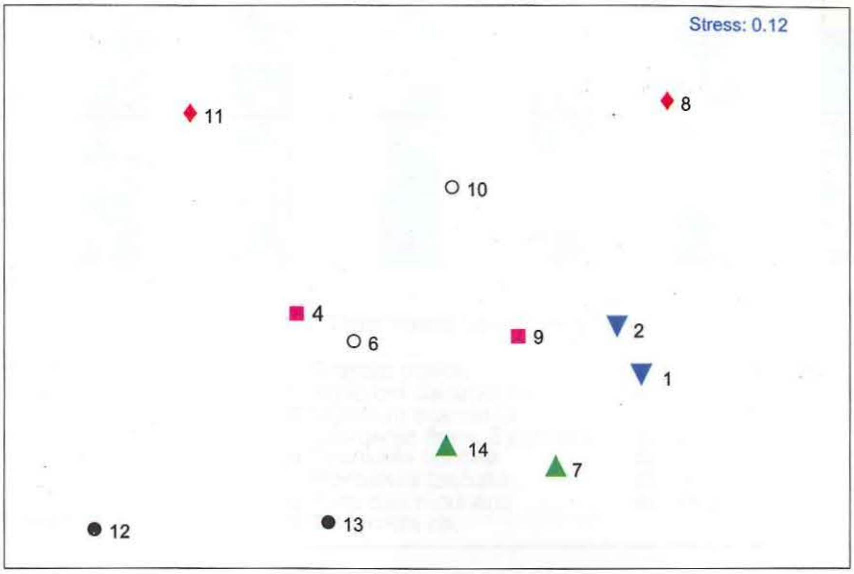

Figure 3.14 MDS ordination illustrating the pattern of different aged plots in relation to species abundance, diversity and composition. Ordination was based on 99 permutations with a stress value of 0.12. Key: ● 1 year old plots (12 and 13), ■ 4 year old plots (9 and 4), ○ 5 year old plots (10 and 6), ♦ 10 year old plots (11 and 8), ▲ 21 year old plots (7 and 14) , ▼ 28 year old plots (1 and 2).

The composition of geophyte species within different aged treatments is illustrated through a percentage composition graph (Figure 3.15). The 1 year old treatments are found to possess *D. menziesii*, *D. erythrorhiza*, *Stylidium sp.* and *Leptoceras menziesii* in high abundance relative to other species, with *Stylidium* sp., *D. menziesii* and *S. juneceum* occurring in large abundances within 4 year old treatments. Species composition within 5 year old treatments proved to very similar to that of 4 year old treatments, although they also contained *P. nigricans* and *Drosera sp.* in high numbers.

D. erythrorhiza and *P. nigricans* are found to occur in high abundance within 10 year old treatments, with *D. menziesii*, *Drosera sp.*, *Stylidium sp.* and *T. crinita* in high abundances in comparison to other species within 21 year old treatments. The 28 year old treatments have *D. menziesii*, *Drosera sp.*, *P. nigricans* and *T. crinita* in high numbers relative to other geophyte species (Figure 3.15).

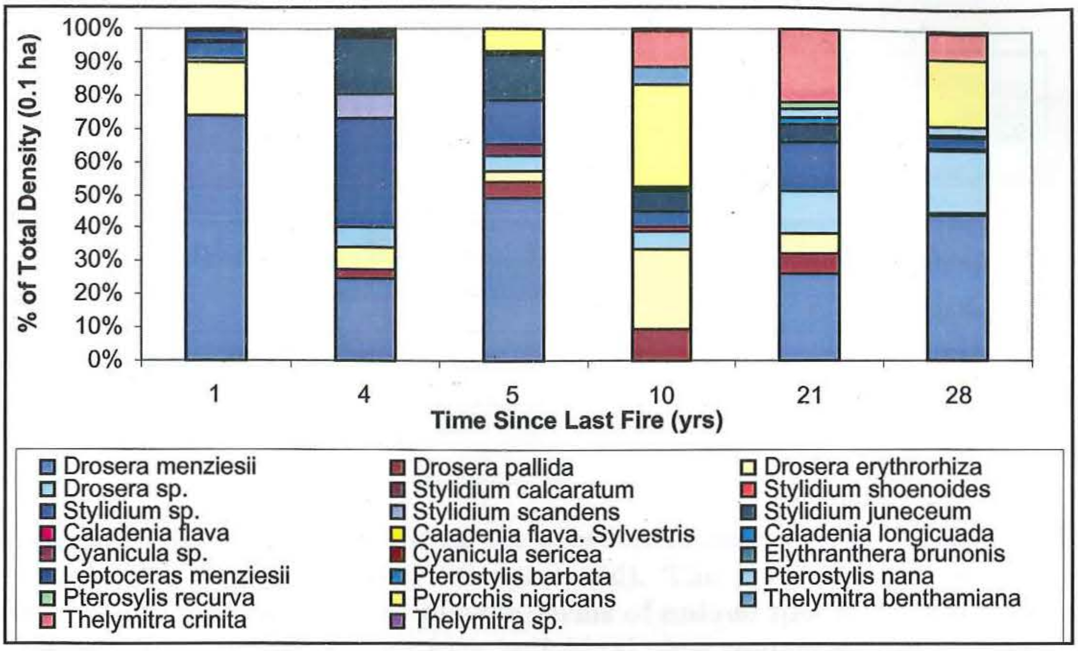

Figure 3.15 Percentage composition of species mean density between 1, 4, 5, 10, 21 and 28 year old treatments.

3.3.3 Individual Species Analysis

CAP analysis shows that replicate treatment plots within the McCorkhill Site group close together and as such contain very similar 'unique species' within treatments. The treatments themselves and their associated 'unique species' are separated along Axis 1 and 2, this is exemplified by the trace statistic value (<0.05) that indicates that the difference in 'unique species' between treatments is significant (Figure 3.16). Of note was the large separation between 10 and 28 year old treatments from the other treatments along Axis 1 and the grouping of 21 year old treatments at the low end scale of Axis 2.

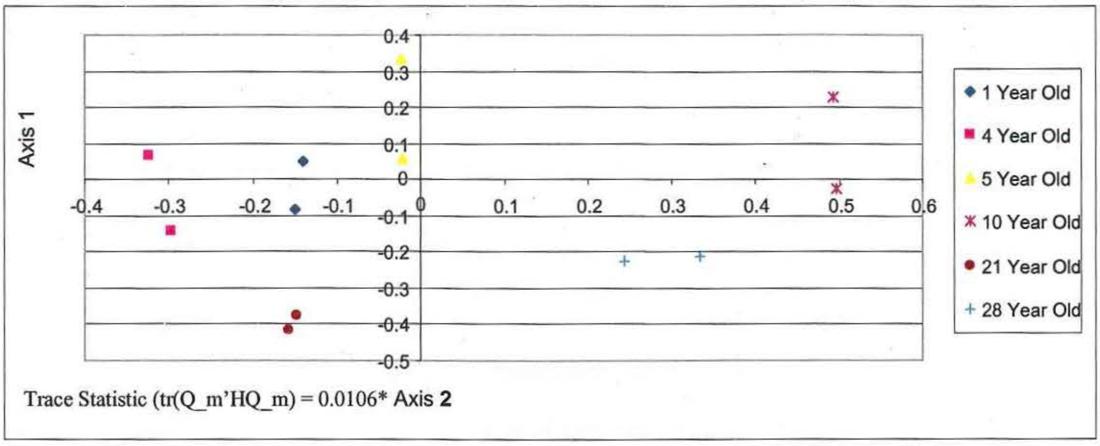

Figure 3.16 CAP analyses illustrating the presence of unique species within different aged plots (1, 4, 5, 10, 21 and 28 years old). The trace statistic indicates the significance of variation between plots in terms of unique species (<0.05) with axis 1 and 2 highlighting the degree to which individual plots contain species unique to that treatment.

Through the analysis of CAP data individual species accounting for the difference in species composition between treatments (Figure 3.16) were extracted and categorised into Early Response Species, Early Intermediate Species, Intermediate Response Species and Late Response Species. Figures 3.17 -3.20 illustrate the trend displayed by species within these three categories. *D. erythrorhiza* was placed within the 'Early Response Species' category as maximum abundance of the species is reached 0-3 years post-fire (Figure 3.17), *S. juneceum* was placed within the 'Early Intermediate Response Species' category since maximum mean abundance is reached 4-5 years post-fire (Figure 3.18), *D. pallida* was placed within the 'Intermediate Response Species' category since the maximum mean abundance of the species is found to occur 9-12 years post-fire (Figure 3.19) and *T. crinita* was placed within the 'Late Response Species' category with the maximum mean abundance of the species reached 18+ years post-fire (Figure 3.20).

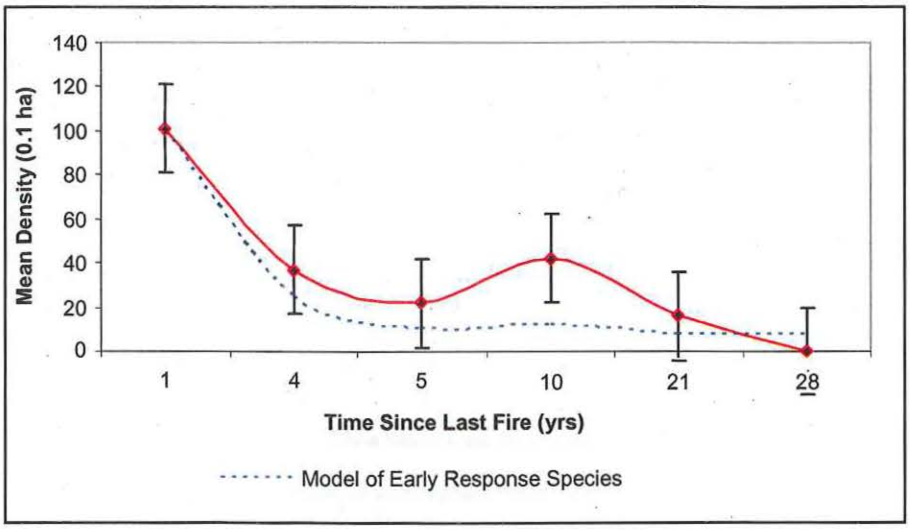

Figure 3.17 Trend in mean density (+/- Std-error) of *Drosera erythrorhiza* 1, 4, 5, 10, 21 and 28 years post-treatment and the trend of Early Response Species that reach maximum mean abundance 0-3 years post-treatment.

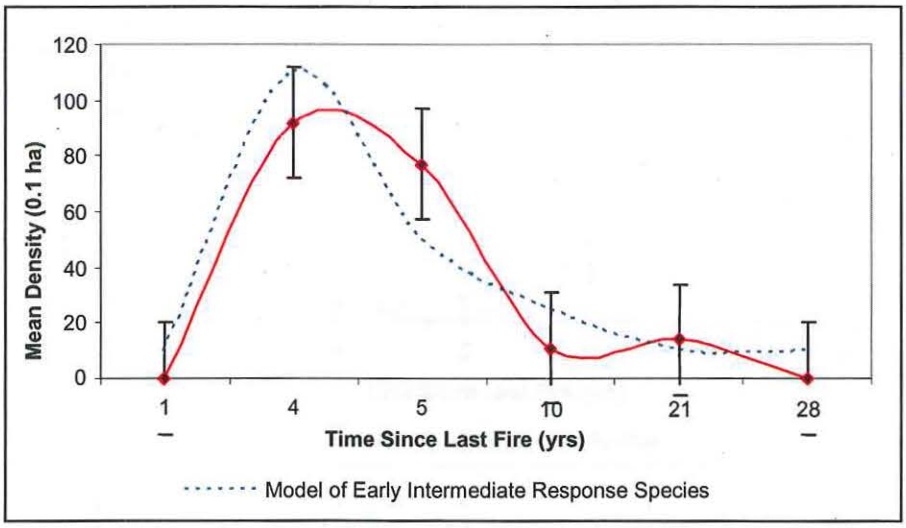

Figure 3.18 Trend in mean density (+/- Std-error) of *Stylidium juneceum* 1, 4, 5, 10, 21 and 28 years post-treatment and the trend of Early Intermediate Response Species that reach maximum mean abundance 4-5 years post-treatment.

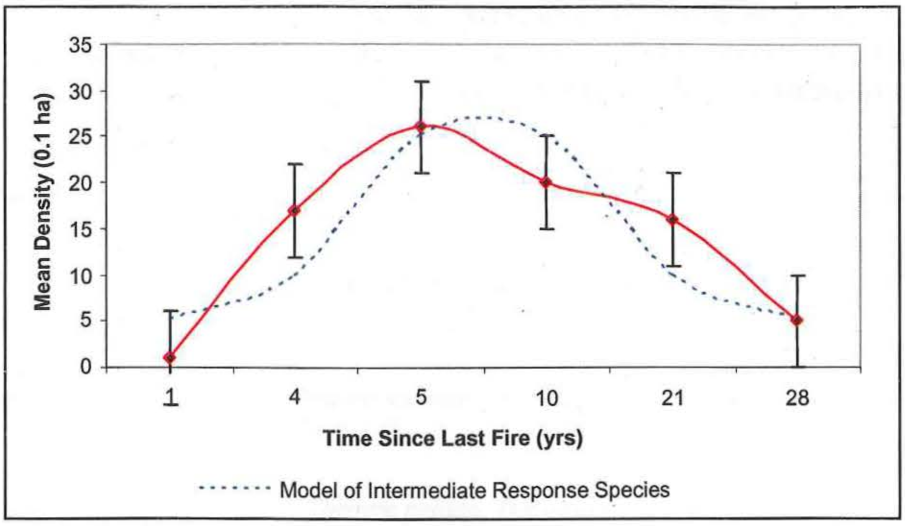

Figure 3.19 Trend in mean density (+/- Std-error) of *Drosera pallida* 1, 4, 5, 10, 21 and 28 years post-treatment and the trend of Intermediate Response Species that reach maximum abundance 9-12 years post-treatment.

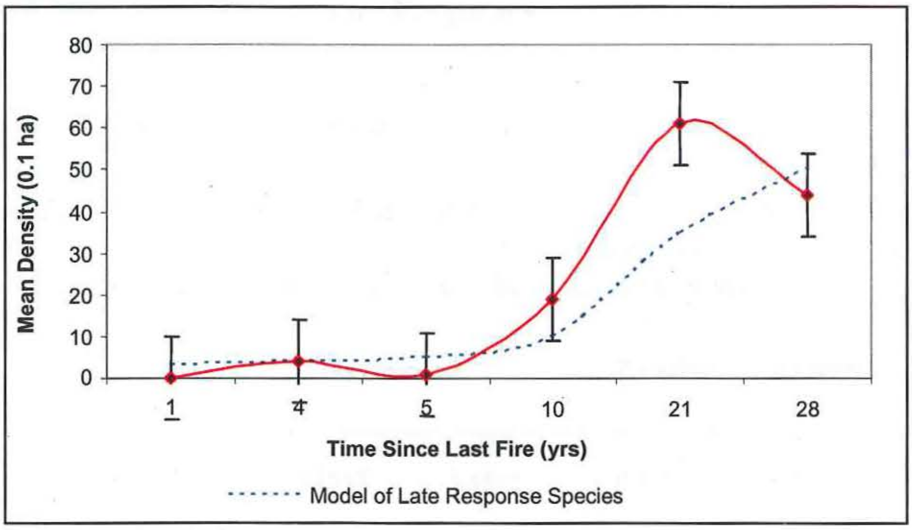

Figure 3.20 Trend in mean density (+/- Std-error) of *Thelymitra crinita* 1, 4, 5, 10, 21 and 28 years post-treatment and the trend displayed by Late Response Species that reach maximum mean abundance 18+ years post-treatment.

The Age Response Category table (Table 3.4) via CAP analysis identifies species that significantly correlated with the mean abundance trend lines (Figures 3.17-3.20) and as such are placed within the appropriate category: Early, Early Intermediate, Intermediate or Late Response Species.

Table 3.4 Age Response Categories for the McCorkhill Treatment Site: Early (0-3 years), Early- Intermediate (4-5 years), Intermediate (9-12 years) and Late (18+ years) and the species that show a significant age response to those treatment ages.

Age Response Category (Post-Fire)	Species
Early (0 - 3 years)	*Drosera erythrorhiza, Leptoceras menziesii, Cyanicula sp.*
Early Intermediate (4-5 years)	*Stylidium scandens, Stylidium juneceum, Caladenia longicuada*
Intermediate (9 - 12 years)	*Drosera pallida, Thelymitra benthamiana, Thelymitra sp.*
Late (18+ years)	*Pterostylis barbata, Pterostylis nana, Pterostylis recurva, Pyrorchis nigricans, Thelymitra crinita*

3.4 McCorkhill Treatment Site: Fire Frequency

3.4.1 Species Abundance and Diversity

Table 3.5 Results of a one-way ANOVA testing difference in species density, species richness and species evenness between different treatment frequencies. (*) Indicates values that show significant difference (P<0.05) between treatments.

	df	Ms	F	P - value	F critical
Species Density	3	0.1517	3.4819	0.1297	6.5914
Species Richness	3	6.4583	3.4444	0.1316	6.5915
Species Evenness	3	0.0335	15.7699*	0.0111*	6.5915

Mean species density decreases between 1 and 2 treatments, with treatments at a frequency of 2 exhibiting the lowest mean abundance within the McCorkhill Site. Mean density increases from 2 to reach a maximum value within treatments with a frequency of 4, mean density is subsequently maintained from 4 to 5 treatments (Figure 3.21). Although a notable decline in mean density took place at a frequency of 2 treatments followed by an

increase in mean density to 4 treatments, ANOVA tests shows that treatment frequency does not significantly (<0.05) impact species density within the McCorkhill Site (Table 3.5).

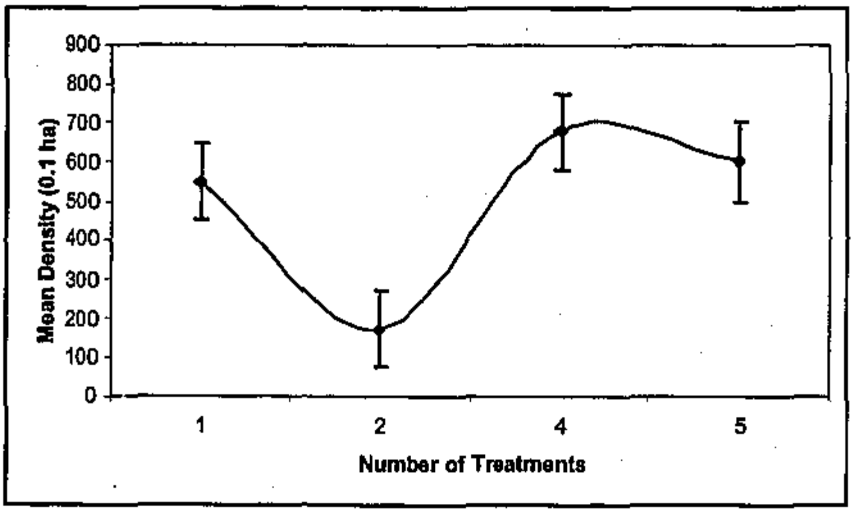

Figure 3.21 Trend between mean abundance (+/- Std-error) and frequency: 1, 2, 4 and 5 treatments since site establishment and known fire records in 1976.

The range in species richness between treatments remains relatively constant from 1 to 5 treatments since 1976, only a small decline within treatments that have had 2 treatments was detected (Figure 3.22). ANOVA test results confirm this and reveal that treatment frequency does not significantly (<0.05) impact species richness within the McCorkhill Site (Table 3.5).

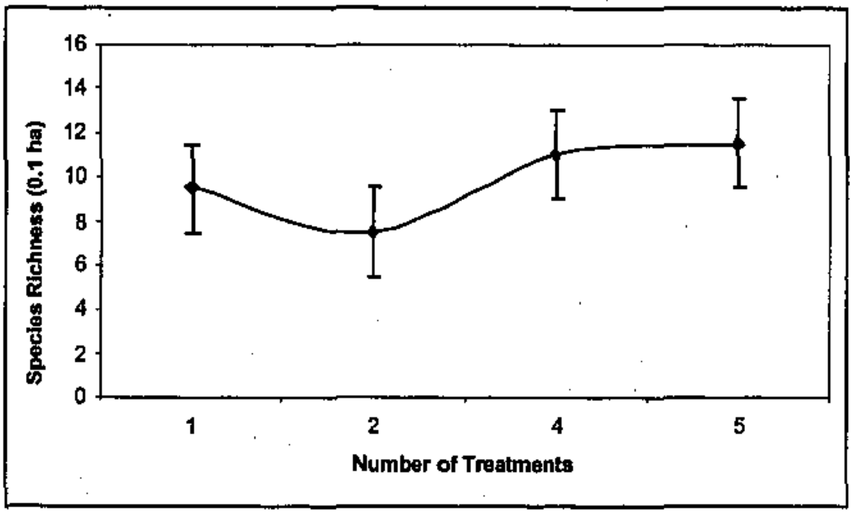

Figure 3.22 Trend between species richness (+/- Std-error) and frequency: 1, 2, 4 and 5 treatments since site establishment and known fire records in 1976.

Species evenness declines from a maximum value within 2 treatment frequencies to a minimum value within 4 treatment frequencies, with a gradual increase from 4 to 5 treatments (Figure 3.23). Given the high species evenness within low frequency treatment compared to high frequency treatments. ANOVA test results revealed that treatment frequency has significantly impacted species evenness within the McCorkhill Site (Table 3.5).

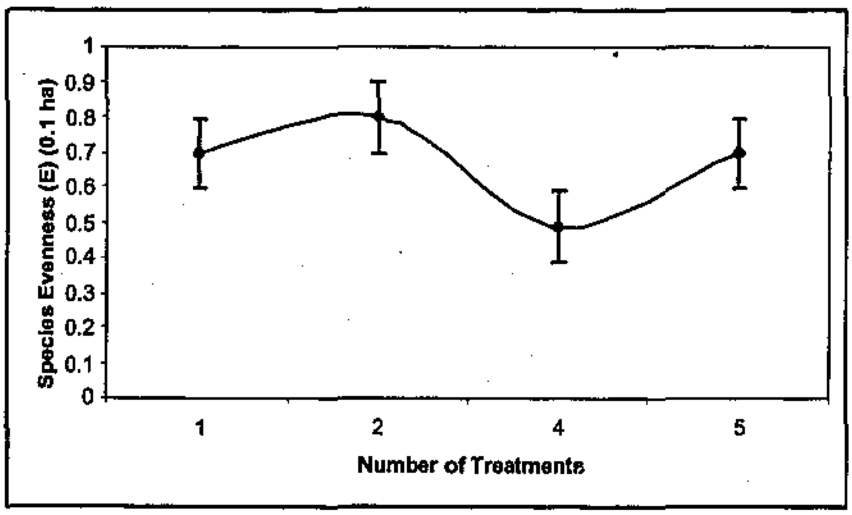

Figure 3.23 Trend between species evenness (E=H/In(R)) (+/-Std-error) and frequency: 1, 2, 4 and 5 treatments since site establishment and known fire records in 1976.

3.4.2 Species Composition

The species composition within treatments of varying frequencies is illustrated through a percentage composition graph (Figure 3.24). *D. menziesii*, *P. nigricans*, *Drosera sp.* and *T. crinita* are present in high abundances in plots treated once since 1976. Similar species were found in treatments that had burnt twice, with the exception of *D. menziesii* which was almost absent. Treatments burnt 4 and 5 times since 1976 contain similar species assemblages with *D. menziesii*, *D. erythrorhiza* and *Stylidium sp.* found in high abundance. Although within treatments burnt 4 times *S. scandens* is very abundant but is not found within treatments burnt 5 times.

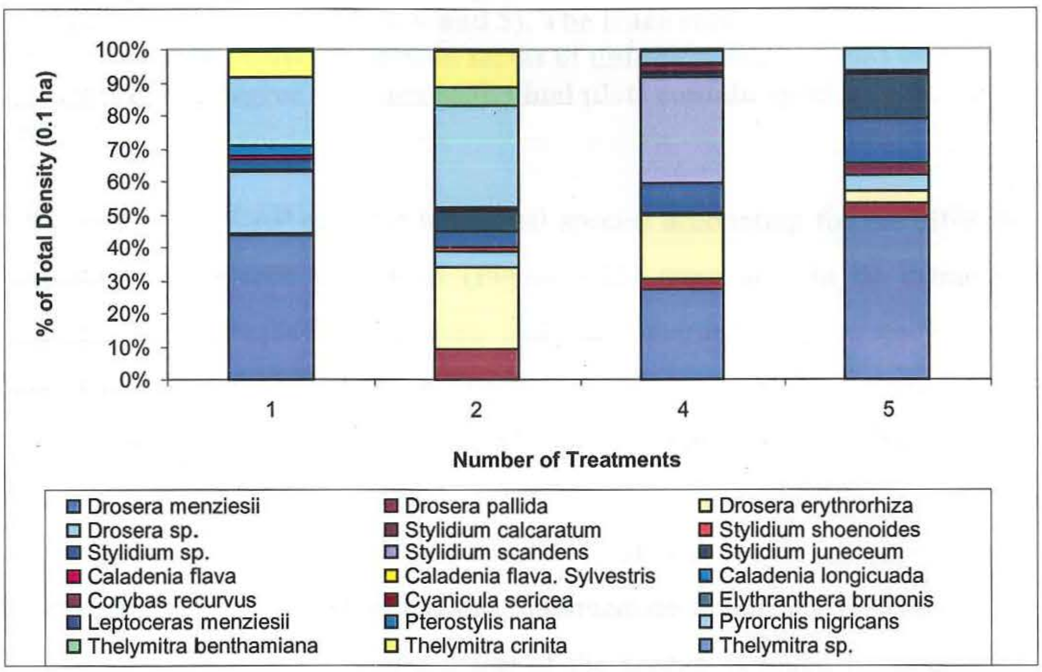

Figure 3.24 Percentage composition of species mean density between different frequencies: 1, 2, 4 and 5 treatments since site establishment and known fire records in 1976.

3.4.3 Individual Species Analysis

CAP analysis shows that treatments with a frequency of 4 group together and are found to occur separate from the other treatments indicating the presence of 'unique species' within this treatment (Figure 3.25). Also of note is the difference in species composition between other treatments within the site with plots and replicate plots containing very different species assemblages. This is further indicated by significance (<0.05) of the trace statistic value.

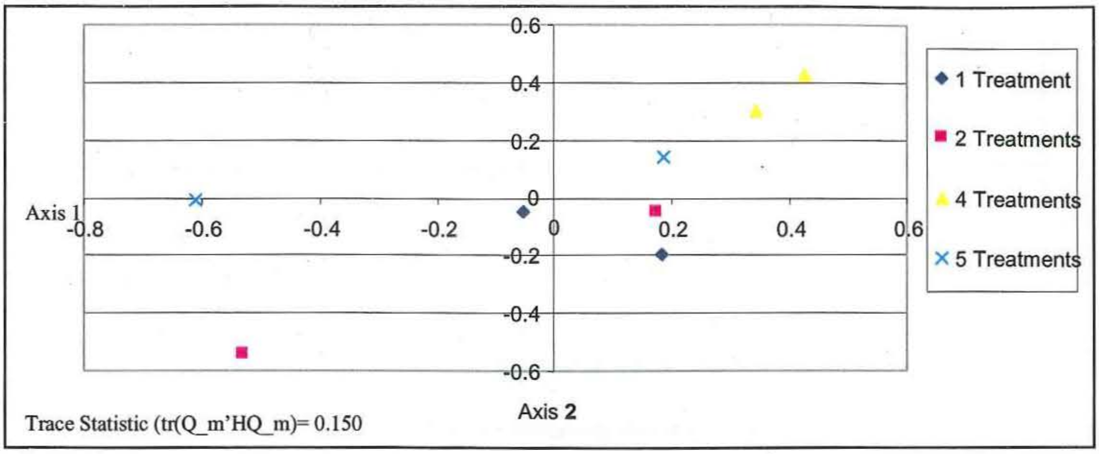

Figure 3.25 CAP analyses illustrating the presence of unique species within plots with different treatment frequencies (1, 2, 4 and 5). The trace statistic indicates the significance of variation between plots in terms of unique species (<0.05) with axis 1 and 2 highlighting the degree to which individual plots contain species unique to that treatment.

Through the analysis of CAP data the individual species accounting for the difference in species composition between treatments (Figure 3.25) were able to be extracted and categorised into Low Frequency Response Species, Intermediate Frequency Response Species and High Frequency Response Species. Figures 3.26-3.28 illustrate the trends displayed by species within these three categories and an example species that correlates to the trend; *Thelymitra crinita* was placed within the 'Low frequency Response Species' category as maximum abundance of the species is found after 1-2 treatment burns (Figure 3.26), *D. erythrorhiza* was placed within the 'Intermediate Frequency Response Species' category since the maximum mean abundance of the species is found to occur after 3-4 burns (Figure 3.27) unlike *S. juneceum* which was placed within the 'High Frequency Response Species' category with the maximum mean abundance of the species being reached after 5+ burns (Figure 3.28).

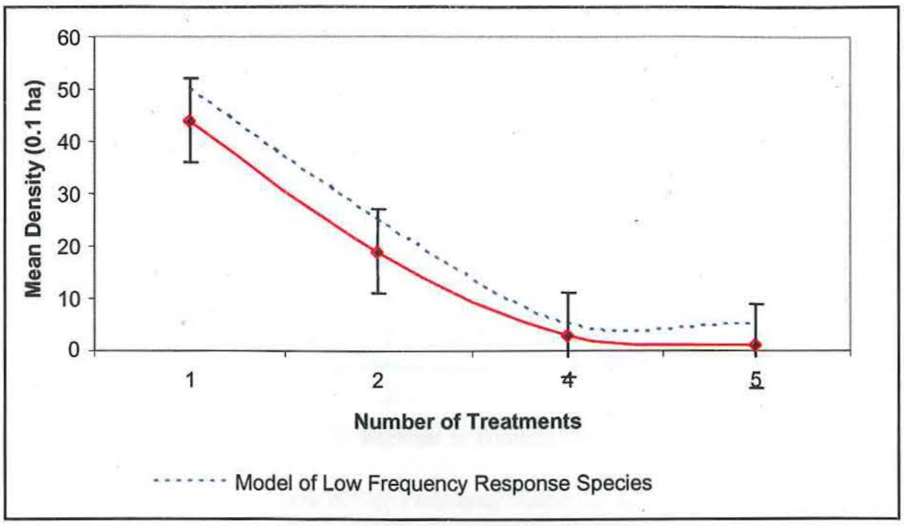

Figure 3.26 Trend in mean density (+/- Std-error) of *Thelymitra crinita* after 1, 2, 4 and 5 treatments since site establishment in 1976 and the trend displayed by Low Frequency Response Species that reach a maximum abundance at 1-2 treatments.

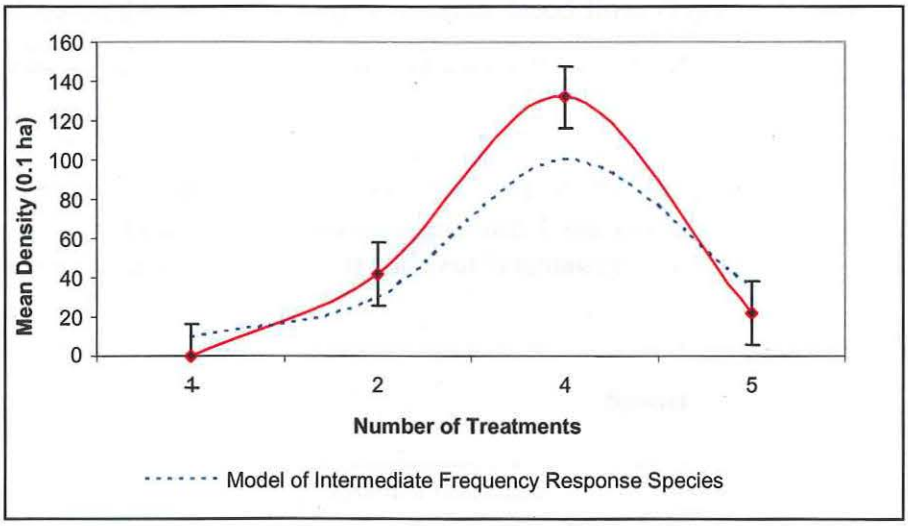

Figure 3.27 Trend in mean density (+/- Std-error) of *Drosera erythrorhiza* after 1, 2, 4 and 5 treatments since site establishment in 1976 and the trend displayed by Intermediate Frequency Response Species that reach maximum at 3-4 treatments.

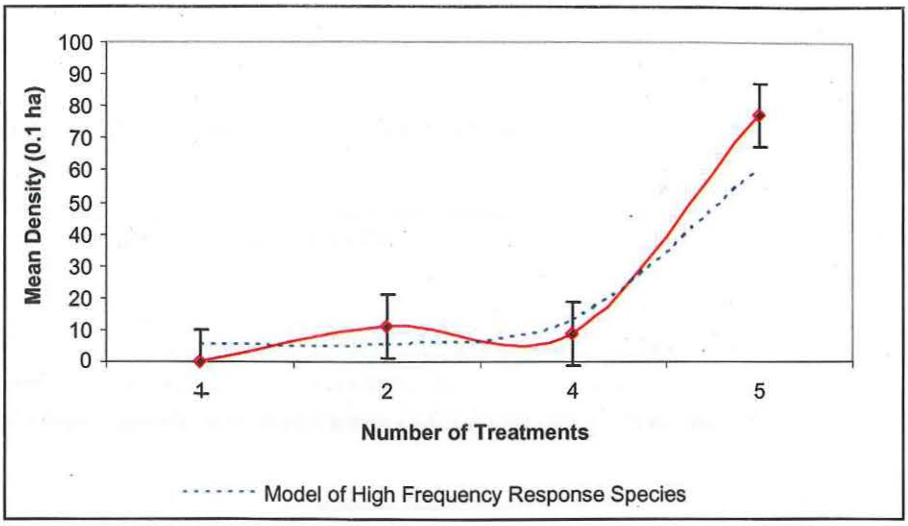

Figure 3.28 Trend in mean density (+/- Std-error) of *Stylidium juneceum* after 1, 2, 4 and 5 treatments since site establishment in 1976 and the trend displayed by High Frequency Response Species that reach maximum mean abundance after 5+ treatments.

The Frequency Response Category table (Table 3.6) via CAP analysis shows species that significantly correlate with the mean abundance trend lines (Figures 3.26-3.28) and these were then placed within the appropriate category (Low, Intermediate or High Frequency Response Species).

Table 3.6 Frequency Response Categories for the McCorkhill Treatment Site: Low (1-2 treatments), Intermediate (3-4 categories) and High (5+ treatments) and the species that show a significant response to treatment frequency.

Frequency Category	Species
Low (1-2 treatments)	*Stylidium shoenoides, Pterostylis nana, Pyrorchis nigricans, Thelymitra crinita, Thelymitra sp.*
Intermediate (3-4 treatments)	*Drosera erythrorhiza, Stylidium scandens, Corybas recurvus, Leptoceras menziesii*
High (5+ treatments)	*Stylidium calcaratum, Stylidium juneceum, Caladenia longicuada, Cyanicula sericea*

3.5 McCorkhill Treatment Site: Fire Season

3.5.1 Species Abundance and Diversity

Table 3.7 Results of a one-way ANOVA testing difference in species density, species richness and species evenness between different seasons of treatment. (*) Indicates values that show significant difference (<0.05) between treatments.

	df	Ms	F	P-value	F-critical
Species Density	3	0.15172	2.9927	0.1585	6.5914
Species Richness	3	6.8333	3.9047	0.1106	6.5914
Species Evenness	3	0.0329	2.91594	0.16399	6.5914

Mean species abundance remains relatively constant through autumn (April), summer (February) and summer (December) treatments, before a marked decrease in mean abundance within the spring (November) treatment (Figure 3.29). Despite the marked decrease in mean abundance within the spring (November) treatment, ANOVA tests show that season of treatment had no significant impact on species mean abundance within the McCorkhill Treatment Site (Table 3.7).

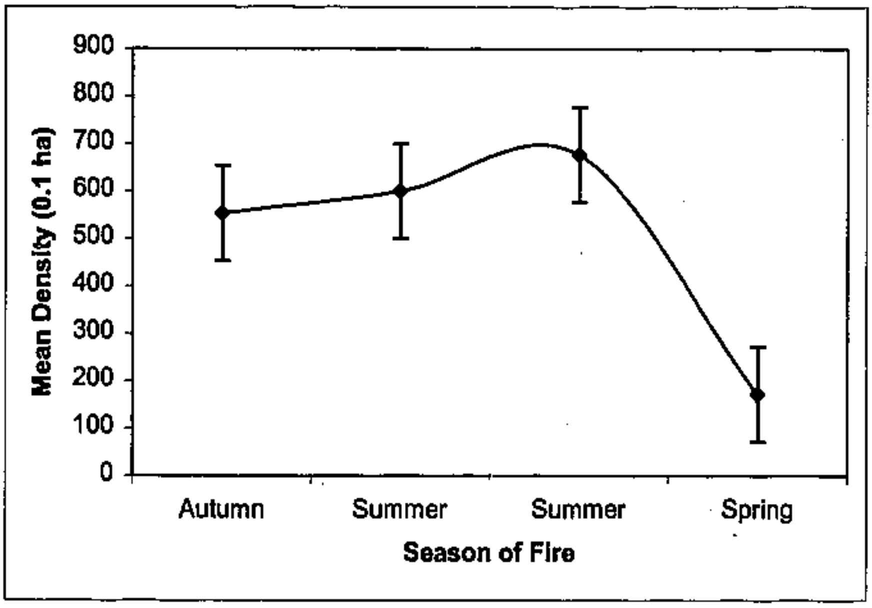

Figure 3.29 Trend between mean density (+/-Std-error) and the season of treatment. Autumn-April, Summer-February, Summer-December and Spring-November.

Species Richness follows a gradual curve with autumn (April) and spring (Nov) treatments exhibiting the lower average species richness in comparison to the summer (Feb) and summer (Dec) treatments (Figure 3.30). Given that there is only a small variation between the highest and lowest values, ANOVA tests show that season of treatment has no significant (<0.05) impact on species richness within the McCorkhill Site (Table 3.7).

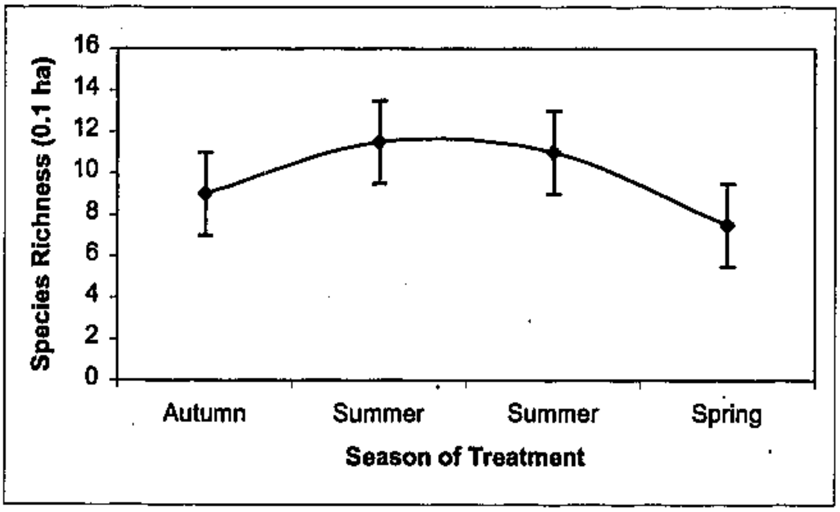

Figure 3.30 Trend between species richness (+/-Std-error) and season of treatment. Autumn-April, Summer-February, Summer-December and Spring-November.

Species evenness remains similar across autumn (April) and summer (February) treatments, while there is a fall in evenness values for the summer (Dec) treatments and subsequent gradual rise in species evenness during spring (Nov) treatments (Figure 3.31). Although trends are displayed descriptively, i.e. a fall in species evenness during Summer (December) treatments, ANOVA test results indicate that season of treatment has not significantly (<0.05) impacted species evenness within the McCorkhill Site (Table 3.7).

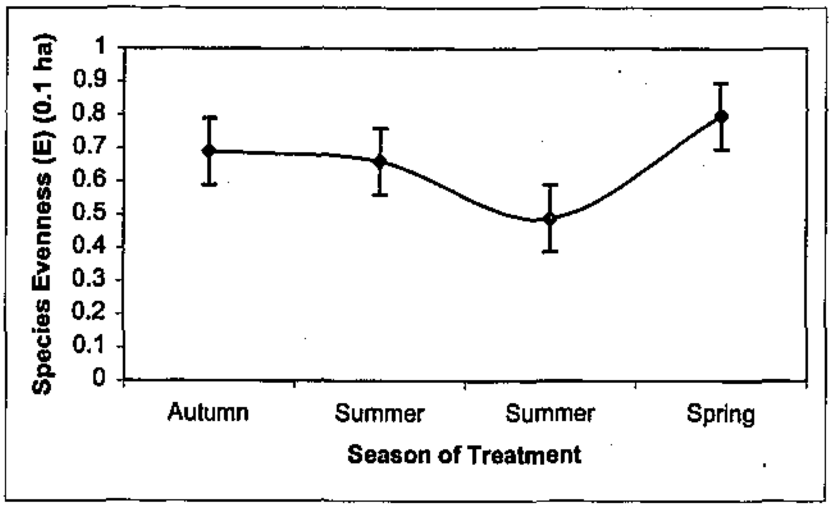

Figure 3.31 Trend between species evenness (E=H/ln(R)) (+/-Std-error) and season of treatment: Autumn-April, Summer-February, Summer-December and Spring-November.

3.5.2 Species Composition

Species composition within different seasonal treatments (Figure 3.32) shows *D. menziesii*, *Stylidium sp.*, *S. juneceum* all present in high abundances within Autumn/April and Summer/Feb Treatments. *P. nigricans* also appears in relatively high abundance within the Sumer/Feb treatment. *S. scandens* is found to be a dominant species within Summer/Dec treatments with *D. menziesii* and *D. erythrorhiza* also occurring in high abundance. Within the Spring/Nov treatment only *D. erythrorhiza* and *P. nigricans* are present in high numbers.

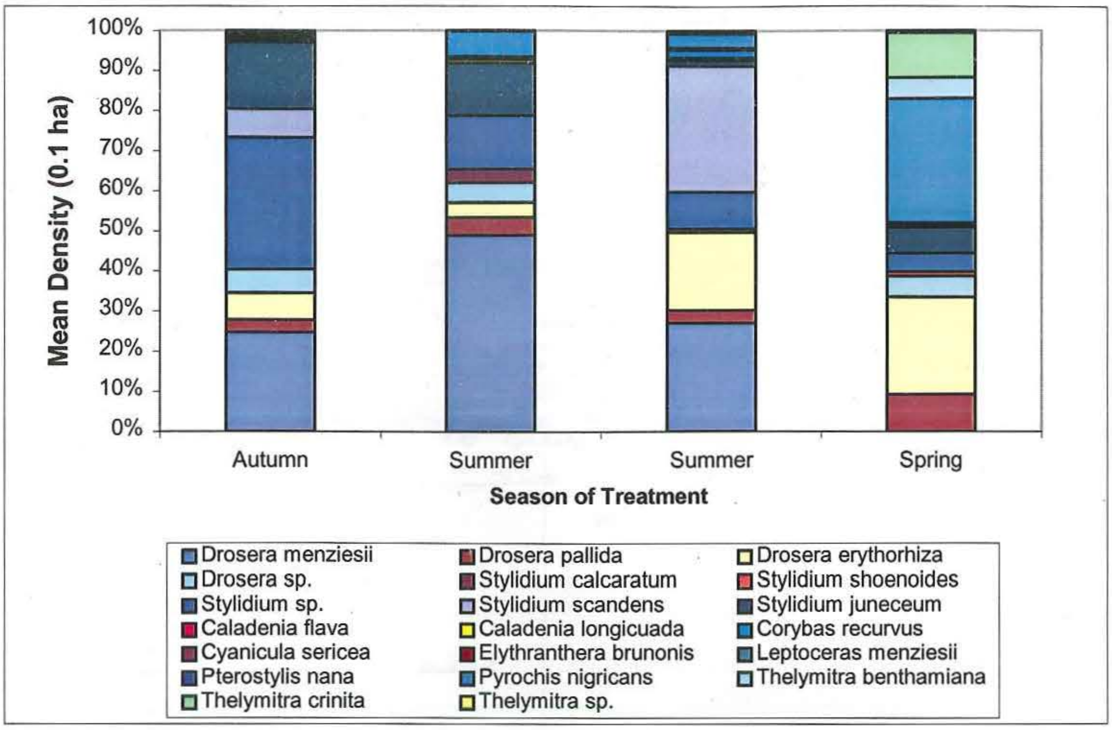

Figure 3.32 Percentage composition of species mean abundance between different seasonal treatments: Autumn-April, Summer-February, Summer-December and Spring-November.

3.6 Environmental Factors

3.6.1 Perup Treatment Site: Micro-scale Analysis

The CCA ordination shows individual species found to correspond with environmental factors at a fine-scale such as litter depth, shrub cover and canopy cover (Figure 3.33). With the exception of outliers, the majority of species cluster together around the central gradient points, indicating that although trends between individual species and environmental factors are present they are at a fine scale in terms of the total geophyte population. In other words the indirect impact of fire treatments on environmental factors may be impacting individual species but has negligible impacts on the total species assemblage within treatments. Species that highly correlated with environmental-gradients were *Pyrorchis nigricans* and *Thelymitra benthamiana* which were abundant in areas exhibiting high pH and low canopy cover and *Cyrtostylis robusta*, *Pyrorchis forrestii* and *Stylidium calcaratum* which were abundant in areas with low litter/shrub cover.

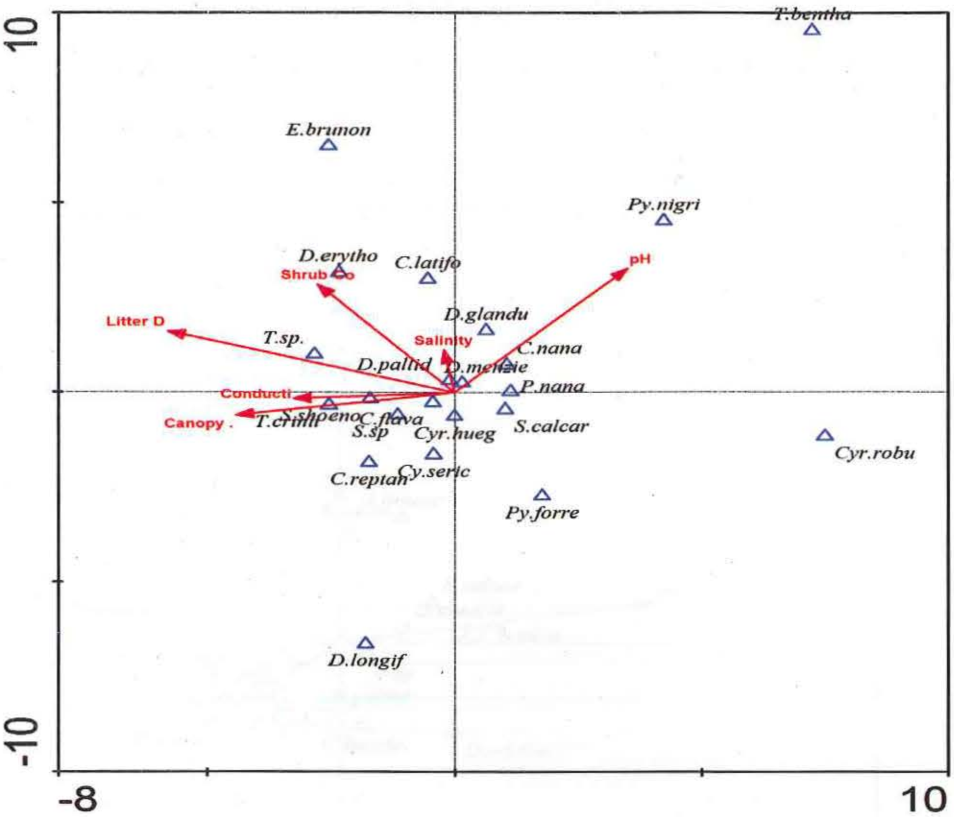

Figure 3.33 CCA species-environment bi-plot of Perup Site plots showing environmental gradients (mean pH, mean canopy cover %, mean salinity TDS, mean litter depth cm, mean shrub cover % and mean conductivity us/cm) and the correlations between individual species and those factors. Note: Length of factor gradient indicates degree of correlation i.e. the longer the gradient line the higher the correlation. Key: ***C. flava=Caladenia flava, C. latifolia=Caladenia latifolia, C. nana=Caladenia nana, C. reptans=Caladenia reptans, Cy. seric= Cyanicula sericea, Cyr. hueg=Cyrtostylis heugelii, Cyr. Robu= Cyrtostylis robusta, D.longif=Diuris longifolia, D. erytho=Drosera erythrorhiza, D. glandu=Drosera glanduligera, D. menzie=Drosera menziesii, D. pallid=Drosera pallida, E. brunon=Elythranthera brunonis, P. nana=Pterostylis nana, Py.forre= Pyrorchis forrestii, Py. nigri= Pyrorchis nigricans, S. calcar=Stylidium calcaratum, S. shoeno=Stylidium shoenoides, S.sp=Stylidium sp., T. bentha=Thelymitra benthamiana, T.crinit=Thelymitra crinita and T. sp=Thelymitra sp.***

3.6.2 McCorkhill Treatment Site: Micro-scale Analysis

The CCA ordination illustrates the strong correlations between shrub cover, litter depth, conductivity and pH with the abundance of certain species within treatments and the minor correlations between species and canopy cover and salinity (Figure 3.38). Many species are shown to have strong interrelationships with environmental factors within the McCorkhill Site. Axis 1 is a pH/shrub/litter gradient with species such as *Leptoceras menziesii*,

Cyanicula sp. and *Drosera erythrorhiza* abundant in high pH and low shrub/litter cover and species such as *Thelymitra crinita*, *Pyrorchis nigricans* and *Pterostylis nana* abundant in low pH and high shrub/litter cover.

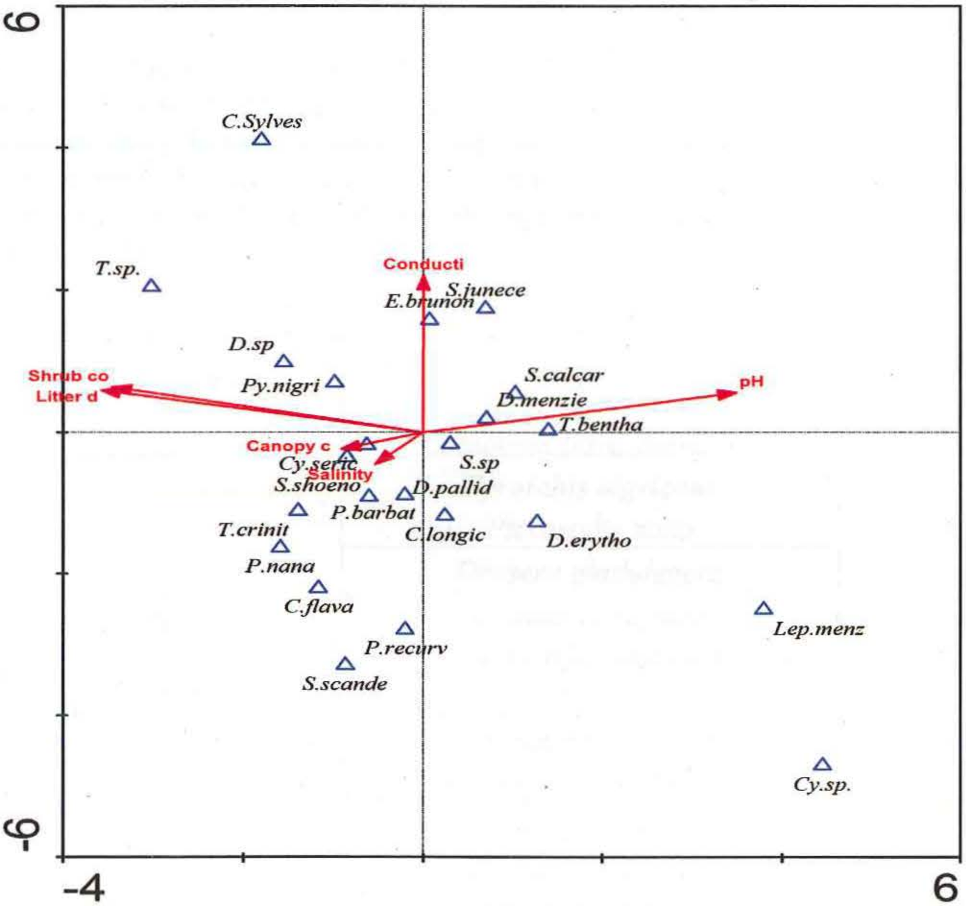

Figure 3.38 CCA species-environment bi-plot of McCorkhill site plots showing environmental gradients (mean pH, mean canopy cover %, mean salinity TDS, mean litter depth cm, mean shrub cover % and mean conductivity us/cm) and the correlations between individual species and those factors. Note: Length of factor gradient indicates degree of correlation i.e. the longer the gradient line the higher the correlation. Key: ***C. flava=Caladenia flava, C. Sylves= Caladenia flava subs. Sylvestris, C. longic=Caladenia longicuada, Cy. seric=Cyanicula sericea, Cy. sp=Cyanicula sp., D. erytho=Drosera erythrorhiza, D. menzie=Drosera menziesii, D. pallid=Drosera pallida, D. sp=Drosera sp., E. brunon=Elythranthera brunonis, Lep. menz=Leptoceras menziesii, P. barbat=Pterostylis barbata, P. nana=Pterostylis nana, P.recurv=Pterostylis recurva, Py. nigri=Pyrorchis nigricans, S. calcar=Stylidium calcaratum, S. junece=Stylidium juneceum, S. scande=Stylidium scandens, S shoeno=Stylidium shoenoides, S. sp=Stylidium sp., T. bentha=Thelymitra benthamiana, T. crinit=Thelymitra crinita and T. sp=Thelymitra sp.***

3.6.3 Microhabitat: Perup Site Log Data

Species that have a significant preference for either Log Habitats or Open Habitats are shown in Table 3.8.

Table 3.8 Habitat Preference for species within the Perup Treatment Site and the ratio to which they occur within that preferred habitat. Logs: Species that were predominantly found to occur either on or under logs or around the ash-bed of selected logs. Neutral: Species that did not show any preference for either Log Habitats or Open Habitats. Open: Species that were predominantly found to occur in Open Habitats.

Habitat Preference	Species	Ratio (Logs:Open)
Logs	*Caladenia flava. sylvestris*	7:1
	Pyrorchis nigricans	5:1
	Pterostylis nana	4:1
	Drosera gladulegera	3:1
	Caladenia reptans	
	Cyrtostylis heugeleii	
	Caladenia nana	
	Caladenia flava	2:1
	Caladenia macrostylis	
Neutral	*Caladenia latifolia*	1:1
	Cyrtostylis robusta	
	Diuris longifolia	
	Pyrorchis forrestii	
	Thelymitra benthamiana	
Open	*Drosera menziesii*	1:37
	Stylidium calcaratum	1:24
	Drosera erythrorhiza	1:7
	Elythranthera brunonis	1:5
	Thelymitra crinita	1:4
	Drosera pallida	
	Stylidium shoenoides	1:2
	Thelymitra sp.	

CHAPTER 4: DISCUSSION

4.1 Fire and Geophyte Communities

In this project conducted in the jarrah forest of south-west Western Australia, geophyte species assemblages were shown to be decidedly different between plots experiencing the same fire treatments at both the Perup and McCorkhill Sites. Ordinations (Figures 3.5 and 3.14) illustrated the degree to which these species assemblages differed and attest to the fact that geophytes are distributed in a very patchy nature across the treatments. The clonal nature and small dispersal distances of many species identified could account for these findings, since many species are able to rapidly and abundantly colonise favourable areas within the sites (Pate & Dixon, 1982). Areas with high colonisation may in fact be a result of the variable nature of fire and possibly even of the fire regimes themselves. Fire within dry sclerophyll forests is known to be a source of spatial and temporal variability even in terms of their impact upon groups such as shrubs and trees (Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003). Geophytes could be responding to fine-scale variations in such things as fire intensity, burn duration, fire effects on the availability of nutrients and fire impaction through changes in community structure (Wardell-Johnson, 2000; Morrison, 2002; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003; Watson & Wardell-Johnson, 2004).

The patchy distribution of species within sites also means that abundance is not a suitable measure when dealing with the impact of fire treatments on geophyte populations. This was observed within 10 and 21-year-old treatments within the McCorkhill Site, where a dramatic drop in mean abundance was recorded (Figure 3.11). Nevertheless, analysis of the species composition (Figures 3.15 and 3.25) and species evenness (Figures 3.13 and 3.23) revealed that the absence or dramatic reduction of *D. menziesii* was the sole cause of this variation, and species assemblages within the two treatments were not significantly different from the other treatments.

It is conceivable the reason mean abundance was found not to be a suitable measurement was that the abundance of the majority of geophyte species could significantly alter from year to year or after fire events, creating biases in measurements. This is because treatment

conditions and fine-scale variations in fire characteristics like intensity and burn duration could in some years and sites favour particular species and allow them to abundantly colonise an area, whereas they would remain dormant or in very low abundance in other years (Pate & Dixon, 1982; Dixon & Barrett, 2003). Despite this phenomenon, the actual presence/absence of the species (regardless of population size) should remain constant.

Species richness proved to be a much more informative measure than did density or evenness for assessing the impacts of different fire treatments on geophytes, since the large discrepancies in abundance due to patchiness between and more importantly within treatments was not an issue. This is in contrast to studies on higher plants where rapid changes in species richness take place post-fire, due to the presence of ephemeral species. Species richness rapidly increases after fire events (0-3 yrs post-fire) mainly due to ephemerals, but the actual assemblage of species across the treatment remains even (Burrows & Wardell-Johnson, 2003).

4.2 Geophyte Response to Time Post-fire, and to Frequency and Season of Fire

The response of geophytes to the three variables of time post-fire, frequency of fire and season of fire was mainly one of variation in species richness. The trends identified in the present study closely followed the trends identified by Burrows and Wardell-Johnson (2003). Time post-fire, frequency and season are intimately related components of both natural and human induced fire regimes (Christensen et al., 1981; Morrison, 2002; Burrows & Wardell-Johnson, 2003). It has been widely accepted that these three factors can alter the forest ecology to varying degrees—in terms of composition, structure and fuel accumulation. The last mentioned, being the amount of organic matter present, is strongly associated with the availability of moisture and nutrients (Christensen & Kimber, 1975; Gill et al., 1981; Morrison, 2002; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003; Watson & Wardell Johnson, 2004). Vegetation changes and survives according to these components (Gill, 1977; Christensen et al., 1981). Geophytes, as speculated earlier, may adapt and exhibit patterns of fine-scale responses to fire variations either by colonising or by remaining dormant under either favourable or unfavourable conditions. Analysis of specific fire regimes has given an indication of those post-fire times, fire frequencies and seasonal fire treatments that have increased the ability of geophytes to colonise suitable areas within the habitat.

Studies conducted by Grant & Koch (2003) in rehabilitated bauxite mines found that the overall orchid density within 20-year-old unmined sites was 1375 individuals per 0.1 hectare. By contrast, the present study identified higher species densities in the Perup and McCorkhill sites due to the collation of all geophyte species not just orchids. Densities of individual species were still similar to Grant and Koch (2003), ranging from 500 to 2200 individuals per 0.1 hectare in similarly aged plots (Figures 3.2 and 3.11). Grant and Koch (2003) concluded that orchid density was lower within young rehabilitated sites due to the lack of mycorrhiza in the soil, which thus inhibited orchid germination. Mycorrhizal requirements are not met until there is sufficient build up of leaf and organic matter. Their study showed these requirements are met 5-10 years post-fire within rehabilitated mine sites. Gardener and Malajczuk (1988) found that rehabilitated mine sites took 3 to 7 years before ectomycorrihzal fungi achieved adequate diversity and abundance. The present study of the Perup and McCorkhill sites showed peaks in species richness 5-10 years post-fire (Figures 3.3 and 3.12) that could be attributed to factors like the increased establishment of mycorrhizal associations in geophytes (Gardner & Malajczuk, 1988; Grant & Koch, 2003).

At the denser vegetation and higher rainfall jarrah complex of the McCorkhill Site, the present study found peaks in species richness approximately 5-7 years post-fire, similar to a comparable site near Nannup. These peaks in species richness follow closely the trends displayed by studies conducted on impacts of fire treatment on woody shrubs, perennial herbs and grasses (higher plants) in sites of similar jarrah complexes (Burrows & Wardell-Johnson, 2003). Richness values between geophytes and higher plants were also similar, with 9-13 species compared to 7-9 species per 0.1 hectare respectively (Figure 3.12). The more open, lower rainfall jarrah complex of the Perup Site, which is comparable to a site east of Manjimup, was shown to peak in species richness 9 years post-fire as studied by Burrows & Wardell-Johnson (2003), similar to that of the Perup Site at 10 years post-fire. Although peak richness was reached at a similar age post-fire, richness in geophyte species was much higher than that recorded for higher plants, namely, 12-16 species per 0.1 hectare (Figure 3.3) compared to 5-6 species per 0.1 hectare respectively (Burrows & Wardell-Johnson, 2003).

Within the McCorkhill site, both frequency of fire and time since last fire produced similar species richness trends. Peak richness was reached after 4 to 5 treatments since 1976

(Figure 3.22). This frequency pattern correlates to maximum richness being recorded every 6 to 7 years, which is almost identical to the pattern of peak species richness (namely, 5-7 years post-fire) found in the present study (Figure 3.12). Correspondingly, Burrows and Wardell-Johnson (2003) identified that species richness in higher plants (woody shrubs, perennial herbs and grasses) was greatest at treatment frequencies of 5-7 and 7-9 years. Peaks in species richness between higher plants and geophytes due to frequency of fire (5-9 years) were similar. However, the number of species identified was significantly different, with approximately 2 higher plant species being recorded compared to 8 to 13 geophyte species per 0.1 hectare (Burrows & Wardell-Johnson, 2003).

Although maximum species richness was recorded 5-10 years post-fire (for both sites) it was surprising that species richness within the 1-year-old treatments was so low (Figure 3.12). At these early post-fire stages individual species were not at peak growth or flowering, in contrast to much of the literature that suggests that fire-stimulated mass flowering often occurs (Hoffman & Brown, 1992; Bond & van Wilgen, 1996; Gill, 2002; Grant & Koch, 2003). This suggests that flowering in the species studied may not in fact be directly triggered by fire, and that other factors such as nutrient availability, presence of mycorrhizal fungi and ecosystem structure need to be taken into account (Dixon & Barrett, 2003; Grant & Koch, 2003). The findings of this study do not support the idea of a specific post-fire flora found only at recently burnt sites. Rather they indicate that the geophyte species are present at most post-fire ages but may vary in terms of abundance, with some species preferring early, intermediate or late aged treatments.

The season of burn, and hence the moisture content of organic matter, affects post-fire recovery primarily due to its impact on fire intensity. This is a prominent issue within dry sclerophyll forests (Gill et al., 1981; Morrison, 2002; Burrows & Wardell-Johnson, 2003; Watson & Wardell-Johnson, 2004). Summer burns produce the highest species richness as compared with autumn (April) and spring (November) burns, although the variation between the treatment seasons was found to be very small—ranging from a minimum of 8 to a maximum of 11.5 species (Figure 3.30). Although fire intensity and soil temperature would be highest in summer, these results are expected given that the treatment seasons coincide with the period of dormancy in the life cycle of geophytes. This survival mechanism would therefore negate the potential impact of the summer fire events on

individual geophytes and thus maintain populations (Pate & Dixon, 1982; Hoffman & Brown, 1992; Dixon & Barrett, 2003).

Burrows and Wardell-Johnson (2003) recorded higher plants (woody shrubs, perennial herbs and grasses) at peak richness (10.1) and abundance (24.7 per 0.5 hectare) after successive mixed spring and autumn condition burns. This effect would be unlikely to occur with geophytes. Fires that coincide with flowering and/or growth phases could significantly impact the population (Brown, pers. comm., 2005), killing individuals during the fire event and limiting the ability of the population to re colonise the area post treatment. Prolonged spring burns that do not coincide with dormancy (winter, early spring) could therefore reduce geophyte species assemblages, potentially limiting the ability of the colony to persist in the ecosystem.

4.3 Fire and Geophyte Responses

The impacts of fire on higher plant species have been well studied and documented, providing knowledge of these plants' specific responses to heat, smoke and nutrient ash: synchronised seed release, epicormic re-sprouting, and heat and smoke germination are all common among fire adapted species (Bradstock, 1981; Christensen et al., 1981; Gill et al., 1981; Bell, 1994; Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003). In contrast, specific responses of geophyte species are not as well understood, with information being speculative or observational.

Species of *Drosera, Caladenia, Leporella, Pyrorchis and Cyanicula* have all been reported to synchronise flowering with fire events (Hoffman & Brown, 1992; Dixon & Barrett, 2003). The release of floral dormancy has been reported within *D. erythrorhiza* as a result of ethylene gas produced from the burning of organic matter (Dixon & Pate, 1978). Although many species have demonstrated 'obligate' responses to fire for flowering and growth stimulus, fire at frequent intervals may not be necessary for the long-term survival of these species (Dixon & Barrett, 2003).

Pyrorchis nigricans was identified in the present study (Table 3.4) as a late post-burn response species, even though this species is seen as having an obligate flowering response to fire (see also Hoffman & Brown, 1992; Dixon & Barrett, 2003). This would agree with

the findings of Dixon & Barrett (2003) that certain species are able to 'bet-hedge' for fire intervals. Species such as *P. nigricans* have adapted to cope with long intervals between fires, and have evolved mechanisms to enable them to flower even after long unburnt periods. Whether this involves an increase in mycorrhizal associations, nutrients from ash-beds or gases released by decomposing organic matter within long unburnt areas is still a topic of conjecture (Gill et al., 1981; Chambers & Attiwill, 1994; Dixon & Barrett, 2003; Robinson & Bougher, 2003). It could then be concluded that although many species possess fire response mechanisms, fire may not be essential for survival. This would correlate to the mosaic or variable impact of fire on the landscape, with areas that have been frequently burnt or remained unburnt for long periods still able to maintain large assemblages of geophyte species (Figures 3.3 and 3.12).

4.4 Fire and Individual Geophyte Species Responses

Even though the two sites were significantly different in terms of geophyte composition (Figure 3.1), the same species between the sites were shown to exhibit very similar responses to fire. Utilising the CAP 'unique species' ordinations, species within the treatment sites that were found only within a certain treatment site—or were found in high abundance relative to other sites—were extracted and placed into categories outlining their response to time since last fire (Figures 3.7, 3.16 and 3.25).

With this in mind the effects of fire on environmental factors were investigated (Section 3.6), in order to ascertain whether the secondary impacts of fire on litter depth, canopy cover, shrub cover, soil salinity and soil pH have more of an impact on geophyte species assemblage than the fire itself. Through CCA bi-plots, both sites were shown to contain individual species influenced by such environmental factors (Section 3.6). Nevertheless, the actual species assemblages of geophytes were on the whole not drastically affected by the secondary impacts of fire, given the spatial patterns observed via CCA analysis and low overall species-environmental correlations. This could be due to variations in fire behaviour across the entire treatment site, with areas of differing structure being created during the fire event (Gill, 1982; Morrison, 2002; Burrows & Wardell-Johnson, 2003). An individual species analysis identified unique species and also those that significantly preferred different treatment ages. Then the preferred post-burn time/burn frequency was compared to environmental factors in order to assess why species prefer these treatment

types. This comparison revealed that species exhibited the same microhabitat preference across the two analyses. For example, a major factor affecting geophyte abundance was the increased litter and shrub cover within long unburnt sites (see section 3.6).

It has been suggested that the impact of fires on mycorrhizal fungi could be one of the factors influencing the presence/absence of certain species, in particular those species that have been shown to prefer older, less frequently burnt treatment sites (Warcup, 1981; Reddell & Malajczuk, 1984; Grove et al., 1986; Robinson & Bougher, 2003). It has also been postulated that growth and flowering responses of geophytes may relate not to the actual fire event but to the presence and proliferation of associate mycorrhiza, and the ability of these fungi to enhance nutrient uptake by the plant (Gill et al., 1981; Smith & Read, 1997; Robinson & Bougher, 2003). The present study supports this postulate: species of *Thelymitra, Pterostylis and Pyrorchis* were significantly more abundant in old, infrequently burnt treatment sites, and this could be due to their intimate relationships with mycorrhizal fungi (Tables 3.4 and 3.6). This in turn would correlate with recent studies by Robinson & Bougher (2003) that indicate mycorrhizal fungi are found in higher abundance within sites containing high amounts of leaf and other organic matter.

Despite the finding that *Thelymitra, Pterostylis and Pyrorchis* were abundant in infrequently burnt sites, little is known about the interactions between mycorrhizal fungi and orchids of the south-west of Western Australia. Further research would prove beneficial in understanding why the species outlined in Tables 3.4 and 3.6 prefer certain habitats, since any changes to the assemblage of fungal species could also impact upon the presence or abundance of geophyte species that are reliant on associate interactions (Warcup, 1981; Hoffman & Brown, 1992, Robinson & Bougher, 2003).

Many of the species identified as abundant in older/infrequently burnt treatments exhibit morphological characteristics that could account for their preference. This is the case with *Pterostylis*, where tubers are close to the soil surface, leaving them vulnerable to the direct impacts of fire (Pate & Dixon, 1982). High intensity burns could heat the soil enough (particularly during summer months) to impact parent tubers, thus killing the *Pterostylis* colony (Pate & Dixon, 1982; Hoffman & Brown, 1992; Dixon & Barrett, 2003). Environmental factors could also account for the abundance of species such as *Thelymitra crinita*, which was found in profusion within older treatment sites (Figure 3.20). Previous

studies of the species indicate that these plants tend to occur within moist, fertile soils that would be indicative of older/less frequently burnt sites (Hoffman & Brown, 1992). Species of *Thelymitra* and *Pterostylis* produce seeds annually in the growing season and do not require fire in order to reproduce and subsequently colonise an area (Pate & Dixon, 1982). Studies conducted in 20-year-old unmined jarrah forests showed that *P. nana* and *T. crinita* were in densities of greater than 1000 individuals per hectare (Grant & Koch, 2003). In summary, factors such as annual seed production, shallow tuber depths and moist/fertile soils allow these species to preferentially colonise areas that have remained unburnt for long periods.

Species that were shown to prefer early post-fire conditions and high frequency treatments were also shown to exhibit morphological characteristics or indeed ecological adaptations to areas of high fire disturbance (Tables 3.4 and 3.6). For example, *D. erythrorhiza*, *S. juneceum* and species of *Cyanicula* produce cutinised sheaths around each tuber or corm through the accumulation of dead epidermal tissue. This protects individuals and colonies from frequent fire events (Dixon & Pate, 1978; Pate & Dixon, 1981). In many instances heat, ethylene gas production and increased nutrient availability caused by the occurrence of fire have been found to stimulate growth and flowering in numerous species of *Drosera* (such as *D. erythrorhiza*) (Gill et al., 1981; Pate & Dixon, 1981 Burrows & Wardell-Johnson, 2003; Dixon & Barrett, 2003). Researchers have also speculated that the same applies for species of *Cyanicula* and other 'fire obligate' orchids (Gill et al., 1981; Hoffman & Brown, 1992). These species were significantly more abundant within recently burnt and frequently burnt plots (Tables 3.4 and 3.6). Species such as *D. erythrorhiza* that contain highly fire resistant sheaths produce daughter tubers post-fire, which enables them to recolonise recently burnt areas quickly (Figure 3.17), as has also been found by Dixon & Pate (1978). The observed regeneration of large patches of individuals post-fire has lead to the widely-held assumption that all geophyte species synchronize flowering with fire events.

4.5 Fire and the Use of Logs as Refuge Areas

Large burnt logs within the Perup Treatment Site were found to be utilised by many species as a refuge area, with many orchid species significantly preferring log habitats to that of open habitats (Table 3.8). In an open, dry jarrah complex such as the Perup Site, large logs

(those greater than 5 m long and 0.5 m in diameter) could provide an environment quite different from that of open areas—possibly due to increases in shading, moisture and nutrient content of the soil under and near the logs (Gill et al., 1981).

Protection from fires could also be a factor influencing some species that have been shown to significantly prefer log habitats to open habitats. The shallow tuber depth of species such as *P. nana* and *C. flava* could be a reason. Logs provide protection for tubers under intense or frequent fire or even wildfire conditions, ensuring that geophyte individuals are able to survive the disturbance and maintain the colony (Pate & Dixon, 1981). Although the burning of dry logs in conjunction with large amounts of leaf litter could create more intense heat in these habitats, particularly in summer where moisture can be uniform across an area. Thus the increased intensity due to the burning of the log itself could have a negative impact on geophyte inhabitants. Increased moisture around the logs also acts as a buffer by lowering the amount of heat transferred into the soil from fire events. The reduction in heat transfer provides additional tuber protection for individuals that have colonised the log habitat (Christensen, 1975; Burrows & Friend, 1998; Burrows & Wardell-Johnson, 2003).

Mycorrhizal fungi that colonise decaying wood could also be a factor contributing to the presence of geophyte species (Robinson & Bougher, 2003). The fertilization effect of the ash-bed produced by these large logs could further enhance the nutrient uptake of geophyte species via their mycorrhizal associations (Gill et al., 1981; Pate & Dixon, 1981; Smith & Read, 1997). Although not tested the presence of fungi around these logs was evident through the observed diggings of small marsupial species (woylies and bandicoots) foraging under the logs.

4.6 Findings with Management Implications

The findings of this study have implications for the use of prescribed burning as a management tool within south-west Western Australian jarrah forests and are as follows:

- Due to the morphology and survival mechanisms of geophytes, species assemblages will persist in the environment despite a wide variety of fire regimes. Nevertheless, individual species have shown preferences either for long unburnt or for frequently

burnt areas. It would therefore prove beneficial for prescribed fire regimes to create mosaic environments both at a local and landscapes scale within the forest to increase variability in forest structure and composition, thus providing areas suitable for a variety of geophyte species.

- Large logs are an important refuge habitat for many geophyte species, particularly orchids. These interactions were proven to be more prevalent within open/dry sclerophyll forests. Management initiatives need to recognise that these microhabitats are essential for the maintenance of geophyte diversity, and as such should be preserved.

- The patchy nature of the occurrence of geophytes could be a result of populations following fine-scale variations in nutrient availability and in ecosystem structure. Therefore geophytes could be utilised as an indicator of microclimate variation within forest ecosystems.

- Maximum diversity for geophyte species is reached 5-10 years post-fire within Southern Jarrah complexes. Hence, fire regimes should be spread across different seasons at intervals of 5 to 10 years in order to ensure sufficient time for mycorrhizal associations to establish.

4.7 Conclusion

The results of this study have shown that overall geophyte species assemblages within southern jarrah forests are not significantly impacted by time post-fire, the fire frequency and the fire seasons, although individual geophyte species displayed a definite preference in fine-scale habitat variations caused by fire events. Geophytes may follow micro-scale variations in habitat structure, composition and nutrient availability, becoming prominent in areas suitable to colonisation of specific species (Gill et al., 1981; Morrison, 2002; Burrows & Wardell-Johnson, 2003; Watson & Wardell-Johnson, 2004). Prescribed fire regimes that create mosaic environments both at a local and landscapes scale could increase variability in forest structure and composition, thus providing areas suitable for a variety of

geophyte species. Implementation of fire management plans should also take into consideration the time taken for mycorrhizal associations to establish, which from these findings may correlate with the peak in geophyte diversity reached 5-10 years post-fire (Figures 3.3 and 3.12).

Geophyte response to microclimate variability, their small dispersal ranges and stochastic nature of plants have contributed to the patchy density and richness of sites within the jarrah forests of south-west Western Australia. This patchiness is more or less maintained regardless of the fire regimes, since dormancy, morphological adaptations, 'bet-hedging', mycorrhizal associations and refuge habitats ensure that species assemblages remain constant in times of both high and low fire disturbance (Dixon & Barrett, 2003; Gill et al., 1981; Pate & Dixon, 1981; Robinson & Bougher, 2003; Smith & Read, 1997). This is not to say that geophytes will not be impacted by fire regimes but rather that geophytes possess various mechanisms to enable them to survive diverse fire regimes and remain within the ecosystem. This makes post-fire population explosions a very site-specific process and not indicative of all burnt areas.

Geophyte species, orchids in particular were shown to use large, fallen burnt logs as refuge areas. These finding were observed primarily within the dry/open sclerophyll forest of the Perup Site. Log habitats could provide suitable conditions in terms of shade, moisture, nutrients, protection and possibly mycorrhizal associations. Species preferring these habitats are able to colonise these preferable conditions quickly and abundantly, making logs an important factor for the maintenance of geophyte diversity within jarrah forest (Hoffman & Brown, 1992; Pate & Dixon, 1981). As such, management plans need to recognise and preserve these microhabitats as they could be essential for the long-term maintenance of geophyte diversity.

4.8 Recommendations for Practice and Further Research

This study set out to achieve five key objectives: The first three interrelated objectives sought to determine the impacts that time since last fire (yrs), and frequency and season of fire have on geophyte abundance, diversity and composition. The fourth objective was to determine whether ecosystem structure and micro-climate are important in controlling geophyte abundance, diversity and composition. Fifthly determine whether microhabitats

such as large, fallen burnt logs are used by geophyte species as refuge areas. In doing so the following measures were found to be appropriate in the event of further study in this area:

- Baseline species data should be collected prior to the initiation of fire treatments in order to better understand the before and after impacts of fire regimes.

- Winter and early spring burns should be conducted to gauge the impact of fire on geophytes at peak flowering and growth phases.

- All plots should be burnt simultaneously in order to better understand the role of fire frequency and to eliminate patterns associated with time elapsed since last fire.

- Refuge habitat analysis should be extended to include the basal area of trees and under shrubs (cycads in particular). The latter would also prove particularly interesting since observations indicated that many orchid species were also utilising those areas (albeit to a lesser extent than logs) as refuges.

- Analysis of soil nutrients (moisture, nitrogen content) around refuge areas would also prove beneficial in better understanding why geophytes utilise logs as refuges.

- Fungi baiting should be carried out within different treatments and log habitats to ascertain whether the presence/absence of mycorrhiza due to variations in fire regime has an effect not only on the germination of geophyte species assemblages but also of individual species.

- Analysis of geophyte storage organ variability should be carried out to determine whether this property could be used as an indicator of fire effects.

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APPENDIX

Table A.1 Combined species data treatments within the Perupm site at a plot-scale.

TREATMENT		2Y OLD, 5 BURNS, Summer/Autumn		8Y OLD, 2 BURNS, Spring/November		10Y OLD, 2 BURNS, Autumn/March		12Y OLD, 1 BURN, Autumn/March		22Y OLD CONTROL	
PLOT NUMBER		2	10	4	5	1	12	7	9	3	6
Species	Code										
Drosera menziesii	D1	286	1231	827	656	495	1121	760	298	414	441
Drosera pallida	D2	49	57	126	42	66	28	31	82	48	60
Drosera glanduligera	D4	0	0	0	0	2	7	0	0	0	0
Drosera erythrorhiza	D5	0	0	14	0	3	17	0	1	0	0
Drosera sp.	D6	0	0	0	0	0	0	0	0	0	0
Stylidium calcaratum	S2	48	639	13	120	35	163	28	60	7	19
Stylidium shoenoides	S3	0	0	0	2	1	4	1	1	0	2
Stylidium sp.	S4	0	5	397	108	0	0	323	3	262	77
Stylidium scandens	S5	0	0	0	0	0	0	0	0	0	0
Stylidium junceum	S6	0	0	0	0	0	0	0	0	0	0
Caladenia flava	O1	24	16	11	1	31	53	4	15	36	9
Caladenia flava. Sylvestris	O2	0	0	0	0	3	0	0	0	0	0
Caladenia latifolia	O3	0	0	0	0	6	2	0	0	0	0
Caladenia longicuada	O4	0	0	0	0	0	0	0	0	0	0
Caladenia macrostylis	O5	0	0	0	0	0	0	0	0	0	0
Caladenia reptans	O6	0	2	0	0	0	4	10	5	10	0
Caladenia nana	O7	0	1	8	0	0	0	2	0	10	0
Corybas recurvus	O8	0	0	0	0	0	0	0	0	0	0
Cyanicula sp.	O9	0	0	0	0	0	0	0	0	0	0
Cyanicula sericea	O10	0	0	0	0	0	0	1	1	0	0
Cyrtostylis robusta	O11	10	0	0	0	0	0	0	0	0	0
Cyrtostylis huegelii	O12	18	30	37	2	6	6	24	48	26	11
Cyrtostylis sp.	O13	0	0	0	0	0	0	0	0	0	0
Diuris longifolia	O14	0	0	0	0	0	2	0	0	0	0
Elythranthera brunonis	O15	0	0	0	0	13	1	3	0	0	7
Leptoceras menziesii	O16	0	0	0	0	0	0	0	0	0	0
Lyperatnthus serratus	O17	0	0	0	0	0	0	0	0	0	0
Pterostylis barbata	O18	0	0	0	0	0	0	0	0	0	0
Pterostylis nana	O19	4	0	3	6	0	6	0	5	19	2
Pterosylis recurva	O20	0	0	0	0	0	0	0	0	0	0
Pyrorchis forrestii	O21	19	28	0	3	0	37	3	12	0	2
Pyrorchis nigricans	O22	11	0	0	4	59	0	0	0	0	0
Pyrorchis scava	O23	0	0	0	0	0	0	0	0	0	0
Thelymitra benthamiana	O24	0	0	0	0	1	0	0	0	0	0
Thelymitra crinita	O25	0	0	11	0	0	0	19	0	18	11
Thelymitra sp.	O26	0	0	0	0	3	0	1	10	0	0
Abundance		**469**	**2009**	**1447**	**944**	**724**	**1453**	**1210**	**541**	**850**	**641**
Number of Species		**9**	**9**	**10**	**10**	**14**	**14**	**14**	**13**	**10**	**11**
Total Treatment Abundance			**2478**		**2391**		**2177**		**1751**		**1491**
Abundance Average for treatment			**1239**		**1195.5**		**1088.5**		**875.5**		**745.5**
No Species Average for treatment			**9**		**10**		**14**		**13.5**		**10.5**

Table A.2 Perup site environmental data at a plot scale.

TREATMENT	2Y OLD, 5 BURNS, Summer/Autumn		9Y OLD, 2 BURNS, Spring/November		10Y OLD, 2 BURNS, Autumn/March		12Y OLD, 1 BURN, Autumn/March		22Y OLD CONTROL	
PLOT NUMBER	2	10	4	5	1	12	7	9	3	6
Plot Average pH	5.225	4.82625	4.80375	5.181	5.2968	4.26925	4.4735	4.3075	4.585	4.30225
Plot Average Salinity	0	0.005	0.005	0	0.01	0.00625	0.005	0.005	0	0.005
Plot Average Conductivity	184.75	211.4225	210.848	208.173	210.98	217.635	218.37	225.283	207.1	223.025
Treatment average ph		5.025625		4.99238		4.783		4.3905		4.4435
Treatment average salinity		0.0025		0.0025		0.008125		0.005		0.0025
Treatment average conductivity		198.085		209.51		214.30625		221.828		215.058
Av. % Canopy Cover for Plot	50	50.25	53	50.75	47.5	50.75	55	55	60.5	57.5
Average Litter Depth for Plot	0.465	0.685	1.245	1.275	1.62	1.275	1.325	1.4	1.385	1.375
Av. % Shrub Cover for Plot	28.6	25.6	29.8	37.6	52	35.4	36.6	45	44.6	55
Av. % Canopy Cover for Treatment		50.125		51.875		49.125		55		59
Av. Litter Depth for Treatment		0.575		1.26		1.4475		1.3625		1.38
Av. Shrub Cover for Treatment		27.1		33.7		43.7		40.8		49.8

Table A. 3 Perup site whole treatment averages: species data, soil data and ecosystem data.

Treatment	2Y OLD, 5 BURNS, Autumn/March	9Y OLD, 2 BURNS, Spring/November	10Y OLD, 2 BURNS, Autumn/March	12Y OLD, 1 BURN, Autumn/March	22Y OLD CONTROL
Total Treatment Abundance	2478	2391	2177	1751	1491
Average Abundance	1239	1195.5	1088.5	875.5	745.5
Average Number of Species	9	10	14	13.5	10.5
Average Ph	5.025625	4.992375	4.783	4.3905	4.4435
Average Salinity	0.0025	0.0025	0.008125	0.005	0.0025
Average Conductivity	198.085	209.51	214.30625	221.8275	215.0575
Average % Canopy Cover	50.125	51.875	49.125	55	59
Average Litter Depth (cm)	0.575	1.26	1.4475	1.3625	1.38
Average % Shrub Cover	27.1	33.7	43.7	40.8	49.8
Age since last burn	2	9	10	12	22
Number of Burns	5	2	2	1	~
Season/Month of Burns	Summer	Summer/Nov	Autumn/March	Autumn March	~

Table A. 4 Log data for the Perup Site: averaged for treatment.

	2 Year Old	9 Year Old	10 Year Old	12 Year Old	22 Year Old
Drosera menziesii	117	74	99	65	7
Drosera pallida	17	47	21	45	40
Drosera glanduligera	13	0	0	0	1
Drosera erythrorhiza	0	0	0	0	0
Drosera sp.	0	0	0	0	0
Stylidium calcaratum	25	4	54	2	2
Stylidium shoenoides	0	0	0	0	0
Stylidium sp.	0	16	0	38	37
Stylidium scandens	0	0	0	0	0
Stylidium juneceum	0	0	0	0	0
Caladenia flava	43	19	125	49	37
Caladenia flava. Sylvestris	0	12	27	0	10
Caladenia latifolia	0	0	5	0	0
Caladenia longicuada	0	0	0	0	0
Caladenia macrostylis	0	0	8	0	0
Caladenia reptans	12	0	3	32	27
Caladenia nana	3	8	0	3	8
Corybas recurvus	0	0	0	0	0
Cyanicula sp.	0	0	0	0	0
Cyanicula sericea	0	0	0	0	0
Cyrtostylis robusta	5	0	0	0	0
Cyrtostylis huegelii	21	143	4	117	239
Cyrtostylis sp.	0	0	0	0	0
Diuris longifolia	0	0	0	0	0
Elythranthera brunonis	0	0	0	3	0
Leptoceras menziesii	0	0	0	0	0
Lyperatnthus serratus	0	0	0	0	0
Pterostylis barbata	0	0	0	0	0
Pterostylis nana	0	27	17	37	178
Pterosylis recurva	0	0	0	0	0
Pyrorchis forrestii	18	0	122	0	1
Pyrorchis nigricans	17	0	14	0	24
Pyrorchis scava	0	0	0	0	0
Thelymitra benthamiana	0	0	0	0	0
Thelymitra crinita	0	0	0	6	27
Thelymitra sp.	0	0	0	7	0
Abundance	291	350	499	404	638
Richness	11	9	12	12	14

Table A. 5 Combined species data for treatments within the McCorkhill site at a plot scale.

Treatment		Fire Break - 1 Year Old		4 Years Old - 4 Burns - Autumn/April		5 Years Old - 4 Burns - Spring/Dec		5 Years Old - 5 Burns - Autumn/Feb		10 Years Old - 2 Burns - Spring/Nov		21 Years Old - Age only plot		28 Years Old - 1 Burn - Control	
plot number		12	13	9	4	3	5	10	6	11	8	7	14	1	2
Species	code														
Drosera menziesii	D1	158	311	65	72	183	0	5	284	0	0	44	29	150	89
Drosera pallida	D2	1	0	12	5	10	11	21	5	12	4	2	14	2	3
Drosera glanduligera	D4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drosera erythrorhiza	D5	91	10	0	37	1	131	3	19	42	0	5	11	0	0
Drosera sp.	D6	0	6	33	0	0	6	19	10	0	9	35	0	75	28
Stylidium calcaratum	S2	3	0	0	0	0	0	9	11	0	0	0	0	0	0
Stylidium shoenoides	S3	0	0	0	0	0	0	0	0	0	2	0	0	0	3
Stylidium sp.	S4	0	30	167	15	56	6	49	30	8	0	27	15	6	13
Stylidium scandens	S5	0	0	0	40	0	214	0	0	0	0	0	0	0	0
Stylidium juneceum	S6	0	0	85	7	5	4	24	53	0	11	1	13	0	0
Caladenia flava	O1	0	0	0	2	0	2	0	0	0	1	0	0	1	3
Caladenia flava. Sylvestris	O2	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Caladenia latifolia	O3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caladenia longicuada	O4	0	0	3	0	1	2	4	1	1	0	0	0	0	0
Caladenia macrostylis	O5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caladenia reptans	O6	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caladenia nana	O7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Corybas recurvus	O8	0	0	0	0	0	13	0	0	0	0	0	0	0	0
Cyanicula sp.	O9	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Cyanicula sericea	O10	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Cyrtostylis robusta	O11	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyrtostylis huegelii	O12	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyrtostylis sp.	O13	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diuris longifolia	O14	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Elythranthera brunonis	O15	0	0	0	0	0	1	2	2	0	0	0	0	0	0
Leptoceras menziesii	O16	0	20	0	0	3	0	0	0	0	0	0	0	0	0
Lyperatnthus serratus	O17	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pterostylis barbata	O18	0	0	0	0	0	0	0	0	0	0	2	3	0	0
Pterostylis nana	O19	0	0	4	0	0	0	0	0	0	0	6	3	6	10
Pterosylis recurva	O20	0	0	0	0	0	0	0	0	0	0	1	4	0	0
Pyrorchis forrestii	O21	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pyrorchis nigricans	O22	0	1	2	1	12	11	22	15	43	10	0	0	83	31
Pyrorchis scava	O23	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thelymitra benthamiana	O24	0	2	0	0	1	1	0	0	6	3	0	0	0	0
Thelymitra crinita	O25	0	0	2	2	3	0	1	0	0	19	34	27	22	22
Thelymitra sp.	O26	0	0	0	0	0	0	0	0	0	1	0	0	3	0
Abundance		253	381	373	181	275	402	160	440	112	60	157	119	348	203
Number of Species		4	8	9	9	10	12	12	11	6	9	10	9	9	10

Table A. 6 McCorkhill environmental data at a plot scale.

Treatment	Fire Break - 1 Year Old		4 Years Old - 4 Burns - Autumn/April		5 Years Old 4 Burns Spring/Dec		5 Years Old - 5 Burns - Autumn/Feb		10 Years Old - 2 Burns - Spring/Nov		21 Years Old Age only plot		28 Years Old - 1 Burn - Control	
plot number	12	13	9	4	3	5	10	6	11	8	7	14	1	2
Average Plot Ph	4.666	4.667	4.58	4.4705	4.17	4.169	4.63	4.7035	4.172	4.3548	4.529	4.868	4.127	4.127
Average Plot Salinity	0	0	0	0	0.01	0.005	0	0	0.003	0	0	0.015	0.01	0.01
Average Plot Conductivity	213.9	213.9	211	198.85	192	192.3	210.7	218.37	196.3	202.19	204.7	237.4	225	225
Average Treatment pH		4.667		4.525		4.17		4.6669		4.2635		4.698		4.127
Average Treatment Salinity		0		0		0.005		0		0.0013		0.008		0.01
Average Treatment Conductivity		213.9		204.9		192.3		214.53		199.27		221		225
Average Plot % Canopy Cover	68.25	66.25	68.75	69.25	72.5	72.5	68.75	66.25	70.75	70.5	71.75	73	72.75	71
Average Plot Litter Depth (cm)	0.65	0.625	1.775	1.6	2.6	2.6	1.575	1.7	2	2.075	2.425	1.9	2.775	2.625
Average Plot % Shrub Cover	15.8	15	42.8	37.2	53.8	56.6	31.2	37.8	39.4	39.6	38.4	34.2	49.6	45.6
Av Treatment % Canopy Cover		67.25		69		72.5		67.5		70.625		72.38		71.88
Av Treatment Litter Depth (cm)		0.638		1.6875		2.6		1.6375		2.0375		2.163		2.7
Av Treatment % Shrub Cover		15.4		40		55.2		34.5		39.5		36.3		47.6

Table A. 6 McCorkhill site whole treatment averages: species, soil and ecosystem data.

Treatment	Fire Break - 1 Year Old	4 Years Old 4 BurnsAutumn/April	5 Years Old 5 Burns Autumn/Feb	5 Years Old 4 Burns Spring/Dec	10 Years Old 2 Burns Spring/Nov	21 Years Old - Age only plot	28 Years Old 1 Burn Control
Total Treatment Abundance	634	554	600	677	172	276	551
Average Plot abundance	317	277	300	338.5	86	138	275.5
Av. Number of species for treatment	6	9	11.5	11	7.5	9.5	9.5
Average Treatment pH	4.66675	4.525	4.666875	4.16975	4.2635	4.698125	4.127
Average Treatment Salinity	0	0	0	0.005	0.00125	0.0075	0.01
Average Treatment Conductivity	213.8775	204.90125	214.525	192.2525	199.26625	221.03125	225
Average Treatment % Canopy Cover	67.25	69	67.5	72.5	70.625	72.375	71.875
Average Treatment Litter Depth (cm)	0.6375	1.6875	1.6375	2.6	2.0375	2.1625	2.7
Average Treatment % Shrub Cover	15.4	40	34.5	55.2	39.5	36.3	47.6
Age since last burn	1	4	5	5	10	21	28
Number of burns	~	4	5	4	2	~	1
Season/Month of burns	~	Autumn April	Autumn/Feb	Spring/Dec	Spring/Nov	~	~