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Life after fire: the response of arthropod communities to fire in different types of forests

Vida após o fogo: a resposta da comunidade de artrópodes ao fogo em diferentes tipos de floresta

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Palavras-chave

Resumo

A resposta das comunidades de artrópodes ao fogo em diferentes tipos de habitats florestais foi estudada.

Os artrópodes do solo foram coletados com armadilhas pitfall, enquanto os artrópodes aéreos foram coletados com armadilhas cromotrópicas montadas no nível do solo e em plataformas elevadas em três tipos diferentes de floresta, floresta de carvalhos velhos, floresta de carvalhos jovens e floresta de eucalipto plantada.

As comunidades de artrópodes, com ênfase nos besouros carabídeos, foram comparadas em termos de abundância e diversidade. A coleta foi realizada imediatamente após a ocorrência do incêndio e em outras duas ocasiões durante o período de um ano.

A comunidade geral de artrópodes mostrou um aumento de diversidade na maioria das parcelas de amostragem de outubro de 2017, dezembro de 2018 e março de 2018, com a floresta de eucalipto a suportar menor diversidade. Os carabídeos mostraram uma tendência semelhante, mas apresentaram uma diversidade elevada na floresta de eucalipto incluindo na amostragem imediatamente após a ocorrência do incêndio. Parâmetros edáficos parecem determinar a resposta de diferentes tipos de artrópodes ao fogo, com o teor de água no solo a ser o principal fator que influencia a diversidade de carabídeos na floresta de eucalipto.

Os resultados deste trabalho enfatizam a importância da seleção criteriosa e adequada das espécies de bioindicadores, pois os resultados obtidos com um táxon podem não ser representativos da comunidade em geral.

Abstract

The response of the arthropod communities to fire in different types of forest habitats was studied.

Ground dwelling arthropods were collected with pitfall traps while aerial arthropods were collected with pan traps set at ground level and in elevated platforms in three different forest types, old oak forest, young oak forest and planted eucalyptus forest.

Arthropod communities, with emphasis on carabid beetles, were compared in terms of abundance and diversity. The collection was carried immediately after the fire occurrence and in other two occasions over one-year period.

The overall arthropod community showed a diversity increase in the majority of the sampling plots from October 2017 to December 2018 and March 2018, with the eucalypt forest supporting lower diversity. Carabids did show a similar tendency but had a high diversity in eucalypt forest in the sampling immediately after the fire occurrence. Edaphic parameters seem to determine the response of different types of arthropods to fire, with soil water content being the major factor influencing carabid diversity in the eucalypt forest.

The results of this work emphasize the importance of choosing an adequate bio indicator species as the results obtained with one taxon may not be representative of the overall community.

CONTENTS

INTRODUCTION

In October 2017 a series of wildfires wreaked havoc across mainland Portugal and Spain, causing the death of dozens of people, destroying a number of houses and other infrastructures and burning large areas of forest. Destructive phenomenon like these fires can occur more frequently in the future as a result of global warming and bad forest management, it is thus of extreme importance to understand the impact that these large-scale perturbations have on forest ecosystems and find ways of mitigating these impacts. Insects represent a great percentage of the biomass in the forest ecosystem and some taxa are important bio-indicators of disturbance. Carabid beetles in particular have been used as indicators of the disturbance effects on insect communities (Nunes *et al.*, 2006; Pearsall, 2007; Rykken, Capen, & Mahabiri, 1997).

The following sections will be dedicated to a review of fire in the Mediterranean basin, with particular focus on Portugal. Important aspects of fire history and plant adaptation will be addressed.

In a latter section it will be addressed the relations between arthropod communities and wildfire and the methodology and results obtained in this work.

Fire and the Mediterranean forest

In many parts of the world, fire is a common phenomenon and is an integral part of many terrestrial ecosystems. Forests are ecosystems prone to the fire hazard as they accumulate up to 15 times more biomass than other terrestrial ecosystems (Pyne, 1997). In the Mediterranean, fire has an important ecological role, both as a disturbance and as a regenerative process. In this ecosystem summer droughts produce an annual fire hazard that contributes to a highly predictable fire regime (Keeley, Bond, Bradstock, & Rundel, 2011). This fire regime has shaped the Mediterranean vegetation for thousands of years and has led to the appearance of adaptations that allow plants to survive and prosper in these ecosystems.

Vegetation adaption to fire

Fire as long been regarded as a man-made destructive force and thus in the past there have been few studies of its role in the evolution of Mediterranean vegetation (Naveh, 1975). The studies that have been made indicate that the Mediterranean "fire bioclimate" (Naveh, 1973), which is characterized by long, hot and dry summer seasons with maximum average daily temperatures around 30°C, and sometimes surpassing 40°C, average relative air humidity of 50-60% and frequent heat waves, has created harsh conditions for plants to survive in a post-fire scenario without proper adaptations like the regeneration from underground bulbs and the development of fire tolerant parts, these plant traits have led to the concept of pyrophytism (plant adaptations to fire) first described by Lorant in 1938 (Trabaud, 2002). The different types of adaptations to fire can be divided in two groups (Naveh, 1975) depending on a feedback response.

Positive feedback will correspond to adaptations that help in the after-effects of the fire hazard by increasing physiological responses of the affected vegetation. The most commonly expressed adaptations of these kind are the fire-stimulated resprouting and/or germination, present in species like *Quercus coccifer* L. and *Pinus halepensii* Miller.*,* respectively.

Negative feedback will be regarded as responsible for defence mechanisms that enable protection from the fire hazard, either by direct fire tolerance of seeds or plant organs or by their reduced physiological activity during critical fire periods. *Quercus suber* L. is a good example of these adaptions as it has a thick, insulating bark that makes it well adapted to forest fires (Trabaud, 2002). It is important to note, however, that plants are not adapted to fire but to the fire regime and their survival is always dependent on the alteration of this regime that can increase or decrease fire frequency. These traits that improve plant survival in certain fire regimes are called fire-adaptive traits (Keeley, Pausas, *et al.*, 2011), some adaptations have originated in response to other ecological factors, but have value in fire prone eco-systems, these traits are called "exaptations". To differentiate between the two types of traits is very difficult and will require more research in the future as these concepts are very important not only in an evolutionary perspective but also in an ecological view of the Mediterranean ecosystem for a proper natural resource management.

The Mediterranean Basin is dominated by evergreen sclerophyllous-leaved shrublands, semi-deciduous scrubs, and woodlands, all of which are prone to wide-spread crown fires (Keeley, Bond, *et al.*, 2011). Associated with this type of fire, the plant community will have two types of main responses, associated with the feed-back mechanism discussed above. This main post-fire regenerative strategies include seeders and propagule-persister species in which the plant population will remain in the form of seed or propagule associated with a seed bank. In (Pausas, Bradstock, Keith, & Keeley, 2004) a proper nomenclature was been developed to illustrate these different strategies: Resprouters (R+) are species in which individuals are able to resprout from any plant structure (e.g., rhizomes, root buds); Non-resprouters (R-) are species without the capacity to resprout after fire; Propagule-persisters (P+) are species in which the population locally persist in propagule form (seed, fruit) Propagule-non-persisters (P-) are species in which the propagule does not persist after fire. In this case recolonization would need to occur from dispersed propagules from neighbourhood populations. In the Mediterranean basin the plant community is dominated by non-seeders (R+ P-) in a late successional state, after a disturbance however, it is expected to register a bigger recruitment of seeder species (R+P+,R-P+) (Saura-Mas, Paula, Pausas, & Lloret, 2010).

Other Mediterranean-type ecosystems (MTEs) occur in different parts of the planet: California; Central Chile; the Cape Region of South Africa; and Southwestern and South Australia (Esler, Jacobsen, & Pratt, 2018). In these different regions plants exhibit different characteristics and traits that also make them adapted to the fire prone habitats of the MTEs.

Four of the five Mediterranean type ecosystems (MTEs) are classified as the most fire affected biomes in our planet (Bond & Keeley, 2005) and the floras of these biomes have a history of convergent evolution in their adaptations to the Mediterranean type ecosystems with many species developing traits known as "fire-syndrome" that benefit the plant after fires (Pausas *et al.*, 2004). In spite of the similarities of the METs, there are important differences in the frequency of species that represent a determinate trait or fire related syndrome, this reflects differences in current or past environmental factors such as fire regime, soil characteristics and anthropogenic disturbances (Keeley, Pausas, Rundel, Bond, & Bradstock, 2011). In Ne'Eman, Lev-Yadun, & Arianoutsou, (2012) the different types of fire related traits and syndromes of vegetation in the METs, are revised.

Resprouting is a characteristic present throughout the different METs as it enables plants to persist after fire and other stress inducing factor like herbivory (Trabaud, 1987). Resprountig plants tend to have very well-developed lignotubers. It is the case for vegetation in Australia, South Africa and California but not for vegetation on the Mediterranean basin where lignotubers were only reported in a few species (Keeley *et al.*, 2011). There is little information about why the occurrence of lignotubers is much lower in the Mediterranean basin but it shows resprouting plants have developed different strategies in the MTEs (Ne'Eman *et al.*, 2012).

Serotiny is the delay in seed dispersal by some plant species. This dispersal strategy is scattered along the METs but is most abundant in Australia, South Africa and North America (Lamont, Le Maitre, Cowling, & Enright, 1991). The flora of the Mediterranean basin has only a few serotinous species mainly in the genus *Pinus* L. and *Cupressus* L. (Lev-Yadun, 1995; Thanos & Dousi, 2000).

Seed banks allow plants to persist through fires. Post-fire obligate seeders, which recruit after fire only by seed germination, have soil or canopy-stored seed banks (Goubitz, Nathan, Roitemberg, Shmida, & Ne'eman, 2004). Few species in the Mediterranean basin are obligate seeders. This is in contrast to Australia and South Africa, where resprouters may have soil- or canopy-stored seed banks (Juli Pausas *et al.*, 2004).

Other traits that characterize the flora of the MTEs are heat and smoke en-hanced germinations. The Fabaceae and Cistaceae families are well represented in the Mediterranean basin (Arianoutsou, 1998). Many species in these families are obligatory seeders that typically have physically dormant seeds that germinate after the fire heat shock. Positive effects of heat and smoke on germination of seed banks were found also in Australia (Read, Bellairs, Mulligan, & Lamb, 2000). The effects of smoke in germination to flora from the Mediterranean basin were tested and it was found that effect of smoke on seed germination in the Mediterranean basin flora is less common than in the floras of other MTEs (except Chile) (Ne'Eman *et al.*, 2012).

Annual plants are the flora most affected by smoke in MTEs other than the Mediterranean basin, where perennial non-woody plants and dwarf shrubs seem to be the most affected (Shmida & Ellner, 1983).

It is thus clear that in spite being geographical regions with similar climate and vegetation cover, the different MTEs show some differences in the adaptations, and their frequency, of vegetation to the fire hazard.

Fire doesn't only affect the biotic components of an ecosystem, abiotic con-ditions like soil properties will also be affected by the fire disturbance as it will be discussed below.

Effects of fire on soil

An increase in the frequency and amplitude of wildfires will lead to the deg-radation of soil and have varied effects on soil composition and structure.

Bulk density (the mass of many particles of the material divided by the total volume they occupy) tends to increase after wildfires, as a result of the breakdown

of soil aggregates following the combustion of organic matter. Bulk density in particular seems to be a parameter that lacks some understanding in a postfire view, so in the future researchers should aim to study in depth the effects of high intensity fires on this soil parameter. Soil texture is also a parameter that can be measured after wildfires, with some studies Afif Khouri & Oliveira Prendes, (2006) and Granged, Jordán, Zavala, Muñoz-Rojas, & Mataix-Solera, (2011) reporting significant changes in sandy loam soil, with the increase of sand content as a result of the formation of unstable aggregates. These observations were nevertheless only the result of ephemeral changes as Afif Khouri & Oliveira Prendes, (2006) reported that the sand content had returned to pre-fire level one week after the first observations. Soil texture like other soil parameters seems to be dependent on other soil characteristics like the texture type (Alcañiz, Outeiro, Francos, & Úbeda, 2018). As in the case for bulk density, soil texture as not been the object of study of many researchers so this parameter is also not very well understood in a post-fire scenario.

Wildfires have been found to alter the abundance of nutrients and distributing them within different trophic levels. At the topsoil layer, ashes tend to increase pH values in fact it has been reported that after fire, pH levels can increase up to 4 or 5 units (Ulery, Graham, Chadwick, & Wood, 1995), this happens especially during the first months after fire (Antunes, Curado, Castro, & Gonçalves, 2009; Debano, Dunn, & Conrad, 1977). Other factors associated with the increase of pH values will include OH-losses, the complete oxidation of organic matter during the fire and the release of cations in the soil (Certini, 2005). Electrical conductivity (EC) tends to increase, as well, in the first moths after fire, this happens because of the release of soluble ions during the combustion of soil organic matter and due to the incorporation of ash into the soil (Alcañiz, Outeiro, Francos, Farguell, & Úbeda, 2016; Alcañiz *et al.*, 2018; Berber, Çağatay, & Turgay, 2015; Certini, 2005; Granged *et al.*, 2011). Organic matter content can increase or diminished, depending on fire intensity (Luis, Tárrega, Calvo, Marcos, & Valbuena, 2000). The alteration of pH, EC and nutrient concentration will have varied effects depending on factors like the frequency of fire, vegetation cover and soil characteristics (Certini, 2005). The effects of fire disturbance on soil properties can last for a very long period of time, as it was shown in Alcañiz *et al.*, 2016, 9 years after prescribe fire treatment, pH, total C, total N and

available P were significantly lower than pre-fire values and nutrients concentrations were higher.

History of fire in Portugal

Fire has been used by man as a management tool since early times, with the first evidence of human-induced changes by fire in the Mediterranean dating back to the Neolithic (Naveh, 1975; J. G. Pausas & Vallejo, 2011). In this same period, the Portuguese forest suffered a great transformation from a pristine natural forest to an environment with ever-greater signs of human intervention. Several centuries of severe human pressure such as, burning, cutting, grazing, clearing, terracing, cultivating, and later abandonment of arable portions, have created a strongly human-influenced landscape (Pausas *et al.*, 1999). Studies in palynology and anthracology showed some variation in the tree cover of the Portuguese forests with the almost extinction of the pine (*pinus*) about 4000 BP (before present) which showed the impact of humans activities even before agriculture practices (Figueiral & Sanches, 1998; Figueiral, 1995; J. E. Mateus & Queiroz, 1993; Ribeiro, 2001).

In the last decades, major social and economic changes have influenced the entire Portuguese territory (Azevedo, Gomes, Mendes, Baptista, & Cabral, 2011; Cordeiro, Alcoforado, & Ferreira, 2014). These changes lead to the abandonment of agricultural areas, to the reduction of herds leading to the accumulation of forest fuels consumed by grazing and firewood gathering (Marques *et al.*, 2011; A. N. Nunes & Lourenço, 2018; A. N. Nunes, Lourenço, & Meira, 2016). Mass exodus to other countries as well as to coastal areas within Portugal together with an aging population explain this population decline in forested areas. Associated to these conditions is an increase on the severity of the fire regime that is evident in Portugal for the past decades (Marques *et al.*, 2011). Impacts of this change in fire regime can be negative to the fauna and flora and sometimes put the population in great danger. As an example *Quercus pyrenaica* Willd. forest, have been heavily affected by an increase in fire this is specially worrying as Pyrenean oak is an ecosystem almost restricted to the Iberian Peninsula (Costa M, Morla C, & Sainz H, 1998). The alteration of the fire regime is due to a variety of factors, and includes the changes in demographic distribution, discussed above, climate change and an inadequate

forest management policy. Climate change is altering the environment all around the world and the fire regime is no exception to these changes. The climate plays a very important role in fire occurrence and propagation as it defines the quantity and type of vegetation of each region and it is also responsible for the seasonal dynamics of the vegetation moister content (Pyne, Andrews, & Laven, 1996). Fire severity will depend in a series of climatic factors including seasonal distribution of precipitation, temperature and relative air humidity. All these factors will define a fire prone environment and will influence the fire regime (Durão & Corte-Real, 2006). With the climatic changes of the last century, we can expect an increase in temperatures and longer drought periods in the Mediterranean summer, which in turn will increase the severity of the fire regime with the occurrence of more fires of big dimensions with large ecological and economic impacts. For the last decades, Portugal has been the European country with the highest number of fire occurrences and the highest burnt area (Antunes *et al.*, 2009). It is thus important to understand the changes that have occurred in our forest and in the impact that this new and more aggressive fire regime has on the ecosystem so that government authorities can implement measures to ensure that negative impacts do not aggravate in the future.

Characterization of native and planted forests

Quercus robur L.

One of the forests in the study sites is composed of oak trees (*Quercus robur)*. The Galicio-Portuguese oak forest is under the protection of the Habitats Directive (Council Directive 92/43/CEE), being listed in Annex I under the designation of ("Galicio-Portuguese oak woods with *Quercus robur* and *Quercus pyrenaica* Willd.", habitat code 9230), and constitutes climatic habitats in their entire area of distribution which includes France (Eurosiberian region), Spain and Portugal (Eurosiberian and Mediterranean region). In Portugal they are distributed in the north of Mondego in the Cantabrio-Atlantic Province and above the 600 m in the Carpetano-Iberico-Leonesa Province, in Alto Alentejo in areas above 450 m in Toledano-Tagano sector and in the rainy Gaditano-Onubo-Algarvia Province (ICNF, 2008).
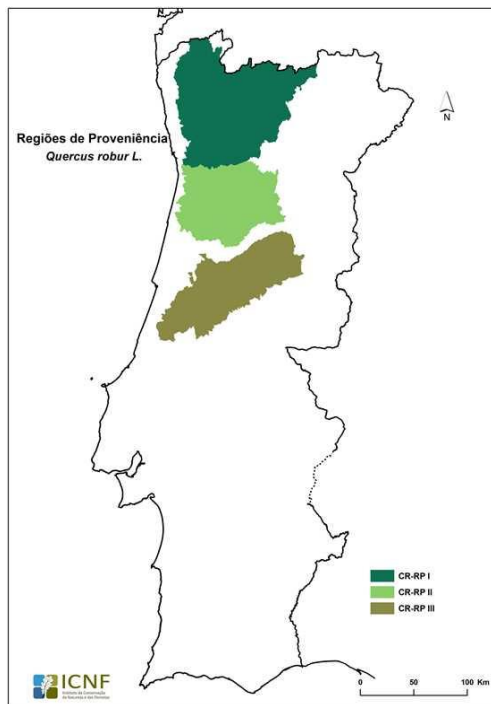
Figure 1 - Delimitation of the regions of provenance of *Quercus robur* L., commonly known as common oak, pedunculated oak or European oak. (Information collected centrally by ICNF and crossed with CAOP - Carta Administrativa Oficial de Portugal).

Galicio-Portuguese forests were formerly abundant in the Portuguese territory, but their area of occupation was severely reduced by human activity. Presently their abundance is slowly increasing due to natural regeneration in abandoned agricultural fields (ICNF, 2008).

Deciduous oak forests are characteristic of hill and montane belts of the Iberian Peninsula. However, these forests were object of severe deforestation due to their presence in areas with special interest for farming and pastures and to their high quality wood (Castro *et al.*, 2011).

Querus robur characterization

Quercus robur oaks show some specific characteristics that make them unique in the Portuguese forest. These trees can grow to be up 12 m and have a canopy which is more or less regular and round shaped. The leaves are deciduous and have round lobes with both the leaves and the twigs being hairless. *Q. robur* has unisexual flower and the flowering occurs between April and May, with the male and female flowers having different times of flowering to avoid self-fertilization. The pollination for these flowers is anemophilous with pollen being distributed by the wind. The fruit is an acorn and it matures in the autumn (Nuno, 2009).

Response to fire

Fire is one of the main causes of deciduous oak forest degradation. Understory communities may suffer compositional changes causing the loss of biodiversity after fire (ICNF, 2013). In Proença (2009), both resistance and resilience of broadleaf woods (*Quercus robur* and *Ilex aquifolium* L.) recorded higher values than in the case of pine plantations (*Pinus pinaster* Aiton. and *Pinus sylvestris*) this was consistent with the results in (Pausas, Llovet, Rodrigo, & Vallejo, 2008). It was hypothesised that the fuel characteristics (amount, shape, arrangement, water content and chemistry in pine plantations, potentiated the fire extent and in the case of the broadleaf woods higher moisture content tended to decrease the fire severity in the study plots (Bond & Midgley, 2001; Castro *et al.*, 2011). Thus, the importance of this broad leaf trees becomes evident as they contribute to a less destructive fire hazard and should be taken into account for their potential to conserve local biodiversity during wildfires. Contributing to this view of *Q. robur* trees as important refuges for wildlife is the knowledge that these trees serve as habitat for many species of animals, other plants and fungi (Carvalho *et al.* 2007).

The *Eucalyptus globulus* forest

The eucalypt or Tasmanian blue gum (*Eucalyptus globulus* Labill) is a tree species native to SE Australia that was introduced in South Europe and in many European colonies in the XIX century. These forest trees make up the greatest area of non-native forest in Europe, mostly located in the Iberian Peninsula (Silva & Tomé, 2016).

In Portugal the expansion of these species occurred extensively in the mid-twenty century driven by the development of the pulp industry (Goes, 1977; Oliveira, Guiomar, Baptista, Pereira, & Claro, 2017).

Eucalyptus globulus continues to expand to this day in Portuguese territory occupying now approximately 26% of the total forest area and 9.1% of Portugal continental surface (ICNF, 2013; Meneses, Vale, & Reis, 2014). Pulp companies manage 20% of the plantations, the remaining being owned and managed by private owners (Silva & Tomé, 2016). This makes *E. globulus* the most widespread tree in the Portuguese mainland.
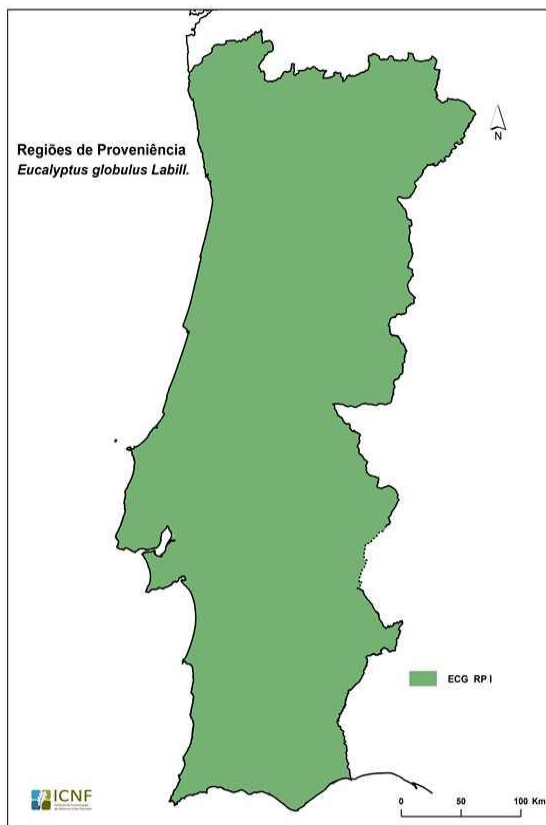
Figure 2 Delimitation of the regions of provenance for *Eucalyptus globulus*.
(Information collected centrally by ICNF and crossed with CAOP - Carta Administrativa Oficial de Portugal).

Risk assessments identifying this species as having a high environmental risk both for the likelihood of invasion and for its socioeconomic and ecological impacts (Gordon, Flory, Cooper, & Morris, 2012).

Eucalypt plantations and fire

The factors contributing for the great expansion of this exotic species include, burning of native forest by wildfire in the past decades (~3% per year) and as a consequence, forest area decreased 10% (Mateus & Fernandes, 2014); Posterior substitution of pine forest by eucalyptus forest; rural exodus since mid-XX century) and the capacity that *E. globulus* trees have of colonizing new habitats after disturbance (Fernandes, Guiomar, & Rossa, 2019)

Fire is often related to an increase in eucalypt recruitment as it was already studied by several authors (Ashton, 1981; Cremer, 1965; Gill, 1997; Mount, 1964) Fire facilitated recruitment in eucalypts is related with: increased seed shed from canopy; increased light availability; ash-bed effect; reduced competition; removal of allelopathic substances and decreased predator activity (Chambers, D.P. & Attiwill, n.d.; Jacobs, 1955; O'Dowd & Gill, 1984; Pryor, 1976; Stoneman, Dell, & Turner, 1994; Wellington & Noble, 1985). Although this species is well adapted to the Mediterranean fire regime, in particular in Portuguese territory, in a recent study Fernandes *et al*., (2019), it is proposed that even with the crescent expansion of eucalypt forest in the last decades in Portuguese mainland, the burned area occupied by this species did not increase. It is known that in Portugal the burned area has increased exponentially in the last decades, but (Fernandes *et al*., 2019) concluded that this burned area is composed mainly of native forest and not of *E. globulus*.

E. globulus and biodiversity

Eucalypt plantations have been shown to arbour less plant biodiversity than native forest. At a local scale biodiversity tends to be lower in eucalypt plantations when compared to native forest, especially at an intermediate stage of development (Calviño-Cancela, Rubido-Bará, & van Etten, 2012). Plant species biodiversity seems to be higher in young stages of eucalypt plantations due to a high turnover when young (when it is similar to shrublands) than in mature eucalypt plantations (when it is more similar to pine plantations) (Calviño-Cancela *et al.*, 2012). Studies that compare biodiversity between native and introduced forests are crucial as the number of exotic plantations grow across the planet as a consequence of higher demand for natural resources (Goes, 1977). As introduced forests grow, the native forests become more fragmented (FAO, 2010; Fernandes, Loureiro, & Botelho, 2004). It is crucial to grow our understanding on the ecology of exotic forests and their potential to harbour native biodiversity, as the only way to favour biodiversity at a regional scale is to improve connectivity between these different forests (Fernandes *et al.*, 2004).

In spite of some studies addressing biodiversity in eucalypt plantations (Calviño-Cancela, López de Silanes, Rubido-Bará, & Uribarri, 2013; Calviño-Cancela *et al.* 2012), these studies focused on plant community and it is important to have a good knowledge of the ecosystem biodiversity in different trophic levels in order to implement good measures for proper natural resources management.

Few studies have focused on the macro-arthropod diversity associated with *E. globulus* plantations (Cordero–Rivera, Martínez Álvarez, & Álvarez, 2017) and since this group of organisms is a crucial part in the food web of any terrestrial eco-system, it is of great value to understand the dynamics between these organisms and eucalypt plantations.

Insects and wildfires

Insects are the most abundant animal group on the planet and have great importance in the ecological organization and processes of almost all terrestrial communities. This group of organisms has adapted to the great majority of terrestrial ecosystems were they have a very wide range of functions such as, pollination, pest control, nutrient cycling or serving as food for other animals. These are collectively known as Ecosystem Services (ES) (Ameixa, Soares, Soares, & Lillebø, 2016). Insects may also be used as bio-indicators of ecosystems health as they have large populations, short generations and are sensitive to habitat changes (Refseth, 1980). Carabid beetles in particular have been shown to be affect by abiotic changes as their distribution is driven by habitat and microclimate effects (Gillingham *et al.*, 2012; Rainio & Niemela, 2003).

Insects and other arthropods can be affected by fire, either directly (e.g. heat and oxygen depletion) or indirectly through changes in soil characteristics and vegetation (Wikars & Schimmel, 2001), in either case, arthropod communities are affected through changes in community structure and composition (Andersen *et al.*, 2005). These changes will affect certain species negatively or positively, depending on the habitat requirements (Taboada, Kotze, Tárrega, & Salgado, 2006).

Among ecological communities insects are important bio-indicators of fire disturbances as they are sensitive to the environmental changes caused by fire (Elia, Lafortezza, Tarasco, Colangelo, & Sanesi, 2012). Assessing the impacts that disturbances like fire have on arthropod assemblages, as well as in single species becomes thus a key complementary need for conservation of insect populations and to address the changes in ecosystem functions that follow such disturbances. Impacts of fire in the arthropod communities are very difficult to predict as it is a complex aggregate of factors like the natural assemblages before the disturbance, the biology of the species that make up the community (e.g., resistance to heat, dispersal capability, life cycle) and the ecological interactions (Antunes *et al.*, 2009; Friend, 1994; Samu *et al.*, 2010). Developing a predictive understanding of how species assemblages respond to wildfire is thus a key conservation goal (Langlands,

Brennan, Framenau, & Main, 2011). Although it is difficult to predict the impacts of fire on arthropod populations there are a series of fire impacts that can be noticed in a large amount of the studies done so far on this subjected which include (1) elimination of individuals, leading to populations extinction; (2) elimination of groups sensitive to fire; (3) reduction of resources which leads to increase in competition pressure; and (4) reduction of resource heterogeneity, leading to extinction of feeding specialists (DeSouza, Albuquerque, Tonello, Pinto, & Reis, 2003; New, 2014). A great effort in being made to understand the population's dynamics after these impacts and the way that arthropod communities recover after such disturbance. To ecological management it is of great importance to understand particularly the circumstances that lead to the extinction of specialists which leads to a community with much less biodiversity and dominated by generalist species.

The recovery of the areas burned by wildfire takes place by the activity of surviving individuals, along with re-colonisation from adjacent areas after the fire (Wikars & Schimmel, 2001). While fire causes mortality in the majority of insect community, post-burned vegetation can be very attractive, especially to localized species of some families of beetles (Buprestidae, Silphidae, Cerambycidae) and flies for example (Empididae and Platypezidae) (New, 2014; Swengel, 2001).

When studying the impact of fire on arthropods there are two concepts to be considered "resistance and resilience" (Moretti, Duelli, & Obrist, 2006). To analise the concept of resistance we most look to the level of similarity, in terms of diversity between the community before and immediately (<1 year) after the fire more resistant communities will show high levels of similarity, meaning that the majority of the individuals did survive and therefore are resistant to this kind of disturbance. Many arthropod species decline markedly after fire showing that they have little resistance to this phenomenon. Resilience reflects the rapid recovery of the species composition to that prior to the fire. (Andersen, 1991; Niwa & Peck, 2002; Saint-Germain, Larrivée, Drapeau, Fahrig, & Buddle, 2005; A. Swengel, 2014; S. Warren, Scifres, & Teel, 1987). In spite of the immediate decline, certain groups seem to be less affected than others, for species below ground, within or beneath unburned wood, or above flames in canopies the decline in abundance is of no expression in the majority of the analysis (Warren & Nichols, 1996), in contrast, the depletion of

floral resources will inevitably lead to changes in pollinator communities (Potts, Dafni, & Ne'Eman, 2001). Factors like post-fire flowering and re-colonization will certainly influence the recovery of the community. Many species have been documented surviving even the most destructive fire, so we must be careful when documenting 100% mortality, as this is an unlikely scenario, sometimes even for very exposed species (Emmel & Daniels, 1997; Panzer, 1988).

The post-fire conditions will certainly influence the survival of individuals that escaped the flames, it is known that fire promotes homogenization of the habitat and only species that can endure the xeric conditions with very little food available, would be capable of colonizing and proliferate in the post-fire scenario (Lamotte, 1975). Influencing the post-fire abundance and arthropod diversity are factors like the fire intensity, the availability of refuges and the possible behaviour of some species that can escape the flames by means of dispersal. Depending on the fire intensity some groups of organisms can show, or not, in the post-fire samples. An example of a group of arthropods that have shown resistance to fire, by means of a refuge, are ground nesting bees from the families Andrenidae and Halictidae. Cane & Neff (2011) hypothesised that bees that nest deeper than 10 cm should be relatively safe from fast moving surface fires. They also analysed data from 445 species of bees for which there is nest deep information and concluded that only 9% of species would be in danger through soil heating. This study shows the great capacity that insects and other arthropods have to survive and recolonize areas affected by wildfire and that seem, to our eyes, to be barren lifeless areas without resources.

Epigeic arthropods tend to be the most affected by fire depending, on the fire intensity. The total recovery of some food webs can be estimated to take decades after the initial disturbance (Pryke & Samways, 2012a). In spite of the disturbance in the ecosystem, caused by fire, some groups tend to increase their abundance immediately after. These species that are benefited by fire tend to prefer open and more homogenized habitats (Moretti *et al.*, 2006). Disturbances such as fire will cause a profound impact on the habitat by altering the availability of niches and resources (Shea, Roxburgh, & Rauschert, 2004). Generalist species will be favoured by more disturbed and homogenized habitats where specialist species seem to be more abundant and diverse before the fire disturbance (Futuyma & Moreno,

1988). Ants (Formicidae) stand out as one of the groups that most benefit from wildfires. These insects are fundamental components of many ecosystems as they play important roles in the food web and contribute significantly to the modification of biotic and abiotic conditions (Hölldobler & Wilson, 1990) since these are considered to be ecosystem engineers. Ants are known to be pioneer organisms and their high mobility allows them to rapidly colonize habitats that have been disturbed by wildfire, several studies have proven this tendency for high abundance of ants after the fire disturbance (Andersen, 1991; Antunes *et al.*, 2009; Lázaro-González, Arnan, Boulay, Cerdá, & Rodrigo, 2013; Pryke & Samways, 2012a; Punttila & Haila, 1996). Although the high temperatures are lethal to ant species that nest on vegetation, these temperatures are not enough to cause high levels of mortality in colonies that nest underground (Arnan, Rodrigo, & Retana, 2006). Therefore, depending on the ant species and biotic and abiotic conditions fire will have a negative or positive effect on ant assemblages. Omnivorous ants, for example, will have an easier adaptation to the new conditions in the early successional stages in the post-fire scenario as they will quickly adapt their diet to the available resources (Lázaro-González *et al.*, 2013).

Other taxa that have also been studied in post fire scenarios as bio-indicator of wildfire disturbance in ecosystems, include spiders (Araneae), Staphilinidae and Carabidae (Coleoptera).

Spiders are an hiper-diverse group of organisms just like insects, and other arthropods (Barriga, Lassaletta, & Moreno, 2010), with Melic (2011) estimating that about 25% or more of Iberian spider species still remained to be discovered. A recent study (Branco, Morano, & Cardoso, 2019) makes an update of the spider species in Portugal as part of a national effort to create a IUCN red list for arthropods.

Spider distribution after wildfires has been related mainly to physical factors being solar radiation considered the most important as it influences ambient temperatures, soil moisture and air humidity having also important roles in the distribution after fire (Huhta, 1971). Their richness on the other hand will be determined by factors like vegetation structure, reflecting the different hunting strategies of ground based and web-building species (Grill, Knoflach, Cleary, & Kati, 2005). In spite of this, the majority of studies focusing on spider assemblage tend to focus on boreal

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forest (Buddle, Spence, & Langor, 2000) and few studies were made in the Mediterranean spiders response to fire (Sachinoglou, Georgiadis, Chatzaki, & Legakis, 2012), this thesis will therefore be important to bring more knowledge about the subject.

For carabid beetles, the taxa subjected to a deeper taxonomic identification in this work, it has been shown that the abundance of specialist species tends to decrease in favour of a crescent abundance of species typical of open habitats (Niemela, Langor, & Spence, 1993; Niemela & Spence, 1992; Niemela, Spence, Langor, Haila, & Tukia, 1993; Werner & Raffa, 2000). Elia *et al* (2012) and Nunes *et al* (2006) showed that carabid beetle diversity decreased in a two year period post-fire. But other studies showed different results, with an increase of carabid beetle and, other soil arthropods, diversity post-fire (see: Warren *et al.*, 1987). Major environmental factors like pre-fire vegetation complexity, microclimatic conditions and soil and litter layers conditions tend to influence the assemblage of coleoptera in a post-fire scenario (Niemela *et al.*, 1993). Different results in terms of biodiversity for carabid beetle and other arthropods can be observed depending on the pre-fire conditions and the species that recolonize or survive in the burned habitat. Colonization by macropterous species is very common as they easily recolonize areas disturbed by fire and can easily that advantage of the resources turned available by fire (Honek, Martinkova, & Jarosik, 2003; Samu *et al.*, 2010). Carabid beetles as well as staphylinid beetles are known to be generalist and opportunistic hunters, but some species are known to be specialized granivores (Honek *et al.*, 2003). These specialized species should be less abundant in burned plots were the habitat would favour much more generalist species. As in the case for spiders little is known about carabid and staphylinid beetles response to wildfires in the Mediterranean, with few works addressing this subject (Nunes *et al.*, 2006; Pryke & Samways, 2012b).

Carabid body mass (MIB – Mean individual body mass) has been successfully used as a variable that shows the response of carabid beetles to a habitat perturbation and to habitat ecosystem maturity. Smaller individuals with higher dispersion abilities are typical of immature habitats, and their abundance is higher, while mature habitat present a higher abundance of bigger brachipterous individuals (Spence J.R., Langor D.W., Niemelä J., 1996).

Objectives

Fire has been a constant in Mediterranean forests, shaping the evolution of many plant and animal species and communities. However, the introduction of exotic an flammable species such as Eucalypt, which due to factors such as global climatic changes has modified the dynamic of fire regimes in Portuguese forests, becoming more frequent and more severe with impacts in plant and animal communities.

This work aimed to test if native forest plots support higher biodiversity and abundance of arthropod fauna than eucalyptus forest plots in the post-fire, and determine if the latter is characterized by species typical of habitats in lower succession stage composed mainly by generalist species.

With this type of analysis the work also aims to evaluate carabid beetles as indicators of disturbance in forest habitats.

The results may lead to important conclusion about forest, fire and arthropod community's dynamics as well as introduce new data in the characteristics of eucalypt forests and their capacity to support native wildfire in a post-fire scenario.

MATERIALS AND METHODS

Study Area

This study was conducted in two localities of the municipality of Vouzela, Portugal, in Póvoa Pequena (40°39'55.0"N 8°06'29.0"W; 860 m above sea level) and in Carvalhal de Vermilhas (40°39'1.75"N 8°8'16.84"W; 880 m above sea level), in 3 different sampling actions on October 2017, March 2018 and December 2018.

The municipality of Vouzela is part of the Galicio-Portuguese oak woods (ICNB, 2006) which presents moderate summer maximum temperatures, high relative humidity throughout the year and abundant rainfall (> 1000mm) (Pereira, Marta, & Peixoto, 2011). In this region, the most common tree species are *Quercus pyrenaica* Willd, *Q. robur* L., *Q. suber* L., *Q. rotundifolia* Lam., *Pinus pinea L.*, *Arbutus unedo* L., *Acer pseudoplatanus* L., *Rhamnus alaternus* L. and *Ilex aquifolium* L. (Telles & Cabral, 1999). However, the vegetation that currently occurs in most of the municipality territory, differs somewhat from the vegetation spontaneous or characteristic in the region. In fact, the present plant community composition in the territory, with low shrub vegetation *Erica scoparia* L., *Ulex europaeus* L. and *Ulex minor* Roth*.*, *Cistus ladanifer* L. and *Pterospartum tridentatum* L., will be directly related and dependent on human occupation (Pereira *et al*., 2011). *Eucalyptus globulus* Labill. stands, are also very abundant in the area and were used as a sampling area.

In Póvoa Pequena transepts 1 and 2 were established in a managed forest of old oak trees (*Quercus robur*) and in Carvalhal de Vermilhas transept 3 was established in a managed stand of eucalyptus (*Eucalyptus globulus*) and transepts 4 and 5 in a young forest of oak trees (*Quercus robur*).The study area presents some heterogeneity in terms of vegetation and terrain with the Carvalhal de Vermilhas area presenting a much more rocky terrain than the Póvoa Pequena area. The area of Carvalhal de Vermilhas also presented a series of streams that covered particularly the Eucalypt forest and make it much wetter. All the areas surveyed in the study, were affected by the wildfires which occurred in October 2017, which left little to no vegetation cover in the sampled plots. These particular wildfires were a series of more than 7,900 forest fires which affected Northern Portugal and North-western

Spain between 13th and 18th of October 2017. The wildfires claimed the lives of 48 people in Portugal and burned half a million hectares of forest and properties, corresponding to 5,6% of the country's territory (Comissão Técnica Independente, 2017).

Fire characteristics

The fire that affected Vouzela on 15th of October 2017 resulted from 2 separated ignition points in Macieira de Alcoba that interacted together (Viegas *et al.*, 2019). The main factor in this fire was the wind although the topography of the burned area with a great amount of dry vegetation did play a crucial part in the fire propagation also.

The sample sites are located at a considerable distance from the ignition points. The plots in Carvalhal de Vermilhas are located approximately 10,64 km from the ignition points and the plots in Póvoa Pequena are located at 13,35 km from the ignition point.

Due to the fire intensity it was not possible to get unburned control areas in any of the sampling periods as the landscape heterogeneity would greatly influence the results. Pre-fire controls were also impossible to obtain as these wildfires were "natural" and not prescribed fires.
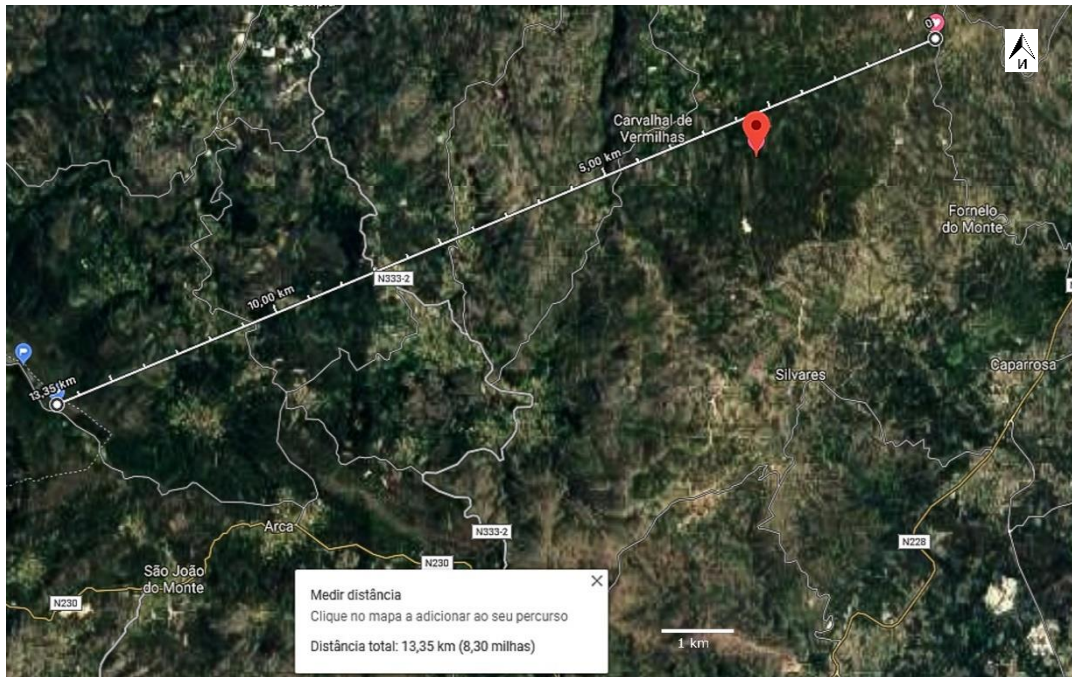
Figure 3 - Distance between ignition points and the sampling area in Póvoa Pequena. *Google maps*, 2019
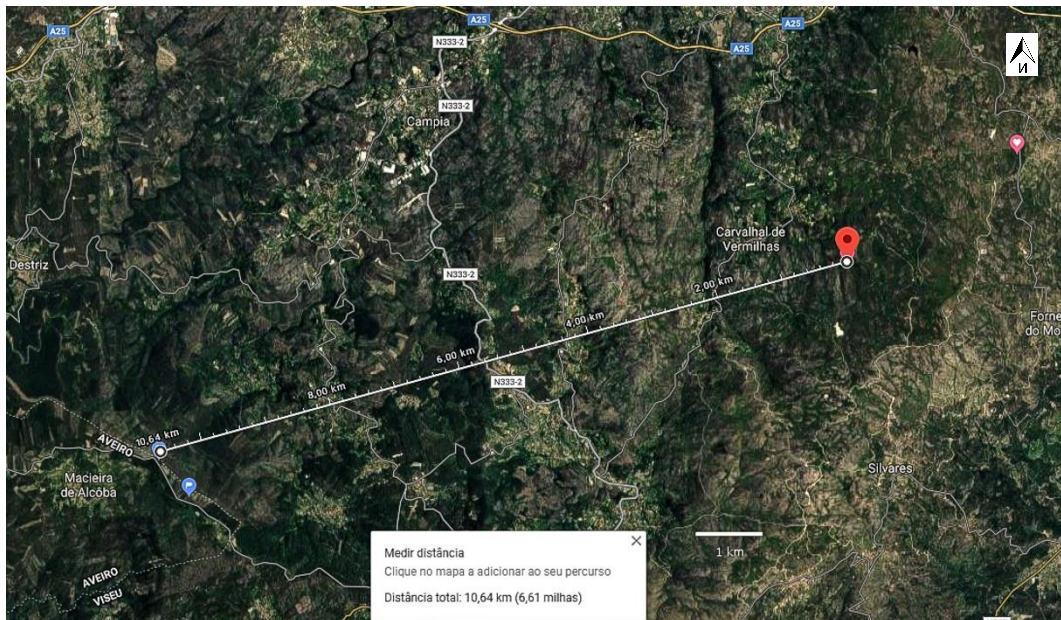
Figure 4 - Distance between ignition points and the sampling area in Carvalhal de Vermilhas. *Google maps, 2019*

Arthropod Sampling

Epigeic arthropods were sampled using pitfall traps, which were placed along 100 m transepts in the areas described above. Traps were placed in plots separated by 20 m each containing 3 pitfall traps, to account with possible lost due to adverse weather conditions or other disturbances, each of these sub-plots was at least 5 m apart. Each pitfall trap consisted of a cylindrical plastic container with an approximate 12 cm diameter and 15 cm height. A mixture of water and 10% of ethylene glycol was placed in the pitfall traps in order to capture and preserve the organisms. Some drops of detergent were added to break the superficial tension of the water, to avoid insects to escape and enable them to drown. To minimise rainfall entrance and prevent the capture of small vertebrates, plastic roofs were added to each pitfall trap (Fig.5). For pollinators, pan traps were placed in the same plots, in the ground 3 plates of different colours (blue, white and yellow, see Fig. 7) with about 5 meters distance from each other. In addition, 3 pan traps with the same colours were placed in a platform, at about 1.5m above the soil (see, Fig.8). All pan trap types were filled with water and a drop of detergent. The pitfalls and pan traps were then left in the field for 7 days before their content was collected.

Figure 5 - Pitfall trap.
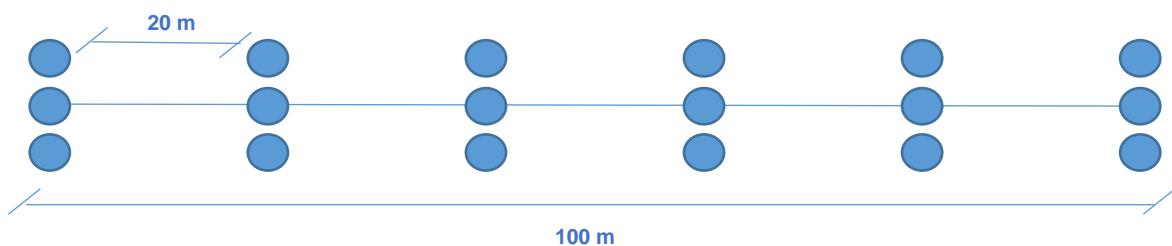
Figure 6 - Pitfall trap sampling scheme

The trapped specimens were collected and preserved in 70% ethanol. Sampled carabid beetles were identified to the species (morphospecies) level using identification keys (Aguiar & Serrano, 2012; Asheley, Springs, & Sinclar, 2017; Barrientos, 1988; Chiney, 2012; Henri Goulet & Huber, 1993; Oosterbroek, 2015). The remaining taxa were identified to the family level. Assemblages were compared between different plots in terms of diversity (Simpson's diversity index) and abundance (Number of individuals).

Figure 7 – Pan traps of different colours set in a sampling plot.

Figure 8 – Aerial pan traps set in a sampling plot.
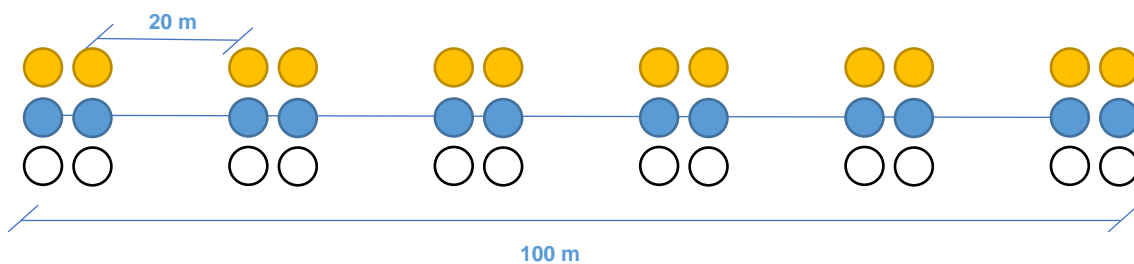
Figure 9 - Chromotropic traps sampling scheme

Soil parameter Analysis

Soil samples were taken from the places were the pitfall traps were placed at a depth of 0-20 cm. Fresh soil weigh was recorded for all soil samples. The soil samples were left to dry at room temperature and weighted until they reached a constant weight, which was recorded as the weight of dried soil. The moist content of the soil samples was obtained by subtracting the dried soil weight from the fresh soil weight.

Other soil parameters (pH, conductivity, salinity and organic matter) were also determined and served as co-variables for the data analysis. First the dried soil samples were sieved with a 2 mm sieve and placed in a solution of 3 g soil and 15 ml distilled water (1 soil:5 water). Flasks with the solution for each of the soil samples were then placed in a mechanical stirrer for 30 minutes at 90 rpm, after this the solutions were allowed to rest for 5 minutes and the readings for conductivity and salinity were taken, the solutions were then allowed to stand for an additional 2 hours and the pH readings were taken. All these measurements were done with a Multiparameter Waterproof Meter HANNA INSTRUMENTS® HI-98194.

Organic matter content was recorded after placing metal crucibles, with dried soil samples (≤2 mm), in a muffle furnace. Sediment that was added to the crucibles was weighted. This weight subtracted to the weight of the metal crucibles represents the fresh weight. After determining the fresh weight, the crucibles containing the soil were placed in a muffle furnace at 105ºC for 48 hours. After cooling off, each crucible with a soil sample was weighted. The dry weight was thus obtained. The same samples were later placed again in the muffle at 500ºC for 6 hours, in order to burn all the organic matter contained in the soil samples. After cooling off, each sample was weighted again and the difference between the dry weight and the weight obtained after burning the organic matter, give us the percentage of organic matter which was lost on ignition (LOI %).

Data Analysis

As there was no control in this study I can only evaluate the samples immediately after the wildfire and compare them with the other samples taken in the remaining time of the study.

Simpson's diversity index was used to calculate the diversity at each sampling point in the correspondent sampling period. This index was previously applied to determine arthropod diversity using families instead of species (Siemann, Haarstad, & Tilman, 1997). The index was also used to calculate carabid diversity separately.

Simpson's index uses the abundance of each group in a sample to determine the proportion that each of these groups of organisms contributes to the total assemblage and can be expressed as:

$$D = 1 - \left(\frac{\sum n(n-1)}{N(N-1)} \right)$$

Where n = number of individuals of each species and N = total number of individuals of all species.

In addition to average individual size (mm) and wing morphology data, the Mean Individual Biomass (MIB, Szyszko, 1983) was also calculated using:

$$Ln\ y = -8.92804283 + 2.5554921 \times ln\ x,$$

where x is the body length of a single carabid individual and y denotes its biomass.

To analyse the correlation between MIB and carabid abundance, habitat type and wing morphology, a spearman rank correlation was calculated using SPSS (Nie, Bent, & Hull, 1970).

To study the relation between soil parameters and carabid diversity, a multi-linear regression analysis was performed, D' diversity of carabid beetle was the dependent variables and soil parameters (pH, Electrical conductivity (EC), water content (%) and organic matter (%)) were de independent variables.

A Prais–Winsten estimation was performed to deal with positive autocorrelation between the residuals.

Distribution of carabid beetles abundances were also analysed using canonical correspondence analysis (CCA) using the software "PAST" (Hammer, Harper, & Ryan, 2001).

Carabid abundance was tested for homogeneity of variance using the Leven's test before being subjected to ANOVA analysis. Whenever homogeneity of variances was not obtained, a Kruskal-Wallis analysis was applied to check for differences in carabid abundances and diversity in the different sampled habitats. Overall arthropod family diversity was subjected to the same analysis as a means of comparison.

RESULTS

Soil Samples

The soil parameters data is exhibit in Table 1.The soil mean electrical conductivity (EC) values in October 2017, immediately after the fire, have shown to be lower in transept 4 (young oak forest) 41.27 ± 21.50 µS/cm and higher in transept 3 (Eucalypt forest) 77.38 ± 48.40 µS/cm. Transepts 1, 2 and 5 presented values of 63.97, 58.97 and 58.67 µS/cm respectively. In the March 2018 samples mean EC values, changed, with the higher values now being recorded in Transept 5 (young oak forest) 71.93 µS/cm and the lowest continuing to be in T4 54.81 µS/cm. Transept 1 to 3 presented values of 68.9, 56.63 and 62.52 respectively. In December 2018, EC values lowered significantly in all sampling points with highest and lowest values being recorded in transepts 1 and 2, with 20.22 and 9.72 µS/cm, respectively. In transepts 3, 4 and 5 the recorded EC values were 12.94, 14.28 and 14.50 µS/cm, respectively.

Water content was apparently influenced by the rainy season after the initial fire disturbance (October 2017) with the highest value recorded in transept 1 (10.85%) and the values of the remaining transepts ranged from 2.75% to 5.11%. On April 2018 the highest water content value was 31.11% in transept 3 and the lowest value was recorded in the transept 1 soil samples, the values of the remaining transepts ranged from 17.09 to 24.23%. On the December 2018 transept 3 remained the transept with highest water content percentage, with 32.87% and the lowest was transept 2 with 12.69%, the reaming values for transepts 1,4 and 5 were 24.84; 21.93 and 19.10 respectively.

Regarding the pH values these were higher in the samples collected in October 2017 ranging from 4.52 to 5.17. These values dropped in the samples collected in April 2018 in all sampling plots except for transepts 2 and 3 which recorded values of 4.92 each. The December 2018 soil samples recorded the lowest values, generalized, for pH levels in all sampling plots, except for transept 1.

Percentage of organic matter content was highest in the October 2017 samples which were the first samples immediately after the fire, the highest value for this parameter was recorded in transept 3 with 28.20% and the lowest was recorded in transept 4 with 16.70%. The values of organic matter percentage in the soil tended to stabilize and become more homogeneous in all sampling plots with the values in the December 2018 sample being around 12% in all transepts.

Table 1 – Soil parameters mean values (±SD).

Forest type	Transept	Month	Electrical Conductivity	pH	Salinity	% Water Content	% Organic Matter
Old Oak Forest	1	October 2017	63.97 (±23.97)	5.17(±0.97)	0.00 (±0.00)	10.85 (±10.84)	21.06 (±13.85)
Old Oak Forest	2	October 2017	58.97 (23.74)	4.52 (±0.61)	0.00 (±0.00)	2.75 (±1.26)	19.11 (±4.21)
Eucalypt Forest	3	October 2017	77.38 (±48.40)	4.57(±0.67)	0.00 (±0.00)	4.77 (±1.09)	28.20 (±26.39)
Young Oak Forest	4	October 2017	41.27 (±21.50)	5.22 (±0.48)	0.00 (±0.00)	4.48 (±2.85)	16.70 (±2.34)
Young Oak Forest	5	October 2017	58.67 (±30.38)	5.35 (±0.53)	0.00 (±0.00)	5.11 (±1.94)	17.85 (±9.04)
Old Oak Forest	1	March 2018	68.89 (±25.87)	4.71(±0.24)	0.00 (±0.00)	16.88 (±5.09)	10.86 (±2.83)
Old Oak Forest	2	March 2018	56.63 (±14.49)	4.92 (±0.24)	0.00 (±0.00)	17.09 (±4.61)	12.52 (±2.68)
Eucalypt Forest	3	March 2018	62.52 (±14.02)	4.92 (±0.17)	0.00 (±0.00)	31.11 (±8.69)	12.82 (±4.06)
Young Oak Forest	4	March 2018	54.81 (±15.39)	4.78 (±0.28)	0.00 (±0.00)	19.24 (±5.14)	10.25 (±2.46)
Young Oak Forest	5	March 2018	71.93 (±25.62)	4.73 (±0.35)	0.00 (±0.00)	24.23 (±4.13)	13.15 (±3.98)
Old Oak Forest	1	December 2018	20.22 (±22.28)	4.98 (±0.43)	0.00 (±0.00)	24.84 (±9.46)	12.52 (±3.97)
Old Oak Forest	2	December 2018	9.72 (±3.46)	4.75 (±0.14)	0.00 (±0.00)	12.69 (±4.70)	11.49 (±3.32)
Eucalypt Forest	3	December 2018	12.94 (±2.78)	4.71 (±0.22)	0.00 (±0.00)	32.87 (±8.38)	12.79 (±3.28)
Young Oak Forest	4	December 2018	14.28 (±6.08)	4.46 (±0.26)	0.00 (±0.00)	21.93 (±5.88)	12.22 (±2.33)
Young Oak Forest	5	December 2018	14.50 (±5.55)	4.49 (±0.28)	0.00 (±0.00)	19.10 (±5.10)	12.11 (±2.31)

Overall arthropod community

Pitfall traps captured a total of 5311 individuals (excluding collembola, psocoptera, mites or thrips) from a total of 86 arthropod families (Appendix A; B and C). Pan traps captured a total of 17669 individuals (excluding collembola, psocoptera, mites or thrips) from a total of 95 families. Carabid beetles were identified to species and a total of 24 species were successfully identified.

In October 2017 (Appendix A), only pitfall traps were used for sampling and the assemblages were dominated by Staphylinidae beetles in all sampling plots with the relative abundance ranging from 54.94% in transept 3 to 66.67% in transept 1. Other abundant arthropod families captured in the samples from October 2017 were spiders from family Dysderidae and ants (Formicidae). Dysderidae greatest abundance occurred in transept 3 and 5 with values around 15% representation and the greatest Formicidae abundance was recorded in transepts 2, 3 and 4 with values of representation of 8.4; 11.59 and 9.35% respectively. Other major abundant groups in this sampling period were spider family Agelenidae (6.2%) in transept 2 and Curculionidae beetles in transepts 3 (7.73%) and 4 (5.76%).

In March 2018 (Appendix B), Staphylinidae beetles remained the dominant taxa in terms of abundance, but only in transepts 1(49%) and 2 (44%). Formicidae became the most abundant taxa in the remaining sampling plots, particularly in transept 5 were they made up to 70% of the pitfall assemblage in this location. Other groups that showed high abundances in these samples were carabid beetles in T1 (12%), Dysderidae in T3 and T4 (14%) and flies from family Ephydridae in T3 (15%). Over this sampling period, the pan traps placed at ground level showed very high abundance of beetles from family Nitidulidae and flies from family Anthomyiidae with the relative abundance of these families sometimes surpassing 30% of the total assemblage (Appendix B), other groups with high representation percentage in these samples were bees from the families Andrenidae and Halictidae in all sampling plots, Staphylinidae beetles in T2 (9.7%) and flies from the family Bibionidae T5 (12.2%). The pan traps placed in the platforms (1.5 m) showed similar results in terms of the relative abundance but a higher absolute

abundance in all the samples. Anthomyiidae flies decreased their abundance in comparison with the samples collected in the pan traps placed at ground level.

The pitfall samples from December 2018 showed a great decrease in terms of absolute abundance for the majority of the taxa that occurred in the previous smapling periods (Appendix C). In these samples it is visible a change in the percentage of occurrence of the different taxa, with some groups which were not very abundant dominating the assemblages, while others were completely absent, these taxa include the family Cicadellidae in T4 (45%) and T5 (36%), spiders from families Linyphiidae in T3 (28%), Lycosidae in T3 (17%) and Agelenidae T1 (21%). Pan traps, at ground level, for this sampling period showed, as well, a decrease in terms of absolute abundance. Anthomyiidae flies continued to be very numerous in the samples particularly in T3 (35.55%), T4 (48.82%) and T5 (40.67%). Syrphyidae flies had similar abundances to Anthomyiidae in T3 (36.19%) and were well represented in T4 (10.19%) and T5 (12.54%) as well. Other taxa with high relative abundances, in these samples, included Muscidae and Cicadelidae. Pan traps on the platforms showed much less abundant assemblages, with T1 for example only containing 50 individuals from 23 arthropod families. The most abundant taxon was Anthomyiidae in all the sampling plots, in particular in T3 (48.85%) with 127 individuals.

Simpson's diversity index was calculated for all arthropod family assemblages collected in each sampling point (Tables 2, 3 and 4). In the pitfall samples, collected in October 2017 the highest Simpson's diversity value was recorded in T3 (0.657) and the lowest in T4 (0.159). However, in March 2018 D' diversity values showed a new pattern with the sampling site in the eucalypt forest (T3) now showing the lowest values (0.186) and T2 presenting the highest diversity (0.759). At ground level, the pan traps in March 2018 showed very similar results between sampling sites (Table 3), while the traps in the platforms recorded higher values in the sampling plots located in the old oak forest habitats (T1 and T2) (Table 4).

The pitfall traps from December 2018 showed the highest D' diversity values for all sampling plots with highest value recorded being 0.851 at T3. Pan traps, at ground level, in this sampling period recorded lower D' diversity values in the old oak forest transepts (T1 and T2) and higher values in young oak forest transepts

(T4 and T5). For the suspended pan traps the results were different with T1 being the most diverse transept (0.965) and T3 being the least diverse (0.742).

Table 2 - Average Simpson's diversity index for each transept in each sampling period (D'= Simpson's diversity value).

SAMPLING PERIOD	TRANSEPT	D'
OCTOBER 2017	T1	0.548
	T2	0.611
	T3	0.657
	T4	0.159
	T5	0.537
MARCH 2018	T1	0.723
	T2	0.759
	T3	0.186
	T4	0.733
	T5	0.496
DECEMBER 2018	T1	0.838
	T2	0.814
	T3	0.851
	T4	0.757
	T5	0.812

Table 3 – Average Simpson's diversity index applied to each sampling point for pan traps (plates).
(D'= Simpson's diversity value).

SAMPLING PERIOD	TRANSEPT	D'
MARCH 2018	T1	0.825
	T2	0.791
	T3	0.773
	T4	0.783
	T5	0.831
DECEMBER 2018	T1	0.672
	T2	0.620
	T3	0.730
	T4	0.819
	T5	0.844

Table 4 – Average Simpson's diversity index applied to each sampling point for pan traps (cups).
(D'= Simpson's diversity value).

SAMPLING PERIOD	TRANSEPT	D'
MARCH 2018	T1	0.912
	T2	0.998
	T3	0.729
	T4	0.708
	T5	0.751
DECEMBER 2018	T1	0.965
	T2	0.855
	T3	0.742
	T4	0.823
	T5	0.864

Carabid species analysis

The distribution of the total carabid beetle species and correspondent abundance for each habitat type for the study period it is shown in Table.11. In total 328 individuals from 26 different species were collected, with the most abundant being Carabus *(Oreocarabus) amplipennis* and *Carabus (Chrysocarabus) lineatus lineatus*. The largest carabid abundance was recorded in the old oak forest sampling points with a total of 229 individuals captured during the study period.

March 2018 and December 2018 sampling periods showed the most diverse carabid assemblages. The eucalypt forest and young forest habitats showed the highest carabid Simpson's diversity index (Fig. 14).

Analysing habitat association (Fig.21), (Tab.11) we can see a prevalence of generalist and individuals (the majority from the genus *Carabus*) and forest specialist (*Leistus* and *Notiopillus*) in the old oak forest while the eucalypt forest and young oak forest habitats don't seem favour any type of habitat association.

Mean individual body mass (MIB) was highest in the old oak forest habitat (Fig.19), for brachypterous and dimorphic species, while macropterous species were most abundant in the eucalypt forest habitat. Overall MIB was highest for *Carabus* species (*Carabus lineatus* and *Carabus amplipennis*) (Tab.11). Spearman's correlation didn't show any significant values between MIB and the carabid abundance in the different sampled habitats (Tab.5), but a negative and significant correlation is shown between MIB and the different wing morphologies ($r=-0.540$; $P<0.05$).

The Levene's test did not assumed homogeneity of variance so non-parametric Kruskal-Wallis, was used. The results of the Kruskal-Wallis test did not show significant results ($R^2=0.290$, $p>0.05$, $df=2$), for carabid abundance, and for carabid beetle diversity ($R^2=2.756$, $p>0.05$, $df=2$), meaning that the distribution of carabid abundance and diversity does not differ between different sampled habitats. The same test was applied to the overall arthropod family diversity. The results didn't show significant results ($R^2=2.000$. $p>0.05$, $df=2$).

Table 5 – SPSS output of Sperman's correlation for carabid abundance, habitat type and wing morphology to MIB (Mean Individual body mass)

			MIB (g)	Wing_morph h	Old Oak forest	Young oak forest	Eucalypt Forest
Correlations							
Spearman's rho	MIB (g)	Correlation Coefficient	1,000	-,540[**]	0,341	-0,076	-0,021
		Sig. (2-tailed)		0,004	0,088	0,712	0,919
		N	26	26	26	26	26
	Wing morphology	Correlation Coefficient	-,540[**]	1,000	-0,333	-0,239	0,142
		Sig. (2-tailed)	0,004		0,097	0,240	0,488
		N	26	26	26	26	26
	Old Oak forest	Correlation Coefficient	0,341	-0,333	1,000	0,362	0,104
		Sig. (2-tailed)	0,088	0,097		0,069	0,614
		N	26	26	26	26	26
	Young oak forest	Correlation Coefficient	-0,076	-0,239	0,362	1,000	,532[**]
		Sig. (2-tailed)	0,712	0,240	0,069		0,005
		N	26	26	26	26	26
	Eucalypt Forest	Correlation Coefficient	-0,021	0,142	0,104	,532[**]	1,000
		Sig. (2-tailed)	0,919	0,488	0,614	0,005	
		N	26	26	26	26	26

**. Correlation is significant at the 0,01 level (2-tailed).

The results from the multiple linear regression (Tab. 6-10) access the relations between the soil parameters and D' diversity. A positive highly significant correlation was found between carabid diversity and water content (R=0.517, P<0.05, the remaining independent variables that showed a significant correlation were pH and organic matter, and both showed negative correlations with D' diver

Table 6 - Summary of multiple linear regression model.

					Estatísticas de mudança					
Model Summary										
Model	R	R square	R square adjusted	Std. Error of the Estimate	R square Change	F Change	gl1	gl2	Sig. Change F	Durbin-Watson
1	,258[a]	0,067	0,062	0,180759	0,067	13,932	1	195	0,000	
2	,262[b]	0,069	0,059	0,181030	0,002	0,417	1	194	0,519	
3	,552[c]	0,305	0,294	0,156797	0,236	65,601	1	193	0,000	
4	,572[d]	0,327	0,313	0,154666	0,022	6,355	1	192	0,013	0,581

a. Preditors: (Constant), pH

b. Preditors: (Constant), pH, E_Conductivity

c. Preditors: (Constant), pH, E_Conductivity, Water_content

d. Preditors: (Constant), pH, E_Conductivity, Water_content, Orgânic_matter

e. Variável Dependente: S'Diversity

Table 7 - Correlation coefficients between variables H' Diversity (Dependent variable), Water content percentage, electrical conductivity, organic matter percentage and pH (Independent variables).

Correlations		S'Diversity	pH	E_Conductivity	Organic_matter	Water_content
Pearson's Correlation	S'Diversity	1,000	-0,258	-0,056	-0,168	0,517
	pH	-0,258	1,000	0,043	-0,208	-0,128
	E_Conductivity	-0,056	0,043	1,000	0,324	-0,049
	Organic_matter	-0,168	-0,208	0,324	1,000	-0,134
	Water_content	0,517	-0,128	-0,049	-0,134	1,000
Sig. (unilateral)	S'Diversity		0,000	0,218	0,009	0,000
	pH	0,000		0,273	0,002	0,036
	E_Conductivity	0,218	0,273		0,000	0,245
	Organic_matter	0,009	0,002	0,000		0,031
	Water_content	0,000	0,036	0,245	0,031	

Table 8 - Coefficients of the final multiple linear regression model with the H' diversity on the labels as dependent variable and their significance, as calculated by SPSS.

Coefficients[a]		Unstandardized Coefficients		Standardized Coefficients			Correlations			Collinearity Statistics	
Model		B	Std. Error	Beta	t	Sig.	Zero order	Partial	Part	Tolerance	VIF
1	(Constante)	1,367	0,171		7,998	0,000					
	pH	-0,134	0,036	-0,258	-3,733	0,000	-0,258	-0,258	-0,258	1,000	1,000
2	(Constante)	1,373	0,171		8,010	0,000					
	pH	-0,133	0,036	-0,256	-3,696	0,000	-0,258	-0,256	-0,256	0,998	1,002
	E_Conductivity	0,000	0,000	-0,045	-0,645	0,519	-0,056	-0,046	-0,045	0,998	1,002
3	(Constante)	1,017	0,155		6,565	0,000					
	pH	-0,100	0,031	-0,194	-3,207	0,002	-0,258	-0,225	-0,192	0,982	1,018
	E_Conductivity	0,000	0,000	-0,023	-0,386	0,700	-0,056	-0,028	-0,023	0,996	1,004
	Water_content	0,010	0,001	0,491	8,099	0,000	0,517	0,504	0,486	0,982	1,019
4	(Constante)	1,169	0,164		7,116	0,000					
	pH	-0,121	0,032	-0,234	-3,786	0,000	-0,258	-0,264	-0,224	0,918	1,089
	E_Conductivity	0,000	0,000	0,031	0,485	0,628	-0,056	0,035	0,029	0,882	1,134
	Water_content	0,009	0,001	0,466	7,702	0,000	0,517	0,486	0,456	0,956	1,046
	Organic_matter	-0,004	0,002	-0,164	-2,521	0,013	-0,168	-0,179	-0,149	0,824	1,214

a. Dependent variable: H' Diversity

The biplot of the Canonical Correspondence Analysis is shown in Appendix D. The model explained 87.21% of the variation (the sum of all eigenvalues was 1.013, with the largest being 0.4956 and 0.3939, respectively for axes 1 and 2. Environmental variables with long arrows were more strongly correlated with the ordination axes than those with short arrows. Water content (WC%) and electrical conductivity (EC) were the two main environmental variables determining the carabid species dispersion in the sample plots from the different areas.

Table 9 - Model fit summary after the Prais-Winsten method was applied.

Model fit summary				
R	R Square	Adjusted R square	Estimate Std. Error	Durbin-Watson
0,211	0,045	0,020	0,079	1,797
The Prais-Winsten estimation method is used.				

Table 10 - Regression Coefficients after the Prais-Winsten method was applied.

Regression Coefficients					
	Unstandardized Coefficients		Standardized Coefficients		
	B	Std. Error	Beta	t	Sig
pH	-0,015	0,017	-0,070	-0,925	0,356
E_Conductivity	0,000	0,000	0,110	1,441	0,151
Water_content	0,001	0,001	0,148	2,015	0,045
Organic_matter	-0,002	0,001	-0,147	-1,848	0,066
(Constant)	0,773	0,100		7,754	0,000
The Prais-Winsten estimation method is used.					
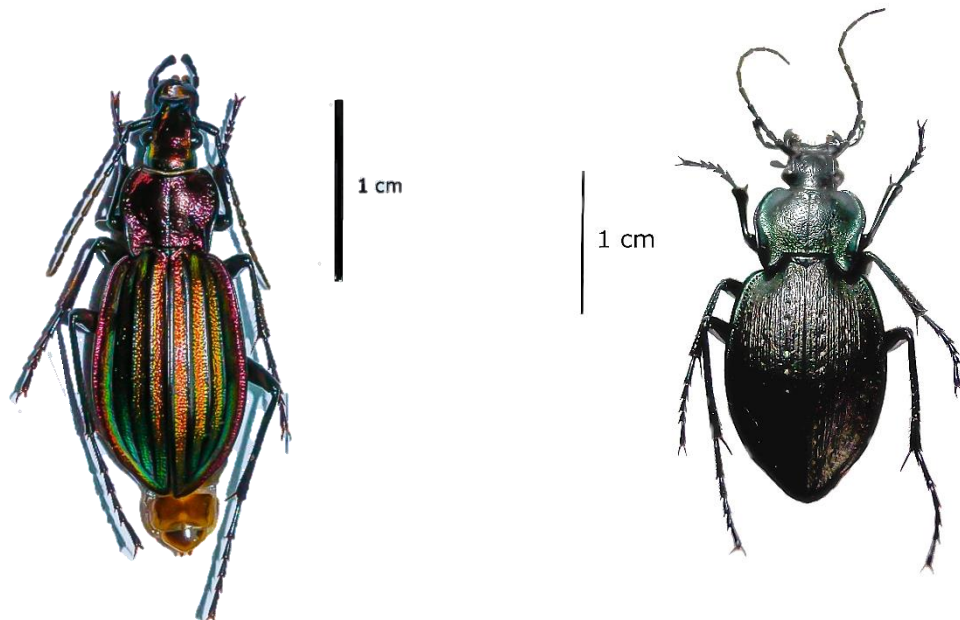
Figure 10 - *Carabus linneatus* and *Carabus amplipennis*, the most abundant carabid species in the sampling plots during the time of the study.
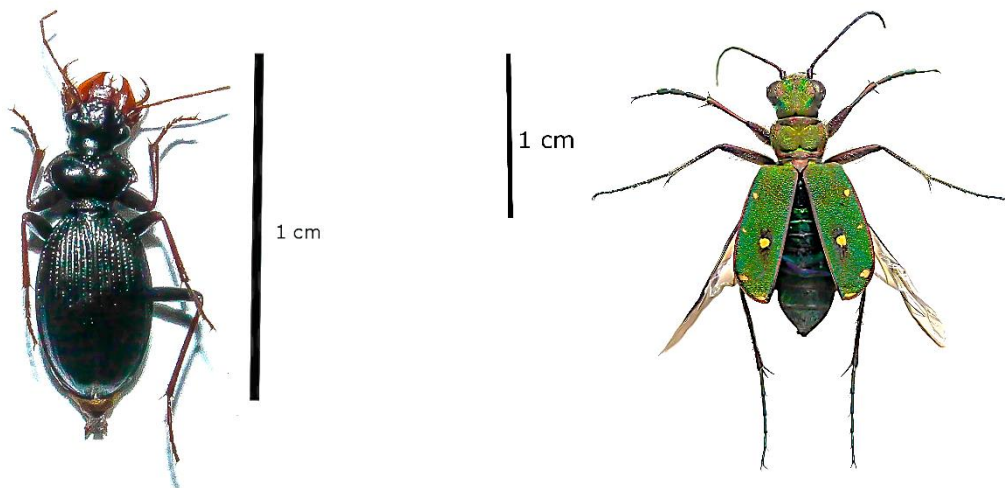
Figure 11 - *Leistus acutangulus*, another abundant species particularly in the oak forest habitats.

Figure 12 - *Cicindela campestris*, was common in the Eucalypt forest during the March 2018 samples.
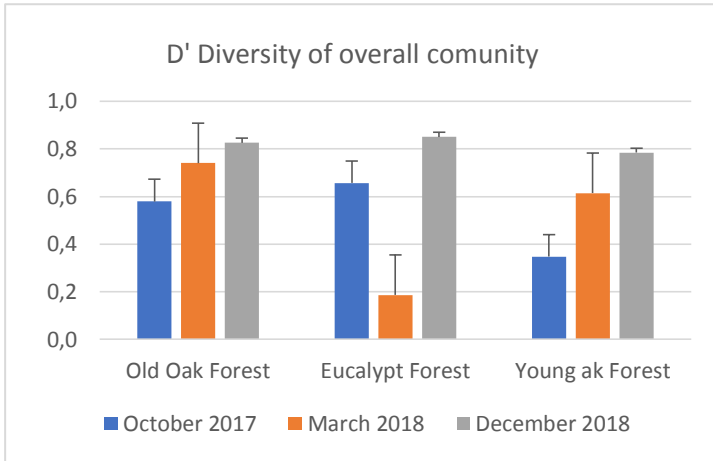
Figure 13 - Simpson's diversity index (± SD) applied to overall arthropod diversity during the study period.
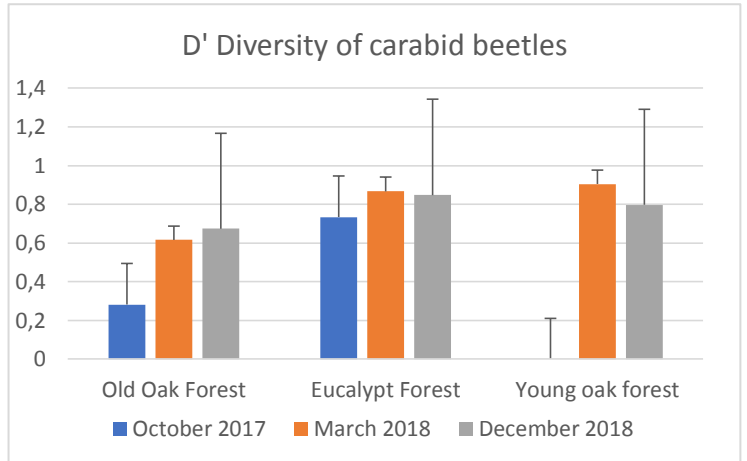
Figure 14 - Simpson's diversity index (± SD) applied to carabid beetles in the different sampled habitats.
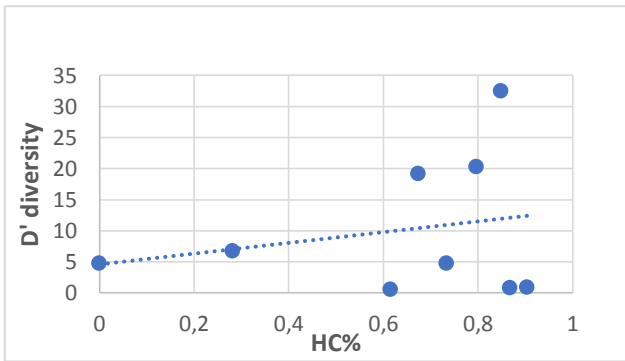
Figure 15 - Average D' Diversity against the average water content (y = 8.6482x + 4.597; R² = 0.0542)
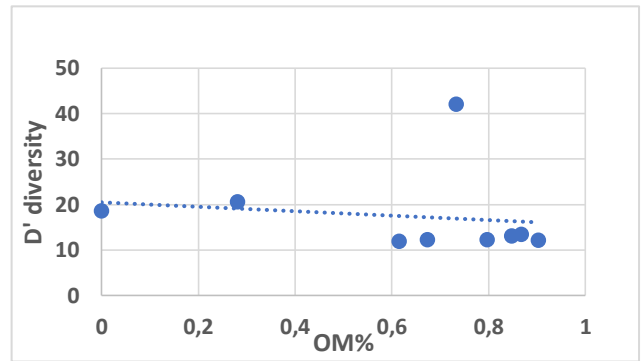
Figure 16 - D' Diversity against the average organic matter (y = -4.8752x + 20.471; R² = 0.0229).
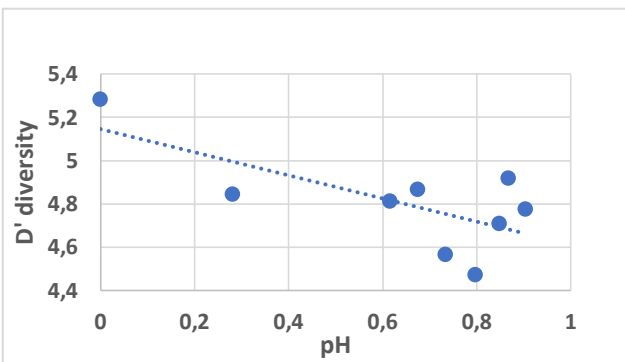
Figure 17 - Average D' Diversity against average pH values (y = -0.5351x + 5.1458; R² = 0.4973).
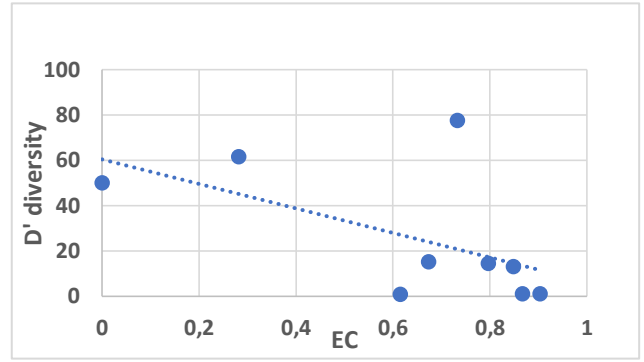
Figure 18 - Average D' Diversity against average Electric Conductivity (y = -54.085x + 60.333; R² = 0.3171).

Table 11 - Carabid beetles collected at each of the sampled forest types. Body size = carabid beetle spe-cies body size range in mm. Wings (W) = wing morphology (b = brachypterous, micropterous or flight-less, i.e. unable to fly; m = macropterous; D == dimorphic). Habitat (H) = habitat association of the species (F = forest, G = generalist, O = open habitat species, W = Hygrophilus species found near water, C= Carvernicolous species that use other animal burrows for example). Information about species characters from: (Assmann, 1997; Den Boer, Van Huizen, Den Boer-Daanje, Aukema, & Den Bieman, 1979; Eversham, 2008; Ghannem, Bejaoui, & Boumaiza, 2017; Ghannem, Bejaoui, Gahdab, & Boumaiza, 2017; Ghannem & Boumaiza, 2017; Gruttke, 1994; Maddison, 2012; Matallah, Abdellaoui-hassaine, Ponel, & Boukli-hacene, 2016; L. Nunes et al., 2006; Ouchtati, Doumandji, & Brandmayr, 2012; Perrault, 1992; Roume, Annie Ouin, Raison, & MARC DECONCHAT, 2006; Serrano, 2002; Taboada, Kotze, Salgado, & Tárrega, 2006; Vanbergen et al., 2010; Walters & Telfer, 2012; Wrase, I.Ruiz-Tapiador, & J.P.Zaballos, 1998; Aguiar & Serrano, 2012).

SPECIES	MIB (G)	W(WINGS)	AVERAGE BODY SIZE (MM)	TYPICAL HABITAT	OLD OAK FOREST	EUCALYPT FOREST	YOUNG OAK FOREST
Acupalpus (acupalpus) brunnipes (Sturm,1825)	0,0027	M	3,25	W	1	7	4
Acupalpus (acupalpus) dubius (Schilsky, 1888)	0,0020	M	2,9	W	0	1	0
Amara aenea (De Geer, 1774)	0,0192	M	7	O	1	0	1
Amara anthobia (Villa, 1833)	0,0129	M	6	O	0	1	0
Amara similata (Gyllenhal, 1810)	0,0315	M	8,5	O	1	0	0
Bembidion (bembidion) quadrimaculatum (Linnaeus, 1761)	0,0018	M	3,75	W	2	1	1
Bembidion (philochthus) biguttatum (Fabricius, 1779)	0,0039	M	3,75	W	0	2	1
Bembidion callossum subconexum (Monte, 1952)	0,0039	B	2,75	O	0	4	1
Carabus (chrysocarabus) lineatus lineatus (Dejean, 1826)	0,4705	B	24,5	G	68	7	6
Carabus (oreocarabus) amplipennis getschmanni (Lapouge, 1924)	0,2801	B	20	G	110	5	2
Cicindela campestris (Linnaeus, 1758)	0,1126	M	14	O	0	10	0
Cymindis (cymindis) alternans (Rambur,1837)	0,0291	B	8,25	O	0	0	2
Laemostenus (laemostenus) complanatus (Dejean, 1828)	0,1232	B	14,5	C	2	0	0
Laemostenus (pristonychus) terricola (Herbst, 1783)	0,1126	B	14	C	2	0	1
Lebia (lamprias) rufipes (Dejean, 1825)	0,0143	M	6	O	4	2	2
Lebia cruxminor (Linnaeus, 1758)	0,0129	M	6,25	O	1	0	0
Leistus (leistus) oopterus (Caudoir, 1861)	0,0315	B	8,5	F	1	0	0
Lesitus (leistus) acutangulus (Perrault, 1979)	0,0315	B	8,5	F	15	5	11
Licinus aequatus reymondi (Colas, 1949)	0,0759	B	12	O	0	0	1
Microlestes abeillei (Brisout, 1885)	0,0018	M	2,75	G	1	0	0
Microlestes luctuosus (Holsaus, 1912)	0,0018	M	2,8	O	0	3	1
Microlestes negrita (Wollaston, 1854)	0,0014	D	2,5	O	0	1	3
Nebria (nebria) salina (Fairmaire & Laboulnénè, 1854)	0,0608	M	11	O	0	2	0
Notiophilus biguttatus (Fabricius, 1779)	0,0092	D	5,25	F	17	4	7
Platyderus beseanus (Jeanne,1970)	0,0228	B	7,5	F	2	0	0
Poecilus (carenostylus) purpuascens (Dejean, 1828)	0,0477	B	10	O	1	0	0
Total					**229**	**55**	**44**
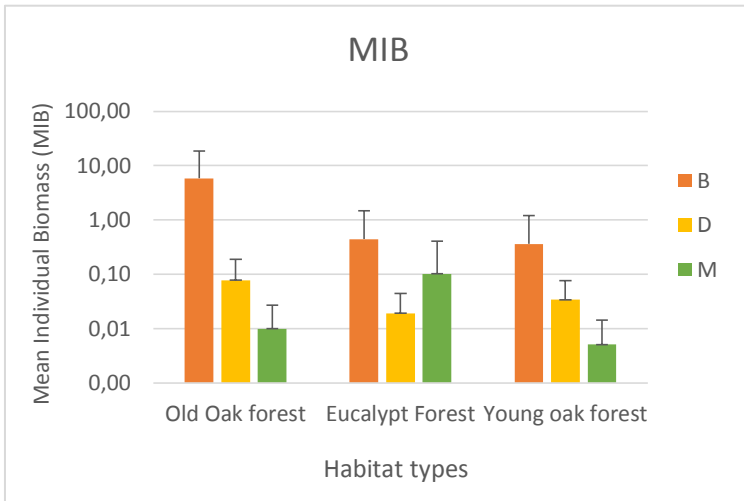
Figure 19 - Mean Individual Biomass (MIB±SD) of the individuals with different wing morphology for each of the habitats (B= Brachypterous; D= Dimorphic: M= Macropterous).

Figure 20 - Young oak forest after the fire, it is visible the large amount of rocks and crevices that may have served as shelter for carabid species during the fire.
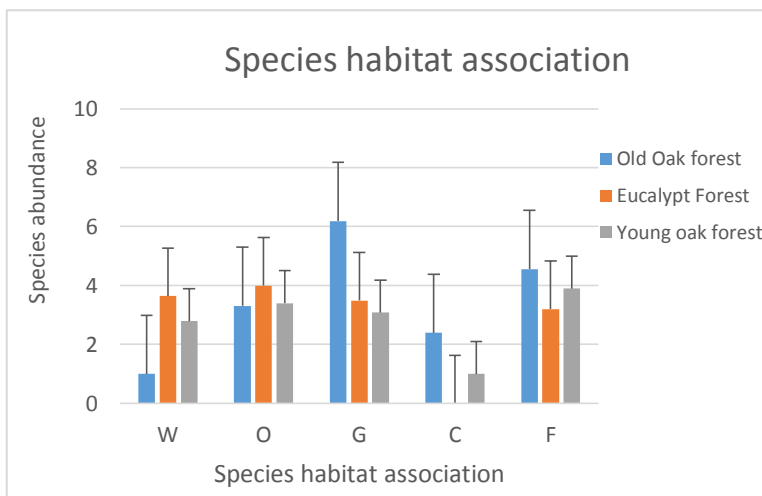
Figure 21 - Species habitat association (±SD). Habitat (H) = habitat association of the species (F = forest, G = generalist, O = open habitat species, W = Hygrophilus species found near water, C= Carvernicolous species that use other animal burrows for example)

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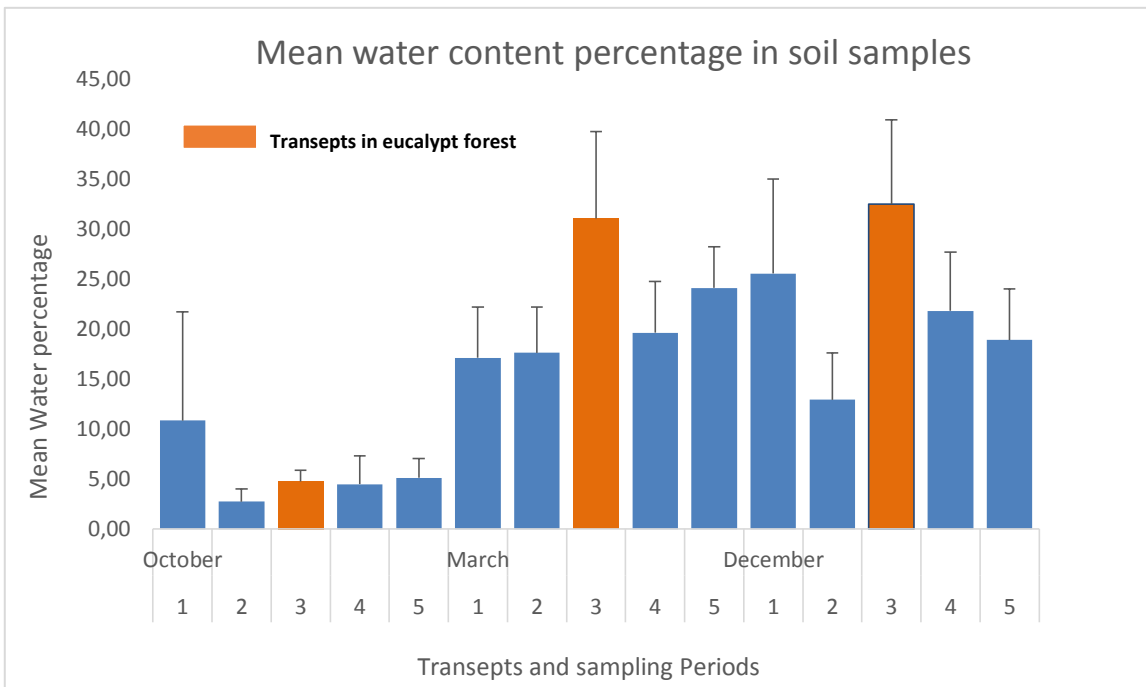
Figure 22 - Mean percentage of water content in soil samples (± SD).

Figure 23 - Photographs taken in the eucalypt forest sampling plots. It is visible water flowing above the soil and also some early regeneration of the flora, after the fire disturbance.

DISCUSSION

It is clear that fire is a phenomenon capable of causing great disturbances in forest habitats (Elia *et al.*, 2012; Kitzberger & Veblen, 1999; Naveh, 1975; Ubeda & Martin, 2012). In spite of fire being a very destructive force capable of leading local populations to extinction (Andersen *et al.*, 2005; Swengel, 2001), the results obtained in this study show the incredible resilience that insect and other arthropod communities have to this kind of natural disasters. Even in the samples obtained immediately after the fire of October 2017 there was a notable assemblage of arthropods (Appendix A), with about 1185 individuals from 45 different families.

Carabid beetles are known to be good bio-indicators of habitat disturbance and are good global representatives of arthropod communities (Elia *et al.*, 2012; Niwa & Peck, 2002; Nunes *et al.*, 2006; Pearsall, 2007; Rykken, Capen, & Mahabiri, 1997; Samu *et al.*, 2010). Because of these characteristics the analysis conducted with this group will be used to drawn general conclusions and comparisons will be made with the overall sampled arthropod communities and their responses to the fire disturbance.

The diversity indexes applied to the carabid assemblages clearly show a diversity increase since the first sampling in October 2017 until December 2018. Carabid diversity was lower in December 2018 than in March 2018 which was probably due to winter climatic conditions, in fact the majority of arthropods, including carabid beetle, do reduce their metabolism in winter to survive the low temperatures (Abdullah, 1961; Heath, Hanegan, Wilkin, & Heath, 1971; Sømme, 1999). However this was still higher than in October 2017, a period where some carabid species are more active and even reproduce (Desender, Dufrene, Lareau, Luff, & Maelfait, 1994), but immediately after the occurrence of fire disturbance (Fig.14). In fact, the climatic conditions at sampling time may have contributed to the lower diversity of overall insects in December 2018 as it was in the first days of winter and temperatures dropped significantly. When analysing the three different sampled habitats it is curious to verify that in terms of diversity the transept in the Eucalypt forest recorded high values of diversity in terms of carabid beetles when compared to the other two sampled habitats in all three different sampling periods.

This seems to contradict the evidence that this type of exotic forest supports lower diversity in comparison with native forests (Calviño-Cancela *et al.*, 2012; Zahn, Rainho, Rodrigues, & Palmeirim, 2010). However, when the overall arthropod community diversity was compared, the transepts in eucalypt forest recorded lowest diversity what is also in accordance with most of the literature. Nonetheless no significant differences were recorded in terms of diversity either analysing the overall community or when analysing the carabid beetle separately, more analysis need to be done in the future to solidify this results.

The highest carabid diversity in the eucalypt forest can be explained by specific environmental conditions. It has been shown that eucalypt forests can have equal or higher diversity values than the native surrounding forest, depending on factors like the eucalypt stand age and water availability (Calviño-Cancela *et al.*, 2012; Sax, 2002). The environmental conditions of this eucalypt plantation may have contributed for this high carabid diversity, as it can be seen in Fig. 22 and 23, water percentage in this sampling site was higher than in the other sampling plots after the first rains and after the fire disturbance, with several hygrophilus species (*Acupalpus brunnipes, Acupalpus dubius Bembidion quadrimaculatum and Bembidion biguttatum*) (Tab. 11) being collected here. Further there was a small stream close to this field which seems to be responsible for maintaining a high soil water content. These results are supported by the multiple linear regression analysis as these show a positive and significant correlation between carabid diversity and soil water content.

Transepts in young oak forest also showed a relatively high carabid beetle diversity when compared to the transepts in old oak forest, what is contrary to previous works (Taboada, Kotze, Tárrega, *et al.*, 2006). The availability of several refuges (Fig.20) in this habitat may have contributed to this outcome, although this was not tested in this work, it his known that the amount of available refuges can contribute to the arthropods diversity in post-fire assemblages (New, 2014).

The CCA biplot showed that the occurrence of some species was related with WC%. Species such as, *A. dubius*, *Microlestes luctuosos*, *B. bigutatum* and *A. brunnipes* showed the greatest relation with this variable and all have their greatest abundances in the eucalypt forest which was the habitat with the highest WC%

(Fig. 22). A second group of species was more related to EC, this group of species included *Carabus lineatus*, *Poecilus purpuascens*, *Amara. aenea*, *Laemostenus. complanatus* and *Amara similata*, and have their highest abundances in the old oak forest sampling plots (Tab.11). However, these transepts did not display the highest values of EC (Tab.1). The distribution of these species may be influenced by the variable pH as they are located close to the vector that represents this variable.

Indeed pH seems to have a great influence in the carabid diversity (Tab.7) (Fig.15). This seems to suggest the presence of species that prefer acid conditions (Lovei, 1996; Magura, Belato, & Elek, 2001; Paje & Mossakowski, 1984). It was shown earlier that pH has increased speciality in the first months after the occurrence of a fire (Antunes *et al.*, 2009; Debano *et al.*, 1977; Ulery *et al.*, 1995), which roughly corresponds to the time period of this study, although we cannot assume that the pH increased because of the lack of a control group, it is clear from this data that local carabid diversity preferred more acid soils. This soil parameter is particularly important to some carabid species specially in their egg stage, which are very sensitive to pH changes (Lovei, 1996; Thiele, 1977). The other soil parameter that showed significant correlation with carabid diversity was organic matter, again a negative correlation was recorded, this soil parameter can either increase or decrease depending on the fire intensity (Luis *et al.*, 2000). Some carabid species are known to be positively associated with organic matter, for instance, *Carabus lineatus* (Taboada, Kotze, Tárrega, *et al.*, 2006), in this case diversity thus seemed to be associated with lower amounts of organic matter.

Analysing the species habitat association preference and their abundance in each habitat (Fig. 21) as led to conclusions that contradict the literature as we find greater abundances of generalist carabid beetle in old oak forest which is a more mature type of habitat (Martínez, Iturrondobeitia, & Goldarazena, 2009; Pryke & Samways, 2012a; Taboada, Kotze, Tárrega, *et al.*, 2006). The majority of these generalist species belong to the genus *Carabus*, evidences seem to show that these generalists do tend to increase in abundance after fire as their habitat becomes more homogenized (Futuyma & Moreno, 1988). This in part can explain

this huge abundance of generalist species, but more concrete analysis would be necessary before drawing any conclusions.

Conclusions

Although carabid beetle have been successfully used as indicators of habitat disturbances in a variety of scenarios, including fire, in this work no differences were found for diversity or abundance of these organisms between different types of forest habitats, which seem to suggest that particular conditions in the sampled habitats may have determined these results. The eucalypt forest habitat did show abnormal values of diversity when compared to similar studies, but conditions such as, the high soil water content seem to be determinant to carabid survival and colonization in the post-fire. Water content was determinant when analysing the carabid distribution in the different sampled forest habitats which further seems to support this hypothesis.

However, carabid diversity did not represent the overall arthropod diversity. As it was previously demonstrated by Cameron & Leather (2012) it is advisable not to trust the results of a single group of bio-indicators to drawn an overall picture of what is really occurring in the sampled habitats, as this can in fact lead to erroneous conclusions of what is occurring with other taxonomic groups.

This particular eucalypt forest did prove to support similar diversity and abundance of carabid beetles in comparison with the other sampled forest habitats. In this way eucalypt forest/plantations can support high values of diversity for some taxa like, carabid beetles, but the overall species diversity seems to be greatly reduced and this tendency seems to be prevalent even after a strong abiotic perturbation, like fire, that can homogenize habitats and transform very different forest types into very similar vegetation matrix.

The importance of soil water content in eucalypt forests/plantations should be addressed in further works since it seems a fundamental feature to support more biodiverse communities and can improve its capacity not only to sustain native species but also to resist to fire occurrences, which are an increasingly frequent phenomenon.

Long-term studies in ecology are becoming increasingly a less common approach in current ecological studies, but in the past, post-fire studies encompass-

ing longer sampling periods sometimes over decades allowed a better understanding of the natural succession after fire. I think in the future this approach should be taken into account as it can result in better understanding of the dynamics between fires, forest and arthropod communities in Portugal. Through this work it has been shown that many variables may influence the post-fire arthropod assemblages in native and exotic forests alike. This helped us conclude that particular characteristics of the sampled site may influence greatly the survival, colonization and dispersion of species in the post-fire scenario. In order for a better understanding of how these variables affect arthropod assemblages after fire, more studies need to be done particularly in eucalypt forest as this can be of great value for the conservation of local biodiversity in this type of human managed forests.

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Appendix A

Arthropods collected during the October 2017 sampling period. Data for the abundance (N) of each family as well as the relative abundance (RA) of each arthropod family is displayed.

Order	Transept	T1		T2		T3		T4		T5	
	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Araneae	Agelenidae	9	3,03%	11	6,2%	2	0,86%	1	0,36%	2	1,01%
Diptera	Anthomyiidae	0	0,00%	0	0,0%	3	1,29%	6	2,16%	1	0,50%
Araneae	Antrodiaetidae	0	0,00%	1	0,6%	0	0,00%	0	0,00%	0	0,00%
Araneae	Anyphaenidae	2	0,67%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Aphodiidae	0	0,00%	0	0,0%	0	0,00%	1	0,36%	0	0,00%
Araneae	Atypiidae	1	0,34%	1	0,6%	1	0,43%	1	0,36%	0	0,00%
Diptera	Calliphoridae	0	0,00%	1	0,6%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Carabídae	13	4,38%	3	1,7%	5	2,15%	6	2,16%	1	0,50%
Diptera	Cecidomyiidae	1	0,34%	2	1,1%	2	0,86%	1	0,36%	0	0,00%
Hymenoptera	Cephidae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Diptera	Chironomidae	1	0,34%	4	2,2%	1	0,43%	0	0,00%	0	0,00%
Hemiptera	Cicadellidae	1	0,34%	2	1,1%	1	0,43%	12	4,32%	10	5,03%
Araneae	Clubionidae	6	2,02%	0	0,0%	1	0,43%	0	0,00%	0	0,00%
Coleoptera	Curculionidae	7	2,36%	1	0,6%	18	7,73%	16	5,76%	7	3,52%
Hymenoptera	Diapriidae	6	2,02%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Araneae	Dysderidae	4	1,35%	12	6,7%	36	15,45%	36	12,95%	31	15,58%
Diptera	Drosophilidae	0	0,00%	0	0,0%	0	0,00%	2	0,72%	2	1,01%
Hymenoptera	Evaniidae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Formicidae	19	6,40%	15	8,4%	27	11,59%	26	9,35%	12	6,03%
Araneae	Gnaphosidae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Halictidae	0	0,00%	1	0,6%	0	0,00%	1	0,36%	0	0,00%
Diptera	Heleomyzidae	0	0,00%	0	0,0%	0	0,00%	1	0,36%	0	0,00%
Coleoptera	Leiodidae	2	0,67%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Siphonaptera	Leptopsyllidae	0	0,00%	1	0,6%	0	0,00%	0	0,00%	0	0,00%
Araneae	Lycosidae	1	0,34%	3	1,7%	1	0,43%	0	0,00%	0	0,00%
Hemiptera	Lygaeidae	4	1,35%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Diptera	Milichiidae	0	0,00%	1	0,6%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Miridae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Diptera	Muscidae	0	0,00%	0	0,0%	0	0,00%	2	0,72%	0	0,00%
Pseudoscorpionida	Neobissidae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Opiliones	Phalangiidae	0	0,00%	1	0,6%	0	0,00%	0	0,00%	0	0,00%
Araneae	Philodromidae	0	0,00%	0	0,0%	0	0,00%	1	0,36%	0	0,00%
Diptera	Phoridae	5	1,68%	1	0,6%	1	0,43%	0	0,00%	0	0,00%
Hemiptera	Piesmatidae	1	0,34%	1	0,6%	0	0,00%	0	0,00%	0	0,00%

Order	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Coleoptera	Pselaphidae	1	0,34%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Araneae	Salticidae	0	0,00%	1	0,6%	1	0,43%	0	0,00%	0	0,00%
Diptera	Sciaridae	0	0,00%	4	2,2%	2	0,86%	4	1,44%	0	0,00%
Hymenoptera	Sepsidae	0	0,00%	0	0,0%	1	0,43%	0	0,00%	0	0,00%
Coleoptera	Staphylinidae	198	66,67%	109	61,2%	128	54,94%	157	56,47%	131	65,83%
Diptera	Syrphidae	0	0,00%	0	0,0%	0	0,00%	1	0,36%	2	1,01%
Araneae	Thomisidae	2	0,67%	2	1,1%	2	0,86%	1	0,36%	0	0,00%
Thysanoptera	Thysanoptera	2	0,67%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
Diptera	Trichoceridae	1	0,34%	0	0,0%	0	0,00%	1	0,36%	0	0,00%
Opiliones	Trogulidae	2	0,67%	0	0,0%	0	0,00%	1	0,36%	0	0,00%
Araneae	Zoridae	3	1,01%	0	0,0%	0	0,00%	0	0,00%	0	0,00%
TOTAL		297	100,00%	178	100,0%	233	100,00%	278	100,00%	199	100,00%

Appendix B

Arthropods collected during the March 2018 sampling period, using pitfall and pan traps. Data for the abundance of each family as well as the relative abundance of each arthropod family is displayed.

- Pitfall Trap samples

	Transept	T1		T2		T3		T4		T5	
Order	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Araneae	Agelenidae	31	4,19	29	3,15%	12	2,93%	18	4,60%	19	2,10%
Araneae	Amaurobiidae	0	0,00	0	0,00%	0	0,00%	2	0,51%	1	0,11%
Diptera	Amisopodidae	0	0,00	0	0,00%	1	0,24%	0	0,00%	0	0,00%
Hymenoptera	Andrenidae	0	0,00	11	1,20%	0	0,00%	6	1,53%	3	0,33%
Heminoptera	Anthocoridae	0	0,00	1	0,11%	1	0,24%	1	0,26%	1	0,11%
Diptera	Anthomyiidae	17	2,30	14	1,52%	20	4,89%	8	2,05%	8	0,88%
Araneae	Antrodiaetidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Araneae	Anyphaenidae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Diptera	Asilidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Araneae	Atypiidae	1	0,14	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Diptera	Bibionidae	0	0,00	0	0,00%	0	0,00%	1	0,26%	17	1,88%
Blattodea	Blattodea	0	0,00	0	0,00%	1	0,24%	0	0,00%	0	0,00%
Hymenoptera	Braconidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Calliphoridae	2	0,27	0	0,00%	3	0,73%	0	0,00%	0	0,00%
Coleoptera	Carabídae	89	12,0	54	5,87%	25	6,11%	6	1,53%	11	1,22%
Diptera	Cecidomyiidae	2	0,27	56	6,09%	11	2,69%	0	0,00%	1	0,11%
Hymenoptera	Cephidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Ceratopogonidae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Cercopidae	0	0,00	2	0,22%	0	0,00%	0	0,00%	0	0,00%

Order	Family										
Coleoptera	Chrysomelidae	3	0,41	3	0,33%	3	0,73%	0	0,00%	3	0,33%
Hemiptera	Cicadellidae	0	0,00	0	0,00%	1	0,24%	0	0,00%	2	0,22%
Araneae	Clubionidae	4	0,54	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Curculionidae	8	1,08	17	1,85%	11	2,69%	12	3,07%	19	2,10%
Hymenoptera	Diapriidae	9	1,22	5	0,54%	0	0,00%	0	0,00%	2	0,22%
Diptera	Dixidae	2	0,27	0	0,00%	1	0,24%	0	0,00%	0	0,00%
Diptera	Drosophilidae	0	0,00	2	0,22%	8	1,96%	1	0,26%	0	0,00%
Araneae	Dysderidae	32	4,32	41	4,46%	59	14,43%	55	14,07%	66	7,30%
Diptera	Empididae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Diptera	Epydridae	15	2,03	6	0,65%	62	15,16%	1	0,26%	2	0,22%
Hymenoptera	Evanidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Formicidae	100	13,5	164	17,83%	87	21,27%	188	48,08%	636	70,35
Geophilomorpha	Geophilidae	1	0,14	0	0,00%	1	0,24%	0	0,00%	0	0,00%
Araneae	Gnaphosidae	0	0,00	2	0,22%	1	0,24%	0	0,00%	12	1,33%
Hymenoptera	Halicticidae	0	0,00	7	0,76%	0	0,00%	3	0,77%	2	0,22%
Hymenoptera	Ichneumonidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	1	0,11%
Coleoptera	Leiodidae	5	0,68	2	0,22%	0	0,00%	0	0,00%	0	0,00%
Araneae	linyphiidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	1	0,11%
Chilopoda	Lithobiidae	1	0,14	2	0,22%	0	0,00%	0	0,00%	0	0,00%
Araneae	Lycosidae	18	2,43	28	3,04%	24	5,87%	0	0,00%	2	0,22%
Hemiptera	Lygaeidae	0	0,00	1	0,11%	0	0,00%	2	0,51%	0	0,00%
Diptera	Milichiidae	3	0,41	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Miridae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Muscidae	1	0,14	7	0,76%	1	0,24%	1	0,26%	1	0,11%
Diptera	Mycetophilidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Nabidae	0	0,00	0	0,00%	1	0,24%	1	0,26%	2	0,22%
Hemiptera	Neobissidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Nitidulidae	2	0,27	15	1,63%	0	0,00%	16	4,09%	6	0,66%
Diptera	Opomyzidae	0	0,00	2	0,22%	0	0,00%	0	0,00%	7	0,77%
Mecoptera	Panorphidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Opiliones	Phalangiidae	0	0,00	0	0,00%	0	0,00%	1	0,26%	1	0,11%
Diptera	Phoridae	4	0,54	7	0,76%	4	0,98%	1	0,26%	2	0,22%
Hemiptera	Piesmatidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Aranea	Pisauridae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Pselaphidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Psychodidae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Rhopalidae	5	0,68	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Rhyparochromidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	1	0,11%
Araneae	Salticidae	1	0,14	1	0,11%	0	0,00%	2	0,51%	3	0,33%
Coleoptera	Scarabidae	1	0,14	5	0,54%	0	0,00%	2	0,51%	0	0,00%
Diptera	Scatophagidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	1	0,11%
Diptera	Sciaridae	4	0,54	10	1,09%	7	1,71%	34	8,70%	39	4,31%
Hymenoptera	Sepsidae	0	0,00	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Sphaeroceridae	1	0,14	2	0,22%	1	0,24%	0	0,00%	2	0,22%
Coleoptera	Sphindidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Staphylinidae	362	48,9	409	44,46%	56	13,69%	26	6,65%	26	2,88%
Diptera	Syrphidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%

Order	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Diptera	Tachinidae	1	0,14	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Tenebrionidae	0	0,00	0	0,00%	0	0,00%	1	0,26%	2	0,22%
Araneae	Tetragnatidae	0	0,00	0	0,00%	3	0,73%	0	0,00%	0	0,00%
Araneae	Thomisidae	3	0,41	4	0,43%	1	0,24%	1	0,26%	0	0,00%
Hymenoptera	Tiphiidae	0	0,00	1	0,11%	0	0,00%	0	0,00%	0	0,00%
Diptera	Trichoceridae	2	0,27	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Opiliones	Trogulidae	2	0,27	2	0,22%	0	0,00%	0	0,00%	2	0,22%
Hymenoptera	Vespidae	0	0,00	0	0,00%	0	0,00%	1	0,26%	0	0,00%
Araneae	Zoridae	0	0,00	1	0,11%	3	0,73%	0	0,00%	0	0,00%
TOTAL		740	100,	920	100,00	409	100,00%	391	100,00%	904	100,0

- Pan trap samples (soil level)

	Transept	T1		T2		T3		T4		T5	
Order	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Araneae	Agelenidae	3	0,17%	2	0,10%	0	0,00%	0	0,00%	2	0,10%
Diptera	Agromyzidae	2	0,12%	1	0,05%	3	0,17%	2	0,17%	1	0,05%
Hymenoptera	Andrenidae	125	7,28%	99	5,02%	123	7,11%	164	13,61%	176	8,98%
Hemiptera	Anthocoridae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Anthomyiidae	301	17,54%	534	27,08%	473	27,34%	408	33,86%	515	26,28%
Hymenoptera	Apidae	3	0,17%	7	0,35%	9	0,52%	8	0,66%	15	0,77%
Diptera	Bibionidae	1	0,06%	7	0,35%	5	0,29%	17	1,41%	239	12,19%
Diptera	Bombyliidae	0	0,00%	0	0,00%	1	0,06%	1	0,08%	0	0,00%
Diptera	Brachystomatidae	0	0,00%	1	0,05%	1	0,06%	0	0,00%	0	0,00%
Hymenoptera	Braconidae	0	0,00%	0	0,00%	4	0,23%	2	0,17%	5	0,26%
Coleoptera	Buprestidae	1	0,06%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Diptera	Calliphoridae	37	2,16%	18	0,91%	11	0,64%	5	0,41%	12	0,61%
Coleoptera	Carabídae	11	0,64%	8	0,41%	3	0,17%	0	0,00%	4	0,20%
Diptera	Cecidomyiidae	13	0,76%	17	0,86%	19	1,10%	4	0,33%	8	0,41%
Hemiptera	Cercopidae	0	0,00%	1	0,05%	2	0,12%	0	0,00%	0	0,00%
Diptera	Chloropidae	0	0,00%	3	0,15%	9	0,52%	8	0,66%	15	0,77%
Coleoptera	Chrysomelidae	1	0,06%	0	0,00%	8	0,46%	0	0,00%	0	0,00%
Hemiptera	Cicadellidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Coleoptera	Coccinellidae	1	0,06%	0	0,00%	0	0,00%	2	0,17%	5	0,26%
Diptera	Conopidae	3	0,17%	1	0,05%	1	0,06%	0	0,00%	2	0,10%
Hymenoptera	Colletidae	2	0,12%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Curculionidae	3	0,17%	2	0,10%	5	0,29%	1	0,08%	4	0,20%
Coleoptera	Dasytidae	0	0,00%	0	0,00%	0	0,00%	7	0,58%	1	0,05%
Hymenoptera	Diapriidae	7	0,41%	1	0,05%	0	0,00%	2	0,17%	0	0,00%
Diptera	Dixidae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Drosophilidae	3	0,17%	5	0,25%	19	1,10%	2	0,17%	10	0,51%
Araneae	Dysderidae	2	0,12%	2	0,10%	0	0,00%	1	0,08%	1	0,05%
Coleoptera	Elateridae	0	0,00%	0	0,00%	1	0,06%	1	0,08%	3	0,15%
Diptera	Empididae	1	0,06%	0	0,00%	39	2,25%	25	2,07%	35	1,79%

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Diptera	Epydridae	2	0,12%	4	0,20%	8	0,46%	1	0,08%	1	0,05%
Diptera	Faniidae	30	1,75%	8	0,41%	6	0,35%	0	0,00%	0	0,00%
Hymenoptera	Formicidae	15	0,87%	9	0,46%	4	0,23%	12	1,00%	49	2,50%
Araneae	Gnaphosidae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Halicticidae	163	9,50%	134	6,80%	66	3,82%	88	7,30%	103	5,26%
Diptera	Heleomyzidae	5	0,29%	9	0,46%	9	0,52%	1	0,08%	0	0,00%
Hymenoptera	Ichneumonidae	13	0,76%	2	0,10%	1	0,06%	5	0,41%	2	0,10%
Coleoptera	Leiodidae	9	0,52%	8	0,41%	0	0,00%	0	0,00%	0	0,00%
Araneae	linyphiidae	0	0,00%	1	0,05%	0	0,00%	0	0,00%	0	0,00%
Araneae	Lycosidae	0	0,00%	0	0,00%	0	0,00%	0	0,00%	1	0,05%
Hemiptera	Lygaeidae	3	0,17%	0	0,00%	0	0,00%	1	0,08%	2	0,10%
Hymenoptera	Megachilidae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Melyridae	1	0,06%	2	0,10%	0	0,00%	1	0,08%	6	0,31%
Diptera	Micropezidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Diptera	Muscidae	71	4,14%	57	2,89%	29	1,68%	11	0,91%	40	2,04%
Diptera	Mycetophilidae	20	1,17%	14	0,71%	19	1,10%	5	0,41%	28	1,43%
Coleoptera	Nitidulidae	599	34,91%	674	34,18%	653	37,75%	336	27,88%	526	26,84%
Diptera	Opomyzidae	9	0,52%	0	0,00%	14	0,81%	3	0,25%	15	0,77%
Mecoptera	Panorpidae	0	0,00%	0	0,00%	2	0,12%	0	0,00%	0	0,00%
Diptera	Pediciidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Opiliones	Phalangiidae	2	0,12%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Phoridae	54	3,15%	26	1,32%	72	4,16%	13	1,08%	40	2,04%
Lepidoptera	Pieridae	0	0,00%	1	0,05%	1	0,06%	3	0,25%	2	0,10%
Diptera	Psychodidae	1	0,06%	4	0,20%	7	0,40%	1	0,08%	0	0,00%
Hemiptera	Rhyparochronidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Araneae	Salticidae	4	0,23%	0	0,00%	1	0,06%	1	0,08%	1	0,05%
Coleoptera	Scarabaeidae	17	0,99%	60	3,04%	31	1,79%	12	1,00%	30	1,53%
Diptera	Scatophagidae	0	0,00%	13	0,66%	8	0,46%	5	0,41%	6	0,31%
Diptera	Sciaridae	28	1,63%	29	1,47%	25	1,45%	16	1,33%	26	1,33%
Diptera	Sarcophagidae	2	0,12%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Araneae	Sparassidae	0	0,00%	1	0,05%	0	0,00%	0	0,00%	0	0,00%
Diptera	Sphaeroceridae	25	1,46%	6	0,30%	7	0,40%	3	0,25%	4	0,20%
Coleoptera	Staphylinidae	110	6,41%	191	9,69%	16	0,92%	19	1,58%	8	0,41%
Diptera	Syrphidae	1	0,06%	3	0,15%	4	0,23%	2	0,17%	3	0,15%
Diptera	Tabanidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Diptera	Tachinidae	0	0,00%	3	0,15%	4	0,23%	3	0,25%	10	0,51%
Hymenoptera	Tenthredinidae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	1	0,05%
Diptera	Tephritidae	0	0,00%	0	0,00%	0	0,00%	1	0,08%	0	0,00%
Araneae	Thomisidae	5	0,29%	4	0,20%	1	0,06%	1	0,08%	3	0,15%
Diptera	Trichoceridae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Vespidae	1	0,06%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
TOTAL		1716		1972		1730		1205		1960	

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- Pan trap samples (plataform level)

Order	Families	T1 N	T1 RA	T2 N	T2 RA	T3 N	T3 RA	T4 N	T4 RA	T5 N	T5 RA
	Transepts	T1		T2		T3		T4		T5	
Araneae	Agelenidae	1	0,08%	1	0,05%	0	0,00%	0	0,00%	1	0,12%
Diptera	Agromyzidae	0	0,00%	0	0,00%	1	0,06%	2	0,21%	0	0,00%
Hymenoptera	Andrenidae	147	11,88%	154	7,64%	128	8,09%	182	19,02%	161	19,83%
Diptera	Anthomyiidae	30	2,43%	59	2,93%	162	10,24%	101	10,55%	72	8,87%
Hemiptera	Alydidae	0	0,00%	0	0,00%	0	0,00%	1	0,10%	0	0,00%
Hymenoptera	Apidae	16	1,29%	35	1,74%	17	1,07%	10	1,04%	15	1,85%
Diptera	Bibionidae	3	0,24%	7	0,35%	2	0,13%	23	2,40%	29	3,57%
Hymenoptera	Braconidae	3	0,24%	1	0,05%	0	0,00%	0	0,00%	0	0,00%
Diptera	Calliphoridae	0	0,00%	3	0,15%	4	0,25%	6	0,63%	6	0,74%
Coleoptera	Cantharidae	0	0,00%	0	0,00%	1	0,06%	1	0,10%	0	0,00%
Coleoptera	Carabídae	4	0,32%	4	0,20%	9	0,57%	2	0,21%	0	0,00%
Diptera	Cecidomyiidae	17	1,37%	50	2,48%	11	0,70%	5	0,52%	9	1,11%
Coleoptera	Cerambycidae	0	0,00%	0	0,00%	5	0,32%	0	0,00%	0	0,00%
Diptera	Chloropidae	1	0,08%	0	0,00%	4	0,25%	0	0,00%	0	0,00%
Coleoptera	Chrysomelidae	5	0,40%	0	0,00%	1	0,06%	0	0,00%	1	0,12%
Hemiptera	Coccinellidae	2	0,16%	1	0,05%	3	0,19%	0	0,00%	0	0,00%
Diptera	Conopidae	0	0,00%	0	0,00%	0	0,00%	0	0,00%	1	0,12%
Hemiptera	Coreidae	0	0,00%	0	0,00%	0	0,00%	1	0,10%	0	0,00%
Coleoptera	Curculionidae	4	0,32%	15	0,74%	5	0,32%	1	0,10%	2	0,25%
Hymenoptera	Diapriidae	3	0,24%	2	0,10%	3	0,19%	2	0,21%	2	0,25%
Diptera	Drosophilidae	1	0,08%	19	0,94%	15	0,95%	3	0,31%	4	0,49%
Araneae	Dictynidae	1	0,08%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Elateridae	4	0,32%	1	0,05%	1	0,06%	3	0,31%	3	0,37%
Diptera	Empididae	17	1,37%	30	1,49%	136	8,60%	29	3,03%	21	2,59%
Diptera	Epydridae	1	0,08%	0	0,00%	1	0,06%	2	0,21%	0	0,00%
Hymenoptera	Formicidae	4	0,32%	7	0,35%	3	0,19%	25	2,61%	61	7,51%
Hymenoptera	Halicticidae	120	9,70%	222	11,02%	135	8,53%	142	14,84%	111	13,67%
Diptera	Heleomyzidae	0	0,00%	1	0,05%	3	0,19%	3	0,31%	4	0,49%
Hymenoptera	Ichneumonidae	4	0,32%	0	0,00%	3	0,19%	1	0,10%	1	0,12%
Hemiptera	Lygaeidae	0	0,00%	0	0,00%	0	0,00%	0	0,00%	1	0,12%
Diptera	Lonchopteridae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Coleoptera	Melyridae	5	0,40%	25	1,24%	3	0,19%	7	0,73%	1	0,12%
Diptera	Micropezidae	0	0,00%	0	0,00%	0	0,00%	1	0,10%	0	0,00%
Diptera	Muscidae	5	0,40%	7	0,35%	8	0,51%	4	0,42%	9	1,11%
Diptera	Mycetophilidae	14	1,13%	23	1,14%	35	2,21%	10	1,04%	7	0,86%
Coleoptera	Nitidulidae	678	54,81%	1207	59,90%	770	48,67%	315	32,92%	232	28,57%
Diptera	Phoridae	18	1,46%	12	0,60%	40	2,53%	7	0,73%	13	1,60%
Diptera	Psilidae	0	0,00%	1	0,05%	2	0,13%	0	0,00%	0	0,00%
Diptera	Psychodidae	0	0,00%	1	0,05%	0	0,00%	0	0,00%	0	0,00%
Hemiptera	Rhyparochronidae	0	0,00%	1	0,05%	1	0,06%	0	0,00%	1	0,12%
Araneae	Salticidae	9	0,73%	0	0,00%	0	0,00%	0	0,00%	1	0,12%
Coleoptera	Scarabaeidae	19	1,54%	61	3,03%	28	1,77%	31	3,24%	17	2,09%
Diptera	Scatophagidae	3	0,24%	1	0,05%	2	0,13%	2	0,21%	1	0,12%

		N	RA	N	RA	N	RA	N	RA	N	RA
Diptera	Sciaridae	27	2,18%	21	1,04%	20	1,26%	12	1,25%	10	1,23%
Diptera	Sarcophagidae	0	0,00%	0	0,00%	0	0,00%	8	0,84%	0	0,00%
Araneae	Sparassidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Diptera	Sphaeroceridae	5	0,40%	3	0,15%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Staphylinidae	54	4,37%	29	1,44%	11	0,70%	8	0,84%	11	1,35%
Diptera	Syrphidae	0	0,00%	1	0,05%	2	0,13%	3	0,31%	1	0,12%
Diptera	Tachinidae	4	0,32%	0	0,00%	1	0,06%	1	0,10%	0	0,00%
Hemyptera	Tenthredinidae	2	0,16%	2	0,10%	1	0,06%	1	0,10%	3	0,37%
Araneae	Tetragnathidae	0	0,00%	0	0,00%	0	0,00%	1	0,10%	0	0,00%
Araneae	Theridiidae	0	0,00%	0	0,00%	1	0,06%	0	0,00%	0	0,00%
Araneae	Thomisidae	4	0,32%	8	0,40%	2	0,13%	1	0,10%	0	0,00%
Hymenoptera	Vespidae	1	0,08%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Zopheridae	1	0,08%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
TOTAL		1237		2015		1582		957		812	

Appendix C

Arthropods collected during the December 2018 sampling period, using pit-fall and pan traps. Data for the abundance of each family as well as the relative abundance of each arthropod family is displayed.

- Pitfall sample traps

	Transept	T1		T2		T3		T4		T5	
Order	Families	N	RA	N	RA	N	RA	N	RA	N	RA
Orthoptera	Acrididae	0,00	0%	0,00	0%	1,00	1%	0,00	0%	0,00	0%
Araneae	Agelenidae	43,00	21%	4,00	3%	1,00	1%	1,00	1%	3,00	2%
Araneae	Amaurobiidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Amisopodidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Andrenidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Anthocoridae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Anthomyiidae	0,00	0%	0,00	0%	1,00	1%	22,00	12%	10,00	6%
Araneae	Antrodiaetidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Anyphaenidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Ditera	Asilidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Atypiidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Bibionidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Blattodea	Blatodea	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Braconidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Calliphoridae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Carabídae	22,00	11%	17,00	14%	10,00	13%	3,00	2%	8,00	4%
Diptera	Cecidomyiidae	1,00	0%	18,00	15%	0,00	0%	4,00	2%	16,00	9%

Order	Family										
Hymenoptera	Cephidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Ceratopogonidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Cercopidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Chrysomelidae	0,00	0%	1,00	1%	0,00	0%	0,00	0%	1,00	1%
Hemiptera	Cicadellidae	5,00	2%	2,00	2%	12,00	16%	80,00	45%	65,00	36%
Araneae	Clubionidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Coccinelidae	1,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Curculionidae	2,00	1%	2,00	2%	1,00	1%	2,00	1%	5,00	3%
Hymenoptera	Diapriidae	13,00	6%	7,00	6%	1,00	1%	0,00	0%	0,00	0%
Araneae	Dictynidae	9,00	4%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Dixidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Drosophilidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Dysderidae	5,00	2%	0,00	0%	0,00	0%	0,00	0%	1,00	1%
Diptera	Empididae	0,00	0%	0,00	0%	1,00	1%	0,00	0%	0,00	0%
Diptera	Epydridae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Evanidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Formicidae	9,00	4%	7,00	6%	3,00	4%	11,00	6%	16,00	9%
Coleoptera	Geotrupidae	2,00	1%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Gnaphosidae	0,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Orthoptera	Gryllidae	0,00	0%	1,00	1%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Halicticidae	0,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Diptera	Heleomyzidae	1,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Hymenoptera	Ichneumonidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Leiodidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Siphonaptera	Leptopsyllidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	linyphiidae	12,00	6%	5,00	4%	21,00	28%	24,00	14%	32,00	18%
Aranea	Lycosidae	4,00	2%	5,00	4%	13,00	17%	8,00	5%	16,00	9%
Hemiptera	Lygaeidae	0,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Diptera	Milichiidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Miridae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Miturgidae	0,00	0%	1,00	1%	0,00	0%	0,00	0%	0,00	0%
Diptera	Muscidae	0,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Diptera	Mycetophilidae	0,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Hemiptera	Nabidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Nemesiidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	1,00	1%
Hemiptera	Neobissidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Nitidulidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Opomyzidae	0,00	0%	1,00	1%	0,00	0%	0,00	0%	0,00	0%
Mecoptera	Panorpidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Opiliones	Phalangiidae	2,00	1%	1,00	1%	1,00	1%	3,00	2%	1,00	1%
Diptera	Phoridae	1,00	0%	0,00	0%	0,00	0%	1,00	1%	0,00	0%
Hemiptera	Piesmatidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Pisauridae	0,00	0%	1,00	1%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Pselaphidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Psychodidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Rhopalidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hemiptera	Rhyparochronidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%

Order	Family										
Araneae	Salticidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Scarabidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Scatophagidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Sciaridae	3,00	1%	3,00	2%	2,00	3%	0,00	0%	0,00	0%
Diptera	Sepsidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Sphaeroceridae	1,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Sphindidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Staphylinidae	65,00	31%	45,00	37%	7,00	9%	6,00	3%	5,00	3%
Diptera	Syrphidae	0,00	0%	0,00	0%	0,00	0%	4,00	2%	0,00	0%
Diptera	Tachinidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Coleoptera	Tenebrionidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Tetragnatidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Theridiidae	1,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Thomisidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Tiphiidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Diptera	Trichoceridae	6,00	3%	0,00	0%	0,00	0%	2,00	1%	0,00	0%
Coleoptera	Trogidae	0,00	0%	0,00	0%	1,00	1%	0,00	0%	0,00	0%
Opiliones	Trogulidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Hymenoptera	Vespidae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
Araneae	Zoridae	0,00	0%	0,00	0%	0,00	0%	0,00	0%	0,00	0%
		208,00	100%	121,00	100%	76,00	100%	177,00	100%	180,00	100%

- Pan trap samples (soil level)

		T1		T2		T3		T4		T5	
Order	Family	N	RA	N	RA	N	RA	N	RA	N	RA
Orthoptera	Acrididae	0	0,00%	0	0,00%	2	0,21%	0	0,00%	0	0,00%
Araneae	Agelenidae	2	1,94%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Agromyzidae	0	0,00%	0	0,00%	11	1,18%	4	0,95%	0	0,00%
Diptera	Anthomyiidae	24	23,30%	7	9,21%	332	35,55%	206	48,82%	133	40,67%
Hymenoptera	Apidae	0	0,00%	0	0,00%	2	0,21%	2	0,47%	4	1,22%
Diptera	Bibionidae	0	0,00%	0	0,00%	5	0,54%	0	0,00%	1	0,31%
Diptera	Bolithophilidae	0	0,00%	1	1,32%	0	0,00%	1	0,24%	2	0,61%
Hymenoptera	Braconidae	0	0,00%	0	0,00%	0	0,00%	2	0,47%	0	0,00%
Diptera	Calliphoridae	0	0,00%	0	0,00%	54	5,78%	6	1,42%	0	0,00%
Coleoptera	Carabídae	3	2,91%	0	0,00%	1	0,11%	1	0,24%	1	0,31%
Diptera	Cecidomyiidae	2	1,94%	1	1,32%	4	0,43%	0	0,00%	1	0,31%
Diptera	Chloropidae	0	0,00%	0	0,00%	6	0,64%	0	0,00%	0	0,00%
Coleoptera	Chrysomelidae	10	9,71%	2	2,63%	2	0,21%	4	0,95%	3	0,92%
Hemiptera	Cicadelidae	3	2,91%	1	1,32%	90	9,64%	82	19,43%	82	25,08%
Coleoptera	Curculionidae	0	0,00%	1	1,32%	1	0,11%	1	0,24%	0	0,00%
Hymenoptera	Diapriidae	4	3,88%	4	5,26%	0	0,00%	0	0,00%	0	0,00%
Diptera	Drosophilidae	0	0,00%	1	1,32%	0	0,00%	0	0,00%	2	0,61%
Coleoptera	Elateridae	1	0,97%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Empididae	0	0,00%	0	0,00%	4	0,43%	0	0,00%	5	1,53%
Diptera	Faniidae	0	0,00%	0	0,00%	1	0,11%	0	0,00%	0	0,00%

Order	Family	N	RA	N	RA	N	RA	N	RA	N	RA
Hymenoptera	Formicidae	9	8,74%	4	5,26%	0	0,00%	3	0,71%	3	0,92%
Araneae	Gnaphosidae	2	1,94%	0	0,00%	1	0,11%	0	0,00%	0	0,00%
Hymenoptera	Halicticidae	2	1,94%	0	0,00%	13	1,39%	3	0,71%	2	0,61%
Diptera	Heleomyzidae	3	2,91%	2	2,63%	1	0,11%	3	0,71%	2	0,61%
Hymenoptera	Ichneumonidae	2	1,94%	2	2,63%	1	0,11%	0	0,00%	1	0,31%
Coleoptera	Laemophloeidae	0	0,00%	1	1,32%	0	0,00%	0	0,00%	0	0,00%
Diptera	Lauxaniidae	0	0,00%	1	1,32%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Leiodidae	1	0,97%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Araneae	Linyphidae	3	2,91%	2	2,63%	2	0,21%	4	0,95%	1	0,31%
Araneae	Lycosidae	1	0,97%	0	0,00%	1	0,11%	0	0,00%	2	0,61%
Diptera	Muscidae	5	4,85%	8	10,53%	26	2,78%	26	6,16%	26	7,95%
Diptera	Mycetophilidae	1	0,97%	0	0,00%	1	0,11%	0	0,00%	4	1,22%
Hemiptera	Nabidae	0	0,00%	0	0,00%	0	0,00%	1	0,24%	0	0,00%
Coleoptera	Nitidulidae	0	0,00%	0	0,00%	2	0,21%	0	0,00%	1	0,31%
Lepidoptera	Nymphalidae	0	0,00%	0	0,00%	2	0,21%	0	0,00%	0	0,00%
Hemiptera	Pentatomidae	0	0,00%	0	0,00%	1	0,11%	0	0,00%	0	0,00%
Opiliones	Phalangiidae	1	0,97%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Phoridae	1	0,97%	1	1,32%	2	0,21%	1	0,24%	1	0,31%
Heiptera	Rhyparochronidae	0	0,00%	0	0,00%	1	0,11%	0	0,00%	0	0,00%
Diptera	Sciaridae	6	5,83%	1	1,32%	12	1,28%	2	0,47%	3	0,92%
Diptera	Sphaeroceridae	1	0,97%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Sphecidae	0	0,00%	0	0,00%	3	0,32%	1	0,24%	0	0,00%
Coleoptera	Staphylinidae	11	10,68%	34	44,74%	6	0,64%	1	0,24%	2	0,61%
Coleoptera	Syrphidae	0	0,00%	0	0,00%	338	36,19%	43	10,19%	41	12,54%
Diptera	Tachinidae	0	0,00%	0	0,00%	3	0,32%	1	0,24%	2	0,61%
Coleoptera	Tenebrionidae	0	0,00%	2	2,63%	2	0,21%	22	5,21%	1	0,31%
Diptera	Tephritidae	0	0,00%	0	0,00%	1	0,11%	1	0,24%	0	0,00%
Araneae	Thomisidae	2	1,94%	0	0,00%	0	0,00%	0	0,00%	1	0,31%
Diptera	Trichoceridae	3	2,91%	0	0,00%	0	0,00%	1	0,24%	0	0,00%
TOTAL		103		76		934		422		327	

- Pan trap samples (plataform level)

	Transepts	T1		T2		T3		T4		T5	
Order	Family	N	RA	N	RA	N	RA	N	RA	N	RA
Diptera	Agromyzidae	0	0,00%	0	0,00%	2	0,77%	0	0,00%	1	1,08%
Diptera	Anthomyiidae	5	10,00%	33	28,45%	127	48,85%	39	38,24%	29	31,18%
Hymenoptera	Apidae	0	0,00%	0	0,00%	7	2,69%	4	3,92%	2	2,15%
Diptera	Asteiidae	0	0,00%	3	2,59%	1	0,38%	0	0,00%	0	0,00%
Diptera	Bibionidae	2	4,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Bolithophilidae	2	4,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Calliphoridae	1	2,00%	3	2,59%	0	0,00%	0	0,00%	1	1,08%
Coleoptera	Carabídae	0	0,00%	1	0,86%	1	0,38%	1	0,98%	0	0,00%
Diptera	Cecidomyiidae	3	6,00%	4	3,45%	0	0,00%	3	2,94%	2	2,15%
Diptera	Chloropidae	0	0,00%	0	0,00%	3	1,15%	0	0,00%	1	1,08%
Coleoptera	Chrysomelidae	0	0,00%	0	0,00%	1	0,38%	0	0,00%	1	1,08%

Order	Family										
Hemiptera	Cicadelidae	2	4,00%	0	0,00%	5	1,92%	1	0,98%	4	4,30%
Coleoptera	Curculionidae	0	0,00%	1	0,86%	2	0,77%	1	0,98%	0	0,00%
Hymenoptera	Diapriidae	1	2,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Drosophilidae	2	4,00%	0	0,00%	1	0,38%	1	0,98%	2	2,15%
Diptera	Empididae	3	6,00%	5	4,31%	8	3,08%	13	12,75%	1	1,08%
Diptera	Ephydridae	0	0,00%	1	0,86%	1	0,38%	1	0,98%	3	3,23%
Hymenoptera	Formicidae	4	8,00%	1	0,86%	0	0,00%	0	0,00%	0	0,00%
Araneae	Gnaphosidae	0	0,00%	0	0,00%	0	0,00%	1	0,98%	0	0,00%
Hymenoptera	Halicticidae	1	2,00%	1	0,86%	3	1,15%	4	3,92%	2	2,15%
Diptera	Heleomyzidae	0	0,00%	2	1,72%	0	0,00%	0	0,00%	0	0,00%
Hymenoptera	Ichneumonidae	3	6,00%	3	2,59%	2	0,77%	1	0,98%	2	2,15%
Araneae	Linyphidae	2	4,00%	1	0,86%	9	3,46%	0	0,00%	0	0,00%
Coleoptera	Melyridae	1	2,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Muscidae	0	0,00%	27	23,28%	27	10,38%	10	9,80%	14	15,05%
Diptera	Mycetophilidae	4	8,00%	4	3,45%	2	0,77%	1	0,98%	2	2,15%
Hemiptera	Nabidae	0	0,00%	0	0,00%	1	0,38%	0	0,00%	1	1,08%
Coleoptera	Nitidulidae	2	4,00%	5	4,31%	10	3,85%	5	4,90%	2	2,15%
Diptera	Phoridae	1	2,00%	3	2,59%	2	0,77%	2	1,96%	2	2,15%
Diptera	Platypezidae	0	0,00%	1	0,86%	0	0,00%	0	0,00%	0	0,00%
Diptera	Psychodidae	0	0,00%	0	0,00%	0	0,00%	1	0,98%	0	0,00%
Hemiptera	Rhyparochromi-	0	0,00%	0	0,00%	2	0,77%	0	0,00%	0	0,00%
Diptera	Sepsidae	0	0,00%	0	0,00%	3	1,15%	0	0,00%	0	0,00%
Diptera	Sciaridae	2	4,00%	9	7,76%	12	4,62%	6	5,88%	5	5,38%
Diptera	Scathophagidae	1	2,00%	0	0,00%	1	0,38%	0	0,00%	0	0,00%
Diptera	Sphaeroceridae	0	0,00%	0	0,00%	1	0,38%	0	0,00%	0	0,00%
Hymenoptera	Sphecidae	0	0,00%	1	0,86%	0	0,00%	0	0,00%	0	0,00%
Coleoptera	Staphylinidae	3	6,00%	2	1,72%	10	3,85%	1	0,98%	1	1,08%
Diptera	Syrphidae	0	0,00%	3	2,59%	12	4,62%	4	3,92%	11	11,83%
Plecoptera	Taeniopterygidae	2	4,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Tachinidae	0	0,00%	0	0,00%	1	0,38%	1	0,98%	2	2,15%
Diptera	Tephritidae	0	0,00%	0	0,00%	3	1,15%	0	0,00%	1	1,08%
Araneae	Thomisidae	2	4,00%	0	0,00%	0	0,00%	0	0,00%	0	0,00%
Diptera	Trichoceridae	1	2,00%	1	0,86%	0	0,00%	1	0,98%	1	1,08%
Coleoptera	Trogossitidae	0	0,00%	1	0,86%	0	0,00%	0	0,00%	0	0,00%
TOTAL		50		116		260		102		93	

Appendix D

Canonical correspondence analysis with sampling times as response variables. Species are represented by dots (●), the nominal variables (time of sampling) are represented by squares (■) for October 2017 samples, triangles (▲) for March 2018 samples and diamonds (◊) for December 2018 samples, the environmental variables, soil parameters (Water content percentage, organic matter percentage, Electrical conductivity and pH) are represented by the lines with the labels at the end.