The influence of prescribed fire treatments on the abundance of Hakea sericea in fire-prone areas in Northern Portugal

Asjra Bosch
5533945

Utrecht University
Sustainable Development – Environmental Change & Ecosystems

Supervisors:
* Ángeles Garcia Mayor (Utrecht University)
* Oscar Gonzalez Pelayo (CESAM, Universidade de Aveiro, Portugal)

Second reader: Mariska te Beest (Utrecht University)
Mentor in Portugal: Paula Maia (Universidade de Aveiro, Portugal)

Abstract

Reinforced by fire occurrence, the invader *Hakea sericea* is increasingly emerging in burned areas in Portugal. Being able to alter ecosystem properties and threatening local biodiversity, management of invasive species should have high priority. Prescribed burning is a widely applicated tool in the Mediterranean regions to reduce fuels loads, and more recently, it seems also a promising way to control particular invasive species. However, research on the effectiveness of prescribed burning regarding this issue is barely documented, especially for the European Mediterranean regions. In this study, the short-term influence of two types of prescribed burning treatments (slash and burn vs. burn) on the vegetation community, soil surface cover, plant community structure and the abundance of the invader *Hakea sericea* is examined. The experiment was carried out with a total of 6 5x2 m plots, with 3 replicates for each treatment. During the burn, fire temperature and duration were recorded in the surface, duff, and soil layer using thermocouples. Soil burn severity was estimated directly after the fire. The study area was visited every 1-2 months, in which development of the soil surface cover was documented together with species abundance. Furthermore, vegetation species diversity was determined in 3 subplots located in the bottom, middle and top of each plot. The main findings of this study show that the relative abundance of *Hakea sericea* is slightly higher in burned plots, but not significant. In contrast to our expectation, soil burn severity was significantly higher in the burned plots. This possibly explains why the species amount was in general lower for the burned plot, as some vegetative buds or seeds might be destroyed. Both treatments seemed to be effective in reducing *Hakea sericea* density in the short term. However, more research to fire characteristics, combined with local soil-, and vegetation development in the long-term, is needed to get a complete overview of the effect of prescribed burns on invasive species such as *Hakea*.

Keywords: *Plant invasions, Hakea sericea, prescribed fires, Mediterranean Basin, fire-prone ecosystems*

Content

1. Introduction

1.1 State of the art & problem definition

Fire has been an important component within Mediterranean ecosystems for millennia. These systems are characterized by vegetation that is well adapted to fire or to a particular fire regime resulting in a balanced and dynamic ecosystem (Pausas et al., 2009; Vallejo et al., 2012). However, these ecosystems are becoming highly disturbed and more prone to invasive species.The Mediterranean is known for its long history of intensive land use and fire has been applied as a management tool already since the Bronze age (4500 BP) for deforestation and pasture improvement (Keeley et al., 2011). However, since 1970, fire regimes have perceptibly shifted towards higher fire frequency and burned area as consequence of land use change (e.g., afforestation and land abandonment) and climate change (Kraus et al., 2016; Keeley et al., 2011; Moreira et al., 2011; Tessler et al., 2016a; Vallejo et al., 2012). Rural depopulation, abandonment of farmland and the establishment of pine and eucalypt plantations increased fuel loads resulting in larger and more frequent fires (Keeley et al., 2011; Pausas et al., 2009). This recent shift in the fire regime has raised concern about the ecological and socio-economic impact of wildfires and their ability to alter landscape patterns, soil properties, vegetation composition and community structure (Moreira et al., 2011; Vallejo et al., 2012). Altered fire regimes can contribute to a higher susceptibility of plant communities to invasive species, and once invaded the invader may alter the disturbance regime to a point that native species cannot cope with anymore. Several invasive species are for example known to increase fuel loads and thus increasing the fire intensity. Such changes on community and ecosystem level have been observed for example in Western North America where non-native grasses (e.g. *Bromus tectorum*) displace the native shrub/steppe vegetation by altering the fire regime and other ecosystem properties (Brooks et al., 2004). Also, Mediterranean shrublands in California suffer from invasion by alien grasses and forbs during postfire years (Keeley et al., 2005), and South-African Fynbos shrublands are invaded by several tree and shrub species including pines, eucalypt, and *Acacia* trees, and *Hakea* woody shrubs (van Wilgen., 2009). In the Mediterranean Basin, invasive plant species are increasingly emerging in burned areas, particularly in the humid coastal regions (Marchante & Marchante, 2016; Vallejo et al., 2012). In central and northern Portugal, the study area of this research, species of the genera *Acacia*, *Hakea*, and *Ailanthus* are increasing in abundance (Vallejo et al., 2012). Such invasive trees and shrubs often appear in dense monopolistic stands supressing other vegetation, decreasing biological diversity, and increasing competition for water resources (Alvarez-Taboada et al., 2017). According to de Almeida & Freitas (2018) around 20% of the approximately 3800 taxa of Portuguese vascular flora is consisting of invasive plant species (Fig. 1).

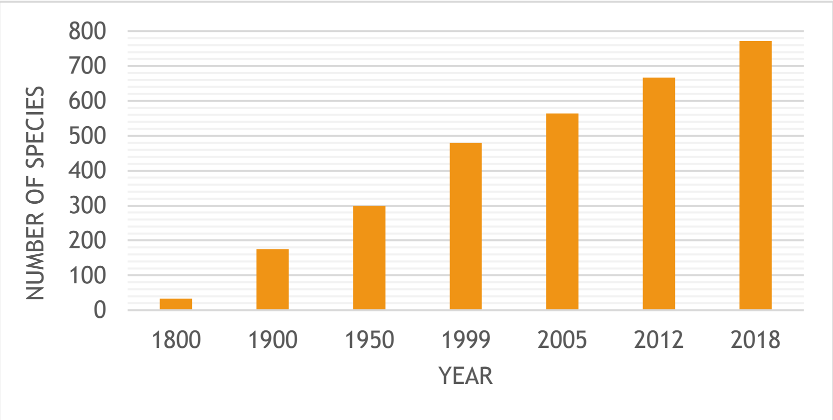


Figure 1: Evolution of invasive species over time in Portugal. Data is derived from Marchante & Marchante (2005), de Almeida & Freitas (2006), de Almeida & Freitas (2012), and de Almeida (2018).

Invasive species thus do not only change the fire regime, but also negatively affect economic, social and ecological values and are posing a threat to both native and global biodiversity (Morais et al., 2017). As stated by IPBES (2019) around one fifth of the Earth's surface is now at risk of plant and animal invasion. Emergence of invasive species is one of the most important drives next to climate change, land use change and pollution exacerbating the negative impacts on nature and affecting nature's contribution to humanity (IPBES, 2019). In Europe, it is estimated that problematic alien species, including plants, animals, fungi and other micro-organisms, have cost around 12 billion euros per year over the past 20 years, and it is expected that this will increase even more in the future (European Commission, 2013). Consequently, the European Union has given high priority to the identification, control and prevention of establishment and spread of invasive species. The Mediterranean is known for its high species richness and high number of endemic plant species. For this reason, it is designated as one of the world's top biodiversity hotspots and conservation of the area has high urgency (European Commission, 2019; Myers et al., 2000). Nevertheless, fire management is often focusing on preventing wildfires and reducing fuel loads, but less attention has been paid to management of invasive species and recovery of fire-prone areas (Vallejo et al., 2012).

Prescribed fires are widely used in the Mediterranean to suppress wildfires, and their effect on soils and shrublands are increasingly studied (Alcañiz et al., 2018; Badía et al., 2017; González-Pelayo et al., 2006; Úbeda et al., 2005), but their effect on the abundance of invasive species in the Mediterranean remains unknown (Fernandes, 2018). For effective management of invasive species it is thus needed to review these practices. Until now, most information regarding the control of invasive species is based on croplands instead of wildlands. Wildlands are more complex, and timing of the fire, fuel characteristics (e.g. fuel moisture), and fire types can be very different from croplands. It is thus important to incorporate these features. More recently, prescribed fires are also used as tool to control invasive species, especially to combat invasive annual grasses. In the western United States for example, prescribed fires have been found to be most effective on summer annuals and some winter annuals as *Taeniatherum caput-medusae* and *Aegilops triuncialis* (DiTomaso & Johnson, 2006). In other ecosystems as savannas, prescribed fires are increasingly used to control woody encroachment in order to sustain open grasslands. Using prescribed fires in combination with clearing has been proven to be successful in reducing biomass of the woody invader *Chromolaena odorata* in South African savanna (te Beest et al., 2012). However, prescribed fires alone were less effective in reducing biomass and preventing re-establishment of this woody invader. Follow-up control by clearing is of main importance in management of this invasive species (te Beest et al., 2012; te Beest et al., 2017). Control by prescribed fires are therefore not always successful, and especially some resprouting woody species or biennials, seem to be difficult to control with this practice. Also, the use of prescribed fires can be at the expense of valuable native species (Bradshaw et al., 2018; DiTamoso & Johnson, 2006; Zavaleta et al., 2001). The effectiveness of prescribed fires is thus highly variable, depending on the target species, timing and location. For this reason, research on the influence of different factors (e.g. climatic variables, soil type, fire characteristics, community structure) on the effectiveness of prescribed fires to control invasive species is needed (DiTomaso & Johnson, 2006).

1.2 Research question

This research aims to gain insight into the use of prescribed burning as a tool to control the spread of invasive plant species. The study focuses on Northern Portugal as this is one of the regions having one of the highest proportions of burned area per year in combination with an alarming rate of spread of invasive plant species, which are both expected to grow as a consequence of increased fire events and climate change (Silva et al., 2011; Vallego et al., 2012). The main focus of the study is the invasive shrub species *Hakea sericea*, which is recorded as invasive in Portuguese law and also seen as one of the most aggressive invasive species in Portugal next to *Acacia dealbata* (Marchante & Marchante, 2016). The ultimate goal of this research is to contribute to the identification of best practices for the management of fire-prone areas in Northern Portugal. The results may also be useful for other Mediterranean countries in Europe or even other Mediterranean climate regions found in southwestern Australia, central Chile, coastal California, and South-Africa. The following research question will be addressed: *What is the influence of different prescribed fire treatments (slash and burn versus burn) on the abundance of the invasive plant species Hakea sericea, and on the structure of the whole vegetation community in fire-prone areas in Northern Portugal?*

1.3 Hypothesis

The hypothesis is that the abundance of *Hakea sericea* will be lower under slash and burn than under burn, because the expected higher fire severity under slash and burn practices will destroy more seeds of *Hakea sericea*. It is also expected that slash and burn slows down the recovery of the whole vegetation community relative to the burn treatment.

1.4 Thesis outline

In the first chapter some background information will be given concerning the concept of fire regimes in relation to plant response. Also the use of fire as management tool to control invasive species will be described in this chapter. The third chapter describes the methodology of this study, including the study location, experimental setup and data collection related to fire characteristics and vegetation cover. Then the results concerning fire characteristics, vegetation development, and soil properties will be presented in chapter 4, followed by the discussion and conclusion in chapter 5.

2. Background and local context

2.1 The concept of fire and ecosystem responses

Ecosystem responses to fire are very context- and site specific. In this paper, ecosystem response refers to all kind of changes in ecosystem processes caused directly or indirectly by fire. Fire regimes influence vegetation community composition and structure but are also influenced by these which results in a complex interaction (Mandle et al., 2011; Vallejo et al., 2012). The fire regime is defined by the average fire characteristics and fire patterns for a given site and time period (Fig 2). This comprises aspects such as fire frequency, intensity, severity, size, duration and seasonality (Mandle et al., 2011). Fire frequency is the amount of fires occurring within a certain period. The fire intensity is the energy released during a fire and often expressed in kJ/m^2. Fire severity is correlated to fire intensity, and describes the impact of the fire on the vegetation and soil (Ansley, 2000; Keeley, 2009). Fire regime aspects which are believed to have the strongest influence on plant response in burned areas are fire severity and -frequency. The relation between these aspects and vegetation will be explained in greater detail in section 2.3, after a short introduction on plant survival strategies in fire-prone areas.

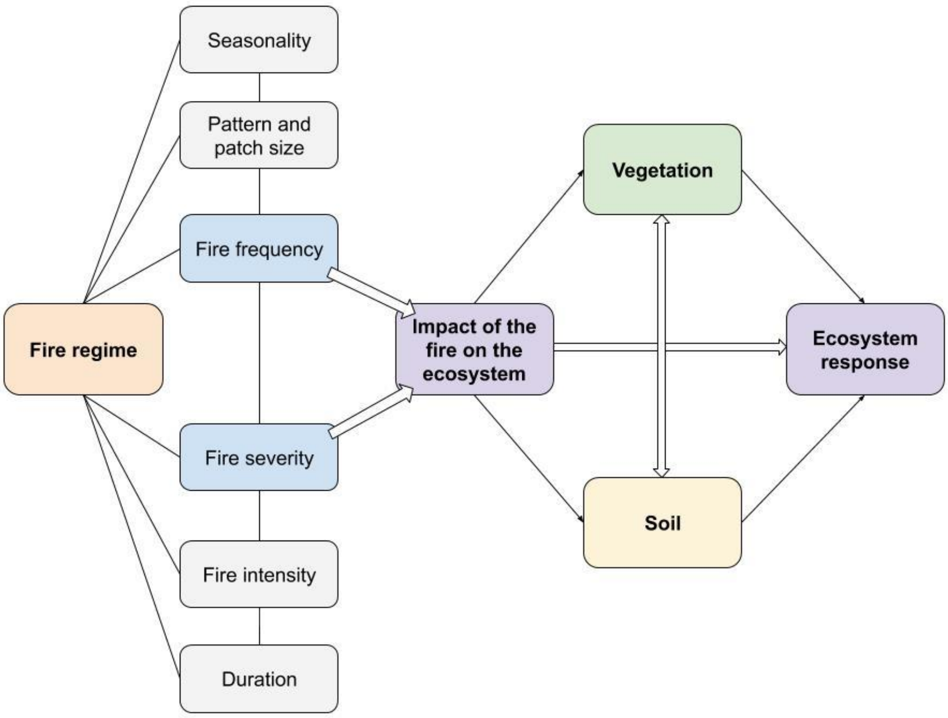


Figure 2: Fire regime as concept and its relation to ecosystem responses.

2.2 Post-fire plant response

Based on plant regeneration mechanisms and post-fire strategies plant species in shrublands can be divided into three groups: obligate seeders, sprouters, and facultative seeders. Facultative seeders have the ability to both resprout and germinate after fire. Obligate seeders germinate only by seeds available in the canopy or soil seed bank. Spouters generate by shoots from existing plant meristems, and different types can be distinguished (e.g. epicormic resprouting, resprouting from roots, rhizomes, root collar). Especially shooting from epicormic buds lying underneath the bark or stem, is associated with fire-prone systems, and applied by many *Eucalyptus* species. Also resprouting from lignotubers, a woody tissue located at the root-shoot transition zone is characterizing for plants exposed by (crown) fires (Paula et al., 2016; Keeley et al., 2011; Pausas & Keeley, 2014). In nutrient rich and moist conditions, regeneration by resprouting is more beneficial compared to seeding. In this case, quick establishment of aboveground biomass from existing plant tissue limits seed recruitment (Pausas & Keeley, 2014). However, in more sever circumstances such as low soil fertility or low, and oscillating water availability postfire resprouting ability is lower offering opportunities to seeders. Especially, high intensity fires favour seeders instead of resprouters because vegetative buds might be killed (Moreira & Pausas, 2012). Seeders are better adopted to droughts and changing postfire conditions, because of their possibility for delayed reproduction, a bigger spatial range, and increased genetic differentiation (Pausas & Keeley, 2014). Combined with evolved traits as rapid seedling growth, and the formation of heat resistant seeds, seeders regenerate well after fire, being dominant in most of nowadays fire-prone plant communities found in the Mediterranean (Granged et al., 2011; Pausas et al., 2009; Pausas & Keeley, 2014).

2.3 Influence of fire severity and frequency on vegetation

2.3.1 Influence of fire severity

As mentioned above, fire severity refers to the effects of a fire, with a focus on soil, and the survival and structure of the vegetation community. It measures the loss or decomposition of organic matter above or below ground by using for example plant mortality or looking to the amount of tree canopy damage (Keeley, 2009; Moreira et al., 2011). Other indicators for fire severity are the diameter of remaining twigs for indicating loss of biomass, or the colour of ashes used for changes in soil related properties (Maia et al., 2012). Fire severity influences the amount of individuals surviving a fire and affects seed germination or resprouting potential (Vallejo et al., 2012). Survival of seeds in the seed bank is linked to soil temperature, but also duration of heating (Gagnon et al., 2015; Keeley, 2009; Maia et al., 2012). Temperatures between 40 to 70 C° are lethal to most plant tissues, but seeds can endure larger temperatures, even above 100 C° (Zouhar, 2008). The duration of soil heating determines the changes in chemical- and physical properties of soils, especially in nitrogen and soil organic matter content. The availability of soil nutrients after the fire in turn affects vegetation regrowth.

When assessment of fire severity is mainly focusing on changes in soil properties such as loss of organic matter and related phenomena also the term soil burn severity can be used (Keeley, 2009). Soil burn severity refers to the effect of a fire on ground surface characteristics such as loss of organic matter, char depth, altered structure of the soil, and reduced infiltration (Parson et al., 2010). Burned areas consist of a wide variability of small scale patches with different degrees of soil burn severity which influences among others post-fire hydrology, erosion and vegetation regrowth (Vega et al., 2013). In general soil burn severity is expected to be higher when the amount of ground fuels, often linked to vegetation density, is higher. Estimation of soil burn severity in the field can be done by looking at ash colours and depth, ground cover, soil structure, roots and soil water repellency. Based on this kind of indicators, the soil burn severity can be classified in five levels (Fig. 3). In case of low soil burn severity, organic layers at the surface are not completely consumed, and roots are not affected. Also, soil structure is not changed as result of the fire. When exposed, the mineral soil has often a brown or black colour (Parson et al., 2010; Vega et al., 2013). Because a low severity fire has little effect on buried plants parts, it can stimulate post-fire resprouting. With increasing soil burn severity the structure of the soil remains the same, but some fine roots might be scorched. Very high degrees of soil burn severity can lead to the loss of structural aggregate stability as consequence of combustion of organic material in the surface soil horizon. Regenerative structures (e.g. rhizomes) in the duff layer, or at the surface of the mineral soil, might be damaged during a severe burn (Ansley et al., 2000; Parson et al., 2010).

SBS levels	Forest floor (Oi+Oe+Oa)	Mineral soil (Ah horizon)
0	No evidence of fire	No evidence of fire
1	Oa layer (lower duff) partially or totally intact.	Undisturbed
2	Oa layer totally charred and covering mineral soil. There may be ash.	Undisturbed
3	Forest floor completely consumed (bare soil). There may be ash.	Undisturbed. Soil structure unaffected. SOM not consumed. Surface fine roots not burned
4	Forest floor completely consumed (bare soil). There is no charred residue. Thick layer of ash.	Soil structure affected. SOM consumed in the upper layer. Surface soil colour altered (grey). Surface fine roots burned
5	Forest floor completely consumed (bare soil). There is no charred residue.	Soil structure affected. SOM consumed in the upper layer. Surface soil colour altered (reddish). Surface fine roots burned

Figure 3: Classification of soil burn severity according to Vega et al. (2013).

2.3.2 Influence of fire frequency

An increase in fire frequency is often related to a decrease in vegetation cover and plant density. However, the effect of recurrent fires on species richness is not completely clear, and both negative and positive effects have been observed dependent on the fire history (Tessler., 2016a). Vegetation recovery is slower after repeated fires compared to single fire events. When intervals between fire events are short, the persistence of a particular woody species, is affected (Hosseini et al., 2016; Tessler et al., 2016a). Plant species require a particular time length to replenish their regeneration capacity. Changes in vegetation composition may occur when a fire took place before maturity of the plant. For example the obligate seeder Pinus Pinaster tree, a characteristic species in the study area, can handle single fire events relatively well but cannot regenerate within short fire intervals because production of sufficient canopy seed banks will take around 15 years. As consequence of this, a shift from trees towards shrubs is observed in Mediterranean ecosystems affected by high fire recurrence (Malkisnon et al., 2011; Mayor et al., 2016; Tessler et al., 2016b; Vallejo et al,. 2012). In contrast to tree communities, the recovery of shrub cover and the return of shrub taxons present before fire take between 1 and 3 years after a fire event (Grangred et al., 2011). Furthermore plant recovery after fire is also dependent on several other factors as water availability, soil loss, the presence of steep slopes, and edaphic conditions (Malkisnon et al., 2011; Mayor et al., 2007).

2.4 Fire regimes and invasive species

For improving fire management towards more resilient fire-prone ecosystems it is from big importance to increase the knowledge concerning the relationship between fires, plant communities and invasion by invasive species (Mandle et al., 2011; Moreira et al., 2013). When invasive species are becoming more abundant, a shift in fuel properties of a plant community can occur and thus altering the fire regime. Fuel properties such as ignitability and the amount of moisture in the plant tissue are factors influencing fire frequency and -intensity. In Mediterranean ecosystems invasive species can influence vegetation structure and fire regimes in a different way (Brooks et al,. 2004; Mandle et al., 2011; van Wilgen et al., 2010).

For some invasive species it is for example known that they increase fire frequency and/ or intensity due to an increase in fuel load and the production of highly flammable material. Exotic grasses for example increase fire frequency by the quick production of fuel (Mandle et al., 2010; van Wilgen et al., 2010). Many plants from the genus Eucalyptus contain flammable oils and thus also increase fire frequency. In this way a positive feedback is created in which fire support occurrence of the invasive species, but the species itself also promotes fire (Mandle et al., 2010; Wyse et al., 2018).

However, other invasive species supress aspects of the fire regime when plant tissues retain a lot of moisture or when woody species out-compete grasses and forbs (e.g. shadowing, competition for water and nutrients). In the last case ground fuel is reduced resulting in a shift from frequent low-intensity ground fires to less frequent but more intense crown fires (Mandle et al., 2011). The woody shrub invader *Hakea sericea*, which is the main focus of this study, is believed to reduce fire intensity and spread by the production of high packed fuels which exclude oxygen when burning (Mandle et al., 2011; van Wilgen & Richardson, 1985). On the other hand, fuel load can strongly increase as consequence of an invasion by this species and under extreme dry and warm weather conditions this leads to increased fire intensity (van Wilgen & Richardson, 1985). The influence of *Hakea sericea* on the fire regime is thus not completely clear, and contains both fire suppressing and - intensifying properties. But it is certain that its abundance is promoted by fire.

2.5 Hakea invasion in Portugal

Hakea sericea is introduced in Portugal as hedge plant around 1930. It is a woody shrub originating from south-eastern Australia and invasive in New Zealand, South Africa, and in some countries of the Mediterranean Basin such as France and Portugal and currently spreading to Spain. It mainly invaded the northern part of Portugal (Fig. 4), probably triggered by the high amount of fires occurring in this region. According to observations in Portugal it seems that the plant prefers schistose bedrock. It is often found on phosphorus-poor soils with open shrub vegetation (Alvarex-Taboada et al., 2017; Martins et al., 2016). By developing proteoid roots it is able to cope very well with nutrient-poor soils. In Australia, it grows in dry sclerophyll forests and heaths. Originally, *Hakea sericea* performs well in climates characterized by warm temperatures and humid climates with a warm summer. The he plant is also able to grow in drier climates and is known as drought resistance (EPPO, 2017).

Hakea sericea is a serotinous species meaning that it releases seeds in response to environmental triggers such as fire. It belongs to the Proteaceae family. The shrub is around 2-4 meter tall with sharp needle-like leaves (Fig 5). It alters the vegetation structure of the native plant community by the formation of dense thickets (Richardson et al., 1987). Seeds of mature plants are retained in woody follicles (fruits) that protects them against heat, making them very resistant to fire (Fig. 6). Each follicle contains two single-winged seeds. Compared to the above-ground dry mass *Hakea sericea* produces relatively a high number of follicles. Seeds are viable for several years. The shrub becomes mature after approximately 2 years, being able to produce viable seeds. It is also able to regenerate from rootstocks after fire, but this is not very often seen in the field. Furthermore, the species is known for its large canopy-stored seed bank which enlarges over time during the lifetime of the plant. Seeds are mainly released after the death of the plant, e.g. due to fire. This explains why a large amount of seeds can be released after a fire event enhancing establishment of this invasive species in fire-prone areas. Next to high seed production and fire resistance, the seeds also have the ability to disperse over a long distance by wind supporting a quick spread of the species (EPPO, 2017; Martins et al., 2016; Richardson et al.,1987).

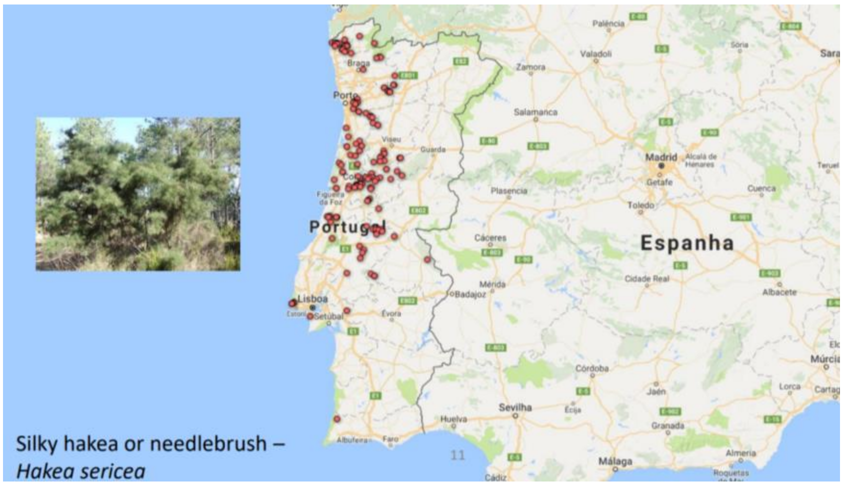


Figure 4: Abundance of Hakea sericea (Invasoras, 2019).

Figure 5: Needles and flower of Hakea sericea (photo by Asjra Bosch).

Figure 6: Wooden follicles of Hakea sericea after fire (photo by Asjra Bosch).

2.6 Fire as management tool

Fire plays an important role in Mediterranean ecosystems by removing above-ground biomass and creating opportunities for regeneration and coexistence of vegetation. Reducing the unwanted effects of wildfires (e.g. damage to crops, livestock and humans) managers of fire-prone systems still need to ensure that fire can keep its crucial role in ecosystem functioning. This is often done by the use of man-made and controlled fires such as prescribed fires (Alcañiz et al., 2018; Vallejo et al., 2012; van Wilgen et al., 2010).

A prescribed fire is a planned, and often a low-intensity fire to achieve a certain management goal. A wide variety of objectives can be pursued, dependent on the local context, weather conditions, fuel characteristics and other topographic factors. Prescribed burning is usually used to decrease wildfire risk by decreasing fuel loads. In this way high-intensity wildfires which often have a much stronger impact on soils and ecosystem degradation are prevented. Secondly, prescribed fires can also be used to support biodiversity and for the regeneration of particular plant species. Third, prescribed fires are used for the transformation of shrublands into grassland for grazing (DiTomaso et al., 2006; Alcañiz et al., 2018). Additionally, it has a fertilizing effect by increasing the soil nutrient availability which is beneficial for agricultural purposes (Alcañiz et al., 2018). Finally, in fire management prescribed fires are in some cases also used to control invasive species. This last application will be further elaborated in the following paragraph (DiTomaso et al., 2006; Mandle et al., 2011; van Wilgen et al., 2010). When executed in a proper way prescribed fires are mainly beneficial to fire-dependent ecosystems, but could have also negative consequences to soils and vegetation when fire severity or intensity is too high (Alcañiz et al., 2018).

Prescribed fires can control or prevent re-establishment of invasive species by depleting the soil seed bank and destroying seedlings, in particular when invasive species are more likely to supress aspects of the fire regime. However, research to the role of prescribed fire as management tool and its characteristics such as frequency and intensity in controlling invasive species is scarce, especially for woody invaders such as *Hakea sericea*. Also, the effects of prescribed fires are very dependent on their location (Mandle et al., 2011). Research to woody species as *Hakea sericea* and its relationship with fire management has mainly carried out in South-African Fynbos (Breytenbach,1989; Esler et al., 2010; Fugler, 1983; Holmes et al., 2000; van Wilgen & Richardson, 1985). The above-mentioned studies conclude that prescribed fire, in the form of slash and burn (also called ‘fell and burn’), is effective in controlling *Hakea sericea* invaded in Fynbos ecosystems. Before burning adult plants are cut down and left for 12-18 months. As well, biological control by insects (e.g. hakea seed weevil) or fungus are applied, but an integrative method combining multiple approaches is recommended. However, in this region there is a limited opportunity to carry out prescribed burns safely and increased fire intensity linked to this technique is not always beneficial for conservation of Fynbos vegetation. Incorporating the effect of fire on the whole ecosystem, rather than only focusing on control of invasive species is thus very important.

3. Methods

3.1 Study area

The study area is located in Talhadas (40°39’58.97’’N, 8°21’53.86’’W), close to the city of Aveiro in Northern Portugal (Fig. 7). The climate type in the study area according to the Köppen classification is Csb, with moderately warm but dry summers, and a mild and wet winter. Annual mean precipitation ranges between 1600 and 2000 mm, with a mean annual temperature around 14 ° C. The minimum temperature is around 9 ° C (IPMA, 2020). Soils in these region are classified as humic cambisols with underlying rocks of schist (ESDAC, 2019). Common vegetation in this region consists of pine (*Pinus Pinaster)*, eucalypt trees, and shrubs such as *Pterospartum tridentatum, Cistus spp.*, and species of the Ericaceae family. Due to a high fire frequency (2 fires in the last 30 years), there are barely trees left and the study area is mainly covered by *Hakea sericea*.

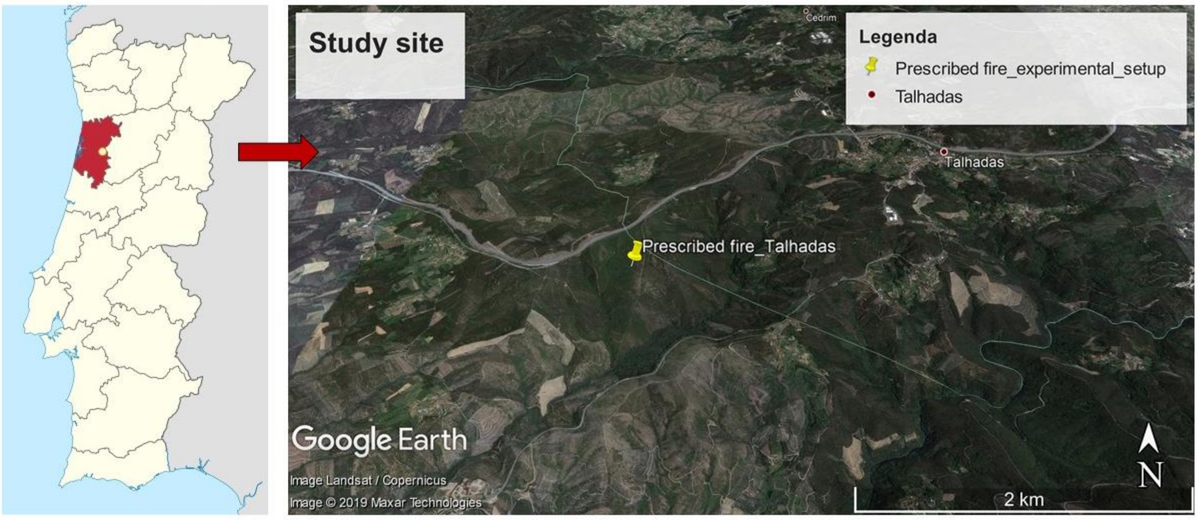


Figure 7: Study area in the district of Aveiro, Portugal.

3.2 Data collection & experimental setup

The ESPteam at the University of Aveiro carried out an experimental fire in May 2019 (Fig. 8), in locations that had been previously burnt by two wildfires (in 1991 and 2013) since 1975 (ICNF, 2013). Cover by *Hakea sericea*, was very dense, as shown by figure 8. Three treatments were applied to 5x2 m plots (including 3 replicates per treatment): no burning (control, C), burning (B) and slash and burn (SB). Each plot is divided into quadrats of 1 m² (Fig. 9), and surrounded with silt fences. In these plots, fire characteristics (e.g. soil burn severity, temperature and duration of the fire), soil surface cover, abundance of *Hakea sericea*, characteristics of the plant community (height, cover, diversity), together with several soil-related properties are examined as explained in the next paragraphs. The study area was revisited every 1-2 months.

Figure 8: Experimental fire in May 2019. (photo by Oscar Gonzalez Pelayo).

3.2.1. Fire characteristics

After the experimental burning the following aspects related to fire characteristics were analysed; (i) the maximum temperature reached during the fire (°C), (ii) the duration above several temperature threshold (s), and (iii) soil burn severity. To record the variation in temperature over time, thermocouples (type K) deployed with dataloggers were installed in the litter, duff and soil layer. The litter layer is (Oi horizon) is consisting of freshly fallen leaves and branches. The duff layer (Oe) consist of partially decomposed material and the Oa layer contains well decomposed material. However, in shrublands distinction between these layers is more difficult than in forest floors, especially in absence of continuous woody cover (Elango, 2011). Because the duff layer was very thin or absent in our study area measurement took place at the top of the litter layer, at the border between the litter layer and soil layer (which thus refers to the duff layer), and at the surface of the soil layer. Prior to the fire, the litter layer in the plots was about 3 cm deep on average. The temperature of the fire was recorded every 5 seconds. Temperatures were recorded at one location per plot. The maximum temperature is the highest value which is recorded in each layer by the thermocouple during burning. The duration above a certain temperature threshold consists of the cumulative time in seconds in which the temperature is higher or equal to the threshold. In total 3 temperature thresholds were used, namely 60 °C, 100 °C, and 300 °C. The first threshold of 60 °C is chosen, because from this point fire might be seriously damaging to plant tissues (Cardoso et al., 2018). Heating above 100 °C alters biological components in the soil. At the threshold of 300 °C, organic matter in the soil is already partly combusted and higher temperatures affect soil texture and composition of soil minerals (Campbell et al., 1995; Terefe et al., 2008). Data of soil burn severity was collected according to the method described in Vega et al. (2013). Determination of the soil burn severity included the examination of the ash colour, ash depth, ash cover, the combustion of the litter layer, burning of understory vegetation, the rate of canopy damage, and flame height in relation to the tree stems. Soil burn severity was determined in the middle, top and bottom of each plot.

3.2.2. Vegetation and soil surface cover assessment

Soil surface cover was measured using 3 randomly selected quadrats located in the corners and in the middle of the plot covering both the left and right side of the plot. Soil surface cover in the quadrats was measured in May 2019, September 2019, December 2019, January 2020, and February 2020. This was done by making pictures of each quadrat and applying a grid corresponding to 1 x 1m, with interception points at 10 cm, comparable to Prats et al. (2019). The following classification for soil surface cover was applied at each contact point of the grid: Stone (bedrock and fragments bigger than 2 mm), bare soil, ashes, litter, vegetation, and biological soil crust. Litter refers to fresh plant material or other material as very slightly scorched branches and leaves which fell to the ground after the fire. Then, the average cover of the quadrats in each plot was calculated and used to determine average cover in the slash and burn and burn treatments.

Vegetation diversity and *Hakea sericea* cover was measured in 3 subplots of 50 x 50 cm selected in the top, bottom and middle of each plot. All seedlings in these subplots were counted and identified in January, February, and May 2020. When plants couldn't be identified on species level, genus or family level were assigned. Relative abundance is based on the absolute abundance of a particular species compared to the total absolute amount of species found in a particular plot and summed up for each treatment. The absolute amounts were converted into a seedling density per m² from which the average density per plot was taken to calculate average density over the whole treatment.

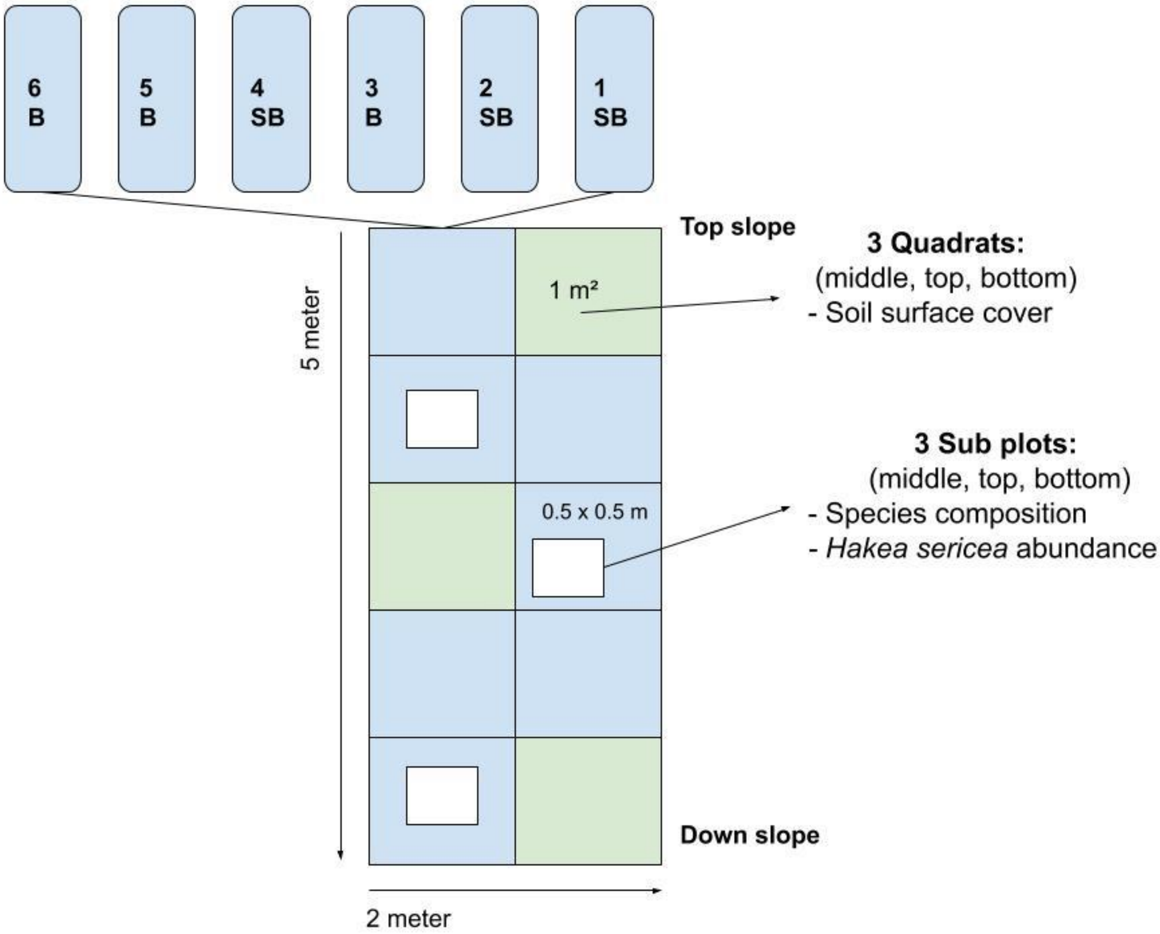


Figure 9: Experimental setup of study plots. Slash and burn plots are indicated with SB, burned plots with B. Picture is not on scale.

3.3 Data analysis

In order to test normality and homoscedasticity of the data the Shapiro-Wilk test and the Levene's test is used. Data concerning the fire characteristics met the requirements, so a T-test was used (table 1). Other data did not met the requirements so a Mann-Whitney U test was used (table 1). To see if there is a significant difference in species relative abundance between treatments, only the most dominant plant species and *Hakea sericea* were tested.

Variable	Statistical test
Temperature exceedance above threshold of 60 °C (s)	T-test
Temperature exceedance above threshold of 100 °C (s)	T-test
Temperature exceedance above threshold of 300 °C (s)	T-test
Mean temperature above 100 °C (°C)	T-test
Mean temperature above 300 °C (°C)	T-test
Maximum temperature (°C)	T-test
Soil burn severity (range between 0-5)	Mann-Whitney U
Species relative abundance	Mann-Whitney U

Table 1: Variables and executed statistical test.

4. Results

4.1 Fire characteristics

4.1.1. Fire behaviour and fuel characteristics

During the burn, high temperatures were reached in all layers (Fig. 10). In general, there are minor differences between the treatments. Logically, temperatures were in general higher in the litter layer and lowest in the soil layer. Some errors were visible in the recording data of one of the thermocouples under the slash and burn treatment (measurement 2, duff layer). In the duff layer, the thermocouple recorded some values below 0 °C. After the peak, the temperature curve of the duff layer shows a big variability. Also starting temperatures were too high, namely around 80 °C. For this reason, values below 0 °C were not taken into account in the analysis of the fire temperatures. Furthermore, the start temperature of the curve was set on 18 °C, based on the starting temperatures of the other measurements.

In certain cases the average temperature above the threshold of 100 °C or 300 °C was higher in the duff layer than the litter layer, for example measurement 3 of the slash and burn treatment (Figure 11a). The thin litter layer in both treatments offered little protection during burning, which explained these high temperatures of the duff layer and heat transfer to the underlaying soil layer. Both mean temperatures and peak temperatures did not differ between treatments. The mean temperature above 100 °C was slightly higher for the slash and burn treatment (Appendix A), but not significantly (p = 0.702). The same applies to the mean temperature above threshold of 300 °C with a not significant difference between the treatments (p = 0.739). In 2 plots of the burn treatment and 1 plot of the slash and burn treatment, the thermocouples didn't record temperatures above 300 °C (Fig. 11b).The mean peak temperature was lower in the burn treatment (Appendix A), but also in this case not significant (p = 0.664).

Concerning the exceedance of different thresholds, some trends are found, but not significantly different. The mean duration of exceeding the threshold of 60 °C including all layers was higher for the slash and burn treatment, with 2026 seconds compared to 1406 seconds (Appendix A), still the difference was not statistically significant (p = 0.1171, t = 1.432). The mean duration of exceedance of 100 °C was 1318 seconds for the slash and burn treatment and 1020 seconds for the burn treatment (Appendix A), but also not significant different (p = 0.400). Likewise, there was not a significant difference for the duration of exceeding the threshold of 300 °C with a p-value of 0.968. Exceedance of 100 °C is comparable in the same layers between and within treatments, with exception of the duff layer in measurement 2 reaching 3480 seconds (Fig. 12b).

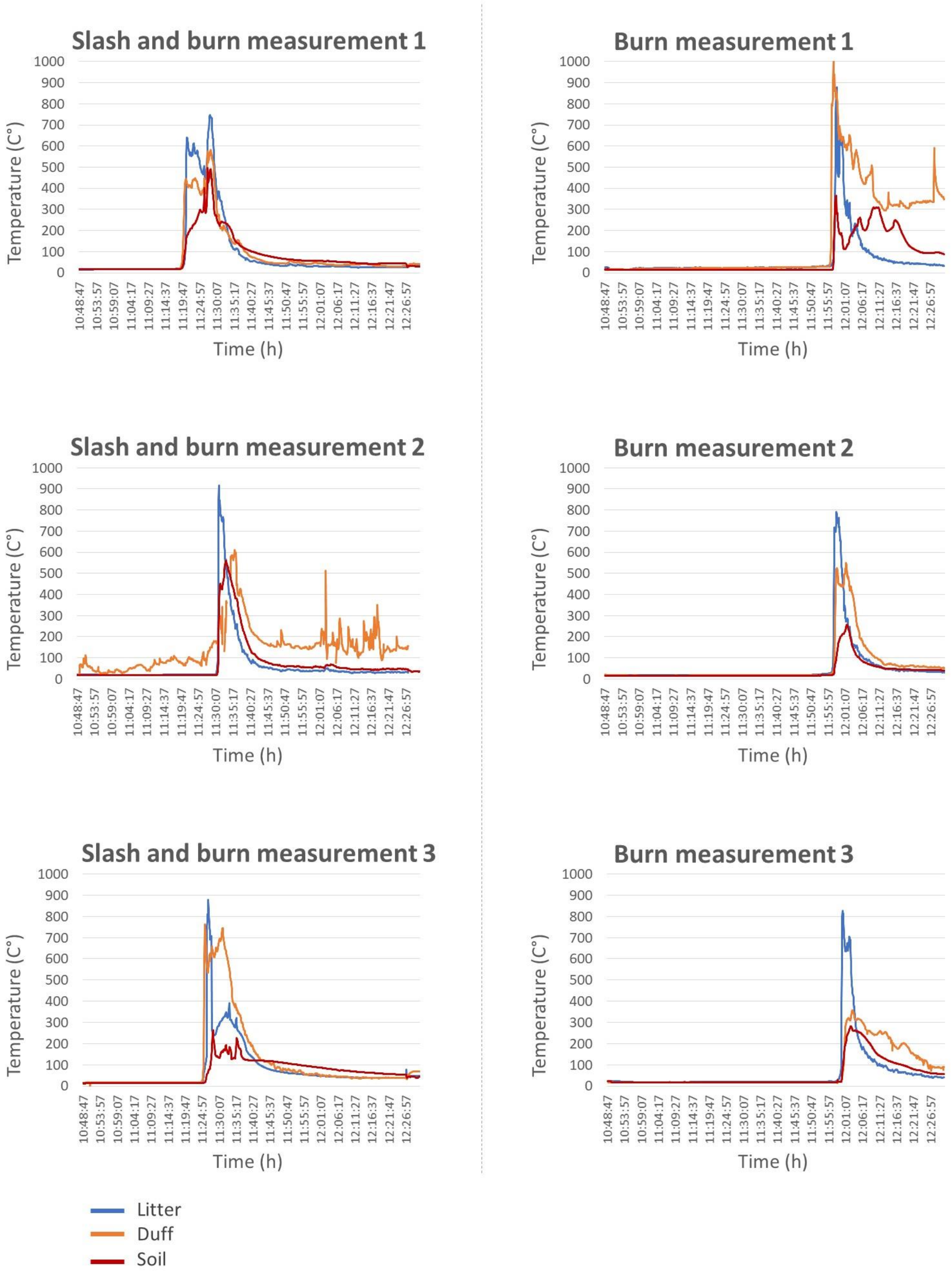


Figure 10: Fire temperatures over time. Slash and burn measurement 1, 2 and 3 belongs to plot 1, 2 and 4 respectively. Burn measurement 1, 2 and 3 belongs to plot 3, 5 and 6 respectively. Times are recorded in the UTC+0 time zone.

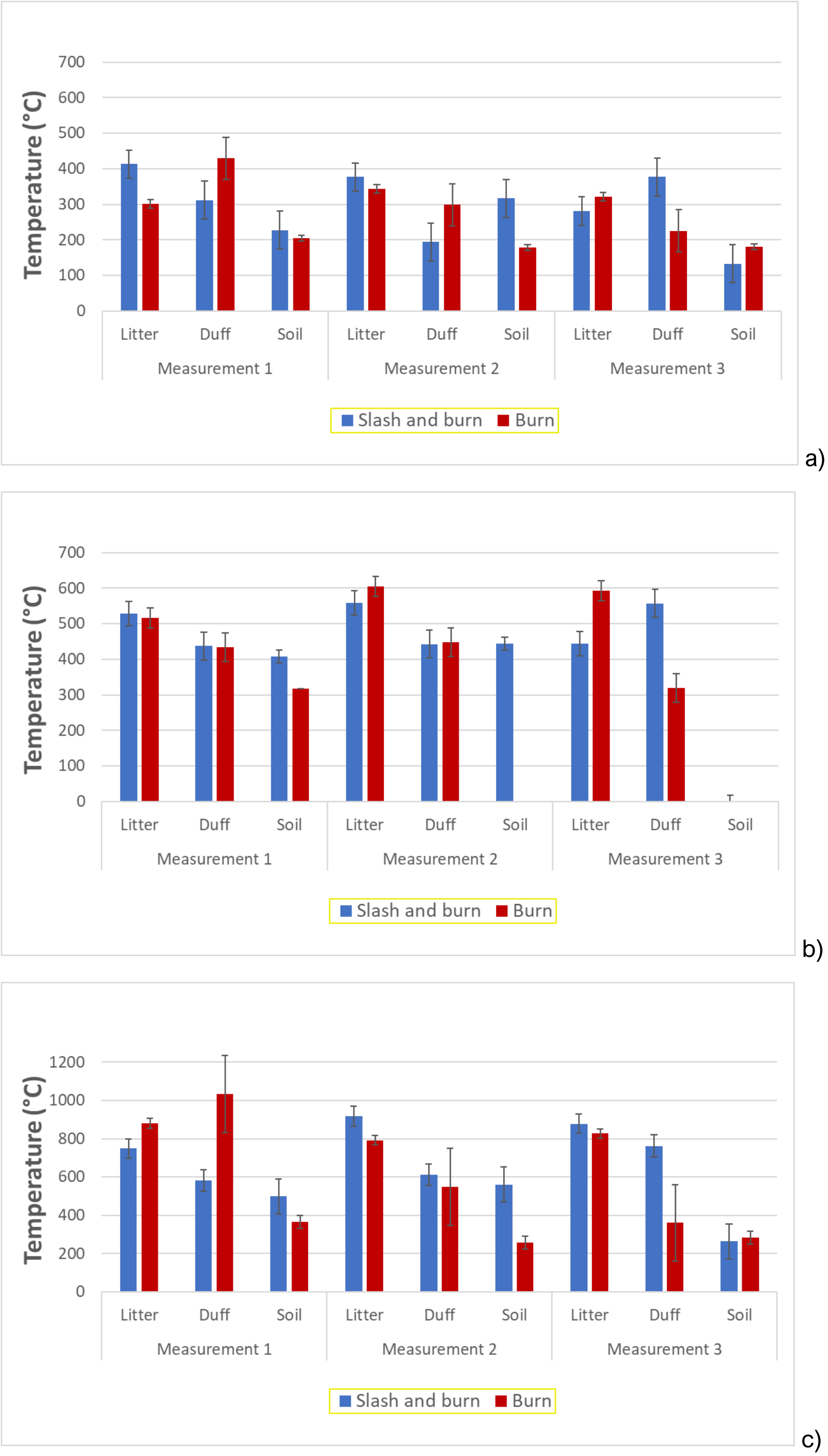
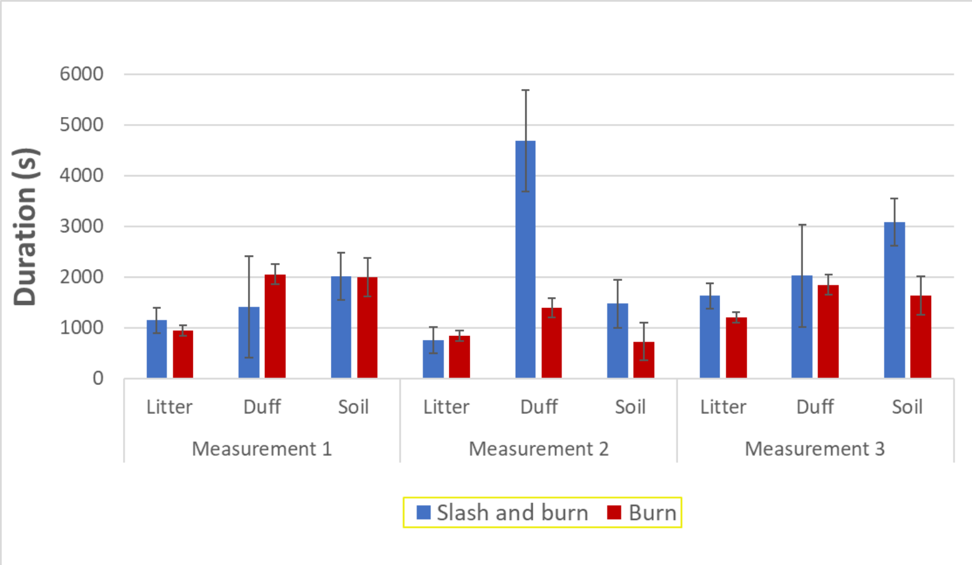


Figure 11: Average temperatures above thresholds (100 °C (a) and 300 °C (b)), and peak temperature (c). Bars represent the standard error (SE).



a)

b)

c)

Figure 12: Duration in seconds above several temperature thresholds; 60 °C (a), 100 °C (b), and 300 °C (c) for both treatments. Bars represent the standard error (SE).

Overall, the soil burn severity was found to be significantly higher in the burned plots, with a *p*-value of 0.011 (Fig. 13). The severity class in the slash and burned plots varied between 1 to 3. In the burn treatment the severity class ranged from 2 to the most severe class of 5. Especially in the burned plots 3 and 5, soil burn severity was higher with a median of 3. Plots of the slash and burn treatment had a median soil burn severity of 1 or 2 (Appendix B).

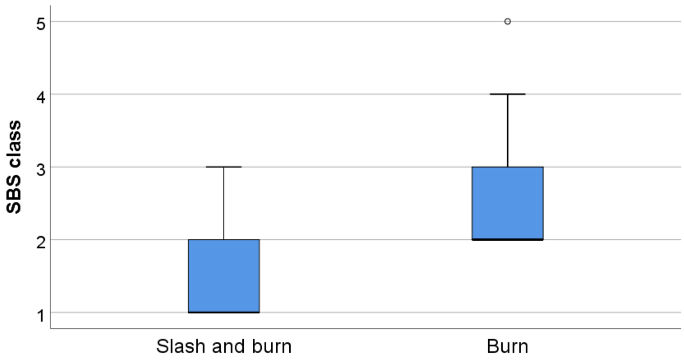


Figure 13: Soil burn severity per treatment.

4.3 Effect of fire treatments on soil surface cover

The temporal changes in soil surface cover during 9 months after the experimental fire are shown in Figure 13 and 14. In general, there were no differences between treatments in the surface cover of ash, stones, bare soil or vegetation during the study period. Shortly after the fire, the soil surface cover consisted mainly of ashes (including charred material) and stones (Fig. 14). While the ash cover was decreasing over time, the cover by stones increased since some were initially covered by ash and could not be seen in the first campaigns. The ash cover was similar between the two treatments one day after the fire in May 2019 and accounted for around 80 % of the total surface. Ash was reduced to approximately 30% in both treatments 9 months after the fire. Soil surface cover by stones was also showing a similar trend in both treatments, with an increase over time to a probably stable cover between 30-40 %. After the fire, the amount of bare soil was slightly higher in the burn treatment, but not significantly (Fig 15). It decreased over time, until a value of 0% is reached in January (8 months after the fire) in both treatments. Cover by litter ranged from 1 to 6% in both treatments, slightly increasing over time. Directly after the fire, vegetation cover was 0% in both treatments and showed a similar increase over time. It shows slightly higher values in the slash and burn treatment compared to the burn treatment from the second campaign onwards and reached a cover of 20% and 13% respectively. This difference was however not significant, as shown by the overlapping error bars in figure 15. The presence of biological soil crust (BSC), mainly consisting of mosses, was also comparable in both treatments, and reached around 10% in the last campaign. Based on visual observations, patches with bare soil are more abundant in the burned plots, while vegetation seems to be more abundant in the slash and burned plots.

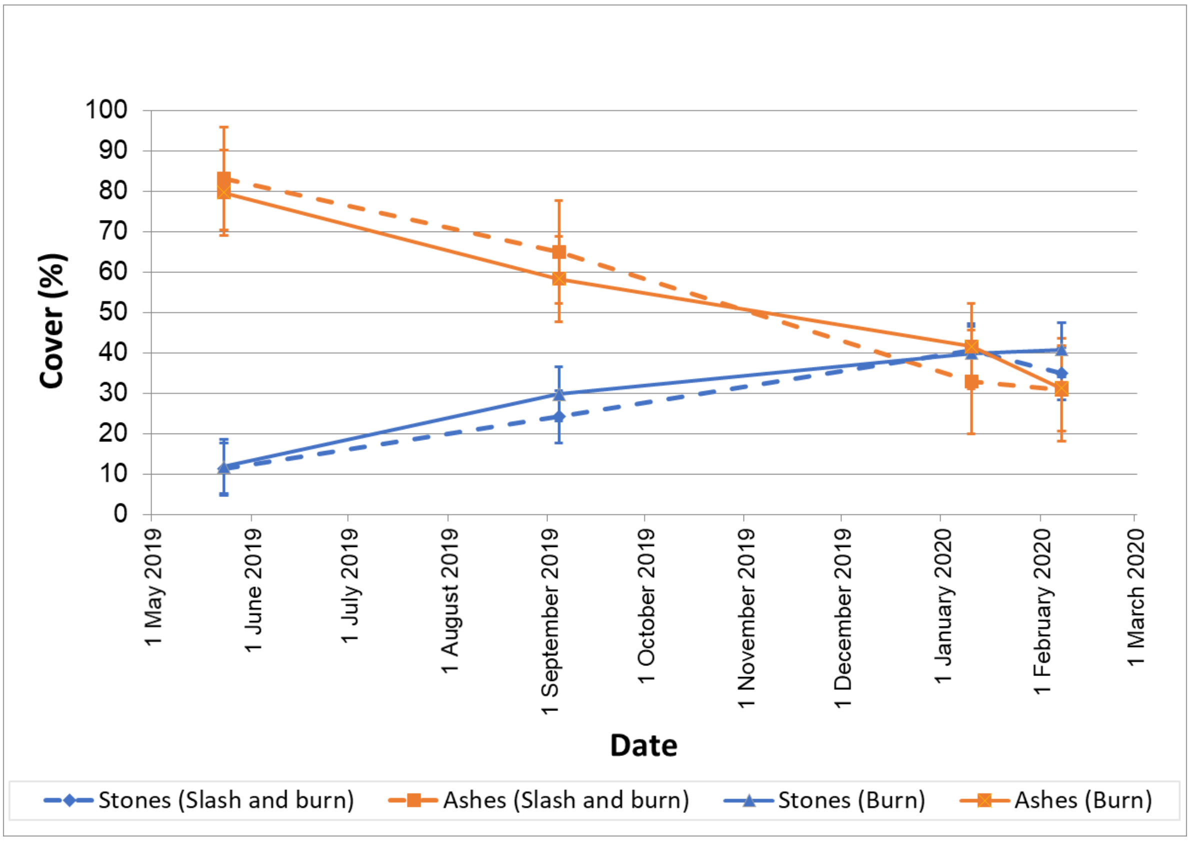


Figure 14: Soil surface cover evolution of stones and ash in both treatments including the standard error (SE).

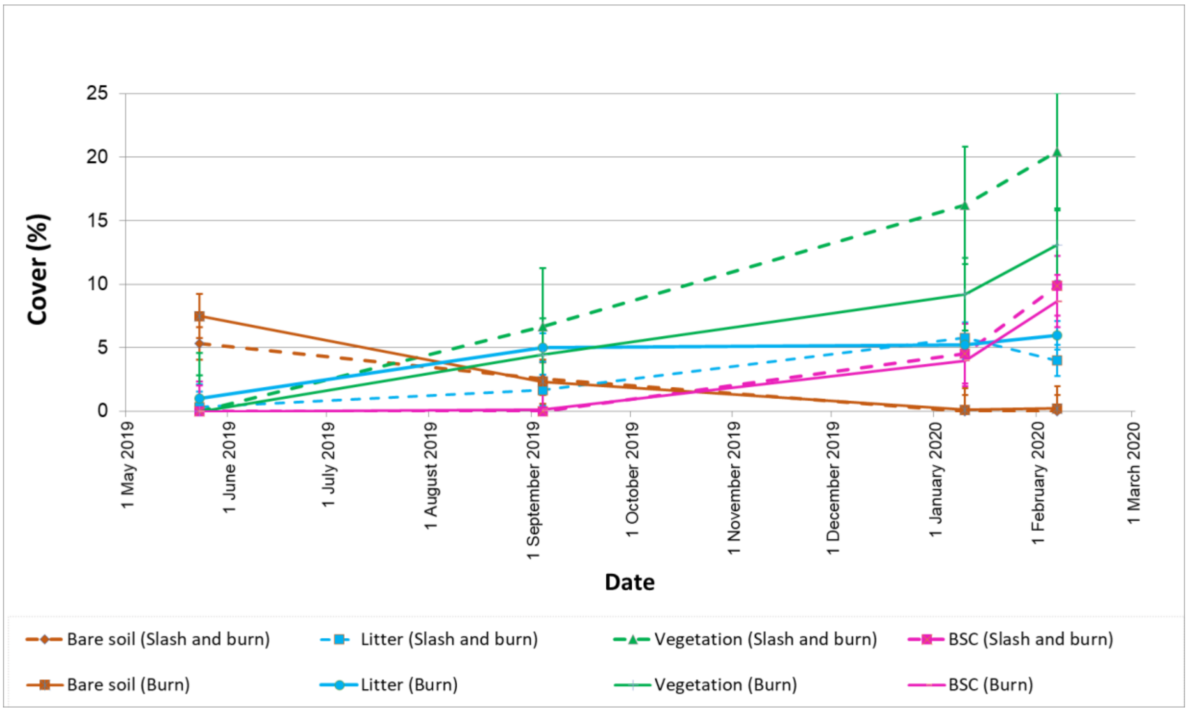


Figure 15: Soil surface cover evolution of bare soil, litter, vegetation and biological soil crust (BSC) in both treatments including the standard error (SE).

4.4 Vegetation composition & abundance of Hakea sericea

4.4.1. Most abundant species

During autumn vegetation started to regrowth, and within one year after the prescribed fire, typical Mediterranean species as *Pterospartum* tridentatum, *Erica spp.* and *Cistus spp.* appeared as main vegetation in the plots. The relative abundance of the counted vegetation in the 8th, 9th and 12th month after the prescribed fire are shown in figure 16 and 17. In the first counts in January, seedlings of the genus *Cistus* were most abundant in both treatments, followed by a variety of bulbous plants (geophytes). Relative abundance of *Cistus spp*. remains quite constant in the slash and burn treatment over the measured period. From al recorded plants genus or families, *Cistus spp.* was the only one from which the relative abundance was significantly higher in the slash and burn treatment over the whole measuring period. The *p*-value was 0.047 for January, and 0.024 for February and May (Table 3). In the slash and burn treatment species of the genus *Cistus* made up to 64% of the plant community, and around 40% in the burn treatment. This corresponds to an average seedling density of 275 seedlings per m² in the slash and burn treatment, and 59 in the burn treatment (Table 2).

The share of the geophytes in the vegetation composition was more equally between the treatments, compared to *Cistus spp.* Over time, the relative abundance of geophytes decreased in both treatments with 12.3% (slash and burn) and 15.1% (burn). In the beginning, relative abundance was higher in the burned plots, but this changed towards a higher relative abundance in the slash and burn treatment in the last campaign of May. Still, these differences were not significant (Table 3). Although the burned plots had a higher relative abundance, the average seedling densities were lower over all campaigns (Table 2).

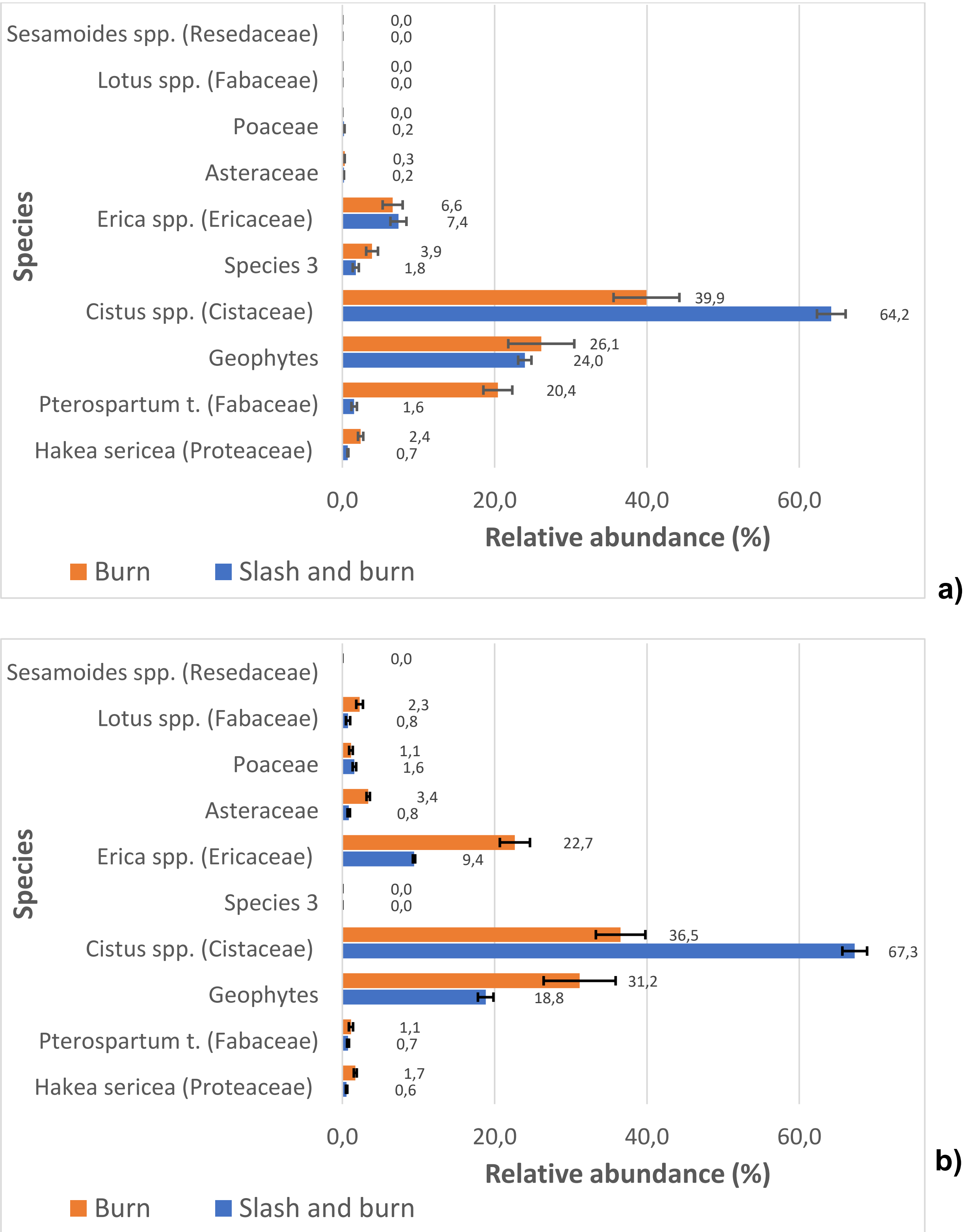
The third most abundant species found under the slash and burn treatment belonged to the genus *Erica*, where it remained one of the most abundant species 1 year after the prescribed fire. The relative abundance increased from 7.4% in January until 11.2% in May, corresponding to 32 and 40 seedlings per m² respectively. The increase of relative abundance of the genus *Erica* was larger in the burn treatment, in which it increased from 6.6% towards 33.5 % (Fig. 16). Also a strong increase in seedling density was found, from 10 to 35 seedlings per m².

After 8 months after the prescribed fire, relative abundance of *Pterospartum tridentatum* (20.4%) was significantly higher in the burned plot (20.4%) compared to the slash and burned plots (1.6%), with a *p*-value of 0.000. The relative abundance of *Pterospartum tridentatum* includes both seedlings and resprouts, but the seedlings accounted for the largest part with more than 20%. Over time relative abundance becomes more equal between treatments (Fig.16c). It increased in total with 6% in the slash and burn treatment, and decreased with 10.5% in the burn treatment. Though, this was not the case for seedling density (Table 4).

4.4.2 Hakea sericea and remaining species

The relative abundance of *Hakea sericea* seedlings was quite low with a range of 0.4 - 0.7% in the slash and burn treatment and a range of 0.8 - 2.4% in the burn treatment. *Hakea sericea* seedling density varied between 1 and 4 seedlings per m² in both treatments (Table 4). Relative abundance of *Hakea sericea* (Fig. 15), was higher in the burned plots, although the difference was not significant (January: p = 0.149, February: p = 0.331, May: p = 0.634). Overall, there was a decreasing trend of *Hakea sericea* density over time (Fig. 16). First, the seedling density was higher in the burned plots, but this was the other way around in May (Table 4).

Furthermore, some other species appeared, but they were in general very low in abundance (<6%) and did not differ in relative abundance between treatments. This includes for example species of the family of the Asteraceae, Fabaceae and Poaceae. Species of the genus *Senico*, probably *Senico* *lividus* (Asteraceae), and Boraginaceae (*Glandora prostrata*) were present in the slash and burn treatment. Asteraceae had a higher seedling density in the slash and burn treatment than in the burn treatment, although their relative abundance was higher in the burned plots (Fig. 16). Species seedling density of the grasses (Poaceae) and Fabaceae, including *Pterospartum tridentatum* and probably *Lotus spp.* was also higher in the slash and burn treatment. However, it need to be noted that identification of *Sesamoides spp*, and *Lotus spp.* was uncertain, and will become clear in a later stage. The total amount of seedlings is higher in the slash and burn treatment than the burn treatment, with respectively 428 and 153 in January, 467 and 157 in February, and 353 and 102 in May.



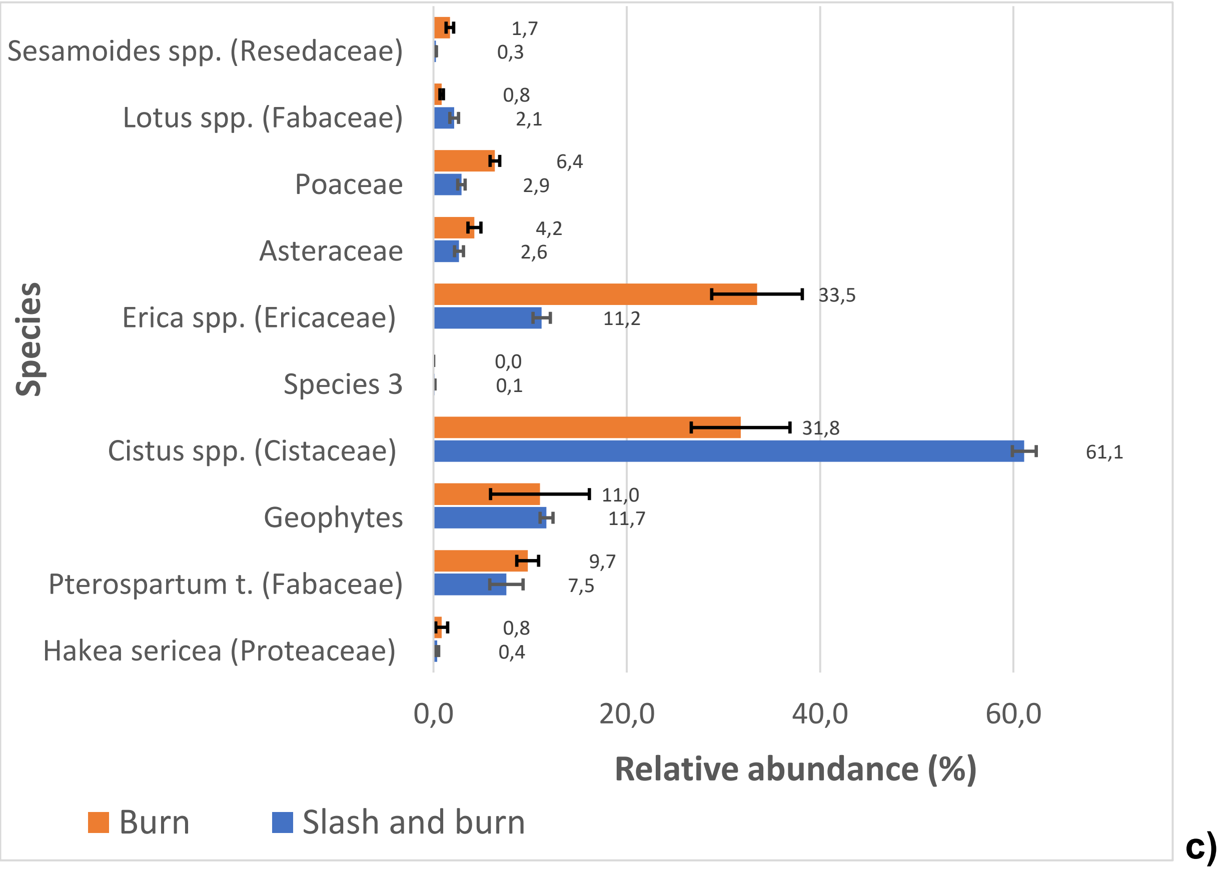


Figure 16: Relative abundance over 3 campaigns (January (a), February (b) and May (c) in both treatments.

Species	**Density (m²)**					
	January 10-01-2020		February 07-02-2020		May 08-05-2020	
	Slash and burn	**Burn**	**Slash and burn**	**Burn**	**Slash and burn**	**Burn**
Hakea sericea	3.1	3.6	2.7	2.7	1.3	0.9
***Pterospartum t.* (Fabaceae)**	5.3	30.2	1.3	0.9	25.3	9.3
Geophytes	102.7	38.7	79.7	48.9	41.3	9.3
Cistus spp.	275.1	59.1	320.9	57.3	216.0	33.3
Species 3	7.6	5.8	0.0	0.0	0.4	0.0
Erica spp.	31.6	9.8	44.9	35.6	39.6	35.1
Asteraceae	0.9	0.4	4.0	5.3	9.3	4.4
Pterospartum t. resprout	1.3	5.8	2.2	0.9	1.3	0.9
Poaceae	0.9	0.0	7.6	1.8	10.2	6.7
***Lotus spp.* (Fabaceae)**	0.0	0.0	3.6	3.6	7.6	0.9
***Sesamoides spp.* (Resedaceae)**	0.0	0.0	0.0	0.0	0.9	1.8
Total	**428.4**	**153.3**	**466.8**	**156.9**	**353.3**	**102.7**

Table 2: Species density over time in both treatments.

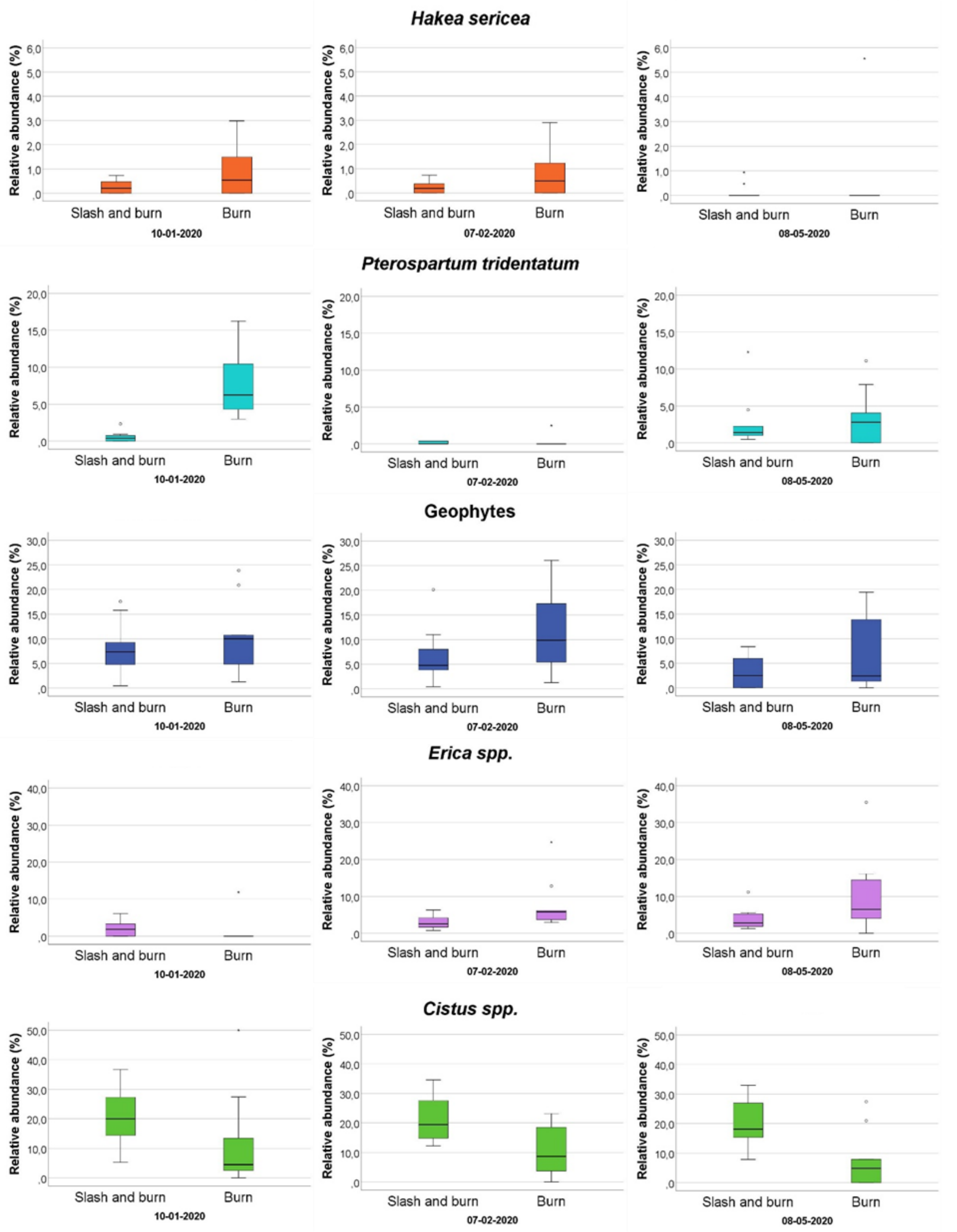


Figure 17: Relative abundance of most dominant species and Hakea sericea over time. Not the difference in scale axes between species.

Most abundant species	**Statistical significance**		
	January 10-01-2020	February 07-02-2020	May 08-05-2020
Hakea sericea	P = 0.149, U=24.5	P = 0.331, U = 30.0	P = 0.634, U = 37.0
Pterospartum t.	**P = 0.000**, U=0.0	P = 0.3611, U = 33.0	P = 0.757, U = 37.0
Geophytes	P = 0.566, U=34.0	P = 0.200, U = 26.0	P = 0.688, U = 36.0
Cistus spp.	**P = 0.047**, U=18.0	**P = 0.024**, U = 15.0	**P = 0.024**, U = 15.0
Erica spp.	P = 0.050, U=21.0	P = 0.058, U = 19.0	P = 0.102, U = 22.0

Table 3: Statistical difference between the both treatments in relative abundance. A Mann-whitney U test was performed, and significant results are indicated in bold.

5. Discussion

5.1 Fire temperatures and soil burn severity

5.1.1 Fire temperatures

The average and peak temperatures reached in the experimental fire are relative high compared to experimental fires described in other literature (e.g. Maia et al., 2016; Penman & Towerton, 2008; Sagra et al., 2017). In most cases, this kind of experimental fires have low fire intensity and severity, with temperatures reaching around 40 - 70 °C in the soil surface layer (2 - 5 cm depth). In this study, thermocouples recorded high temperatures (>300 °C) even in the soil surface layer, which are more comparable to experimental fires described by DeBano et al. (1997) and Soto et al. (1995). Prove of these high temperatures can also be observed in the field. White coloured ashes, particularly in the burned plots, indicated high temperatures with complete consumption of fuel and the litter layer. Locally, some patches of bare soil were present. This is in line with the data of the soil burn severity, which are higher in the burn treatment, especially at the locations with white ashes and bare soil. The higher soil burn severity found in the burned treatment might be related to the lower moisture content of the soil and litter. In particular, soil moisture content is known to have a big influence on temperature regulation in the soil during a fire (Badía et al., 2017; DeBano et al., 1979; Stoof et al., 2011). Litter- and soil moisture content were lower in the burn treatment, but moisture content of the dead fuel was lower in the slash and burn treatment (Table 4). Therefore, in contrast to the hypotheses, temperatures were not significantly higher in the slash-and-burn treatment, despite the presence of more fuel on the ground surface. This result was probably linked to the lower moisture content of the litter and soil surface layers in the slash and burn treatment.

Furthermore, the discrepancy between the lack of temperature differences between treatments but still a higher soil burn severity in the burn treatment may be explained by the fact that only one thermocouple was used to measure temperatures within plots. The use of thermocouples is a very sufficient way for the continuous recording of fire temperature and duration (Gimeno-Garcıa et al., 2004), but represents in this case only one particular location per plot. Based on the data of the soil burn severity data, there was spatial diversity between and within plots. So fire temperatures or duration could have been different locally. This spatial diversity can result in a local difference in soil damage, which in turn also influences vegetation appearance (Gimeno-Garcıa et al., 2004; Vega et al., 2013). Furthermore, one of the thermocouples (slash and burn, measurement 2) showed some errors, and recorded very high temperatures from the start. Although this was partially corrected, the installation setting was probably affected during the prescribed fire, causing incorrect measures of the temperatures in the duff layer. Furthermore, it needs to be keep in mind that the types of thermocouple used (e.g. type of metal or thickness) influence temperature measurement.

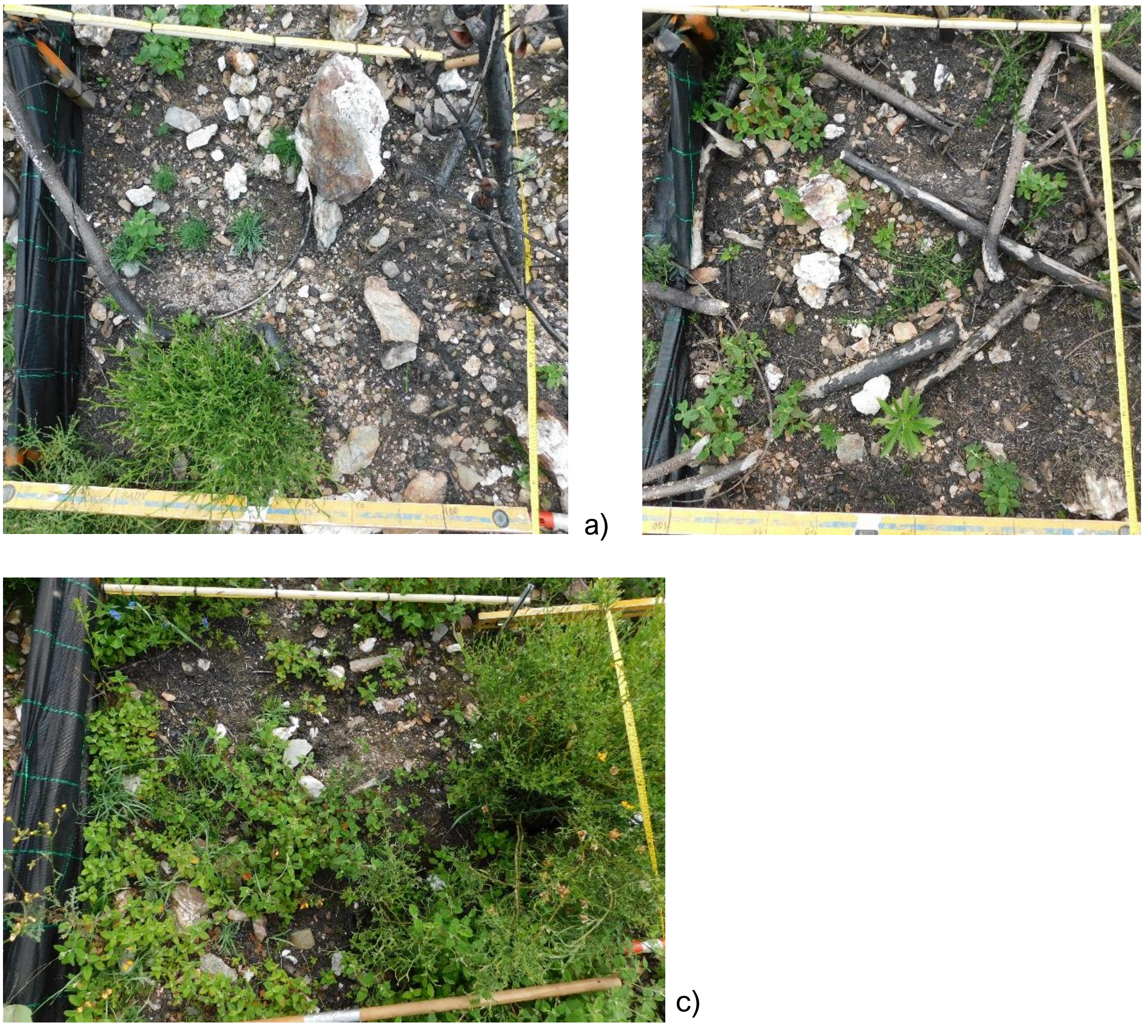
	Moisture content soil (%)	Moisture content litter (%)	Moisture content dead fuel (%)
Slash and burn	21	23	25
Burn	16	18	37

Table 4: Moisture content of the soil, litter and dead fuel (Data collected by the ESPTeam, CESAM).

5.2 Soil surface cover

The most important finding concerning the soil surface cover was the trend towards a higher vegetation cover in the slash and burn treatment, although differences were not yet significant in the last measurement included (February 2020, 9 months after the fire). However, based on this trend and recent observations from May, it can be expected that vegetation cover will be significantly higher in the slash and burn treatment 1 year after the fire (Fig.18). Patches of bare soil seemed to be more present in the burn treatment, according to visual observations, which may be partly explained by the higher soil burn severity found in this treatment (Parson et al., 2010).

The overall trend shown by both treatments was in accordance with other studies and the dynamics of the system follows a logical pathway. Ashes decreased over time, because they are redistributed by water or wind, or taken up by the soil under influence of physical processes and bioturbation (Bodí et al., 2011). The general trend, and distribution along soil surface cover classes are in accordance with Esposito et al. (1999). In this study, cover by bryophytes, bare soil, shrub and herbs after a high intensity slash fire with soil surface temperatures around 900 C°, created by wood piles, and a low intensity wildfire with soil surface temperatures around 200 C° is compared. Although fire intensity used by Esposito et al. (1990) is not the same as the fire severity used in this study, the two indicators are closely related to each other, and still gives usable insights. Comparable to this study, a higher dominance of vegetation (herbs) is found in the low intensity plots compared to high intensity plots. After 2.7 years cover by herbs is recovered towards 50% and bare soil is disappeared, which can also be expected in this case. However, the post-fire development of bryophytes (e.g. mosses) is contrasting to the results found in this study. Where bryophytes are more dominant in high intensity plots according to Esposito et al. (1999), this study does not really showed a positive relation between higher burn severity and the cover by mosses. Instead, moss cover seems to be related to vegetation growth. By the provision of shadow and development of substrates around vegetation, moss growth can be supported and explaining the higher moss cover in plots where vegetation cover is also higher.



a) b) c)

Figure 18: Representative examples of cover in May in the burn treatment (a) and (b) and slash and burn treatment (c). (Photos by Oscar Gonzalez Pelayo).

5.3 Vegetation abundance

The results of this study showed the appearance of typical Mediterranean plant species after the experimental fire in both treatments, including species of the genus *Cistus*, *Erica* and *Pterospartum* in combination with members of the Asteraceae family. The peak temperatures reached in the soil layers were high enough to break physical seed dormancy of above mentioned species, and thus stimulate post-fire regeneration (Moreira & Pausas, 2012). For some seedlings species is was very difficult or sometimes impossible to distinguish them in this early stage. This might explain differences in abundance between January and February in particular. Some seedlings identified as *Erica spp.* are probably *Ulex spp.,* a typical evergreen Mediterranean shrub in the family of the Fabaceae. Consequently, the abundance of *Erica spp.* was probably affected by this. Furthermore, it needs to be mentioned that some resprouts of *Pterospartum tridentatum* showed signs of being eaten, probably by rabbits. According to the hypothesis, it was expected that recovery of the vegetation would be slower in the slash and burn treatment with a lower abundance of *Hakea sericea* due to higher fire temperatures in combination with a higher soil burn severity. However, the results are opposite to the expectation. The relative abundance of *Hakea sericea* did not differ between treatments and more seedlings (absolute amount) were found in the slash and burn treatment. Nevertheless, the presence of *Hakea sericea* at the end of this study was very minor in both treatments and differences may still appear in the near future.

5.3.1 Influence of fire treatments on Hakea sericea

In both treatments, absolute numbers of *Hakea sericea*, relative abundance and seedling density was decreasing over time. One year after the fire, seedling density was around 1 seedling per m². This was lower than expected. Based on visual estimation and burned stands in the burned plots, *Hakea sericea* density ranged from 0 to 6 adults shrubs per m² prior to the fire. This corresponds to a mean density of 3 - 4 shrubs per m². However, it can be assumed that not all seedlings survive and will reach maturity. Also, the amount of seeds released in such dense stands should be at least 100 to 200 seeds/ m² as found by Richardson & Van Wilgen (1984). For this reason seedling density was expected to be higher. So, from this point of view both treatments seemed to be effective in reducing *Hakea sericea* density in the short term. These results are probably explained by the high temperatures reached during the fire in both treatments, and the increase in availability of light and resources after the fire.

Earlier research to *Hakea sericea* by Fugler (1983) reported that most successful eradication measures in South-Africa consist of mechanical clearing with follow-up measures. Burning is applied 12 to 18 months after clearing to destroy appearing seedlings. After the fire still surviving seedlings are pulled out. Eventually, biological control can be additionally used to reduce viable seed amounts of mature plants. Different than in this study, burning was thus only used in combination with other treatments (felling and biological control) to successfully control *Hakea sericea.* Long term experiences have shown that burning alone can be ineffective and even facilitate regeneration of *Hakea spp.* (Richardson & Van Wilgen, 1984), which is contrasting to the results found in our shorter-term study. According to Esler et al. (2010), the long-term clearing program in South-Africa using this combination of clearing and burning was able to reduce high density stands of *Hakea sericea* (from ≥10.000 to 10-100 individuals/ha). Clearing measures are costly, in particular for dense stands. Costs will decrease over time with decreasing densities, but follow-up treatments several years after burning need to be ensured for successful control (Esler et al., 2010).

5.3.2 Influence of fire intensity & severity on vegetation

Another research in relation to invasive species and different ways of burning by Cilliers et al. (2004) in South Africa showed a clear relation between fire intensity and the appearance of seedlings from both indigenous and invasive species. Areas covered by several woody invasive species (*Pinus* *pinaster, Acacia spp.*, and *Hakea drupacea*) were investigated after occurrence of a wildfire. Comparable to *Hakea sericea*, the serotinous species *Hakea drupacea* and *Pinus Pinaster* also store seeds in the canopy. The study of Cilliers et al. (2004) included different treatments: burning of standing aliens, burning of felled and stacked aliens (formation of piles), burning of an area which is completely cleared, burning of native vegetation (Fynbos), and burning of felled and stacked aliens by controlled burning. Cilliers et al. (2004) thus mainly investigated the effect of a wildfire, but these associated fire characteristics are also representative for the prescribed fire in our study. In invaded areas, emergence of native and invasive propagules was highest in the area with prescribed burning (87 individuals/ m^2 and 504 individuals/ m^2 respectively), but indigenous species richness was low. Lower densities of invasive species were found in the other treatments burned by the wildfire, due to damage to the seedbank as consequence of high temperatures and intensity. In the treatment with highest fire intensity (burning and stacking), seedbanks of both native and invasive species were even almost completely eliminated towards very low densities (< 2 individuals/ m^2), one year after the wildfire. Positive results were found for the clearing, removing and burning treatment which resulted in relative high seedling density of native vegetation, but lower densities of invaders, and a species richness almost equally to the uninvaded Fynbos. This result thus clearly displays that finding a balance between fire severity and killing seeds in the soil seedbank is very important for recovery of native vegetation and development of a resilient ecosystems (Cilliers et al., 2004). This is also supported by Richardson & Van Wilgen (1986), who stated that too high fire intensities in invaded areas by *Hakea sericea* should be avoided in order to promote recovery of native vegetation, and preventing soil damage. For this reason, fireline intensities around 7000 kW/m should be aimed and can be manipulated by the amount of fuel and moisture content of the fuel (Richardson & Van Wilgen, 1986). In Holmes et al. (2000) similar results where shown. Application of burning treatments only in invaded areas by *Hakea sericea* and *Pinus spp.* led to lower native density and higher invasive density compared the other treatments (felling + burning with and without removing plant material). Though, differences between these treatments were not significant. In general, seeders became more abundant and sprouting plants decreased (Holmes et al., 2000).

Set against the lower soil burn severity found in the slash and burn treatments, the trend in seedling density in our study in this treatment can thus be explained with the above mentioned studies. Contrary to our expectation, the amount of native seedlings were higher in this treatment compared to the burn treatment. This is also in agreement with De Luis et al. (2005). The soil moisture content in the soil, litter and dead fuel probably minimized damage to deeper soil layers and ensured pre-existence of the native seedbank which is also the case in the clearing, removing and felling treatment described above by several authors. The soil burn severity partly also explained the community structure and plant regeneration strategies found in this study. Most plants species which were found to be most abundant are regenerating by the productions of seeds (e.g. *Cistus spp.* and *Erica spp.*). Only *Pterospartum tridentatum* uses both germination and resprouting as post-fire strategy, although germination is the dominant strategy. Soil burn severity classes ranging from 4 to 5 and found in the burn treatments, are expected to have a damaging effect on vegetative buds and thus negatively effect regeneration by *Pterospartum tridentatum.* Results of Fernández et al. (2013) show a negative effect of soil burn severity on resprouters as *Pterospartum tridentatum*, as well. Surprisingly, the density of *Pterospartum tridentatum* resprouts in our study was higher in the first counts in the burn treatment, but did not differ anymore in the longer term. Still, the high soil burn severity may explain why a larger share of *Pterospartum tridentatum* came from germination instead of resprouting. It also may explain the increasing trend of *Pterospartum tridentatum* in the slash and burn treatment. However, monitoring in the longer term is needed to see if density of *Pterospartum tridentatum* is indeed affected by soil burn severity and/ or other factors.

5.3.3 Fire characteristics in relation to erosion and surface wash of seeds

Soil burn severity also affects plant establishment in a more indirect way. As described earlier it affects the availability of soil nutrients and the amount of carbon. Furthermore, high burn severity is positively related to erosion, which can cause a potential loss of the soil seedbank and soil fertility (Parson et al., 2010). Data (unpublished yet) collected by the ESPteam (CESAM, University Aveiro) in the same study plots shows considerable higher erosion amounts in the burn treatment compared to the slash and burn treatment (Fig. 19). Stored in the upper part of the soil, seeds are vulnerable to surface wash removal (Cerdà & Garcia-Fayos, 2002; Garcia-Fayos & Cerdà, 1997; Jiao et al., 2013). Seed removal is dependent on several factors including the angle of the slope, rainfall intensity, surface roughness, seed weight, seed shape and vegetation cover. Estimating 83 plants species in South-eastern Spain, a the study of Cerdà & Garcia-Fayos (2002) reveals that average seed losses due to erosion approach 11% with a rainfall intensity of approximately 55 mm/h. Under normal conditions this is not problematic for the seed bank, as it will be replenished the following year. Though, the combination of high erosion rates together with severe fires damaging the soil seed bank, makes systems as addressed in this study vulnerable. The influence of surface runoff on potential seedbank loss is not extensively studied. For this reason, an additional small scale experiment was set up to see if erosion played a role in seed bank loss of the native vegetation and if erosion material contained viable seeds of *Hakea sericea* as well. This was done according to the indirect seedling method described in Maia et al. (2016), by putting soil samples of the eroded material on a substrate in the greenhouse. For more details concerning the method see Appendix C. These results didn't give a sign of a large amount viable seeds in the eroded soil, and very few seedlings appear. Nevertheless, incorporating this kind of features can be interesting in follow-up research to vegetation recovery. Where locally depletion of the soil seedbank can occur due to surface runoff, local seed loss will be converted into seed redistribution on a larger scale such as a hillslope or river catchment (Bochet, 2015). Although this is not specifically applicable for *Hakea sericea*, other invasive species in fire-prone Mediterranean areas as *Acacia spp.* have seeds stored in the soil seedbank and can be redistributed by water (Souza-Alonso et al., 2017).

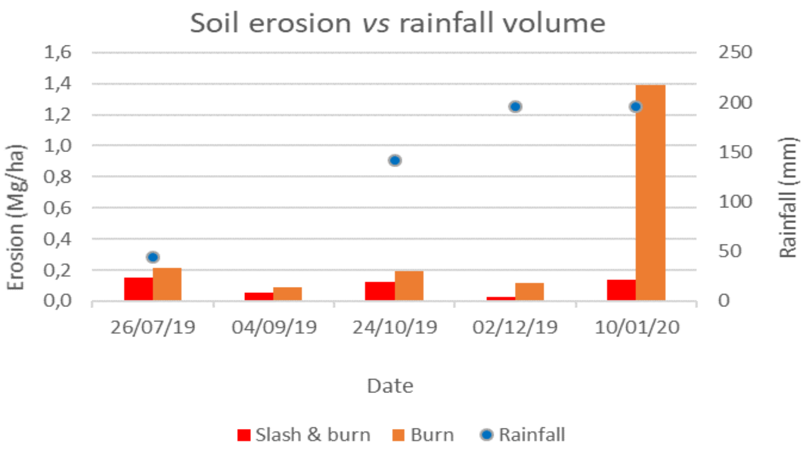


Figure 19: Amount of erosion combined with rainfall in both treatments.

5.3.4 Vegetation recovery in the longer term

Research to the effect of burning with equivalent vegetation composition in relation to soil properties in the longer term is performed by among others Granged et al., (2011). Colonization by herbaceous species, and species as *Erica australis*, *Pterospartum* and *Cistus* takes place within 1 year after the fire which is in line with the findings of our study. Pre-fire cover is most of the time returned to the pre-fire state in 4 years. Only short-term vegetation responses are assed in this study, but it can be expected that vegetation will follow a similar trend as described by the findings of Granged et al. (2011), if the invader *Hakea sericea* does not become the most dominant species. Following this trend, herbaceous species increase in the first years after the fire, but are outcompeted in the longer term by shrubs and other woody species. This is also supported by Calvo et al. (2002), who performed a long term study to vegetation responses in a Mediterranean shrubland after different perturbations; burning, cutting and ploughing. Seeders from the Cistaceae familiy, thrive well in the first few years after burning in which they benefit from the reduced competition by bigger plants. Furthermore, some resprouting species as *Erica Australis* regenerates quickly within 1 year after burning, while others as *Calluna vulgaris* needs more time (12 years) to return to a pre-disturbance state (Calvo et al., 2011).

Above mentioned studies of Calvo et al. (2002) and Granged et al. (2011) are based on shrublands with comparable ecosystem functioning as the one addressed in this study. However, these shrublands are not completely dominated by invaders as *Hakea sericea.* The short term results show low density and relative abundance of *Hakea sericea* in both treatments compared to other vegetation. At this point it is impossible to say if the small *Hakea sericea* seedlings are able to outcompete the other species in the longer term. Despite the high temperatures and in some cases high soil burn severity, native seeds in the seedbank are still viable giving opportunity to the native vegetation to flourish in this short period after the fire.

For now, both treatments seems to be effective in reducing abundance of *Hakea sericea*. Although not significant yet, there are some indications that vegetation cover consisting of native species is recovering quicker in the slash and burn treatment and contains higher amounts of seedlings. Higher cover will reduce erosion, which is beneficial to the soil. This is a positive side effect of the slash and burn treatment in this case. However, the use of prescribed fires should always be carefully considered, incorporating the local characteristics and limits of the area. More time is needed to see how *Hakea sericea* develops and if more frequent or severe burns are needed to control this species. In such a case, this will unfortunately negatively affect the native seedbank as well. So, control measures always need to be well though out incorporating the effect on the whole ecosystem (Holmes et al., 2002), as recovery is often highly dependent on the maintenance of native seedbanks or surrounding resources of native plants (Souza-Alonso et al., 2017).

5.4 Recommendations for management

Managing invasive species is very complex, and time-consuming. It has to deal with highly interlinked ecosystem components, and several other challenges such as conflicting interests of stakeholders, contrasting management objectives, lack of political will, and scarcity in resources (e.g. money) (Buckley, 2008). Management programs will be in general more effective if invasive species are detected in an early stage. Detection and monitoring the spread of invasive species are of main importance to minimize effects of invasive species. The use of scenario's and distribution models can be useful to detect vulnerable areas in order to prevent new introductions. It can also facilitate decision-making in prioritizing valuable areas with high ecological- or socio-economic interests (Souza-Alonso et al., 2017). Only eradication measures might not be enough to recover ecosystems, and control of invasive species should be done in an ecosystem-whole context (Zavaleta et al., 2001). The appearance of invaders is a symptom of underlaying changes in the ecosystem properties, in this case caused due particular way of land-management and changed fire regime. The management goal is thus not only focusing on the invasive species, but on restoring for example ecosystem services, restoration of a particular plant community structure, or conservation of endangered or species. This involves multiple steps including; 1) Detection, assessing spatial distribution and impact assessment of the invasive species, 2) Development of management strategies and identification of vulnerable areas (Alvarez-Taboada, 2017), 3) Development and evaluation of tools to achieve management objectives (e.g. eradication tools and monitoring), 4) Cost-benefit assessment in the broadest sense, i.e. monetary costs of tools and ecosystems loss, and 5) Involvement of stakeholders including landowners, private companies, local governments, scientist and raising public awareness (Buckley, 2008). In Portugal for example, a large share of the country is owned by the private sector and incorporating their views and interests is from main importance in the management of land and soils in a more sustainable way. To my knowledge, some mapping of invasive species such as *Hakea sericea* has been done in Portugal (Alvarez-Taboada, 2017), but not extensively yet. This study is a small piece of a puzzle searching for the best management practices to control invasive species, and support native vegetation. It this way it makes a contribution to the next step in the management of invasive species on both a local, and global scale. Further research, including a multifactor analyses on the relation between geographic characteristics, soil properties, fire characterises and abundance of plant species and invasive species would be very useful. Using this kind of information from several cases and locations will enable the creation of a toolkit which can be used by land managers worldwide which are facing the problems associated with invasive plant species such as *Hakea sericea*.

6. Conclusion

According to the hypothesis it was expected that the slash and burn treatment would be more effective in reducing the abundancy of Hakea *sericea* than the burn treatment, but would also slowed down the recovery of the whole plant community due to higher temperatures in the soil layer. However, this study did not found significant differences between both treatments in duration and peak temperatures reached during the fire in the litter, duff and soil layer. Actually, opposite to expectation, soil burn severity was higher in the burned plots, probably in relation to the lower moisture of the litter and soil surface layers measured in these plots right before the start of the experimental fire. The dynamics of the soil surface cover were also not significantly different between both treatments, yet there was a trend towards a higher vegetation cover in the slash and burnt treatment. Minor differences were observed between the two treatments in plant species abundance and diversity. In both treatments, seeders of *Cistus spp.* and *Erica spp.* were most abundant, but total seedling density was higher in the slash and burned plots. The relative abundance and seedling density of *Hakea sericea* was rather low and similar between the treatments. Thus, both treatments seemed to be able to successfully reduce *Hakea sericea* density towards 1 seedling per m² (Fig. 20) and a relative abundance less than 3%, one year after the fire. However, research in the longer term is needed to fully examine the effectiveness of these treatments regarding control of invasive species as *Hakea sericea.*

Figure 20: Hakea seedling. (Photo by João Pinho, ICNF).

7. Acknowledgement

This thesis was written as an end result of the master programme Sustainable Development within the track Environmental Change and Ecosystems, at Utrecht University. It was done with help of the facilities provided by the Department of Environment and Planning (DAO) at the University of Aveiro-CESAM, in Portugal. This research is carried out with support of the FOGO e INVASORAS (PDR2020-101- 030919) and the SusPiRe (PTDC/ASP-SIL/30983/2017) projects, and fits within recent research investigating the use of prescribed fires to control invasive species in the Mediterranean. I am grateful for all information I have acquired during the writing process of my thesis and the fieldwork in Portugal. I really enjoyed the trips to the field, and the lunches in the local restaurants during the breaks. And I am also very curious how the study area I investigated will develop in the future.

First, I will thank my supervisors Ángeles Garcia Mayor and Oscar Gonzalez Pelayo for the supervision during my thesis. Also, my mentor Paula Maia helped me a lot, especially with identifying plant species in my study area. And of course I am happy with Mariska te Beest as second reader of my thesis. Furthermore, I will thank all members of the ESPteam for sharing their knowledge and for the great time I experienced during my stay in Portugal.

8. References

Alcañiz, M., Outeiro, L., Francos, M., & Úbeda, X. (2018). Effects of prescribed fires on soil properties: a review. *Science of The Total Environment*, *613*, 944-957.

de Almeida, J. D., & Freitas, H. (2006). Exotic naturalized flora of continental Portugal-A reassessment. *Botanica complutensis*, *30*, 117.

de Almeida, J. D., & Freitas, H. (2012). Exotic flora of continental Portugal–a new assessment. *Bocconea*, *24*, 231-237.

de Almeida, J. D. (2018). New additions to the exotic vascular flora of continental Portugal. *Flora Mediterranea, 28,* 259–278*.*

Alvarez-Taboada, F., Paredes, C., & Julián-Pelaz, J. (2017). Mapping of the invasive species Hakea sericea using unmanned aerial vehicle (UAV) and WorldView-2 imagery and an object-oriented approach. *Remote Sensing*, *9*(9), 913.

Ansley, R. James; Arno, Stephen F.; Brock, Brent L.; Brose, Patrick H.; Brown, James K.; Duchesne, Luc C.; Grace, James B.; Gottfried, Gerald J.; Haase, Sally M.; Harrington, Michael G.; Hawkes, Brad C.; Hoch, Greg A.; Miller, Melanie; Myers, Ronald L.; Narog, Marcia G.; Patterson, William A. III; Paysen, Timothy E.; Ryan, Kevin C.; Sackett, Stephen S.; Wade, Dale D.; and Wilson, Ruth C. (2002). Wildland Fire in Ecosystems Effects of Fire on Flora. *JFSP Synthesis Reports, 9.*

Badía, D., López-García, S., Martí, C., Ortíz-Perpiñá, O., Girona-García, A., & Casanova-Gascón, J. (2017). Burn effects on soil properties associated to heat transfer under contrasting moisture content. *Science of the Total Environment*, *601*, 1119-1128.

te Beest, M., Cromsigt, J. P., Ngobese, J., & Olff, H. (2012). Managing invasions at the cost of native habitat? An experimental test of the impact of fire on the invasion of Chromolaena odorata in a South African savanna. *Biological Invasions*, *14*(3), 607-618.

te Beest, M., Howison, O., Howison, R., Dew, L., Poswa, M., Dumalisile, L., . . . Terblanche, C. (2017). Successful Control of the Invasive Shrub Chromolaena odorata in Hluhluwe-iMfolozi Park. In J. Cromsigt, S. Archibald, & N. Owen-Smith (Eds.), *Conserving Africa's Mega-Diversity in the Anthropocene: The Hluhluwe-iMfolozi Park Story* (Ecology, Biodiversity and Conservation, pp. 358-382). Cambridge: Cambridge University Press.

Bochet, E. (2015). The fate of seeds in the soil: a review of the influence of overland flow on seed 149 removal and its consequences for the vegetation of arid and semiarid patchy ecosystems. *SOIL, 1*, 131–146.

Bodí, M. B., Mataix-Solera, J., Doerr, S. H., & Cerdà, A. (2011). The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content. *Geoderma*, *160*(3-4), 599-607.

Bradshaw, S. D., Dixon, K. W., Lambers, H., Cross, A. T., Bailey, J., & Hopper, S. D. (2018). Understanding the long-term impact of prescribed burning in mediterranean-climate biodiversity hotspots, with a focus on south-western Australia. *International Journal of Wildland Fire*, *27*(10), 643-657.

Breytenbach, G. J. (1989). Alien control: Can we afford to slash and burn hakea in fynbos ecosystems?. *South African Forestry Journal*, *151*(1), 6-16.

Brooks, M. L., D'antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., ... & Pyke, D. (2004). Effects of invasive alien plants on fire regimes. *BioScience*, *54*(7), 677-688.

Buckley, Y. M. (2008). The role of research for integrated management of invasive species, invaded landscapes and communities. *Journal of Applied Ecology*, *45*(2), 397-402.

Buhk, C., Meyn, A., & Jentsch, A. (2007). The challenge of plant regeneration after fire in the Mediterranean Basin: scientific gaps in our knowledge on plant strategies and evolution of traits. *Plant ecology*, *192*(1), 1-19.

Calvo, L., Tárrega, R., & De Luis, E. (2002). The dynamics of Mediterranean shrubs species over 12 years following perturbations. *Plant ecology*, *160*(1), 25-42.

Campbell, G. S., Jungbauer Jr, J. D., Bristow, K. L., & Hungerford, R. D. (1995). Soil temperature and water content beneath a surface fire. *Soil Science*, *159*(6), 363-374.

Cardoso, A. W., Oliveras, I., Abernethy, K. A., Jeffery, K. J., Lehmann, D., Edzang Ndong, J., ... & Malhi, Y. S. (2018). Grass Species Flammability, Not Biomass, Drives Changes in Fire Behavior at Tropical Forest-Savanna Transitions. *Frontiers in Forests and Global Change*, *1*, 6.

Cerdà, A., & Garcıa-Fayos, P. (2002). The influence of seed size and shape on their removal by water erosion. *Catena*, *48*(4), 293-301.

Cilliers, C. D., Esler, K. J., & Boucher, C. (2004). Effects of alien plant management and fire on soil seed banks and regeneration in the Cape Peninsula National Park, South Africa. *South African Journal of Botany*, *70*(5), 705-712

DeBano, L. F., Rice, R. M., & Eugene, C. C. (1979). Soil heating in chaparral fires: effects on soil properties, plant nutrients, erosion, and runoff. *Res. Paper PSW-RP-145. Berkeley, CA: US Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 21 p*, *145*.

De Luis, M., Raventós, J., & González-Hidalgo, J. C. (2005). Factors controlling seedling germination after fire in Mediterranean gorse shrublands. Implications for fire prescription. *Journal of Environmental Management*, *76*(2), 159-166.

DiTomaso, J. M., Brooks, M. L., Allen, E. B., Minnich, R., Rice, P. M., & Kyser, G. B. (2006). Control of invasive weeds with prescribed burning. *Weed technology*, *20*(2), 535-548.

DiTomaso, J.M. and D.W. Johnson (eds.). 2006. The Use of Fire as a Tool for Controlling Invasive Plants. Cal-IPC Publication 2006-01. California Invasive Plant Council: Berkeley, CA. 56 pp.

Elango, L. (Ed.). (2011). *Hydraulic Conductivity: Issues, Determination and Applications*. InTech, Croatia.

EPPO. (2017). Pest risk analysis for Hakea sericea. EPPO, Paris.

European soil data centre (ESDAC). (2019). *National Soil Maps (EUDASM)*. Retrieved December 9, 2019, from https://esdac.jrc.ec.europa.eu/resource-type/national-soil-maps-eudasm?field_data_continent_tid_selective=All&field_data_country_country_selective=PT&field_data_cont_coverage_value=#

Esler, K. J., Van Wilgen, B. W., Te Roller, K. S., Wood, A. R., & Van Der Merwe, J. H. (2010). A landscape-scale assessment of the long-term integrated control of an invasive shrub in South Africa. *Biological Invasions*, *12*(1), 211.

Esposito, A., Mazzoleni, S., & Strumia, S. (1999). Post-fire bryophyte dynamics in Mediterranean vegetation. *Journal of Vegetation Science*, *10*(2), 261-268.

European Commission. (2013). Commission Staff Working Document. *Executive Summary of the Impact Assessment accompanying the document 'Proposal for a regulation of the European Parliament and of the Council on the prevention and management of the introduction and spread of invasive alien species'. SWD (2013)*, *321*.

European Commission. (2019, September 7). The Mediterranean Region. Retrieved September 21, 2019, from https://ec.europa.eu/environment/nature/natura2000/biogeog_regions/mediterranean/index_en.htm

Fernandes, P. M. (2018). Scientific support to prescribed underburning in southern Europe: What do we know?. *Science of the Total Environment*, *630*, 340-348.

Fernández, C., Vega, J. A., & Fonturbel, T. (2013). Does fire severity influence shrub resprouting after spring prescribed burning?. *Acta oecologica*, *48*, 30-36.

Fugler, S. R. (1983). The control of silky hakea in South Africa. *Bothalia*, *14*(3/4), 977-980.

Gagnon, P. R., Passmore, H. A., Slocum, M., Myers, J. A., Harms, K. E., Platt, W. J., & Paine, C. T. (2015). Fuels and fires influence vegetation via above-and belowground pathways in a high-diversity plant community. *Journal of Ecology*, *103*(4), 1009-1019.

Garcia-Fayor, P., & Cerdà, A. (1997). Seed losses by surface wash in degraded Mediterranean environments. *Catena*, *29*(1), 73-83.

Gimeno-Garcıa, E., Andreu, V., & Rubio, J. L. (2004). Spatial patterns of soil temperatures during experimental fires. *Geoderma*, *118*(1-2), 17-38.

González-Pelayo, O., Andreu, V., Campo, J., Gimeno-García, E., & Rubio, J. L. (2006). Hydrological properties of a Mediterranean soil burned with different fire intensities. *Catena*, *68*(2-3), 186-193.

Granged, A. J., Zavala, L. M., Jordán, A., & Bárcenas-Moreno, G. (2011). Post-fire evolution of soil properties and vegetation cover in a Mediterranean heathland after experimental burning: A 3-year study. *Geoderma*, *164*(1-2), 85-94.

Holmes, P. M., Richardson, D. M., Van Wilgen, B. W., & Gelderblom, C. (2000). Recovery of South African fynbos vegetation following alien woody plant clearing and fire: implications for restoration. *Austral Ecology*, *25*(6), 631-639.

Hosseini, M., Keizer, J. J., Pelayo, O. G., Prats, S. A., Ritsema, C., & Geissen, V. (2016). Effect of fire frequency on runoff, soil erosion, and loss of organic matter at the micro-plot scale in north-central Portugal. *Geoderma*, *269*, 126-137.

ICNF. (2013). *Mapas de áreas ardidas entre 2019, 2018, 2017, 2016, 2015, 2014, 2013, 2012, 2010, 2000-2009, 1990-1999.* Retrieved November 20, 2019, from http://www2.icnf.pt/portal/florestas/dfci/inc/mapas

IPBES. (2019). Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. E. S. Brondizio, J. Settele, S. Díaz, and H. T. Ngo (editors). *IPBES secretariat, Bonn, Germany*.

IPMA. (2020). *Climate normals – 1971-2000.* Climate station 102, Aveiro Universidade. Retrieved January 16, 2020, from https://www.ipma.pt/en/oclima/normais.clima/1971-2000/#102

Jiao, J., Han, L., Jia, Y., Lei, D., Wang, N., & Li, L. (2013). Seed morphology characteristics in relation to seed loss by water erosion in the Loess Plateau. *SpringerPlus* (Vol. 2, No. S1, p. S9). Springer International Publishing.

Keeley, J. E., Baer-Keeley, M., & Fotheringham, C. J. (2005). Alien plant dynamics following fire in Mediterranean-climate California shrublands. *Ecological Applications*, *15*(6), 2109-2125.

Keeley, J. E. (2009). Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire*, *18*(1), 116-126.

Keeley, J. E., Bond, W. J., Bradstock, R. A., Pausas, J. G., & Rundel, P. W. (2011). *Fire in Mediterranean ecosystems: ecology, evolution and management*. Cambridge University Press.

Kraus, D., Silva, J. S., Held, A., Castellnou, M., & Krumm, F. (2016). 4.7 Natural disturbances and invasive introduced plants–a specific view on fire. In Vítková, L., & Krumm, F. (Eds.), *Introduced tree species in European forests: opportunities and challenges* (pp.1-425). European Forest Institute.

Maia, P., Pausas, J. G., Arcenegui, V., Guerrero, C., Pérez-Bejarano, A., Mataix-Solera, J., ... & Keizer, J. J. (2012). Wildfire effects on the soil seed bank of a maritime pine stand—the importance of fire severity. *Geoderma*, *191*, 80-88.

Maia, P., Vasques, A., Pausas, J. G., Viegas, D. X., & Keizer, J. J. (2016). Fire effects on the seed bank of three Mediterranean shrubs: implications for fire management. *Plant Ecology*, *217*(10), 1235-1246.

Malkisnon, D., Wittenberg, L., Beeri, O., & Barzilai, R. (2011). Effects of repeated fires on the structure, composition, and dynamics of Mediterranean maquis: short-and long-term perspectives. *Ecosystems*, *14*(3), 478-488.

Mandle, L., Bufford, J. L., Schmidt, I. B., & Daehler, C. C. (2011). Woody exotic plant invasions and fire: reciprocal impacts and consequences for native ecosystems. *Biological Invasions*, *13*(8), 1815-1827.

Marchante, E., & Marchante, H. (2016). Engaging society to fight invasive alien plants in Portugal—One of the main threats to biodiversity. In *Biodiversity and Education for Sustainable Development* (pp. 107-122). Springer, Cham.

Martins, J., Richardson, D. M., Henriques, R., Marchante, E., Marchante, H., Alves, P., ... & Vicente, J. R. (2016). A multi-scale modelling framework to guide management of plant invasions in a transboundary context. *Forest Ecosystems*, *3*(1), 17.

Mayor, A. G., Bautista, S., Llovet, J., & Bellot, J. (2007). Post-fire hydrological and erosional responses of a Mediterranean landscape: Seven years of catchment-scale dynamics. *Catena*, *71*(1), 68-75.

Mayor, A. G., Valdecantos, A., Vallejo, V. R., Keizer, J. J., Bloem, J., Baeza, J., ... & de Ruiter, P. C. (2016). Fire-induced pine woodland to shrubland transitions in Southern Europe may promote shifts in soil fertility. *Science of the Total Environment*, *573*, 1232-1241.

Morais, M., Marchante, E., & Marchante, H. (2017). Big troubles are already here: risk assessment protocol shows high risk of many alien plants present in Portugal. *Journal for nature conservation*, *35*, 1-12.

Moreira, F., Viedma, O., Arianoutsou, M., Curt, T., Koutsias, N., Rigolot, E., ... & Mouillot, F. (2011). Landscape–wildfire interactions in southern Europe: implications for landscape management. *Journal of environmental management*, *92*(10), 2389-2402.

Moreira, B., & Pausas, J. G. (2012). Tanned or burned: the role of fire in shaping physical seed dormancy. *PLoS One*, *7*(12).

Moreira, F., Ferreira, A., Abrantes, N., Catry, F., Fernandes, P., Roxo, L., ... & Silva, J. (2013). Occurrence of native and exotic invasive trees in burned pine and eucalypt plantations: implications for post-fire forest conversion. *Ecological engineering*, *58*, 296-302.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*(6772), 853.

Parson, A., Robichaud, P. R., Lewis, S. A., Napper, C., & Clark, J. T. (2010). Field guide for mapping post-fire soil burn severity. *Gen. Tech. Rep. RMRS-GTR-243. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 p.*, *243*.

Paula, S., Naulin, P. I., Arce, C., Galaz, C., & Pausas, J. G. (2016). Lignotubers in Mediterranean basin plants. *Plant Ecology*, *217*(6), 661-676.

Pausas, J. G., Llovet, J., Rodrigo, A., & Vallejo, R. (2009). Are wildfires a disaster in the Mediterranean basin?–A review. *International Journal of wildland fire*, *17*(6), 713-723.

Pausas, J. G., & Keeley, J. E. (2014). Evolutionary ecology of resprouting and seeding in fire-prone ecosystems. *New Phytologist*, *204*(1), 55-65.

Penman, T. D., & Towerton, A. L. (2008). Soil temperatures during autumn prescribed burning: implications for the germination of fire responsive species?. *International Journal of Wildland Fire*, *17*(5), 572-578.

Prats, S. A., González-Pelayo, Ó., Silva, F. C., Bokhorst, K. J., Baartman, J. E., & Keizer, J. J. (2019). Post-fire soil erosion mitigation at the scale of swales using forest logging residues at a reduced application rate. *Earth Surface Processes and Landforms*, *44*(14), 2837-2848.

Richardson, D. M., & Van Wilgen, B. W. (1984). Factors affecting the regeneration success of Hakea sericea. *South African Forestry Journal*, *131*(1), 63-68.

Richardson, D. M., & Van Wilgen, B. W. (1986). The effects of fire in felled Hakea sericea and natural fynbos and implications for weed control in mountain catchments. *South African Forestry Journal*, *139*(1), 4-14.

Richardson, D. M., Van Wilgen, B. W., & Mitchell, D. T. (1987). Aspects of the reproductive ecology of four Australian Hakea species (Proteaceae) in South Africa. *Oecologia*, *71*(3), 345-354.

Sagra, J., Moya, D., Plaza-Álvarez, P. A., Lucas-Borja, M. E., Alfaro-Sánchez, R., De Las Heras, J., & Ferrandis, P. (2017). Predation on early recruitment in Mediterranean forests after prescribed fires. *Forests*, *8*(7), 243.

Silva, J. S., Vaz, P., Moreira, F., Catry, F., & Rego, F. C. (2011). Wildfires as a major driver of landscape dynamics in three fire-prone areas of Portugal. Landscape and Urban Planning, 101(4), 349-358.

Soto, B., Basanta, R., Perez, R., & Diaz-Fierros, F. (1995). An experimental study of the influence of traditional slash-and-burn practices on soil erosion. *Catena*, *24*(1), 13-23.

Souza-Alonso, P., Rodríguez, J., González, L., & Lorenzo, P. (2017). Here to stay. Recent advances and perspectives about Acacia invasion in Mediterranean areas. *Annals of Forest Science*, *74*(3), 55.

Stoof, C. R., De Kort, A., Bishop, T. F., Moore, D., Wesseling, J. G., & Ritsema, C. J. (2011). How rock fragments and moisture affect soil temperatures during fire. *Soil Science Society of America Journal*, *75*(3), 1133-1143.

Terefe, T., Mariscal-Sancho, I., Peregrina, F., & Espejo, R. (2008). Influence of heating on various properties of six Mediterranean soils. A laboratory study. *Geoderma*, *143*(3-4), 273-280.

Tessler, N., Sapir, Y., Wittenberg, L., & Greenbaum, N. (2016a). Recovery of Mediterranean vegetation after recurrent forest fires: insight from the 2010 forest fire on Mount Carmel, Israel. *Land Degradation & Development*, *27*(5), 1424-1431.

Tessler, N., Wittenberg, L., & Greenbaum, N. (2016b). Vegetation cover and species richness after recurrent forest fires in the Eastern Mediterranean ecosystem of Mount Carmel, Israel. *Science of the Total Environment*, *572*, 1395-1402.

Úbeda, X., Lorca, M., Outeiro, L. R., Bernia, S., & Castellnou, M. (2005). Effects of prescribed fire on soil quality in Mediterranean grassland (Prades Mountains, north-east Spain). *International Journal of Wildland Fire*, *14*(4), 379-384.

Vallejo, V. R., Arianoutsou, M., & Moreira, F. (2012). Fire ecology and post-fire restoration approaches in Southern European forest types. In *Post-fire management and restoration of southern European forests* (pp. 93-119). Springer, Dordrecht.

van Wilgen, B. W., & Richardson, D. M. (1985). The effects of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: a simulation study. *Journal of applied Ecology*, 955-966.

van Wilgen, B. W. (2009). The evolution of fire and invasive alien plant management practices in fynbos. *South African Journal of Science*, *105*(9-10), 335-342.

van Wilgen, B. W., Forsyth, G. G., De Klerk, H., Das, S., Khuluse, S., & Schmitz, P. (2010). Fire management in Mediterranean-climate shrublands: a case study from the Cape fynbos, South Africa. *Journal of Applied Ecology*, *47*(3), 631-638.

Vega, J. A., Fontúrbel, T., Merino, A., Fernández, C., Ferreiro, A., & Jiménez, E. (2013). Testing the ability of visual indicators of soil burn severity to reflect changes in soil chemical and microbial properties in pine forests and shrubland. *Plant and Soil*, *369*(1-2), 73-91.

Wyse, S. V., Perry, G. L., & Curran, T. J. (2018). Shoot-level flammability of species mixtures is driven by the most flammable species: implications for vegetation-fire feedbacks favouring invasive species. *Ecosystems*, *21*(5), 886-900.

Zavaleta, E. S., Hobbs, R. J., & Mooney, H. A. (2001). Viewing invasive species removal in a whole-ecosystem context. *Trends in Ecology & Evolution*, *16*(8), 454-459.

Zouhar, K. (2008). *Wildland fire in ecosystems: fire and nonnative invasive plants*. Government Printing Office.

Invasoras (2019). *Háquea-picante*. Retrieved January 10, 2020, from http://www.invasoras.uc.pt

9. Appendix

Appendix A: *Table with details concerning fire characteristics; 1) Exceedance over several thresholds, 2) Mean temperature above thresholds, and 3) Maximum temperature*.

Layer	Time (s) > 60 °C	Time (s) > 100 °C	Time (s) > 300 °C	Mean temperature (°C) > 100 °C	Mean temperature (°C) > 300 °C	Maximum temperature (°C)
Slash and burn						
Measurement 1 (Plot 1)						
Litter	1150	965	655	413	529	749
Duff	1410	1125	585	312	437	581
Soil	2015	1210	165	227	408	498
Measurement 2 (Plot 2)						
Litter	755	535	280	377	559	918
Duff	4685	3480	335	194	443	611
Soil	1475	700	370	317	444	560
Measurement 3 (Plot 4)						
Litter	1630	985	310	281	444	878
Duff	2025	1270	685	377	558	763
Soil	3085	1595	0	132	-	263
Mean slash and burn	2026	1318	376	292	478	646
Burn						
Measurement 1 (Plot 3)						
Litter	945	620	225	301	516	880
Duff	2055	2050	1985	429	434	1034
Soil	2000	1545	145	205	316	365
Measurement 2 (Plot 5)						
Litter	840	535	205	343	604	792
Duff	1395	765	365	298	447	548
Soil	730	405	0	178	-	256
Measurement 3 (Plot 6)						
Litter	1205	680	250	321	593	827
Duff	1850	1525	290	226	320	359
Soil	1635	1055	0	181	-	282
Mean burn	1406	1020	385	275	461	593

Appendix B; Soil burn severity per plot

	B	B	SB	B	SB	SB
Top	2	4	1	5	2	1
Middle	2	2	2	3	3	1
Bottom	2	3	2	2	1	1
	Plot 6	Plot 5	Plot 4	Plot 3	Plot 2	Plot 1
Median	(2)	(3)	(2)	(3)	(2)	(1)

Appendix C; *Method description small scale experiment 'surface wash of seeds'*

Eroded material from each plot was collected from the catchment and done each campaign. The catchment is cleaned and soil is collected in plastic bags and stored in fridge or freezer. For the seed viability experiment the soil is first mixed and put in plastic cups with a volume of 50 ml. Soil samples were available from Ro1, Ro3, Ro4 and Ro5. Remaining soil is sieved with a 2mm sieve to collect seeds for identification under the microscope. Sieved material is stored in paper bags and put in freezer to prevent germination. In order to see what kind of seeds are available in the seedbank (composition) and to measure Hakea seed density, the seedling emergence method (indirect method) is used (Maia et al., 2016)[1]. Each sample was spread out on a wetted layer of 5 cm substratum in an aluminium tray with and put in the greenhouse. The substratum is a mixture of perligran with potting soil. The aluminium trays were perforated to ensure water drainage. Seeds of Ro1, Ro3, Ro4 were sown on 22 January. Monitoring of seedling emergence took place every day, except for the weekend. Emerged seedlings were counted, and marked with a metal pin. Additional, photos were taken from each seedling.

[1] Maia, P., Vasques, A., Pausas, J. G., Viegas, D. X., & Keizer, J. J. (2016). Fire effects on the seed bank of three Mediterranean shrubs: implications for fire management. *Plant Ecology*, *217*(10), 1235-1246.

Appendix D; *Absolute & relative amount of species per plot for January, February and May.*

Plot	Hakea_m2	Hka_rel_abundanc	#Pterospart	Pte_rel_abu	#Bulb_m2	Bulb_rel_a	#sp2_Cistu	sp2_rel_al	#sp3_m2	sp3_rel_al	#sp4_Erica	sp4_rel_al	#sp5	sp5_rel_al	#sp6	sp6_rel_al	#Pte_resp	Pte_rsprt_rel_abundance_plot	
SB1_B	4	0.366300366	8	0.7326007	52	4.761905	152	13.91941	4	0.3663	20	1.831502	4	0.3663	0	0	0	0	
SB1_M	8	0.732600733	4	0.3663004	192	17.58242	196	17.94872	8	0.732601	36	3.296703	0	0	0	0	4	0.3663	
SB1_T	0	0	4	0.3663004	80	7.326007	228	20.87912	32	2.930403	48	4.395604	0	0	8	0.732601	0	0	
SB1_Total	**12**	**1.098901099**	**16**	**1.4652015**	**324**	**29.67033**	**576**	**52.74725**	**44**	**4.029304**	**104**	**9.52381**	**4**	**0.3663**	**8**	**0.732601**	**4**	**0.3663**	100
SB1_AV	**4**	**0.366300366**	**5.3333333**	**0.4884005**	**108**	**9.89011**	**192**	**17.58242**	**14.66667**	**1.343101**	**34.66667**	**3.174603**	**1.333333**	**0.1221**	**2.666667**	**0.2442**	**1.333333**	**0.1221**	
SB2_B	0	0	0	0	136	7.142857	520	27.31092	0	0	8	0.420168	0	0	0	0	0	0	
SB2_M	4	0.210084034	0	0	176	9.243697	100	5.252101	12	0.630252	56	2.941176	0	0	0	0	4	0.210084	
SB2_T	4	0.210084034	0	0	84	4.411765	668	35.08403	12	0.630252	116	6.092437	4	0.210084	0	0	0	0	
SB2_Total	**8**	**0.420168067**	**0**	**0**	**396**	**20.79832**	**1288**	**67.64706**	**24**	**1.260504**	**180**	**9.453782**	**4**	**0.210084**	**0**	**0**	**4**	**0.210084**	100
SB2_AV	**2.666667**	**0.140056022**	**0**	**0**	**132**	**6.932773**	**429.3333**	**22.54902**	**8**	**0.420168**	**60**	**3.151261**	**1.333333**	**0.070028**	**0**	**0**	**1.333333**	**0.070028**	
B3_B	4	0.537634409	24	3.2258065	36	4.83871	32	4.301075	36	4.83871	0	0	0	0	0	0	0	0	
B3_M	0	0	40	5.3763441	80	10.75269	204	27.41935	0	0	0	0	0	0	0	0	0	0	
B3_T	4	0.537634409	32	4.3010753	44	5.913978	100	13.44086	16	2.150538	88	11.82796	4	0.537634	0	0	0	0	
B3_Total	**8**	**1.075268817**	**96**	**12.903226**	**160**	**21.50538**	**336**	**45.16129**	**52**	**6.989247**	**88**	**11.82796**	**4**	**0.537634**	**0**	**0**	**0**	**0**	100
B3_AV	**2.666667**	**0.358422939**	**32**	**4.3010753**	**53.33333**	**7.168459**	**112**	**15.05376**	**17.33333**	**2.329749**	**29.33333**	**3.942652**	**1.333333**	**0.179211**	**0**	**0**	**0**	**0**	
SB4_B	4	0.465116279	8	0.9302326	4	0.465116	316	36.74419	0	0	0	0	0	0	0	0	4	0.465116	
SB4_M	0	0	4	0.4651163	64	7.44186	172	20	0	0	0	0	0	0	0	0	0	0	
SB4_T	4	0.465116279	20	2.3255814	136	15.81395	124	14.4186	0	0	0	0	0	0	0	0	0	0	
SB4_Total	**8**	**0.930232558**	**32**	**3.7209302**	**204**	**23.72093**	**612**	**71.16279**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**4**	**0.465116**	100
Sb4_AV	**2.666667**	**0.310077519**	**10.666667**	**1.2403101**	**68**	**7.906977**	**204**	**23.72093**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1.333333**	**0.155039**	
B5_B	8	2.985074627	40	14.925373	56	20.89552	12	4.477612	0	0	0	0	0	0	0	0	0	0	
B5_M	0	0	8	2.9850746	64	23.8806	16	5.970149	0	0	0	0	0	0	0	0	0	0	
B5_T	4	1.492537313	28	10.447761	28	10.44776	0	0	0	0	0	0	0	0	0	0	4	1.492537	
B5_Total	**12**	**4.47761194**	**76**	**28.358209**	**148**	**55.22388**	**28**	**10.44776**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**4**	**1.492537**	100
B5_AV	**4**	**1.492537313**	**25.333333**	**9.4527363**	**49.33333**	**18.40796**	**9.333333**	**3.482587**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1.333333**	**0.497512**	
B6_B	4	1.25	20	6.25	4	1.25	0	0	0	0	0	0	0	0	0	0	0	0	
B6_M	8	2.5	52	16.25	4	1.25	8	2.5	0	0	0	0	0	0	0	0	0	0	
B6_T	0	0	28	8.75	32	10	160	50	0	0	0	0	0	0	0	0	0	0	
B6_Total	**12**	**3.75**	**100**	**31.25**	**40**	**12.5**	**168**	**52.5**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	100
B6_AV	**4**	**1.25**	**33.333333**	**10.416667**	**13.33333**	**4.166667**	**56**	**17.5**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	
specific_sp	**60**		**320**		**1272**		**3008**		**120**		**372**		**12**		**8**		**16**		
all_counte	5188																		

Plot	#Hakea	Hka_rel_abu	#Pterospa	Pte_rel_abun	#Bulb	Bulb_rel_abu	#sp2_Cist	sp2_rel_abun	#sp3	sp3_rel_abundance_plot	#sp4_Eric	sp4_rel_abun	#sp5	sp5_rel_abun	#sp6	sp6_rel_abundar	#Pte_resp	Pte_rsprt_rel	#sp7	sp7_rel_al	#sp8	sp8_rel_abundance_plot	
SB1_B	4	0.36630037	0	0	52	4.76190476	212	19.4139194	0	0	16	1.46520147	4	0.36630037	0	0	4	0.366300366	8	0.732601	0	0	
SB1_M	8	0.73260073	0	0	120	10.989011	192	17.5824176	0	0	68	6.22710623	0	0	0	0	4	0.366300366	12	1.098901	0	0	
SB1_T	0	0	0	0	72	6.59340659	236	21.6117216	0	0	28	2.56410256	0	0	4	0.366300366	4	0.366300366	8	0.732601	36	3.296703	
SB1_Total	**12**	**1.0989011**	**0**	**0**	**244**	**22.3443223**	**640**	**58.6080586**	**0**	**0**	**112**	**10.2564103**	**4**	**0.36630037**	**4**	**0.366300366**	**12**	**1.098901099**	**28**	**2.564103**	**36**	**3.296703**	100
SB2_B	0	0	0	0	84	3.91061453	724	33.7057728	0	0	16	0.74487896	4	0.18621974	0	0	0	0	4	0.18622	0	0	
SB2_M	0	0	0	0	172	8.00744879	284	13.2216015	0	0	36	1.67597765	4	0.18621974	16	0.744878957	4	0.186219739	0	0	0	0	
SB2_T	4	0.18621974	0	0	52	2.42085661	592	27.5605214	0	0	136	6.33147114	4	0.18621974	8	0.372439479	0	0	0	0	4	0.18622	
SB2_Total	**4**	**0.18621974**	**0**	**0**	**308**	**14.3389199**	**1600**	**74.4878957**	**0**	**0**	**188**	**8.75232775**	**12**	**0.55865922**	**24**	**1.117318436**	**4**	**0.186219739**	**4**	**0.18622**	**4**	**0.18622**	100
B3_B	4	0.49261084	0	0	44	5.41871921	24	2.95566502	0	0	24	2.95566502	24	2.95566502	8	0.985221675	0	0	0	0	0	0	
B3_M	0	0	0	0	80	9.85221675	168	20.6896552	0	0	40	4.92610837	0	0	0	0	0	0	12	1.477833	0	0	
B3_T	4	0.49261084	0	0	64	7.8817734	188	23.1527094	0	0	104	12.8078818	0	0	0	0	4	0.492610837	20	2.463054	0	0	
B3_Total	**8**	**0.98522167**	**0**	**0**	**188**	**23.1527094**	**380**	**46.7980296**	**0**	**0**	**168**	**20.6896552**	**24**	**2.95566502**	**8**	**0.985221675**	**4**	**0.492610837**	**32**	**3.940887**	**0**	**0**	100
SB4_B	4	0.38022814	4	0.38022814	4	0.38022814	364	34.6007605	0	0	36	3.42205323	12	1.14068441	0	0	4	0.380228137	0	0	0	0	
SB4_M	0	0	4	0.38022814	40	3.80228137	156	14.8288973	0	0	44	4.18250951	4	0.38022814	0	0	0	0	0	0	0	0	
SB4_T	4	0.38022814	4	0.38022814	121	20.1520913	128	12.1673004	0	0	24	2.28136882	4	0.38022814	0	0	0	0	0	0	0	0	
SB4_Total	**8**	**0.76045627**	**161**	**1.14068441**	**256**	**24.3346008**	**648**	**61.5969582**	**0**	**0**	**104**	**9.88593156**	**20**	**1.90114068**	**0**	**0**	**4**	**0.380228137**	**0**	**0**	**0**	**0**	100
B5_B	8	2.89855072	0	0	68	24.6376812	24	8.69565217	0	0	8	2.89855072	0	0	0	0	0	0	0	0	0	0	
B5_M	0	0	0	0	72	26.0869565	12	4.34782609	0	0	16	5.79710145	4	1.44927536	0	0	0	0	0	0	0	0	
B5_T	0	0	0	0	40	14.4927536	0	0	0	0	16	5.79710145	4	1.44927536	0	0	4	1.449275362	0	0	0	0	
B5_Total	**8**	**2.89855072**	**0**	**0**	**180**	**65.2173913**	**36**	**13.0434783**	**0**	**0**	**40**	**14.4927536**	**8**	**2.89855072**	**0**	**0**	**4**	**1.449275362**	**0**	**0**	**0**	**0**	100
B6_B	4	1.2345679	8	2.4691358	4	1.2345679	12	3.7037037	0	0	12	3.7037037	8	2.4691358	0	0	0	0	0	0	0	0	
B6_M	4	1.2345679	0	0	12	3.7037037	28	8.64197531	0	0	20	6.17283951	0	0	0	0	0	0	0	0	0	0	
B6_T	0	0	0	0	56	17.2839506	60	18.5185185	0	0	80	24.691358	8	2.4691358	8	2.469135802	0	0	0	0	0	0	
B6_Total	**8**	**2.4691358**	**8**	**2.4691358**	**72**	**22.2222222**	**100**	**30.8641975**	**0**	**0**	**112**	**34.5679012**	**16**	**4.9382716**	**8**	**2.469135802**	**0**	**0**	**0**	**0**	**0**	**0**	100
specific_sp	**48**		**169**		**1248**		**3404**		**0**		**724**		**84**		**44**		**28**		**64**		**40**		
all_counte	5853																						

Plot	#Hakea	Hka_rel_abu	#Pterospa	Pte_rel_abun	#Bulb	Bulb_rel_abu	#sp2_Cisti	sp2_rel_abun	#sp3	sp3_rel_abundance_plot	#sp4_Erica	sp4_rel_abun	#sp5	sp5_rel_abun	#sp6	sp6_rel_abundar	#Pte_resp	Pte_rsprt_rel	#sp7	sp7_rel_al	#sp8	sp8_rel_al	#sp9	
SB1_B	4	0.46511628	12	1.39534884	20	2.3255814	132	15.3488372	0	0	16	1.86046512	0	0	0	0	0	0	4	0.465116	8	0.930233	0	0
SB1_M	8	0.93023256	4	0.46511628	72	8.37209302	156	18.1395349	0	0	96	11.1627907	8	0.93023256	16	1.860465116	4	0.465116279	8	0.930233	8	0.930233	0	0
SB1_T	0	0	8	0.93023256	0	0	180	20.9302326	0	0	36	4.18604651	0	0	20	2.325581395	0	0	28	3.255814	12	1.395349	0	0
SB1_Total	12	1.39534884	24	2.79069767	92	10.6976744	468	54.4186047	0	0	148	17.2093023	8	0.93023256	36	4.186046512	4	0.465116279	40	4.651163	28	3.255814	0	0
SB2_B	0	0	20	1.24688279	92	5.73566085	436	27.1820449	0	0	20	1.24688279	0	0	0	0	0	0	0	0	20	1.246883	0	0
SB2_M	0	0	16	0.99750623	96	5.98503741	200	12.4688279	4	0.24937656	32	1.99501247	8	0.49875312	4	0.249376559	4	0.249376559	20	1.246883	4	0.249377	0	0
SB2_T	0	0	32	1.99501247	40	2.49376559	432	26.9326683	0	0	84	5.23690773	28	1.74563591	0	0	0	0	8	0.498753	0	0	4	0.249377
SB2_Total	0	0	68	4.2394015	228	14.2144638	1068	66.5835411	4	0.24937656	136	8.47880299	36	2.24438903	4	0.249376559	4	0.249376559	28	1.745636	24	1.496259	4	0.249377
B3_B	0	0	0	0	8	1.61290323	24	4.83870968	0	0	20	4.03225806	4	0.80645161	4	0.806451613	0	0	8	1.612903	0	0	0	0
B3_M	0	0	0	0	12	2.41935484	136	27.4193548	0	0	32	6.4516129	0	0	8	1.612903226	0	0	0	0	4	0.806452	0	0
B3_T	0	0	20	4.03225806	0	0	104	20.9677419	0	0	80	16.1290323	4	0.80645161	24	4.838709677	4	0.806451613	0	0	0	0	0	0
B3_Total	0	0	20	4.03225806	20	4.03225806	264	53.2258065	0	0	132	26.6129032	8	1.61290323	36	7.258064516	4	0.806451613	8	1.612903	4	0.806452	0	0
SB4_B	0	0	32	4.46927374	0	0	236	32.9608939	0	0	20	2.79329609	8	1.11731844	0	0	4	0.558659218	0	0	0	0	4	0.558659
SB4_M	0	0	16	2.23463687	0	0	116	16.2011173	0	0	40	5.58659218	28	3.91061453	0	0	0	0	0	0	0	0	0	0
SB4_T	0	0	88	12.2905028	52	7.26256983	56	7.82122905	0	0	12	1.67597765	4	0.55865922	0	0	0	0	0	0	0	0	0	0
SB4_Total	0	0	136	18.9944134	52	7.26256983	408	56.9832402	0	0	72	10.0558659	40	5.58659218	0	0	4	0.558659218	0	0	0	0	4	0.558659
B5_B	8	5.55555556	16	11.1111111	28	19.4444444	0	0	0	0	4	2.77777778	0	0	0	0	0	0	0	0	4	2.777778	4	2.777778
B5_M	0	0	4	2.77777778	24	16.6666667	8	5.55555556	0	0	8	5.55555556	4	2.77777778	0	0	0	0	0	0	0	0	0	0
B5_T	0	0	0	0	20	13.8888889	0	0	0	0	0	0	8	5.55555556	0	0	4	2.777777778	0	0	0	0	0	0
B5_Total	8	5.55555556	20	13.8888889	72	50	8	5.55555556	0	0	12	8.33333333	12	8.33333333	0	0	4	2.777777778	0	0	4	2.777778	4	2.777778
B6_B	0	0	8	2.63157895	4	1.31578947	4	1.31578947	0	0	20	6.57894737	0	0	0	0	0	0	0	0	0	0	0	0
B6_M	0	0	12	3.94736842	0	0	24	7.89473684	0	0	44	14.4736842	0	0	0	0	0	0	0	0	0	0	0	0
B6_T	0	0	24	7.89473684	8	2.63157895	0	0	0	0	108	35.5263158	20	6.57894737	16	5.263157895	0	0	0	0	0	0	12	3.947368
B6_Total	0	0	44	14.4736842	12	3.94736842	28	9.21052632	0	0	172	56.5789474	20	6.57894737	16	5.263157895	0	0	0	0	0	0	12	3.947368
specific_sp	20		312		476		2244		4		672		124		92		20		76		60		24	
all_counte	4124																							