Universidade de Aveiro Departamento de Biologia
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Estudo do efeito das queimadas sobre a dinâmica das plantas nas florestas.

The effects of fire on soil plant dynamics in forests, a review

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The effects of fire on soil plant dynamics in forests, a review

dissertação apresentada à Universidade de Aveiro para cumprimento dos requisitos necessários à obtenção do grau de Mestre em Ecologia, Biodiversidade e Gestão de Ecossistemas, realizada sob a orientação científica de Newton Carlos Marcial Gomes, PhD, Investigador Auxiliar do Departamento de Biologia da Universidade de Aveiro e da Professora Doutora Celeste De Oliveira Alves Coelho, Professora Catedrática do Departamento de Ambiente e Ordenamento da Universidade de Aveiro.

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palavras-chave efeitos do fogo, vegetação, solo, fungus, mycorrhiza

resumo

Este trabalho é uma revisão bibliográfica sobre os efeitos do fogo na dinâmica solo-planta nas florestas. A perda de área florestal leva ao empobrecimento da biodiversidade, da componente biótica do solo e a alterações no funcionamento do ecossistema. Após os incêndios, a perda de nutrientes do solo por escorrência superficial e a consequente erosão dos solos influencia a camada de repelência, a estabilidade dos agregados, o conteúdo de matéria orgânica do solo, a infiltração e a cobertura de manta morta.

É importante considerar o efeito do fogo no banco de sementes e na vegetação dada a sua importância para minimizar os efeitos dos processos hidrológicos e de erosão do solo, assim como da intercepção da chuva.

As plantas têm diferentes mecanismos de regeneração após o fogo por via seminal (espécies arbustivas comuns do *maqui* mediterrânico) ou por via vegetativa (raízes, rizomas ou tecidos aéreos).

O efeito do fogo nos fungos e micorrizas do solo é estudado pela importância que estas últimas têm na regeneração de áreas degradadas e na estabilização do solo sendo importantes na retirada de nutrientes do solo pela associação fungo-planta o que irá influenciar o crescimento diferencial da planta. São descritos diferentes tipos de micorriza (ECM, AM), a importância dos fungos no sistema solo - planta na floresta, a função dos fungos na estrutura da floresta, a importância das micorrizas antes e após o fogo e a sua importância como possíveis indicadores de perturbação como o fogo. Por fim são abordados possíveis métodos de pesquisa para fungos e micorrizas e diferentes técnicas de micorrização.

Quando ocorrem fogos de intensidade alta pode ocorrer destruição das raízes das plantas daí a importância dos fungos pioneiros no processo inicial de recuperação do solo.

As diferentes intensidades do fogo (baixa, média e alta) ou os diferentes tipos de fogo (controlado, experimental ou natural) foram analisados sobre os processos de hidrologia e erosão do solo e para os diferentes elementos biológicos.

Áreas que necessitam uma pesquisa mais continuada são os processos de hidrologia e de erosão do solo e a compilação de informação sobre os efeitos do fogo na camada superior e inferior do solo na tentativa de relacionar estes efeitos.

Algumas técnicas de mitigação da erosão foram também discutidas ao nível das encostas e dos canais de drenagem por forma a minimizar os efeitos do fogo nas encostas.

keywords

fire effects, vegetation, soil processes,fungus,mycorrhiza.

Abstract

This thesis is a literature review about the effects of fire on soil-plant dynamics in forests. The loss of forest areas leads to the impoverishment of biodiversity of the biotic components of soil and changes in ecosystem function. After the passage of a fire the loss of nutrients on the soil by runoff and by soil erosion processes influences the repellent layer, soil aggregate stability, the organic matter content, the soil water content, the infiltration rates and the percentage of litter coverage.

It is important to consider the effects of fire on the seed bank and on vegetation due to its importance to minimize the processes of hydrology, soil erosion and rainfall interception.

The different mechanisms of vegetation to regenerate after fire are seminal and vegetative processes. In the first case (common shrub species of the Mediterranean *maqui*) and in the second case (roots, rhizomes or aerial tissues).

The effects of fire on mycorrhiza and fungi on the soil is studied due to their importance in the regeneration of degraded areas and on soil stabilization by their importance in nutrient uptake by the soil-plant system.

When high intensity fire occurs it could happen the destruction of root plants and pionner fungi that are very important in the initial process of soil recovery.

The different fire intensities (low, medium and high), or the different type of fire (controlled fire, experimental and natural fire) were analysed by hydrological and soil erosion processes and to the different biological elements.

Areas of study that need further research about hydrological and erosion processes and the joint information about the effects of fire in the superior and inferior soil layer try to relate such effects.

Some erosion mitigation techniques were studied to slope and in water dreanage channels in order to minimize the effects of fire on the slope.

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CHAPTER 1

Forests and Fires

CHAPTER 1 FORESTS AND FIRES

1.1. Introduction

The general importance of studying the effects of fires on forests is related to the increasing occurrence of fires, over the last decades. The acidity of forest soils is related with the intensity of fire and as the organic matter increases the ph diminish. After fire there is a general impoverishment of biodiversity, loss of the soil biota and changes in the ecosystem function. Changes on such components alters the available supplies for the invertebrates in the soil and to the soil plant continuum. Vegetal diversity is the base of food to soil invertebrates and vertebrates. In an ecosystem, animals depend on plants because it's their base of food. General patterns of post-fire events are reviewed to the soil-vegetation dynamics in Mediterranean and temperate forests of pine and eucalyptus trees, shrubland or "montado" areas and the advantages and disadvantages of prescribed and wildfire according to fire intensity (Úbeda *et al.*,2005). The ecological effects of fire on the seed bank and on vegetation are also important considering the biology of these elements and on their overall importance to the forest regeneration. The changes on these communities could be indicators of ecosystem disturbance. Then the effects of fire on soil properties and processes as soil erosion and soil water repellence independently of post-fire changes in litter and vegetation destruction (Shakesby & Doerr, 2006). The soil water repellent layer could affect the hydrological response on a basin area, which consequently will cause soil erosion, a decreased on infiltration rates and an increase on overland flow. Finally, the effects of fire on fungus and the occurrence of pioneer and intermediary fungus and the important interactions with vegetation on aggregate stability and on forest regeneration after disturbance.

According to several authors (Claridge *et al.*, 2009; Román & Miguel, 2005 and Goicoechea *et al.*,2009), pioneer fungi occurred immediately after fire and grows within forest regeneration, helping to minimize the movement of soil in the absence of plant roots. To Ingleby *et al.* (1998) "intermediate" fungus occurs after pioneer fungus in the succession sequence.

An area of study that needs further investigation tests the importance of post fire fungi in the capture of nutrients through the mycorrhizal network system and the importance to the reestablishment of vegetation (Goicoecha *et al.*, 2009, Claridge *et al.*, 2009). To an expedite study of post-fire fungus ecology, recent developed techniques as the quantitative analyse of mycelia contents on soil by biochemical analysis to fungus-specific compounds can be done (Wood & Goodenough, 1977; Anderson & Cairney, 2007).

According to Cerdá & Doerr (2005), it is needed a long term monitoring in soil hydrology and erosion because the major changes in these two soil processes occurred after the first two years of fire. The objective of this literature review is to compile recent information about the effects of fire on soil-plant dynamics and the behaviour of the ecosystem after fire considering the above-ground components, the below-ground components, soil biota and soil processes and how they influence each other in the ecosystem function (Wardle, 2002) in particular the effects of fire on:

1) Vegetation

2) Soil and water erosion processes and associated processes of runoff and overland flow; Application of mitigation techniques after fire.

3) The function of fungus to the resilience of vegetation, in soil erosion mitigation processes, and in soil stabilization.

According to Wardle (2002), few attempts were made to bring together the dispersed literature on the aboveground and belowground communities to interpret the entire ecosystem. Moreover, most of the studies about the effects of fire on Mediterranean ecosystems focus on each compartment most of the times, in a separately analyse (Capogna *et al.*, 2009), so it is necessary to relate the information about fire effects on the above-ground (vegetation), soil physical processes (erosion, runoff, soil-water repellence and soil aggregate stability) and in the below-ground components (mycorrhiza and fungus community).

1.2. The forests on the planet

Before the glaciations, Portuguese territory was covered with forests of woody evergreen trees *Laurisilva* (*silva* means forest) similar to the existent vegetation in some areas of Açores, Madeira (Portugal) and Canárias (Spain) islands. This type of evergreen forests included several species of Lauraceae family as *Laurus azorica* and *Persea indica* of which remains some relics in the post glaciation period as *Laurus nobilis*, *Rhododendron ponticum and Prunus lusitanica*. The post-glaciations "new climate" brought more competitive woody species predominantly of *Fagaceae* family as *Quercus sp., Castanea sativa* and *Fagus sylvatica* in areas left by the pre-glaciation niches of vegetation. These are native trees. This mixed forest of evergreen trees (relics from *Laurisilva*) and deciduous woody trees is *Fagosilva* (Paiva, 1998). When *Homo sapiens sapiens* settled in Europe he cooperated in the development of Fagosilva but as the practice of cereals cultivation (wheat, barley) and animals domestication (goat, sheep and pork) began, 8-7 million years ago the degradation of Fagosilva started slowly (Paiva,1998). The destruction of forest ecosystems by this time was very slowly not devastating and man used fire to conquest non-forestry areas. In the middle age the intensive wood, coal and timber demand and the enlargement of grazing and agriculture practices started to fragment the Portuguese primitive forests. Latter other factors accelerated the destruction of Fagosilva forests and soil impoverishment: the intense pastorate activity, the abandonment of land practices as agriculture and silviculture, soil management with heavy machinery to grow mono-specific species and the use of fire (bushfire) as the occurrence of wildfires (Paiva, 1998; Coelho *et al.*, 2002 and Ferreira *et al.*, 2008). The loss of soil organic matter and soil destruction results on a natural replacement by heathers of *Erica* sp., and *Calluna vulgaris*, brooms of *Cytisus* sp., gorses of *Ulex* sp. and *Pterospartum tridentatum* and the predominant habitats became shrub-lands and brushwood instead of Fagosilva forests.

In the Mediterranean arid areas the mixed oak woodlands has been replaced on the hillslopes by *Cistus* matorral and by cultivated dryland farming on the plateaux. The dominant vegetation type is mattorral and common species include *Cistus landanifer*, *Cistus laurifolius* and *Rosmarinus officinalis*.

From one point the bush stands started to be reforested with *Pinus pinaster* and *Pinus pinea* (naturalized species). The first tree is well adapted to the atlantic climate of Portugal and it was widely planted or sown comparing to Pinus pinea (Paiva, 1998).To understand the continuous degradation of the woody cover (forest relics and pine areas) as well as agricultural areas is important to refer the high intense fires occurred since 1975 where thousands of hectares were burned. In the consequence of pinewood devastation there was an increased area occupied by eucalyptus and acacias that were indiscriminately planted with an invasive behaviour (Ferreira *et al.*, 2008; Paiva, 1998). A similar situation occurred in the forests of northwest Spain a contiguous region with the north of Portugal, Galicia: in the past Galician forests had fire-resistant deciduous species, due to tissue tolerance to heat, to leaf moisture, to the protecting buds and to cork protection, as occurs with *Quercus robur, Quercus pyrenaica, Castanea sativa* and *Betula pendula*. However, in the last 30 years the incidence of forests fires in Galicia reached alarming levels causing soil erosion and vulnerability and during the last years and at the same time evergreen growing trees as *Pinus pinaster* and *Eucalyptus globulus* were planted (Varela *et al.*, 2005). Carballas *et al.*, (2009) reported in a local newspaper in 2007, that Galicia and the north of Portugal were the European zones most affected by forests fires with the highest number of fires per ha and inhabitant.

The development on the forestry sector was a consequence of forest management tools adapted to the private forest domain since they represent a considerable proportion of the national forest. It is important to refer that Portuguese forest belongs to private and public domain. In the continent 84.2% of the total wooded forestry area is private and 6.5% belongs to industry. The forest public domain is 15.8% of the total and 2% of this percentage is private state domain. In Açores 67% of the production forest is private and 33% is public domain. Until 1974 the portuguese politics do not favour agriculture practices and consequently forest activities increased with the establishment of "*montados*", pine areas and since the 50´s the increase of eucalyptus areas in place of pines as a consequence of fires. *Eucalyptus globules* have long extensive roots, few nutrient demands and fire resistant seeds that allow a rapid growth in disturbed areas after fire. During the 20th century, Portuguese forests had a high economical production of timber materials as wood and cork but the non-timber products were the highest sources from the forests as fruits, mushrooms, honey, aromatic plants and the "silvopastorícia" practices. Pasture and leisure activities as sportive hunting and fishing increased much since 1980 until 2005 as an intention to approach the rural world (Estratégia Nacional para as Florestas, 2006). According with the cited source wood, cork and paper industries had high levels of production since the eighties until 2003 but the paper industry was the most profitable one. The "silvicultural" practices, woddy and cork extraction were more a less constant and similar since 1977 until 2003. The timber production is mainly from *pinus* spp.

and *eucalyptus* spp. and from cork activity. The production of dried fruits as pinion and chestnut fruits had increased during 1990 until 2004, the highest rates of pinion production was from 1997 until 2004 while chestnut production increased in the last decade. During the last 30 years the forestry sector has faced some management risks and it is necessary to improve the quality and the efficiency of multifunctional management areas with low productivity woody areas which potentiate the exploration of non-timber resources (cork, fruits, honey, etc.), grazing areas and leisure areas. In the process of forest certification the timber and wood production in the forests is certified according to standard parameters. The consumers could identify if wood products are from a well managed forest. It is also important to identify the owners of forestry areas to include all adjacent forest areas into a standard management and productivity methods. Attribute reforestation supports to encourage the conversion of eucalyptal and pine areas into mounted of corktrees, chestnut, oak trees and other endemic species as well as shurblands of naturalized species (Estratégia Nacional para as Florestas, 2006).

As a conclusion of this analyse it is suggested reforestation programs with native trees and shrubs and the conversion on shrubland of naturalized species in degraded areas to retard soil erosion in desertified lands that are a result of human action and climatic variations. To a better management programme the forest owners and producers created the called “ZIF areas” which means that these areas have common plans of forest management and protection. Other conclusion is about the importance of environmental monitoring programs to a better knowledge of native and naturalized vegetation of trees, shrubs and herbaceous vegetation and to the awareness about the forests.

1.3. Fires and forest disappearance

Historical facts confirm that “Os Descobrimentos” had a strong contribution to the decline of European forests and also to the Portuguese forest (Paiva, 1998). The study of Nicole Devy-Vareta (1986), a geographer, clearly exemplifies this problem and she reported, referring to the XIV century in Portugal, “…The need of national wood will intensify and forest regeneration will be increasingly difficult …” On the other hand there was a big exchange of plants between the “new and the old world” with important economic and ecological changes. New flora species were bring and an exchange of plants occurred with plants of high ecological plasticity (Otero, n.d). Some woody trees and shrubs were naturalized or planted in Europe to feed, for wood, for clothes as *Pyrus* sp., *Morus* sp., *Cupressus* spp., *Juglans* sp., *Celtis* sp. and others.

Another influential factor of forest disappearance is the occurrence of fire with a mean annual area burned of 1070 km^2 since 1980, approximately 1% of the total area of the country and the highest fire incidence in Europe (Nunes *et al.*, 2005).

Data reported by Ferreira *et al.* (2008), comparing the Portuguese burned area with other in southern european countries (Spain, France, Italy and Greece) demonstrates that since 2000 the burned area is superior to 100000ha/year, in 2003 reached 400000 ha/year and in 2005 300000 ha/year, a tragic number also reported in several foreign newspapers around Europe. In a period of two years a total of 750000 ha were burned, in a country with a forest surface of more than 3 million ha (Figure 1). This burned area is seven times higher than the burned area in Spain, 24 times more than France and six times more comparing to Greece (Silva & Catry, 2006).

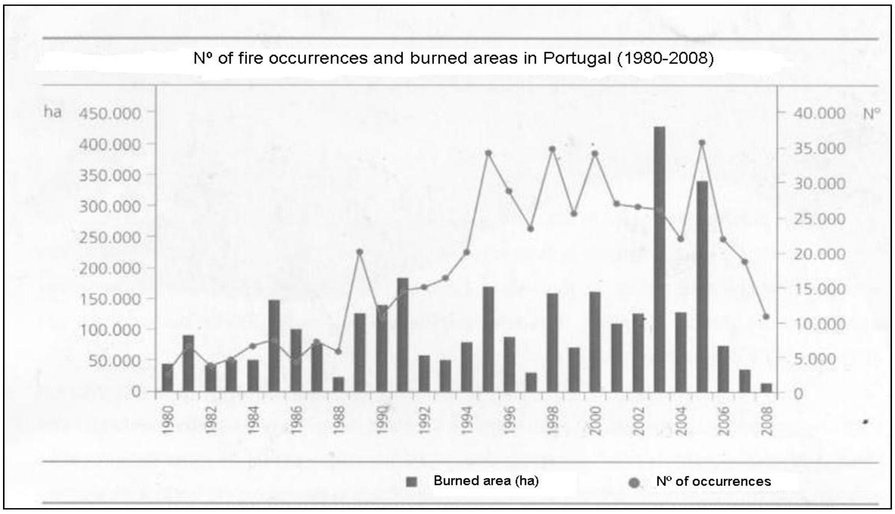

Figure1.3- A. Number of fire occurrences and burned area in Portugal, between 1980 and 2008 (from Silva, 2008).

The distribution of forest fires in different Portuguese regions shows that forest fires normally occur in the northern half part of the country and in the interior whereas regions closer to the sea have smaller fires due to the type of landscape that is continuous in the interior and fragmented near the sea (Silva & Catry, 2006).

In general after the passage of fire the shrubland areas occupied by spontaneous vegetation of *Erica* sp., *Cytisus* sp., *Crataegus* sp., *Cistus* sp*.,* and *Hallimium* sp., forming bushwoods in poor soils are a consequence of grazing and wood collection practices abandonment. In the first summer after fire, the annual herbaceous species and the *cistaceae* occupied areas left by the *ericaceae* in the shrubland but the exotic vegetation re-sprouted and germinate from the seed bank stored in the soil and quickly germinate comparing to the spontaneous vegetation because fire alter the behaviour of exotic species (Domingues, P., personal communication).

According to the report of 2008 from “DGRF” the burned area decreased much since 2007 due to preventive actions near populations, forest vigilance and structural prevention as the cleaning of brush-woods. The number of fire occurrences per/day decreased much in 2007 and 2008. The national plan to the forests establishes to the 2012 year less than 100.000 ha of burned area /per year.

1.4. Defining fire: prescribed fire and wildfire

Fire is a chemical combustion process of the organic material with the release of flames. There are four types of fire: accidental, intentional, natural and negligent as bushfire and bonfire.

In the Mediterranean ecosystems fire was considered a natural ecological factor, because since the Neolithic era, humans used fire as a tool to increase the disposable land area for cultivation and pastoralism, eliminating the natural vegetation with controlled fires (Cabello *et al.*, 2007).

The natural occurrence of fires in the Portuguese territory is very small compared to diverse human activities; however it can be originated by electric discharges, flakes and lightning. Forest fires are frequently described by their recurrence intervals, fire size, burn severity and frequency (Silva *et al.*, 2007; Shakesby & Doerr, 2006).

Some consequences of fire are the increased soil temperatures, changes in chemical and physical properties of forest soil in the soil biota that also depends on fire severity and consequently affecting the hydrological and erosive processes on the burned watershed (Moody & Martin, 2001). The severity of fire depends on two components: intensity and duration. Intensity is the rate at which a fire produces thermal energy and is quantified by the scorch height, flame length and other (Silva *et al.*, 2007; Certini, 2005). In a low-severity fire soil heating on the mineral soil does not exceed 100ºC at the surface, and 50ºC at 5cm depth. Hot fires in soils with thick organic horizons, raise the soil surface at temperatures around 275ºC and in a severe fire the soil heating temperatures can be nearly 700ºC on the surface, above 250ºC at a depth of 10cm and exceed 100ºC at a depth of 22cm belowground. In a forest watershed, temperatures above 450ºC increase overland flow and erosion inducing water repellent properties and decreasing infiltration rates (Ferreira *et al.*, 2007). The frequency of forest fires is dependent on vegetation type (on its resilience capacity), on climate changes, on the long-term intensive land uses and on deep land transformations occurred during this century (Silva *et al.*, 2007). The conversion of the burned shrublands and pine areas in eucalyptus areas and land abandonment are some examples of land transformation (Silva *et al.*, 2007).

Basically, there are two types of forest fires. A controlled combustion that is known as a prescribed fire, leaving the soil surface as in a low severity fire (Robichaud, 2000) and wildfire is as an uncontrolled combustion. Bush fire is a prescribed fire on bush stands, used to clean the land for an agriculture use as crops or grazing formation (Macedo, 1993).

In spite of the varying temperatures and fire intensity, in prescribed fires and wildfires usually the entire litter layer and most of the vegetation are burnt and the remaining wood debris had little effect on the hydrological and erosive processes (Ferreira *et al.*, 2008).In general wildfire and prescribed fire has physical, chemical and biological effects on the soil, some advantageous and other disadvantageous (Macedo, 1993).

The majority of prescribed fires are low to moderate severity fires that take place under less dry and warm weather conditions at lower temperatures than wildfires. The fuel load is less than that of wildfires and is done in stripped down-slope direction (Coelho *et al.*, 2002; Certini, 2005). This traditional practice used in the mountain areas of north and central Portugal reduce the risk of wildfire in drought months, partly due to the dwindling of forest understory management, which creates breaks to the forest fires progression. This technique burn fuel in a controlled manner during winter and is used to prevent forest fires during summer (Silva, 2002). The advantages are the reduction of fuel load, the low cost per/ha and the facility to be used in forest stands. Other applications of prescribed fire are fauna habitat management and the maintenance of grazing areas (Silva, 2002). It should be taken into account climate, wind, humidity and the minor possible impact in vegetation and on the soil (Silva, 2002).

The adequacy of this practice in the combat of wildfire depends on its effectiveness to diminish wildfire and on its proper environmental effects (Coelho *et al.*, 2002; Ferreira *et al.*, 2008). In a low intense prescribe fire soil temperatures remain high from few minutes to an hour but in severe fires with entire organic horizon consumed surface mineral soil temperature remain elevated for months or years due to the direct solar radiation (DeBano, 2000; Ferreira cited Neary *et al.*, 1999).

A negative aspect of prescribed fire is that a long term application of burning (2 years interval during 20 years) conduces to a reduction in fine root length density and on mycorrhizal root biomass in the upper 15cm of mineral soil, with the resulting changes in nutrient cycling respectively on the amount of nitrogen and phosphorus (Hart *et al.*, 2005).

Wildfire is an important disturbance factor in vegetation zones throughout the world and is more or less common since the late Devonian times. In Portugal the spread of wildfires occurs since the 80's of last century with catastrophic patterns. Wildfires are a threat to the economic viability of commercial forestry and to the ecological health of diverse ecosystem, since fire affects soil micro-organism populations that plays a critical role to maintain soil health and fertility (Ferreira *et al.*, 2008; Capogna *et al.*, 2009). According to Úbeda *et al.* (2006), the changes in a burnt forest are highly dependant on fire intensity and can influence vegetation recovery in a short and long timescale and wildfire (uncontrolled or naturally occurring fire) can be the major important cause of hydrological and geomorphological change in landscapes (Shakesby & Doerr, 2006). Ferreira *et al.* (2008), reported wildfire as the most important agent of land cover change in Portugal, in part as a result on the lack of control of the amount of biomass accumulation in forests and shrub-land areas in the Mediterranean regions, which are fuels and drivers of fire spread (Nunes *et al.*, 2005). Severe fires also cause changes in the successional dynamics, in the above and belowground species composition generating volatilization of nutrients and ash deposition.

Portugal has a warm temperate climate mostly Mediterranean, characterized by hot summers and cool wet winters, areas of rugged terrain with natural evergreen vegetation resistant to drought and pyrophytic (Nunes *et al.*, 2005).

1.5. The effects of fires in forests

The understanding of the ecological role of fire is based in the deep knowledge of the forest origin, structure and evolution and in the complex interaction of forest elements as vegetation, animals, soil biota, soil properties, topography and climate which makes the forest a live and dynamic system (Macedo, 1993). The extend and duration of fire effects depends on several environmental factors that affects combustion process, such as amount, nature, and moisture of live and dead fuel, air temperature and humidity, wind speed and topography (Certini, 2005).

To analyse the effects of fire in biological diversity, re-colonization processes are analysed in a medium and long term, comparing the post-fire community with the pre-fire community, considering the number of species (floristic richness for vegetation) or the relative abundance of individuals for each specie (Marques, n.d.).

Fires can be beneficial or deleterious to the forests considering the severity and the changes in the above and belowground components. The severity of fire has different effects on a forest depending on combustion intensity, in the live and dead fuel on the soil, soil texture and moisture and time remained since the last burnt (Certini, 2005). Other effects of fire depend on the temperature and duration of fire that can induce, enhance or destroy soil water repellence, normally reducing soil aggregate stability and possible related to organic carbon content. These changes have implications in infiltration rates, overland flow and rain splash detachment and on soil erosion (Shakesby & Doerr, 2006; Ferreira *et al.*, 2008; Varela *et al.*, 2009).

After a low intense fire there is an increasing occurrence of the herbaceous flora in overcrowded forests, the dominance of vegetation through the elimination of undesired species and the increasing nutrients availability (Certini, 2005), or the release of seeds stored for decades as is the case of *Pinus* sp. (Silva *et al.*, 2007; Ferreira *et al.*, 2008). No irreversible ecosystem change occurred but the enhancement of hydrophobicity can decrease infiltration rates and increase erosion rates. Some characteristics can identify a low intense fire: a thick layer of litter, a black layer of humus, trees with some leaves and a large number of branches. On the other hand in a medium intense fire, trees had no leaves but retained a substantial number of branches and small quantities of litter remain on the soil (Úbeda *et al.*, 2006; Prats *et al.*, 2008).

The severe fires, such as wildfires, normally have negative effects on the soil. They cause significant removal of organic matter, deterioration of structure and porosity, loss of nutrients due to volatilization, ash deposition and a marked alteration on the quantity and specific composition of soil-biota and on soil-invertebrate communities (Certini, 2005). On a long term, a direct geomorphical change of wildfire is the weathering of bedrock surface (Shakesby & Doerr, 2006). Other effects of wildfire are the increased soil erosion and considerable hydrological changes that has also been recognized (Coelho *et al.*, 2002; Ferreira *et al.*, 2008; Varela *et al.*, 2009). Evidences of a high intensity fire are trees with no leaves and no branches with only trunks remaining. The soil surface is covered by an extensive layer of grey and white ashes (Úbeda *et al.*, 2006; Prats *et al.*, 2008).

In general, forest fires had more destructive effects on forests of fast or evergreen growing trees (ex: *eucalyptus*, *pinus*) than on forests with fire-resistant deciduous species. For example, Díaz-Fierros *et al.*,1982, showed that in *Pinus* and *Eucalyptus* forests the limits of tolerable soil loss is generally far exceeded with rates over 50 ton/ha/ano recorded in extreme cases. The rates of erosion are exceeded by steep relief and high erodibility from rainfall (Varela *et al.*, 2005). Another influential factor is the soil water repellent layer (hydrophobicity) that is frequent under pine and eucalyptus forests.

In the next chapters is a detailed analyse about the effects of fire on soil physical properties that normally reduces soil aggregate stability and can induce, enhance or destroy soil water repellence that has effects on infiltration and overland flow (Shakesby & Doerr, 2006). Taking into account the biological elements, fire can affect soil biota and species composition to varying degrees and soil fungi that are the main decomposers of organic residues in soil, depending on the intensity of fire, soil-water content and maximum temperatures (Capogna *et al.*, 2009).

1.6. REFERENCES

Anderson I.C., Cairney J.W.G. (2007). Ectomycorrhizal fungi: exploring the mycelia frontier. *Microbiology reviews*, 31: 388-406.

Cabello-Pérez F., Benlloch P.I., Fernández J.R., Echeverría M.T., Llovería R.M., Martín A.G. (2007). Impacto de los incendios forestales en comunidades vegetales sub-mediterráneas: Evaluación multitemporal de la diversidad del paisaje utilizando imágenes landsat T. *Cuadernos de Investigación Geográfica*, 33: 101-114.

Capogna F., Persiani A.M., Maggi O., Dowgiallo G., Puppi G., Manes F. (2009). Effects of different fire intensities on chemical and biological soil components and related feedbacks on a Mediterranean shrub (Phillyrea angustifollia L.). *Plant Ecology*, 204 (2):155-171.

Carballas T. (2007). Los incendios forestales, un desastre ecológico y económico para Galicia. En: Voz Natura. 10 años de compromiso medioambiental. Fundación Santiago Fernández Latorre. La Voz de Galicia, A Coruña. pp. 97-106.

Carballas T., Andrade M.I., Alonso-Betanzos A., Carballo E., Caselle V., Díaz-Raviña M., Gago A., Jiménez E., Legido J.L., Martín A., Mato M.M., Varela A., Galiñanes V.A. (2009). Advances in forest fire effects on soils In: Book of Abstracts II International Meeting on Forest Fire Effects on soils. (Instituto de Recursos Naturales y Agrobiología de Sevilla), pp.

Cerdá A., Doerr S. H. (2005). The influence of vegetation recovery on soil hydrology and erodibility following fire: an eleven-year investigation. *International Journal of Wildland Fire*,14 (4).

Certini G. (2005). Effects of fire on properties of forest soils: a review. *Oecologia,* 143: 1-10.

Claridge A.W., Trappe J.M., Hansen K. (2009). Do fungi have a role as soil stabilizers and remediators after forest fires? Forest Ecology and Management, 257:1063-1069.

Coelho C.O.A., Ferreira A.J.D., Baake M. & Keizer J.J. (2002). Impacts of prescribed shrubland fire and forest wildfire on overland flow and soil erosion generating processes. In Rubio J.L., Asins S. &

V. Andreu V. (Eds), Man and Soil at the Third Millennium: Proceedings of the third International Congress of the European Society for Soil Conservation, Valencia–Spain.

DeBano L.F. (2000). The role of fire and soil heating on water repellency in wildland environments: a review. Journal of Hydrology, 231-232: 195-206.

Estratégia Nacional para a Floresta (2006). Available on:
http://www.afn.min-agricultura.pt/portal/gestao-florestal/ppf/enf

Ferreira A.J.D., Coelho C.O.A., Boulet A.K., Keiser J.J., Ritsema C.J. (2008). Soil and water degradation processes in burned areas: lessons learned from a nested approach. *Catena*, 74 (3): 273-285.

Ferreira A.J.D., Silva J.S., Coelho C.O.A., Boulet A.K., Keizer J.J. (2009).The Portuguese experience in managing fire effects In: Cerdá A. & Robichaud P.(Eds.), Fire Effects on soils and restoration strategies: Land Reconstruction and Management, (pp. 401- 424).

Hart, S.C., Classen, A.T., Wright, R.J. (2005). Long-term interval burning alters fine root and mycorrhzal dynamics in a ponderosa pine forest. Journal of Applied Ecology, 42:752-761.

Macedo F. W. & Sardinha A.M. (1993). Fogos Florestais, volume I. 2ª edição. Publicações Ciência e Vida, Lda. Lisboa.

Marques, A. (n.d.). O Fogo e a Biodiversidade. Available on:
http://naturlink.sapo.pt/article.aspx?menuid=3&exmenuid=76&bl=1&cid=10148&viewall=true#Go_1
(Norma APA para web page citation)

Moody A.J., Martin D.A. (2001). Initial hydrological and geomorphic response following a wildfire in the Colorado Front Range. *Earth Surface Processes and Landforms*, 26 (10): 1049-1070.

Nunes M.C.S., Vasconcelos M.J., Pereira J.M.C., Dasgupta N., Alldredge R.J., Rego F.C. (2005). Land cover type and fire in Portugal:do fires burn land cover selectively? *Landscape Ecology*, 20: 661-673. (só li abstract na net)

Otero S. (n.d). Os descobrimentos e a descoberta de novas plantas. Available on:
http://naturlink.sapo.pt/article.aspx?menuid=2&cid=12838&bl=1&section=2&viewall=true#Go_2

Paiva J.A.R. (1998). *A Crise ambiental, Apocalipse ou Advento de uma Nova Idade*. Liga de Amigos de Conimbriga.

Prats S.A., Amaral L.P., Coelho C.O.A., Ferreira A.J.D., Pinheiro A., Barragán F., Boulet A.K. (2008). Mitigation techniques and strategies to reduce soil and water degradation immediately after forest fires. In: Final Cost 634 Conference, (Aveiro, Portugal), pp.106.

Relatório Provisório da Floresta contra Incêndios (2008). DGRF DFCI, Available on:
http://www.afn.min-agricultura.pt/portal/gestao-florestal/ppf/enf

Robichaud P.R., Beyers J.L., Neary D.G. (2000). Evaluating the effectiveness of postfire rehabilitation treatments (GTR):63,85pp. Rocky Mountain Research Station.

Shakesby R.A., Doerr S.H. (2006). Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviews*, 74: 269-307.

Silva J.S. (2002). O Fogo controlado. In: Manual de silvicultura para a prevenção de incêndios. Direcção-Geral das Florestas, Lisboa.

Silva J.S., Catry F. (2006). Forest fires in cork oak (*Quercus suber* L.) stands in Portugal. *International Journal of Environmental Studies,* 63 (3): 235-257.

Silva J.S., Ferreira A.D., Sequeira E.M. (2007). E depois do fogo. In Silva J.S., *et al.* (Eds.),

Proteger a Floresta: Incêndios, pragas e doenças. vol.VIII: 93-128. Colecção Árvores e Florestas de Portugal. Jornal Público/Fundação Luso-Americana para o Desenvolvimento/ Liga para a Protecção da Natureza. Lisboa.

Úbeda X., Outeiro L.R., Sala M. (2006). Vegetation regrowth after a differential intensity forest fire in a Mediterranean environment in northeast Spain. *Land Degradation & Development*, 17:429-440.

Varela M.E., Benito E., Blas E. Impact of wildfires on surface water repellency in soils of northwest Spain. *Hydrological Processes*, 19: 3649- 3657.

Varela M.E., Benito E., Keizer J.J. (2009).Wildfire effects on soil erodibility of woodlands in NW Spain. Land Degradation and Development.

Vareta-Devy N. (1986). Para uma geografia histórica da floresta portuguesa. Revista da Faculdade de Letras-Geografia,I:5-37.

Wardle D.A. (2002). Communities and ecosystems: linking the above-ground and below-ground components. Princeton University Press, New Jersey.

CHAPTER 2

Effects of fire on vegetation

CHAPTER 2 EFFECTS OF FIRE ON VEGETATION

2.1. Regeneration Strategies

Fire has played a decisive role in Post-Glacial biological and cultural evolution in the Mediterranean Region. Its evolutionary impact on plants has been manifested by feedback responses, in which the fire and its after effects select plants by physiological and other mechanisms that enables fire tolerance or permit vegetative and reproductive/seminal regeneration (Naveh, 1975). Other effects of fire includes the removal of standing biomass and litter (Úbeda *et al.*, 2006), the disappearance of vegetation recovery (Moreno *et al.*, 2004) and the activation of seeds of fire resistant species (Ferreira *et al.*, 2008) stored in the soil.

Plants use two different strategies to regenerate after a fire:

1) The seminal or reproductive species depend on the germination capacity of seeds produced before fire. This seeds can be protected by an external and rugged cap (pericarp), can be buried in the soil or have lignified structures (the pines). Soil affords a good protection to seeds from the heat because only the upper centimetres are affected by lethal temperatures (Lloret & Zedler, 2009), the persistence of a seed bank or the capacity to produce enough seeds to incorporate the soil between fire occurrence is also essential to re-establish a population although human activity can cause changes in vegetation composition due to the introduction of aggressive species (Zedler *et al.*, 1983). Fire is considered an extreme event even tough it has a relatively high frequency because it brings changes, such as the germination of seeds that otherwise would be dormant (Zedler *et al.*, 1983) as the case of some Mediterranean species that are stimulated by the high temperatures that breaks the seed coat and dormancy and are exposure to nutrients, light and heat inducing germination (Moreno *et al.*, 2004). If fire occurs immediately after seed production and seed dispersion this specie has a better chance to establish seedlings than latter reproductive species (Moreno *et al.*, 2004). The ecology of these fire resistant species determines the resilient capacity after fire. The seeds of typical Mediterranean shrubs (*Erica* sp., *Arbutus unedo*, *Cistus* salvifolius, *Cistus ladanifer*, *Hallimium* sp.) are very small, produced in high number and then incorporated on the soil until a favourable germinating condition (Moreno *et al.*,2004). *Acacia* species produces a large quantity of seeds with high longevity and seed germination is stimulated by fire and has huge growth rates (Marchante, E., 2005). In the Pinaceae (*Pinus pinaster*, *Pinus halepensis*) and in *Eucaliptus* species the seeds are stored in the canopy. The Pinaceae produces a high quantity of seeds and are protected by pines, which are closed during a long time and open due to the high temperatures produced after fire, which allows a rapid regeneration after fire. In *Eucalyptus* the resilience capacity depends on fire severity. This specie also regenerates by vegetative tissues mostly due to basal log and root regeneration (Fernandes, 2009). In general in the Mediterranean *maqui*, the Cistaceae quickly occupies the soil after disturbance as the ericaceae and are able to recover damaged areas affected by fire (Domingues,

2010). Úbeda *et al.* (2006) also reported the increasing number of *Cistus sp*. after fire, partly in response to light but a decrease of *Erica arborea* and *Arbutus unedo*, in a typical Mediterranean shrubland. Some species only regenerate by seeds they are obligatory species: *Pinus halepensis, Rosmarinus officinales, Cistus salviifolius, Cistus albidus* (Pausas, 2004).

The morphological and ecological aspects of these species include plants from exposed areas as the post fire areas with high rates of seed production and shallow roots. The leaves have adjustments to dryness with small leaves, leathery consistency and pelagy consistence (Catry *et al.*, 2007).

2) The other strategy of regeneration is the vegetative via that includes species with protecting buds or shoots in specific tree structures (lignotubes, basal log with roots, aerial structures) that confers protection against fire as other characteristics for example, the protective cork and the soil, that *per se* gives protection to roots and rizhomas (Pausas, 2004). These species regenerate immediately after fire from new shoots released from the roots, basal log and aerial structures on the canopy, but recurrent fires affect the vigour of plants and increases nutrient loss that affect vegetative regeneration conditions. In fact, less nutrient supply increases mortality after fire (Pausas, 2004). Vegetative regeneration depends on individual characteristics of vegetation, frequency of fire regime and post-fire conditions (Lloret & Zedler, 2009).

The below-ground organs (roots, rizhoma) experience lower temperatures than the aerial part of the plant and annual herbaceous plants regenerates from these reproductive structures as some woody plants (*Arbutus unedo*, *Phillyrea latifolia* and *Quercus* coccifera) that preferably regenerates from shoots in the roots and in the basal log with roots, vegetative regeneration. In the initial vegetative stage the underground organs (root, rizhoma) should have carbon reserves to allow root respiration and to generate the new aerial tissues (Moreno *et al.*, 2004). Also species from ericaceae family (*Erica arborea, Erica australis*, Arbutus unedo), *Eucalyptus* (not obligatory, with high rates of basal sprouting) and the native woody plants, *Phylirrea angustifolia*, *Quercus roble*, *Quercus suber* and *Quercus ilex* of the Mediterranean maqui regenerates by vegetative tissues after a fire (Domingues, 2010; Moreno *et al.*, 2004; Catry *et al.*,2007). The deciduous hardwood trees have high survival rates (>80%) and aerial sprouting increases due to cork thickness and reduced fire severity (Fernandes, 2009).

Vegetative regeneration is common in trees or shrubs of long longevity that produce less seeds than seminal species, have deep roots and are characteristic of advanced stages of ecological succession (Catry *et al.*, 2007).

Species of *Eucalyptus* sp., *Olea* sp., *Quercus* sp., *Ericacea* sp., have the two described regeneration strategies: 1) seeds stored in the soil that resist to fire or stored in the canopy and 2) from new shoot tissues, these species are facultative (Moreno *et al.*, 2004; Pausas, 2004).

Generally, spontaneous vegetation as *Erica umbelata*, *Calluna vulgaris*, *Frangula alnus*, Pterospartum tridentatum, Myrtus communis have few physiological supplies or some morphological adaptations that allows the recovery of burned soils as occurred in a fire in 2005 in the north-central Portugal region of Águeda district (Domingues, 2010). These spontaneous species support drought and uncertain climacteric conditions (uncertain rains and high temperatures), have fire-resistant mechanisms and it´s the reason for the quickly growth after fire as the characteristic vegetation of exposure areas in the initial steps of the ecological succession (Úbeda *et al.*,2006) that is delayed due to invasive species such as *Eucalyptus globulus* and *Acacia longifolia* widely planted in central Portugal (Domingues, 2010).This statement is also supported by the Department of Sustainability and environment in Australia which founds that *Acacia* seedlings grows rapidly after fire taking advantages of increase light and soil nutrients. *Eucalyptus* initially is slower growing than *Acacia* sp. but grow to dominate the upper canopy (Úbeda *et al.*, 2006).

The intensity of fire determines the success of vegetation reestablishment, in low intense fires the temperatures leave vegetative structures able to regenerate after fire and species with bark protection improve their survival rates after low intense fires. The vegetative regeneration is an adaptation to the occurrence of frequent fires and few species as *Amplodesmus mauritanica* and *Quercus coccifera* increase their occurrence after fire (Pausas, 2004). High fire intensities diminish the success of vegetative regeneration and the post-fire conditions including temperature, stress, water and nutrient availability, and biotic interactions, such as herbivores are important determinants of vegetative regrowth comparing to the pre-fire vegetation (Lloret & Zedler, 2009). After an intense fire some species are not capable of vegetative regeneration and their seeds do not resist to the high temperatures after fire so these species disappear temporarily after fire although seeds from non-burned places can colonize the burned area if they have a good colonizing capacity (Pausas, 2004).

Different data was compiled with different information about vegetative and seminal regeneration. Vesk & Westoby (2004) reported that vegetative regeneration is widely distributed around different vegetation types in a high-level phylogenetic tree but it is not common in the Gymnosperms. Increased temperatures and time of exposure to fire decrease vegetative regeneration and germinating capacity (Lloret & Zedler, 2009). Habrouk *et al.* (1999) studied the viability of pinaceae seeds and concluded that seeds from the canopy (inside cones) have a higher percentage of germination than those free on the ground.

The effects of fire upon vegetation was reported by Cerdá & Robichaud (2009), Úbeda *et al.* (2006), to let changes in soil processes as a consequence of vegetation and ground cover destruction with increased runoff rates and soil erosion processes and reduced infiltration rates and the importance of vegetation reestablishment after fire to minimize the hydrological and erosional processes and rainfall interception (Cerdá & Robichaud, 2009). As a consequence there is a reduction in soil moisture content and loss of nutrient-rich ash and A horizon sediments. Other effects of fire is the destruction of shallow roots that contributes to soil strength (Cerdá cited Hyde *et al.*, 2007). It is important to explore the mechanisms that allow species to survive after fire as the protecting buds or the existing bank seed in the soil and factors limiting this process as the competition with exotic or introduced species (Lloret & Zedler, 2009). The success of vegetation establishment and regeneration depends on ecological factors as climate, since young plants do not support low soil water content (Baeza, 2004) and farther on the below-above ground characteristics of the biota and soil system. Soil and vegetation are a continuum so abiotic factors as hydrophobicity, soil structure and porosity, available nutrients and the biotic components as fungus and invertebrates communities control the resilience of the soil system after fire (Certini, 2005). In this sense, abiotic processes of soil erosion, soil water repellence, soil structure and biotic process as fungus-plant symbiosis (mycorrhiza and fungi), plays an important role in the establishment of vegetation after disturbance. For example a mycorrhizal plant has a larger soil volume to explore than a non-mycorrhizal one and the possibility to uptake more water and nutrients (Capogna *et al.*, 2009, Goicoecha *et al.*, 2009, Claridge *et al.*, 2009). Vegetation, climate and topography control the resilience of the soil system (Certini, 2005).

To illustrate the communication between above-below ground components of abiotic and biotic factors to the importance of ecosystem resilience, Capogna *et al.*, 2009, studied the resilience of *Phillyrea angustifolia* after fire, a representative mediterranean shrub, under different fire conditions (low and high intensities) and the relation between biological (mycorrhizal plant infection) and abiotic elements. The increased availability of phosphorus (P) due to the combustion of organic matter and consequent mineralisation, after low and intense fires increased the rates of mycorrhizal infection on this specie because plants with higher phosphorus concentration are more vigorous (De Marco *et al.*, 2005). The vegetative regeneration of *Phyllyrea angustifolia* L., increases with phosphorus availability and the influx of this nutrient is increased by mycorrhizal colonization (Capogna *et al.*, 2009). Soil fungi abundance increased in rainy months, in the first spring after fire soil fungi increased due to higher soil water content. There is a relation between biotic and abiotic soil components and growth of *P. angustifolia*, which allows to hypothesis the relationship on above-below ground system resilience (Capogna *et al.*, 2009).

2.2. The importance of vegetation recovery after fire

The re-establishment of vegetation after fire is a result of an auto-succession process done by early post-fire species that differs of species from mature communities. Soon after fire, vegetation regeneration and ecosystem function are potentially restored and the ability of recovery is determined by the impact of such disturbance on plant population as the strategies of these plants to re-establish in the new post-fire conditions (Lloret & Zedler, 2009).

Vegetation recovery, in the Mediterranean basin, could be very rapid due to low competition for sunlight, increased nutrient availability from ashes deposition and reduced water losses by transpiration (Naveh, 1975). In the other hand Mayor (2007), found in a Mediterranean dry area, a slowly recover of vegetation indicating a short term ecosystem resilience in the first two years after fire. In the study of Silva, Rego & Mazzoleni (2006), an intense wildfire burned a control shrubland plot, and no regeneration of *Erica* sp. *Crataegus* sp., *Ulex* sp. and *Rubus* sp. vegetation occurred during a 3 month period after the fire occurred in September.

As explained the resilience of soil system also depends on other factors as meteorological conditions, vegetation type, fire intensity, soil characteristics and topography of the burnt area (Certini, 2005; Silva et al., 2006).

A model of regeneration was described by Úbeda *et al.* (2006), to a typical evergreen Mediterranean forest dominated by *Quercus ilex* and *Quercus suber*. These communities have good vegetative regenerators and a fire resistant seed bank. Herbs play an important role in the initial stages protecting against rainfall impact. Then comes the typical woody species of shrub areas which regenerates their aerial parts from fire survival structures whereas others regenerate from the seed bank that has survived (Gordi *et al.*, 1996).

But not all species in the Mediterranean basin can display an auto-succession pattern after fire: in a Mediterranean gorse community De Luis, Raventós & González (2006), reported that the dominance of Fabaceae change to Cistaceae seedlings with a decrease in the emergences of Lamiaceae (restricted to *Rosmarinus officinalis*, an obligate seeder), after fire.

The resilience of species may diminish due to various factors: 1) the continuous accumulation of fuel that increases fire intensity and severity; 2) the absence of species (bank seed and alive buds unavailable); 3) human-induced changes in species composition could influence the resilience capacity at the community level since introduced species have different strategies of regeneration from the native species which also change the historical fire regime; and 4) granivory that decreases the post-fire seedlings establishment and variability of species in a plant community (Lloret & Zedler, 2009).

According to Doerr *et al.* (1984), to a successful auto-succession pattern, the above-belowground biological interactions as the fungi forming vesicular-arbuscular mycorrhiza (VAM) a symbiotic interaction with plant roots that is very important.

The importance of soil biota-plant interactions are going to be explained in chapter 4, but here is a short explanation about the importance of fungi to the regeneration of degraded areas: the infectivity of AM inoculums on plant is high and persists in the succession, providing a possible explanation for the initial dynamism of mycorrhized plants on disturbed habitats, Doerr *et al.* (1984) reported that above ground biomass production increased with mycorrhizal inoculum infection on plant because this symbiosis improve the uptake of phosphorus especially in nutrient-poor soils (Wilson & Harnett,1998; Gange *et al.*,1990) reported that AM fungi are important in the seedling stage (the earlier stage of plant succession); Wilson & Harnett (1998) and Barni & Siniscalco (2000), reported the benefits of this association during reproduction that latter affects the differential growth, the demographic response and the relative abundance of species in a certain plant community. When high intense fires occur, could happen a destruction of plant roots, but there are pivotal / pionner fungus species in an early system recovery, which helps to minimize the movement of soil in the absence of plant roots, as described by Claridge *et al.*,2009. An example is *Anthracobia* sp. fungi, which have mycelial mats aggregating soil particles, in the dry areas of New South Wales. This fungi forms symbioses in the roots of *Eucalyptus albens*, *Eucalyptus viminalis* and white cypress pine, *Callitris glaucophylla*. Other functional role of early post-fire fungus includes nutrient acquisition improving the reestablishment of vegetation. As plant root system re-establishes, the importance of these fungus in soil stabilization may diminish.

The recovery of ground vegetation, shrubs and trees, protects the soil and a lack of vegetation means that rainfall impacts directly on soil surface after fire, as a consequence the processes of soil degradation, such as destruction of soil aggregates and infiltration capacity, possibly ends in soil erosion (Úbeda *et al.*, 2006).

Also Doerr & Thomas (2000), reported that the removal of vegetation and litter after wildfire disturbance increase runoff and soil erosion and Cerdá & Doerr (2005), reported that vegetation cover is a key factor on the aboveground ecosystem contributing to the decline in the post-fire runoff and soil erosion rates.

To a further understand of this statement, different types and densities of spontaneous vegetation of a community, following a severe wildfire in eastern Mediterranean, from four different groups, trees, herbs, shrubs and dwarf shrub, were studied. Some of this vegetation regenerates from shoots after fire: *Quercus coccifera*, *Pistacia lenticus*, *Erica multiflora*, while other vegetation grew from seedlings after successful germination (*Brachypodium retusum, Cistus albidus, Pinus halepensis, Ulex parviflorus)*, Cerdá & Doerr (2005). The herbs-dominated plots rapidly grow with a dense cover over 50% in the first six months after fire. Three years after fire the shrubs *Thymus*, *Cistus* and *Quercus coccifera*, exceeded 100% of cover while *Ulex* and *Erica* have the lowest coverage percentage. The trees reached 60-80% of coverage in these three years after fire. The different time response of the different groups of vegetation influences the rates of runoff and soil loss (Cerdá & Doerr, 2005). To consider the effects on soil hydrology and erosive processes of climateric conditions on the studied area, were done simulated rainfall experiments to collect data independently of natural rainfall variations. A further view on this is presented in chapter 3 (3.2).

Lloret & Zedler (2009), consider the importance of some management strategies to improve forest resilience in the post-fire conditions, to find equilibrium between the expansion of high resilient non-native plant communities and the spontaneous plants from old forests with depleted species to regenerate after fire. The frequent application of prescribed fire could be done, due to the possibility of accelerated erosion as a result of mortality of less resilient plants replaced by species with lower capacity to control erosion (Coelho *et al.*, 2002). The exotic species could be cut of on the basal log or pulverized with herbicide to dissect and defoliate vegetation and favour the development of less resilient species. However, it’s necessary to eliminate accumulated fuel again; this is why this is an expensive technique (Baeza, 2004).

Biotic interactions as herbivory, granivory and pests also deteriorate resilience capacity at the species level. To improve fire resilience of forests is desirable the manipulation of fuel, cutting and thinning, planting of native species and chemical control of undesirable exotic species.

Azul & Freitas (2005) reported that intense fire causes reduction of the vegetal cover and initiates physico-chemical and biological changes in the soil that could be traduced in the acceleration of erosion processes. Adding to this idea, the changes in composition and structure of the main species of a plant community population could affect the behaviour of soil system with some consequences on the hydrological and erosive processes and consequently on the ecological role of forests as explained in the next chapter. The introduction of commercial seeds leads to a less availability of water, light and nutrients to the native plants and the increase of *Ulex* sp. and *Cytisus* sp., also decreases the successful establishment of native species and the elimination of highly productive grazing areas (Baeza, 2004). Another consequence related with erosive processes is sedimentation due to the introduction of pioneer conifers in order to introduce the successional latter species.

2.3. REFERENCES

Baeza M.J. (2004). El Manejo del Matorral en la Prevención de Incendios Forestales. In: Avances en el estudio de la gestión del monte mediterrâneo. Vallejo V.R., Alloza J.A. (Eds.). Fundación CEAM, pp.65-92.

Barni E.& Siniscalco C. (2000). Vegetation dynamics and arbuscular mycorrhiza in old-field successions of the western Italian Alps. Mycorrhiza, 10 (2): 63-72.

Catry F., Bugalho M., Silva J. (2007). Recuperação da Floresta após o Fogo, O caso da Tapada de Mafra. Centro Ecologia Aplicada. Instituto Superior de Agronomia, Lisboa.

Catry F.X., Rego F.C., Bugalho M.N., Lopes T., Silva J.S., Moreira F. (2006). Efeitos do Fogo na sobrevivência e regeneração das árvores num ecossistema mediterrânico. In: Viegas D.X. (Ed.).

Cerdá A., Doerr S.H. (2005). The influence of vegetation recovery on soil hydrology and erodibility following fire: an eleven-year investigation. International Journal of Wildland Fire, 14 (4): 423-437.

Cerdá A., Robichaud P. (2009). Fire Effects on Soil Infiltration. In: Cerdá A. & Robichaud P. (Eds.), Fire Effects on soils and restoration strategies. Land Reconstruction and Management, (pp. 81-103).

Doerr T.B., Redente E.F., Reeves F.B. (1984). Effects of soil disturbance on plant succession and levels of mycorrhizal fungi in a sagebrush-grassland community. Journal of Range Management, 37(29):135-139.

De Luis M., Raventós J. & González-Hidalgo J.C. (2006). Post-fire vegetation succession in Mediterranean gorse shrublands. Acta Oecologica, 30 (1): 54-61.

Fernandes, P., 2009/2010. Fichas-tipo das relações entre o fogo e a floresta. Eucalipto, Pinheiros.

Gange A.C.,Brown, V.K., Farmer, L.M. (1990). A test of mycorrhizal benefit in an early successional plant community. New Phytologist, 115: 85-91.

Gordi J., Pintó J., Vila J. (1996). L´ estudi dels incendis en el món mediterrani. Doc. Anàl. Geogr., 28:135-151.

Habrouk A., Retana J., Espelta J.M. (1999). Role of heat tolerance and cone protection of seeds in the response of three pine species to wildfire. Plant Ecology, 145: 91-99.

Lloret F., Zedler, P.H. (2009). The effect of forest fire on vegetation. In: Fire Effects on soils and restoration strategies. Cerdá A. & Robichaud P. (Eds.). Land Reconstruction and Management, pp. 257-295.

Marchante, E., Marchante, H. (2005). Plantas Invasoras em Portugal, ficha 15. Escola Superior Agrária de Coimbra: Available on: http://www1.ci.uc.pt/invasoras/files/15acacia-de-espigas.pdf

Marco A., Gentile A.E., Arena C., De Santo A.V. (2005).Organic matter, nutrient content and biological activity in burned and unburned soils of a Mediterranean maquis area of southern Italy. International Journal of Wildland Fire, 14 (4): 365-377.

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

Moreno J.M., Cruz A., Fernández F., Luna B., Pérez B., Quintana J.R., Zuazua E. (2004). Ecología del Monte Mediterráneo en relación con el Fuego: El Jaral Brezal de Quintos de Mora (Toledo). In: Avances en el estudio de la gestión del monte Mediterráneo. Vallejo V.R. & Alloza J.A. (Eds.). Fundación CEAM, pp.65-92.

Naveh Z. (1975). The evolutionary significance of fire in the Mediterranean regions. Vegetation, 29: 199-208.

Pausas J.G. (2004). La recurrencia de incêndios en el Monte Mediterráneo. In: Avances en el estudio de la gestión del monte Mediterráneo. Vallejo V.R. & Alloza J.A. (Eds.). Fundación CEAM, pp.47-64.

Silva J.S., Rego F.C., Mazzoleni S. (2006). Soil water dynamics after fire in a Portuguese shrubland. International Journal of Wildland Fire (15): 99-111.

Vesk P.A. & Westoby M. (2004). Sprouting ability across diverse disturbance and vegetation types worldwide. Journal of Ecology, 94: 1027-1034.

Wilson G.W.T & Harnett D.C. (1998). Interspecific variations in plant response to mycorrhizal colonization in tallgrass prairie. American Journal of Botany, 85 (12): 1732-1738.

CHAPTER 3

Effects of fire on soil processes

CHAPTER 3 EFFECTS OF FIRE ON SOIL PROCESSES

3.1. Introduction

A forest soil develops under the influence of a forestry tree or shrub cover with an organic layer and as a consequence of the deep rooting during long periods of trees and shrubs and to specific organisms associated with forest vegetation. The organic soil layer consists on a litter layer and in an organic cover upon the mineral horizon and the associated microflora and microfauna of the organic cover are the dynamic phase of the soil, essential to maintain the nutrient cycling of nitrogen of phosphorous and sulfur (Serrasolses, Llovet & Bautista, 2004).

The world soil research, published by FAO in the 90´s, indicates that in the majority of European countries the main causes of soil degradation are inadequate agriculture practices and deforestation (Bot, Nachtergaele & Young, 2000). Since immemorial times, soils from temperate forests were used for agriculture, grazing, wood collection and coal production. In the Mediterranean basin the soils in the valleys and the deeper soils on the hillslopes were used in agriculture practices while the woody and bush forests grew in high sloped mountains in superficial, discontinuous and stony soils (Domingues, 2010; Serrasolses et al., 2004). The thickness of the surface soils contributes to the erodibility and degradation of forest soils. On the other hand organic horizons protect the soil from extreme temperatures, confering mechanical protection against erosive agents improving the infiltration rates. In general Mediterranean soils have shallow organic layers, discontinuity and the organic matter content in the soil is low which means that is a mineral soil. In a temperate forest the soils have similar characteristics but the humification rate is high while the mineralization rate is low comparing with Mediterranean soils (Serrasolses *et al*., 2004). According to Jenny (1941) there is a huge diversity of soil types in the Mediterranean basin as a consequence of climactic conditions, rock composition, topography and organisms activity (microorganisms, plants and higher animals), that influences the properties and the diversity of forest soils.

The diversity of forest soils provides different responses to fire depending on the characteristics of hillslope length, steepness and shape; on the intensity and duration of rainfall and on the vegetation cover, but also on soil properties as resistance to penetration and torsion (Scott & Van Wyk, 1990) which determines the consequently post-fire erosion and the hydrological response. Prescribed fires attain lower temperatures than wildfire with distinct effects on the soil erosive response because temperatures above 450ºC induce water repellence on soils and different soil and water degradation processes (Coelho *et al.* 2002). Soil erosion is a consequence of detachment, transport and deposition of soil particles in response to a wildfire (Varela *et al*., 2009), and depends on structure and on the granulometric composition of the soil as on the dimensions and on the cohesive forces between soil particles and thus on its resistance to detachment and transport. The soil constituents like clays, organic matter, Fe, Al oxides and carbonates are keys to indirectly estimate soil erodibility (Varela *et al*. 2009). Ferreira *et al.* (2008), supported the idea that the detachment and transport of soil particles are influenced by the increasing slope steepness, by the erosive forces and also an increase in overland-flow. Fire disturbance leads to high erosion rates comparing to a non disturbed area due to the high erosion rates produced on hillslopes and also in consequence of rills and gullies formation (stream channels), which generate a high concentration of sediments since rainfall does not infiltrate producing a flow downhill under the action of gravity (Scott & Van Wyk, 1990). The existence of rills and gullies on the hillslope area increase the rates of water erosion and soil detachment because the output of sediments will be higher and consequently the erosion yields and runoff coefficients will increase (Ferreira *et al.,* 2008). Some alterations on the land surface also influences soil particle detachment and the downslope transport after the passage of fire (Varela *et al*., 2009).

The heat of a fire is other variable causing soil erosion due to the physico-chemical and biological disruptions that can be controlled under laboratory conditions. For example, the destructive distillation and combustion of about 85% of the litter layer occurs in the range of 180-300ºC and N volatilization begins when temperatures climb to 200-400ºC (Giovannini,1994). When fire consumes vegetation and the underlying litter layer, the bare soil surface can seal off under the impact of raindrops resulting in water erosion (Ferreira *et al*., 2009).

Varela *et al.* (2009), studied the effects of wildfire on soil properties comparing to non-burned sites and these authors selected some soil properties as aggregate size distribution (determined by sieving samples with different mesh sizes and expressed according to the dry mean weight diameter (MWD) and water aggregate stability (measured using the water drop impact (WDI) releasing about 200 water drops of 0.1g from a height of 1m) in accordance to fire severity. Wildfire had negative effects on aggregate size distribution and a variable effect on aggregate stability due to the effects on soil organic matter that negatively affects soil erodibility. The organic carbon content is important to the formation of structure and to a good hydrological function.

The hydrological erosion processes after a fire are characterized by overland and rill flow that are the main hillslope erosion processes after wildfire with a consequently problem of ashes deposition over the soil surface representing part of the nutrient stock and a serious soil degradation risk (Ferreira *et al.* 2009).

Changes on the hydrological properties of the soil are reduced infiltration rates due to the formation of a water repellent layer, the condensation of the organic soil compounds and the combustion temperatures of the available fuels (Ferreira *et al.,* 2008). The deterioration of the hydrologic function as a consequence of fire leads to a decrease in the ecosystem sustainability (Ferreira *et al.* 2007) and these changes are a dependency on the severity of fire and on the spatial variability of water repellence.

Some popular techniques to assess soil and water degradation processes are:

1) Soil profiling devices: a method consisting on the loss and gain of soil (erosion rates) that allows to measure erosion rates, investigating the changes in aspects of microtopography as surface stone lag development and soil surface roughness. The mean soil loss is negative denoting a soil level lowering. The gain of soil is positive denoting a soil level rise (Ferreira *et al*., 2008).

2) Quantification of soil and water degradation processes at micro-plot, plot and catchment scale: they consist in a given area that are partially or fully delimited draining to a collecting device called plots that allows the monitoring of overland flow and erosion yields. Some restrictions are related with the monitoring of water and sediment processes due to the variability of the spatial distribution of water repellence that could be restricted to small and hydrological isolated areas and as a consequence overland flow produced on repellent soils could well infiltrate at hydrophilic areas down-slope which could explain the decrease in the overland flow amounts, sediment and solute transport from the micro-plot scale to the $16m^2$ scale and wider scales (slope and catchment). The advantages are the control of variables inside the plot allowing the quantification of overland flow and erosion rates at a given period.

Some variability could occur depending on the location of the plots in the slope and with vegetation composition which alters the magnitude of runoff and overland flow. In areas with gullies and rills formation the erosion yields and runoff coefficients are high (Ferreira *et al*., 2008). The cited author, also explained the differences on overland flow and erosion yields from rainfall simulations with the same intensity higher in micro-plots than in $16m^2$ bounded plots, due to the increasing degrees of entropy on a short period however it is not clear the risk of peak flows and the mobilization and deposition of sediments downstream. At wider scales (for ex: slope and catchment), erosion rates and overland flow decreases significantly and the decrease is more pronounced in prescribed fires than in wildfires due to the existing hydrophilic areas coexisting with hydrophobic areas of soil repellence in low intense and prescribed fires forming an heterogeneous pattern. According to this idea the use of prescribed fire is a good tool to prevent fire. In severely burned areas soil water repellence is extreme and homogeneously distributed forming an uniform pattern and to Shakesby & Doerr (2006), wildfire can induce, enhance or destroy soil water repellency, depending on the temperatures reached and on its location. These authors stated that the heterogeneity on the initial conditions of soil properties as soil moisture, vegetation and litter layer patches and infiltration are also important to the over scale results.

3) Rainfall simulators: give data on time and reproduce the characteristics of natural rain as rainfall intensity, size and distribution of rainfall drops. The rainfall simulators use plots with various sizes and are often used to study overland flow.

4) To monitor small catchments: sediment traps have to be build in the stream channels to trap the sediments transported and special sampling equipment must be installed to collect the suspended and dissolved loads.

3.2. Fire impacts on soil and water erosive processes

The generation of soil and water degradation processes depends on the initial conditions of the soil affected by fire severity. It is possible to say that the direct effects of fire are upon the litter layer and vegetation, on the organic matter content and on soil aggregate stability. Fire indirectly affects infiltration rates, the soil-water repellent layer and the soil moisture content.

In the burned areas occurs fast soil and water degradation processes, the presence of a water repellent layer interferes with soil and water degradation processes as explained in b) due to the formation of a hydrophobic layer that could intercalate with hydrophilic layers and with soil macropores (a result of the burned roots of shrubs and trees). In the last two situations infiltration occurs and it rates depends on the referred conditions as depend on the variables described in 3.2 a), b), c), e) and f) (Coelho *et al*., 2002; Coelho *et al.,* 2004; Ferreira *et al.,* 2008). The differences on soil and water degradation processes after a prescribed fire or a wildfire is primarily due to the spatial variability on soil water repellence that leads the most important soil degradation impacts during fires. Coelho *et al*. (2002), compared the effects of a prescribed fire on a shrubland area and the effects of a wildfire on a mature pine forest comparing with the respective control sites. These authors found amounts of eroded sediments in the prescribed shrubland fire 10 times higher comparing with the 20 years shrubland control site and a striking impact on soil degradation in the case of wildfire comparing to the pre fire conditions. Litter layer and vegetation cover destructed by the passage of a wildfire creates a repellent layer limiting infiltration to hydrophilic and macropore areas and leads a little control on overland flow and on sediment transport at burned areas and the combustion of the root system forming an extensive macropore system (Ferreira *et al*., 2008)

The soil water content (SW) decrease and increase in different magnitudes after the passage of a fire especially at the surface layers, influencing infiltration and the soil water degradation processes. It decreases less during the dry season (June-September) and increases more in the wetting season (October-December), influencing the soil and water erosive processes (Silva, Rego & Mazzoleni, 2006).

The soil/land use can create a high magnitude of soil and water degradation effects, for example, in the study of Coelho *et al.* (2005), in the Águeda basin the soil erosion rates were an order of 2 ton/ha front to 50 ton/ha in a rip ploughed area. According to this results Ferreira *et al*. (2008), found high erosion rates in rip-ploughed plots than in plots burned by wildfires.

The prolonged dry and hot conditions, typical of the Mediterranean summer, increase soil and aggregate transport downslope, due to the uninterrupted particle accumulation on the soil surface and the air trapped in dry aggregates which makes them more vulnerable to breakdown during rapid wetting (Cerdá & Doerr, 2005).

The post-fire erosion processes have been studied at small rather than large scales and now is reviewed the direct effects of fire on soil properties that are going to influence the soil and water erosive processes as a) the litter layer, vegetation cover and soil water content; b) water repellent layer; c) soil aggregate stability, and the indirect effects on d) organic matter content; e) infiltration capacity; f) overland flow. A special focus is put in a) and b) and how these variables influence overland flow and erosion processes.

a) Litter layer and vegetation cover

In this item is focused the effects of fire on litter and vegetation destruction and the immediately effects occurred after this disturbance directly on the soil surface and on the below-ground soil. The elimination or reduction of the aerial parts of vegetation and litter layer has two basic consequences: a) Increase in soil water content as a result of reduced plant transpiration, a reduction in canopy interception and throughfall increase; the elimination of water uptake by plants leading to an increase in SW content immediately at surface layer and lately delayed to deeper soil layers (Silva *et al*., 2006) b) low soil water content due to the seal of surface soil pores and to the low buffer capacity generating lower infiltration rates, lower soil water content and higher runoff rates. The exposure of soil to different meteorological agents (solar radiation, wind and rain) decreases soil water retention (Silva *et al*., 2006).

The soil water content recharge influences the amount of basal flow and surface flow contributing to the overall flow in the watershed and to the post-fire hydrological effects. The different situations on soil water content could be a result of the initial different existent conditions (ex. fire intensity, vegetation type, soil characteristics, meteorological conditions) and in terms of the used methodology as is the depth at which soil water measurements are done (Silva *et al.,* 2006) and the calibration of the used equipment.

Other consequence of fire over vegetation is the deposition of ashes on the soil surface as a consequence of the burnt vegetation representing a seriously degradation risk in down-slope areas (Ferreira *et al*., 2008). Silva *et al*. (2006), studied vegetation development by measuring the shoots of *Erica* sp. and the fronds of *Pteridium aquilinum*, which were the first signs of vegetation recovery and they measured plant height and the resprouting capability of plants that started in two weeks after fire. At the end of the second growing season the shoots of *Erica* sp. almost attained the height of *Erica* sp. shoots on the burned plots. It is important the SW content recharge in order to the recovery of vegetation during the post-fire period. In the burned shrubland plots Silva *et al.* (2006), reported high soil water content from 30 to 170cm deep on the surface which contributes to plant recovery. The spatial distribution of the hydrophobic layer influences the available surface water and the basal flow water (Ferreira *et al.,* 2009). On the other hand Cerdá (2009), reports that vegetation loss and other ground cover reduces rainfall interception, rainfall storage and the flow resistance (Cerdá, 2009) and the net effect is a reduction in soil moisture content and erosion of nutrient–rich ash and A horizon sediments. Also Shabesky and Doerr (2006), reported a reduced water storage capacity despite of reduced plant transpiration and reduced rainfall interception due to soil exposure to thermal regime as solar heating. Soil water content beyond other physical soil properties as porosity and hydrophobicity are the better tools to understand the whole process that leads to soil loss. Other morphological character of vegetation are the roots, could be studied to understand the other effects of fire along a studied profile as root length and the distance until root apex due to the influence of roots on soil water uptake. The root mats are combusted in temperatures above 300ºC (Shakesby & Doerr, 2006) and the available space left by the burned roots allows water recharge. The study of Silva *et al*. (2006), found a decrease on root length along a studied soil profile.

The time of burning is an important role to determine the possibilities of vegetation recovery and its effects on soil water storage. The vegetation cover composition is also important when there is a sudden decrease on the soil water content and so an important variable character (for example, the type of roots, fleshy or leathery leaves, trunk thickness and inflammability).

According to Cerdá and Robichaud (2009), vegetation is the variable with the largest impact on infiltration rates which also depends on fire severity with a better or worst control on overland flow generation and on sediment transport in the burned areas influencing surface flow and the rate of water infiltrated. Vegetation and litter layer mitigates the impact of rainfall on the soil and due to the destruction of soil surface and the sealing off surface under raindrop impacts results in higher surface runoff and in decrease infiltration (Ferreira *et al.,* 2008). Vegetation provides in the below ground layers biological activity, soil aggregation, micro-macro pore development and water storage. The recovery of vegetation, normally, begins quickly after fire and is often re-established within several years, due to the high resilient capacity of Mediterranean vegetation.

The presence of vegetation increases litter layer deposition (are natural barriers to the left leaves from the canopy and litter fall) improves the formation of an organic layer, improving soil structure due to the high aggregability and porosity that allows the soil to hold large quantities of water and as a consequence the improvement of infiltration rates (Cerdá & Robichaud, 2009).

Variables contributing to different effects on soil properties as (infiltration rates and soil water content): fire intensity, vegetation type, soil characteristics and meteorological conditions. It is important to study soil water storage content after fire beyond physical soil properties (for ex. porosity and hydrophobicity) to understand the whole processes leading to soil loss.

According to Silva *et al.* (2006), more studies are required to measure soil water content down to levels of deep rooting in order to understand the soil water dynamics vs. deep rooting once they are important in water uptake during the dry season. As it was described and as a consequence of fire the removal of vegetation and litter layer due to fire and other important feature as the occurrence of changes on soil water repellent layer and the magnitude effects depends on fire severity as wildfire vs low intense fire with impact on soil erosion and on the hydrological effects.

b) Water repellence

Water repellence is a discreet layer of soil parallel to the soil surface acting as a barrier to water infiltration and diminishing soil evaporation. After a rainfall event or rainfall simulation a wetting front is formed moving through the wettable layer reaching the water repellent layer and infiltration drops to the water repellent soil and then again to the wettably front layer when soil infiltration rates increase again (DeBano, 2000).

After the passage of fire in a burned watershed there is a wettable layer formed by ashes deposition and beneath a water repellent layer formed by the passage of a wildfire, that as been already described as an hydrophobic area intercalating with hydrophilic and macropore areas that occurred after low and prescribed fires (Ferreira *et al*., 2008).

The hydrophobic layer is a consequence of organic substances volatilization in the litter, humus and on top soil layer and the coating of mineral particles occurs at temperatures around 176ºC and are destroyed at temperatures above 288ºC, due to the long periods of heating once the organic substances are destroyed and water does not wet aggregates and infiltration declines (DeBano, 2000; Mataix-Solera, 2004; Ferreira *et al*., 2008; Ferreira cited Neary *et al*.,1999).On the other hand, the reduction or elimination of repellence is caused by the volatilization and combustion of organic compounds (De Bano, 2000).

The degree of soil water repellence is a result of different times and temperatures of heating and on the organic matter content over the soil surface (DeBano, 2000).The decomposed vegetation on the soil surface and the mycelium fungal growth is a source of hydrophobic substances in the upper soil horizon (DeBano, 2000; Claridge et al., 2009) influencing the erosional and hydrological processes and aggregate stability.

Ferreira *et al.* (2008) reported that burning caused widespread destruction of repellence in the mineral soil (0.5-5cm) but bellow the wettable layer is present a repellent layer in the surface and subsurface (Ferreira *et al*., 2008). Robichaud *et al*. (2007) reported that in areas of moderate and high soil burn severity the induced repellence is most often detected at 1 to 3 cm below surface and immediately below the ash layer is absorbent. Other data accordant to this statement from a study in the woodland types of Galicia with burned soils exhibited a strong or very strong water repellent layer in the upper soil layers and below soil surface a decrease from strong to moderate repellent severity layer (Varela *et al*., 2009).

It’s possible to deduce the temperature on the soil surface affected by a fire due to the fact that the soil heating regime have more markedly effects on water repellence than on soil properties (Varela *et al*., 2005; 2009). To examine the changes in water repellence due to soil temperature, the unburned soil samples from woodland types of Galicia were subjected to a controlled heating experiment with temperatures ranging from 25ºC to 460ºC and an exposure time of 30 minutes and the results evidence that water repellence increases between 25º and 220ºC and the maximum values of repellence or peak values occurred between 220º and 240ºC. The hydrophobic soil layer is destroyed in a range of temperatures between 260º to 280ºC confirming the previous studies of De Bano *et al*. (1976) and De Bano (1981) according to the study of Varela *et al.* (2005).

The longevity of fire-induced water repellence and the degree and distribution of the water repellent layer depends on combined interactions of soil properties as organic matter content, soil moisture, texture, depth and on the temperatures reached during a fire, that increases with more fuel load and decreases with the increasing of soil moisture content (Varela *et al*., 2005; De Bano, 2000) and the longevity of fire induced repellence varies according to fire severity: a low or moderate severity fire is normally of short duration and a high severe fire normally is of long duration (De Bano, 2000).

To understand the soil repellent behaviour according to soil characteristics Varela *et a*l. (2005) compared pine, eucalyptus and shrubland areas of unburned and burned soils with similar characteristics of ph, vegetation type and parent material and they found more hydrophobic initial conditions on eucalyptus and pine forests comparing to shrubland areas with a wide water repellent layer. The soil characteristics conferring hydrophobicity were soil carbon content due to its low solubility and the strong soil acidity that favours the biomass of fungi content. The same author reported that repellent layer increases after burning and the repellent layer disappears on the surface layer increasing with depth according to the temperatures reached and described above.

Depending on burn intensity Dyrness (1976), found that wettability characteristics of soils on areas that burned in a low severity fire recover fast than the repellent layer of soils severely burned and also McNabb *et al.* (1989), reported that the water repellent layer formed by a low severity burn in the southwest Oregon forests, allows infiltration at a nearly normal rate after the rain period. Other studies at wildfire areas and after prescribed burning in the Central of Portugal evidence the presence of an extreme repellent layer in the first situation and a slight hydrophobicity in samples collected at (0-5 cm) in the second case. Also the effects of prescribed and wildfires on water repellence suggests that the wettability of soils burned in low and high severe fires approaches the pre-fire soil conditions by the sixth year after fire (as cited in DeBano, 2000) and soils burned in wildfires have strong hydrophobic characteristics (MED>13) with a uniform pattern distribution.

Some consequences of water repellent soils are rill formation that occurs when rainfall exceed the infiltration rates occurring surface runoff and in this cases hydrophobic soils are affected by raindrop detachment because this soils are dry, non-cohesive and displaced by splashing when raindrops break the surrounding water film (De Bano, 2000) however according to Shakesby and Doerr (2006), the wettable post-fire soils indicate that repellency is not a pre-requisite for rill erosion.

c) Soil aggregate stability

Soil aggregate stability is an important factor with a high influence in soils susceptibility to erosion and in soil water infiltration capacity that changes after the passage of a fire due to vegetation and litter removal which is a key characteristic in soil water management and in nutrients and soil loss materials (Mataix-Solera y Guerrero, 2007). Furthermore the occurrence of fire can induce water repellence in previously hydrophilic soils or in previously hydrophobic soils and the soil water repellence can be increased or diminished depending on temperature, time of burning, quantity and type of organic matter consumed and soil moisture content (DeBano *et al*.1976; Doerr *et al*. 1998, 2004; Cerdá & Doerr 2005; Shakesby *et al*. 2007; Arcenegui *et al*.2007).The water repellency induced or favored by the passage of a fire as been described as the principal cause of increased superficial runoff and soil erosion in burned areas (DeBano 2000; Doerr *et al.* 2000; Shakesby & Doerr 2006).

Varela *et al*. (2010) found that the effect of fire on aggregate stability was highly variable. Aggregate stability was clearly lower at the burned than unburned site in about a third of the studied cases and in the remaining cases either basically the same at both sites or higher at the unburned site. The differences in aggregate stability appear to be associated with changes in organic carbon content. The combustion on organic matter content favours dry disaggregation and at the same time abruptly reduces water aggregate stability because it is an important factor contributing to surface soil structure and porosity that is profoundly affected by fires (Ferreira *et al*., 2009). The organic matter in forest soils reaches a maximum stability and maturity between 100 and 160ºC (Almendros *et al*., 1984).

The repellency induced by fires can influence aggregate stability and erodibility in opposite manners. The strong repellent conditions, immediately below the surface, protect the aggregates against slaking by the impact of water drops and thus increasing stability, but on the other hand increases erodibility by reducing infiltration and enhancing overland flow (Varela *et al*., 2010).

The effects of fire in the soil biota and the consequently effects on soil aggregates lack of some studies as the mechanisms by which AMF and extra-radical hyphae help to stabilize soil aggregates that are still not clearly understood (Rillig, 2004). The AMF are thought to have an important contribution to the stabilization of macroaggregates, a basic building block of soil structure (>250 µm) where they are hypothesized to help the stabilization of aggregates via hyphal enmeshment (the string-bag mechanism) and by deposition of organic material since AMF hyphae forms glomalin, that is a glico-protein deposited in outer hyphal walls reducing macroaggregate disruption during wetting and drying events and by retarding water movement into the pores within the aggregate structure (Miller & Jastrow, 2000).

d) Organic matter content

The amount of organic matter has influence in erodibility processes of soil erosion and overland flow and the heat of a fire may transform soil organic components to become mobile coalescent to mineral particles giving rise to hydrophobicity in the upper soil layers (Coelho *et al*., 2002 cited Giovannini,1994; Varela *et al*., 2005).

The distillation of organic matter starts in a range of 200-315ºC. However the substantial loss of organic matter occurs at lower temperatures. The litter scorch and the distillation of volatile organic compounds occurs at (180-200ºC). Nutrient volatilization occurs in a range of 200-400ºC and at temperatures above 300ºC the surface organic horizon of the soil is consumed (Ferreira *et al.,* 2008). Heat also affects the organic matter composition and in this sense the studies in Atlantic forests reveal that cellulose, hemicellulose and water-soluble compounds declined significantly after fire, whereas lipids did not vary (Varela *et al*., 2005 cited Fernández *et al*., 1997). Another evidence after fire on forestry soils with different groups of herbaceous species in Portugal, reveal that the medium weight of organic content decreases when is analyzed the humyfied layer (Macedo, 1993).

In the previous study reported made by Coelho *et al.* (2002), in the central region of Portugal (Góis, Cadafaz and Caratão), the organic matter contents were higher in shrubland and in the prescribed burning than on wildfires and in mature pine plots and the organic matter was mainly composed of ashes and nutrients mobilized on the degradational processes of fire-induced changes.

Other studies concerning fires of low severity explained the elevated organic matter contents with the incorporation of partially burned ashes into the soil (Varela *et al.,* 2009 cited Boix Fayos, 1997), and the deposition and incorporation of unburned or partially burned plant material (Varela *et al*., 2009 cited Hungerford *et al.,* 1991). Ludwig *et al*., 1998 reported that the partial decomposition of the roots due to the heating of the soil can also contribute to the increase of soil organic carbon content (as cited in Varela *et al.,* 2009).

In the studies of Varela *et al*., 2009, at the unburned sites of *pinus* and *eucalyptus* forest and shrubland, there is an elevated organic carbon content and at a same similar location of parent material and vegetation of recently burned soils and the amounts of organic matter doesn´t differ in a consistent manner from the neighbouring unburned soils. There is much variability in the amounts of organic matter with some positive peaks corresponding to higher organic matter contents after fire and the negative peaks corresponding to great amounts of organic matter at the unburned areas.

Also Chandler *et al*. (1983), demonstrate that the effect of fire on soil organic matter is highly variable ranging from a total destruction of organic matter of up to 30% in the uppermost soil layers (as cited in Varela *et al.*, 2009) due to the combustion of the organic matter (Varela *et al*., 2009 cited Bará and Vega, 1983; Fernández *et al*., 1997; Mataix-Solera *et al*., 2002). To Giovannini and Lucchesi (1997), the combustion of organic matter increases proportionally with the severity of fire (as cited in Varela *et al*., 2009).

Other results of soil characteristics concerning soil resistance to both penetration and torsion, right after burning are higher at the wildfire than at the prescribed burning site (Coelho *et al*., 2002).The associated soil density was lower in the shrubland and prescribed fire and higher in the pine and wildfire site.

Other aspect of the study described before, in the mountain forests of Portugal, both the organic matter content (mainly ashes) and eroded soil are higher in wildfires than in prescribed fires. The organic matter content is above 50% in wildfires and about 30% in prescribed fires on the ashes and nutrients mobilized on the degradational processes of fire-induced changes (Coelho *et al*., 2002).

e) Infiltration

Infiltration plays a key role in portioning the rainfall into surface runoff moving in overland flow and subsurface flow and is a key process on the hydrological cycle.

After a fire the soil surface is sealed by ashes and fine soil particles, the soil is compacted, the capillarity forces are reduced and water just fills the fine soil pores so the volume of water infiltrated per time is low and declines comparing to pre fire conditions reaching a constant value after several minutes of rain. The amount of water infiltrated is available for vegetation regrowth, to the soil biota and this is the water in the aquifer in the other hand the amount of water over the soil surface becomes surface runoff and may cause erosion (Cerdá & Robichaud, 2009).

The infiltration rate after fire is directly related with the repellent layer and this variable depends on the organic matter layer combusted. If the fuel load over the soil and the soil moisture content before fire is similar not patchy, the infiltration rates will have a low spatial variability in the post fire conditions, however the variability on the infiltration rates will increase during time as a consequence of the surface ash layer, the repellent layer, vegetation recovery and soil water recharge (Ferreira *et al*., 2009).

While in the first days or weeks after fire the ash layer is thick enough to absorb rainfall after several weeks the climacteric effects of wind/rain the ash layer is moved downslope, while the raindrop impact and surface runoff clogs the soil pores, causing surface sealing and infiltration reach the lowest values (Cerdá and Roubichaud, 2009). The increased sediment and ash in runoff and lack of vegetation cover contributes to a decrease in infiltration rates. Fire consumes vegetation roots that contributes to soil strength and to the diminished runoff and soil erosion potential and as vegetation re-establishes the soil is protected from raindrop impacts and the litter layer deposited reduces the surface runoff and the infiltration rates increases (Cerdá and Robichaud, 2009). The development of vegetation creates micro and macro pores that influence the soil infiltration capacity.

Other experience takes place before and after the removal of ashes downslope and the soil surface crust occurred. The $0.25m^2$ plots had ashes on the soil surface during 9 weeks and after rainfall simulations the infiltration rates were high but after four months due to the washed down of ashes infiltration values decreased to values of 17mm/h similar to infiltration rates on bare road embankments (Cerdá, 2009).

To accurate the rates of infiltration on the post fire period and the consequent runoff on burned soils Cerdá and Robichaud (2009), propose the use of rainfall simulations, as described, to easly reproduce the values of rainfall intensity, size and distribution of rainfall drops. The effects of soil burned severity on infiltration rates were measured under simulated rainfall and prescribed burning and were obtained values of 60-80mm/h on low burn severity, 30-84mm/h in moderate fire severity and 23-55mm/h in high severity fires after repeated measurements (Cerdá and Robichaud, 2009 cited Robichaud *et al.* (2000)).

The rainfall intensity, according to Cerdá (2009), is a driving factor in the post-fire runoff which means that is the non-infiltrated water and according to Prats *et al.* (2008), the rain intensity is a key factor in soil erosion.

f) Overland flow

To several different authors cited by Ferreira *et al.* (2008), in burned forest environments overland flow responses can be enhanced by reduced infiltration capacities, the development of a water repellent layer and destruction of vegetation. There is also a relation between overland flow and the patterns of soil moisture (Coelho *et al*. 2002) as described above.

Cerdá and Doerr (2005), studied the influence of vegetation recovery and soil hydrophobic patterns on soil hydrology and erodibility. They noticed considerable variation according to vegetation type with minimum values of erosion and overland flow 2 years after fire in plots re-colonized by herbs and shrubs, whereas under trees and dwarf shrubs plots, there was appreciable overland flow and soil loss until five years after fire. Plots dominated by dwarf shrubs exceed 100% of covered in year 6 but overland flow never reached zero as in the other two types of coverage. In the pine-dominated plots the relationship between vegetation recovery and post-fire overland flow and erosion is more complex than under other vegetation types.

Pine regrowth and hydrophobicity were associated and the repellent layer was responsible by overland flow and soil erosion compared to non-hydrophobic plots. Hydrophobicity (related with dry soil conditions) seems to influence more overland flow and soil erosion than the soil moisture content during winter. It is also important to consider the variability of hydrological and erosional response typical of Mediterranean ecosystems according to the climacteric conditions and during long wet and dry periods and where vegetation growth is associated with soil hydrophobicity because an increase in vegetation cover does not necessarily reduce overland flow (Cerdá and Doerr, 2005).

The overland flow response at plot level and catchment runoff of burned areas is higher than on mature forests and this is a consequence of the hydrophobic layer and vegetation destruction due to the destroyed obstacles. During the first year after fire the runoff at the catchment represented 50% of the total rainfall amount. The runoff in mature forests is 3,2% of the total rainfall amount and in the burned areas are tenfold those values (Ferreira *et al.,* 2008). According to this results the study of Coelho *et al.* (2002), using rainfall simulators, compared overland flow rates of non-burned, prescribed fire and wildfire areas. In non-burned areas there is no overland flow production and soil moisture content increased. In prescribed burned areas overland flow was 20% of the rainfall amount while in wildfires there are peaks of overland flow of 80% of the rainfall producing changes at the catchment runoff. Comparing overland flow rates of burned areas with rip-ploughed areas, in the first case there were higher rates of overland flow compared to rip ploughing due to the buffer zone around the catchment channel. At the plot level rip-ploughed areas presented higher runoff amounts.

3.3. Mitigation techniques to control soil degradation after forest fires

The application of emergency treatments is widespread in the United States since the 30´s. Since the 70´s the forest services in Spain have a permanent emergency programme to burned areas that evaluates the adequate treatments according to fire severity (Serrasoles *et al.*, 2004). In Portugal the research about the causes of fire began in 1989 by “Direcção-Geral das Florestas” and in 2004 was formed “Fundo Florestal Permanente” with the intent to support forestry and the public and communitarian brushwood (Silva, Deus & Saldanha, 2008).

To protect and conserve the soil immediately after forest fires some techniques could be done to mitigate soil erosion and runoff. Some of these techniques are herbaceous seeding, mulch application of forest residues or straw (to imitate the effect of organic and litter layer on soil protection) and discontinuous log or straw barriers on the hillslope (Serrasoles *et al.,* 2004).

The mulch technique effects are soil moisture conservation, evaporation reduction, increased infiltration, protection of superficial aggregates from the rain and effectiveness in reducing sediment production rates (Macdonald & Robichaud, 2007). Forest residue mulching is better than straw mulch because they incorporate organic materials richer in nutrients comparing to straw and are more resistant to hydrological and wind erosion. Herbaceous seeding is the most common emergency rehabilitation treatment used but in the last years in the united states the effectiveness of this treatment has been though due to the seeds incorporated in the seeding process that could interfere with the regeneration of natural vegetation. Other uncertainty is about the quick and significantly increase of vegetal cover in the control of erosion immediately after fire. It is possible to improve the seedling mixture with sub-shrub and shrub species contributing to a medium/long term ecossistem improvement (Serrasoles *et al.,*2004). Baeza (2004) point the problem about the introduction of commercial seeds leads to a less availability of water, light and nutrients to the native plants and the increase of *Ulex* sp. and *Cytisus* sp., also decreases the successful establishment of native species and the elimination of highly productive grazing areas. Another consequence that is related with erosive processes is sedimentation due to the introduction of pioneer conifers in order to introduce successional latter woddy species.

In stream channels small dams of straw, wood, foliage and rock could be applied as a complement of hillslope treatments immediately and in a medium term after fire (Robichaud *et al*., 2000).

Other strategies to mitigate soil and hydrological degradation after fire in a medium/long term, were described by Vallejo & Alloza (2004) and started in the middle XIX century, where pioneer conifers were introduced in degraded lands to promote sedimentation and a posterior colonization with natural or artificial leafy trees mainly eucalyptus. The wood of *Pinus* species had a high economic value and *Eucalyptus globulus* had a fast growth in order to productive expectations. Latter the measures of “Política Agrária Comum” encouraged the use of a vast number of species including shrubs to the afforestation of agriculture lands in accordance with the evolution in mount practices and the social ecological awareness. In this sense in the mid XIX century was introduced a larger concept of restoration, denominated hydrological forestry restoration that the main priority was the protection of the drainage basin instead of restore with exploitable trees. In the 80´s the new ecological concepts includes the importance of the landscape and give rise to the term ecological restoration and to a new organism denominated Society of Ecological Restoration. The objectives were wood production, pasture, fruit trees and berries; the conservation of hydrographic basins; the social and multifunctional use of the mount and the placement of rural populations. The introduction of native species is other objective but there is much uncertainty about the ecology and growth of woody and native species due to their recent introduction to understand the success in a medium/long term of their reproductive and dissemination capacities. The annual herbaceous vegetation and seeder species activated by fire (*Cistaceae* and *Ericaceae*) have a successful regrowth after fire as well as some species of woody shrublands. The problem of introduced exotic species competing with native species to the necessary management plans of cutting and herbicide application in invasive species completed with plantation of native species from a native origin insures a maximum adaptation to a restored habitat (Domingues, P., 2010). According to Vallejo *et al.* (2003), to the best selection of compatible species in a determined habitat is important to analyze the environmental ecological conditions, climacteric and edaphic characteristics and the dominant species nearby, however in transformed habitats the identification of original species could not be easy. The wide bibliography of Rivaz Martínez (1987), describes the potential vegetation in a territory in his phytosociological studies. The ecosystem is considered restored when there is enough biotic and abiotic resources and it is resilient against disturbance, interacting with the biotic and abiotic conditions of contiguous ecossistems. The abiotic resources were described and biotic conditions are ecological interactions in soil mycobiota and this live soil component is a set of roots, bacteria, fungi, water plant, nematodes and micro-arthropod. In particular the microorganisms of soil-plant system are mainly bacteria and fungus responsible by mineral nutrition, plant establishment and development in the process of revegetation in degraded environments (Barea & Honrubia, 2004). To achieve the similarity of the original ecosystem (in strictly sense) there is some management situations that could be expansive and impracticable due to some biotic and abiotic conditions (Alloza, Bautista & Vallejo, 2004). Some biological constraints are related with plague risk and trees diseases another is the introduction of woody species alone with a possible lack of symbionts, pollinators and disseminators in the surrounding habitat. In the last years became current the artificial inoculation with mycorrhizal fungi and nitrogen-fixing bacteria in woody species since symbiotic bacteria fixes N_2 in the rhizosphere of plants and mycorrhizal fungus colonizes the roots and develops an external mycelium connecting with soil microhabitats (Barea & Honrubia, 2004).

After the passage of fire there is some consequences on the hillslope as the deposition of ashes over the soil surface differences on the soil water content due to the occurred changes on vegetation but on the other hand influenced by the changes on the surface soil properties that will influence the basal and surface flow on the watershed. Depending on temperature, the intensity and duration of a fire it will influence the organic soil layer combusted over the surface (mainly composed of ashes and nutrients mobilized on degradational processes of fire-induced changes) and the formation of an hydrophobic layer (destroyed in a range of temperatures between 260º to 280ºC) that will influence soil structure, aggreggability and porosity, the infiltration rates and the amount of surface runoff on the watershed (Cerdá & Robichaud, 2009; Ferreira *et al*., 2008). The amount of infiltrated water is available for vegetation regrowth, to the soil biota and the water in the aquifer is surface runoff that can cause erosion (Cerdá & Robichaud, 2009). Since the 90´s there has been an interest to understand the spatial variability of water repellence (Doerr and Thomas, 2000; Varela *et al.,* 2005) and the relationship between the spatial distribution of water repellency and the patterns of erosion. Other less often cited changes on soils affected by fire are important to the role of soil erosion, hydrologic processes and aggregate stability as described and the overland flow response at plot level and catchment runoff of burned areas is higher than on mature forests and this is a consequence of the hydrophobic layer and vegetation destruction due to the destroyed obstacles.

Under *Cistus* matorral the stability of soil aggregates is high and under laboratory rainfall simulation conditions the aggregates show a slow breakdown and this may be due to the hydrophobic substances from *Cistus* vegetation comparing to a mature *Pinus* forest were the organic carbon content and soil aggregate stability were lower. At the present time, a mature matorral is in a delicate state of equilibrium (Coelho et al., 1995).

Forest management practices as rip-ploughing at catchment plot present a buffer zone around the catchment channel with high runoff amounts.

In the next chapter is described the importance of microorganisms of the soil biota particularly fungus and mycorrhizal symbiosis to the forest, particularly in the soil-plant system and the effects of fire on such components.

3.4. REFERENCES

Alloza J.A., Bautista S. &Vallejo V.R. (2004). La evaluación de resultados en las repoblaciones forestales. In V.R. Vallejo, J.A. Alloza (Eds), *Avances en el estudio de la gestión del monte mediterráneo* (pp.437-482). Valencia: CEAM.

Almendros G., Polo A., Ibáñez J.J. & Lobo M.C. (1984). Contribución al estudio de la influencia de los incendios forestales en las características de la materia orgánica del suelo. Transformaciones del humus por ignición en condiciones controladas de laboratorio. Révue d'Ecologie et Biologie du Sol, 8, 79-86.

Arcenegui V. Mataix-Solera J., Guerrero C., Zornoza R., Mayoral A.M. y Morales J. (2007). Factors controlling the water repellency induced by fire in calcareous Mediterranean forest soils. European Journal of Soil Science, 58: 1254-1259.

Barea, J.M. & Honrubia, M. La micorrización dirigida de la planta forestal. In V.R. Vallejo, J.A. Alloza (Eds), *Avances en el estudio de la gestión del monte mediterráneo* (pp.215-260). Valencia: CEAM.

Bot A.J., Nachtergaele F.O. & Young A. (2000). Land Resource Potential and Constraints at Regional and Country Levels. Available on ftp://ftp.fao.org/agl/agll/docs/wsr.pdf

Cerdá, A. & Robichaud, P.R. (2009). Fire effects on soil infiltration. In: Cerdá A. & Robichaud P. R. (Ed.), *Fire effects on soils and restoration strategies* (Oxford Brookes University, UK. (pp.81-103).

Cerdá A. & Doerr S.H. (2005). Influence of vegetation on soil hydrology and erodibility following fire: an 11-year investigation. International Journal of Wildland Fire, 14: 423-437.

Claridge A.W., Trappe J.M., Hansen K. (2009). Do fungi have a role as soil stabilizers and remediators after forest fire? *Forest Ecology and Management*, 257 (3): 1063-1069.

Coelho C.O.A., Shakesby R.A, González del Tánago M., Ternan J.L., Walsh R.P.D., Williams A.G. (1995). Land Management and Erosion Limitation in the Iberian Peninsula, IBERLIM. Project EV5V-004.

Coelho C.O.A., Ferreira A.J.D., Baake M. & Keizer J.J. (2002). Impacts of prescribed shrubland fire and forest wildfire on overland flow and soil erosion generating processes. In: Rubio J. L., Morgan R. P. C., Andreu V. & Asins, S (Eds.), Proceedings of the third International Congress of the European Society for Soil Conservation (ESSC), Valencia, Spain, 28 March -1 April, 2000, *Man and Soil at the Third Millennium*. Geoforma Ediciones, Logroño (pp.485-1496).

DeBano L.F., Savage S.M. & Hamilton D.A. (1976). The transfer of heat and hydrophobic substances during burning. Soil Science Society of America Proceedings, 40: 779-782.

Doerr S.H. (1998). On standardising the “water drop penetration time” and the “molarity of an ethanol droplet” techniques to classify soil water repellency: a case study using medium textured soils. Earth Surface Processes and Landformos, 23: 663-668

DeBano L.F. (2000). The role of fire and soil heating on water repellency in wildland environments: a review. Journal of Hydrology, (231-232): 195-206.

Doerr S.H., Shakesby R.A. & Walsh R.P.D. (2000). Soil water repellency: its causes characteristics and hydro-geomorphological consequences. Earth-Science Reviews, 51: 33-65.

Domingues, P. (2010). Personal communication.

Doerr S.H., Blake W.H., Shakesby R.A., Stagnitti F., Vuurens S.H., Humphreys G.S., Wallbrink P. (2004). Heating effects on water repellency in Australian eucalypt forest soils and their value in estimating wildfire soil temperatures. International Journal of Wildland Fire, 13:157-163.

Ferreira A.J.D., Silva J.S., Coelho C.O.A. Boulet A.K. Keizer J.J., Carreiras M.A.M. (2009). The portuguese experience in managing fire effects In: Cerdá A. & Robichaud P.R. (Eds.), *Fire effects on soils and restoration strategies*, Oxford Brookes University, UK (pp. 401-422).

Ferreira A.J.D., Coelho C.O.A.,. Ritsema C.J, Boulet A.K., Keizer J.J. (2008). Soil and water degradation processes in burned areas: Lessons learned from a nested approach, *Catena*, 74 (3): 273-285.

MacDonald L.H. & Robichaud P.R. (2007).Post-fire Erosion and the Effectiveness of Emergency Rehabilitation Treatments over Time. Department of Forest, Rangeland and Watershed Stewardship (JFSP Final Report). Colorado.

Mataix-Solera J. & Guerrero C. (2007). Efectos de los incendios forestales en las propiedades edáficas. In: Mataix-Solera (Ed.). Incendios forestales, suelos y erosión hídrica. Caja Mediterráneo CEMACAM. Font Roja-Alcoi, Alicante.

Moddy J.A., Martin D.A. (2009). Forest fire effect on geomorphic processes. In Cerdá A. & Robichaud P.R. (Eds.), *Fire effects on soils and restoration strategies*. Oxford Brookes University, UK (pp.41-79).

Robichaud P.R., Beyers J.L., Neary D.G. (2000). Evaluating the effectiveness of postfire rehabilitation treatments. Department of Agriculture, Forest Service, (General Technical Report), GTR: 63, 85pp. Fort Collins: Rocky. Mountain Research Station.

Robichaud (2009). Post-Fire Stabilization and Rehabilitation. Section II: Rehabilitation and Restoration Strategies. In: Cerdá & Robichaud P.R. (Ed.), *Fire effects on soils and restoration strategies*, Oxford Brookes University, UK. (pp. 299 - 319).

Serrasolses I., Llovet J. & Bautista S. (2004). Degradación y restauración de suelos forestales mediterrâneos In: Vallejo V.R., Alloza J.A. (Eds), *Avances en el estudio de la gestión del monte mediterráneo,* Valencia: CEAM. (pp.93 -131).

Shakesby R.A., Doerr S.H. & Walsh R.P.D. (2000). The erosional impact of soil hydrophobicity: current problems and future research directions. Journal of Hydrology, 231-232:178-191.

Shakesby R.A. & Doerr S.H. (2006). Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviews*, 74: 269-307.

Silva J.S., Rego F.C., Mazzoleni S. (2006). Soil water dynamics after fire in a Portuguese shrubland. International Journal of Wildland Fire (15): 99-111.

Silva J.S., Deus, E. & Saldanha, L. (2008). Incêndios Florestais, 5 anos após 2003. Liga para a Protecção da Natureza, Autoridade Florestal Nacional.

Society of Ecological Restoration available on: www.ser.org

Vallejo, R., Cortina, J., Vilagrosa, A., Seva, J.P. y Alloza, J.A. (2003). Problemas y perspectivas de la utilización de leñosas autoctonas en la restauración forestal. In: Rey, J.M., Espigares, T. & Nicolau, J.M. (Eds), *Restauración de ecosistemas mediterrâneos*. Servicio de Publicaciones de la Universidad de Alcalá. Alcalá. (pp.11 – 42).

Vallejo V.R. & Alloza J.A. (2004). La selección de especie en la restauración forestal. In V.R. Vallejo, J.A. Alloza (Eds), *Avances en el estudio de la gestión del monte mediterráneo*, Valencia: CEAM (pp.195-214).

Varela M.E., Benito E. de Blas (2005). Impact of wildfires on surface water repellency in soils of northwest Spain. *Hydrological Processes*, 19: 3649-3657.

Varela M.E, Benito E., & Keizer J.J. (2010). Wildfire effects on soil erodibility of woodlands in NW Spain. Land Degradation and Development 21: 75-82 DOI: 10.1002/ldr.896

CHAPTER 4

Effects of fire on fungus and mycorrhiza

CHAPTER 4 EFFECTS OF FIRE ON FUNGUS AND MYCORRHIZA

The microorganisms developing in the soil surrounding plant seeds and rhizosphere (the soil profile that interact directly with plant roots) are beneficial to the soil-plant system. These organisms stimulate seed germination due to the production of hormones and vitamins, improve rooting due to the production of phyto-active substances, increase nutrient availability to plants and contribute to soil aggregate stability, increase the plant resistance against pathogen organisms (antagonistic bacteria), eliminate xenobiotic compounds (biodegradation) and protect the plant against abiotic stress (such as salinity and dryness), therefore contributing to the ecophysiological resistance of plants (Barea & Honrubia, 2004). In chapter 4, a review of the effects of fire on soil fungi, their ecological role on soil-plant system in forests and their importance to forest sustainability are presented.

4.1. The Fungus and Mycorrhiza

The fungi are a group of heterotrophic organisms that belongs to Domain *Eukarya* and Kingdom *Fungi*. Currently, seven phyla are proposed: *Microsporidia*, *Chytridiomycota*, *Blastocladiomycota*, *Neocallimastigomycota*, *Glomeromycota*, *Ascomycota*, and *Basidiomycota* (Hibbett *et al.*, 2007). The fungi group has similar characteristics with plants and with animals, the storage of energy is glycogen and the cellular wall is mainly formed by chitin, a carbohydrate (polymer: N-acetyl-D-glucosamine) also present in the exosqueletum of artrophoda animals. Reproduction in fungi takes place by bud formation, fragmentation and sporulation. The spores have thick walls conferring resistance and accumulation of reserve substances benefiting the germination, development and reproduction ensuring subsistence (Carlile & Watkinson, 1994).The fungi occur in all environments and are essential to the mineralization of complex organic molecules and to recycle nutrients.

The fungi (*myco*) forming symbiotic relation with plant root (*riza*) are designated mycorrhiza and is an example of mutualism. These fungi normally inhabit the soil, the roots and other subterranean organ and more than 70% of vascular plants are mycorrhizal including the angiosperm, gymnosperm, pteridophytes and some bryophytes (Azul & Freitas, 2005). The mycorrhizal association is beneficial to plant mineral nutrition and to the maintenance of soil productivity. The mycorrhizas and plants have mutualistic associations in which the fungus and plant exchange nutrients required for their growth and survival but myco-heterotrophic plants have "explorative" mycorrhizas where transfer process will only benefit plants (Brundrett, 2004). The mycorrhizal symbioses improve plant nutrient uptake (e.g. P, Nh4+, and organic nutrients) creating "mycorrhizosphere" communities by the hyphae. In the ecological association the fungus get photosynthetic sugar from the plants and in turn the fungus mediated the uptake of mineral nutrients. The photosynthetic sugars from the plant depend on the photosynthetic rates, translocation of sugars from the leaves and mobilization of sugars stored in different organs (Hibbett *et al*., 2000). There are different types of mycorrhiza, including arbuscular mycorrhiza (AM), dominant in grassland, tropical and temperate forests; ectomycorrhiza (ECM), dominant in boreal, temperate and tropical forests, and ericoid mycorrhizal (EM), in temperate and boreal forests and heathlands (Dickie, 2006). The different types and nutritional strategies is a result of a prolonged and selective process on the group of fungus forming mycorrhiza (Honrubia, 2009). The co-evolution between mycorrhizal fungus and plant roots began in the Palaeozoic with the first terrestrial plants, the antecedents of bryophytes and ferns which had structures similar to the actual AM and also Wang & Qiu (2006) reported in their study that AM were present with the most ancestral plants the hepatics in pteridophyts and in the primitive gymnosperms that use the glomeromycota AM as a nutritional strategy to colonize the changing environment from Jurassic and Triassic. Following this idea, Hibbet (2000), suggested that mycorrhizal symbioses are unstable and dynamic associations that evolved from long-term interaction between plant and fungus and after a phylogenetic analyses this author concluded that the evolutionary stability of mycorrhiza comes from saprotrophic precursors suggesting that mutualism are the stable derivates between plants and parasitic fungus.

Table 4.1-A– Mycorrhizal trophic group, common plants forming mycorrhiza, responsible fungus and sources.

Trophic Micorrhiza	Plant	Phylum	Fungus	Author
Ectomycorrhiza	Pinaceae (Pinus sp., Abies sp., Tsuga sp.) Fagaceae (Quercus sp., Fagus sp.)Betula sp., Ulmus sp., Salix sp.	Basidiomycota (1), Ascomycota (2)	(1) - Laccaria sp., Boletos sp., Pisolithus sp., Hebeloma sp., Russula sp., Amanita sp., Thelephora terrestris, Russula sp. (2) - Tuber sp., Wilcoxina mikolae, Cenococcum sp.	Smith & Read, 1997; Mason, 1983; Molina et al., 1992; Brundrett, 2004, Barea & Honrubia, 2004; Azul & Freitas, 2005; www.deemy.de
Endomycorrhiza (arbuscular mycorrhiza formed by zygomicete)	Salix sp., Populus sp. , Acer sp., Fraxinus sp., Ulmus sp., Thymus sp., Rosmarinus sp., Lavandula sp., Salvia sp., Rosa sp., Genista sp., Juniperus sp.	Glomeromycota	-	Smith & Read, 1997; Brundrett, 2004; Barea & Honrubia, 2004; www.deemy.de

The concept of mutualism with rhizoids, gametophic tissues and with the incipient roots of pteridophyt, began with the terrestrial colonization by the ancestral *Glomeromycota* fungus. Throughout time, in biomes of extreme climacteric conditions with variations of temperature and precipitation (water supply in the soil), both gymnosperms and angiosperms (on the trees) developed radical structures that enabled them to adapt to the ecosystems. The radical system incorporates a new root type, a secondary structure with a limited growth that is colonized by fungus that became specialized in the exchange of nutrients with the host plant (Brundrett, 2002). These are the ecto-mycorrhizas (ECM) represented by *Ascomycota* and *Basydiomicota* fungi appearing latter than AM and developed in a parallel evolution independently of AM with their origin in saprophytic forms but also have been multiple reversals to a free-living condition (Hibbet *et al.*, 2000). The six most ecologically important groups of ectomycorrhizal plants are *Fagaceae*, *Salicaceae*, *Betulaceae*, *Dipterocarpaceae*, *Pinaceae* and *Myrtaceae* (Hibbet *et al.*, 2000). Other plant families with flower as *Rosaceae*, *Mirtaceae*, *Fabaceae*, *Euphorbiaceae* and *Poligonaceae* that normally forms AM symbiosis, have ectotrophic mycorrhiza (Brundrett, 2002). Besides the *Ascomycota* and *Basydiomycota* representing a potential form of symbiosis the concept of mutualism was established between the ancestral of the actual glomeromycota fungi and the first terrestrial plants (Honrubia, 2009). Brundrett (2008), reported that AM provided the primary conditions for angiosperm roots to spread in the Tertiary and Quaternary, contributing with more evidence about the anciently, abundance and ecological importance of AM.

There are different types of endo-mycorrhiza according to the association with the myco-symbiont establishment. a) endo-mycorrhiza formed by basydiomicete, b) endo-mycorrhiza formed by ascomycete and c) endo-mycorrhiza formed by zygomicete (arbuscular mycorrhiza). Basydiomicete mycorrhiza has been described in *Ericaceae* (*Arbutus* sp., *Erica* sp.) with the responsible fungus, *Pisolithus*, *Lactarius*, *Laccaria*, *Thelephora*, forming a mantle, a hartig net and intracellular hyphae (Smith & Read, 1997). The ascomycete fungus form septate hyphae and in some cases forms a mycelium similar to the ectomycorrhiza mycelium that penetrates in the interior of the cells from the radical cortex by haustorium structures (Smith & Read, 1997).

Vesicular-arbucuslar mycorrhiza (VAM) or endomycorrhiza also called glomeromycotan mycorrhiza (Brundrett, 2004), are completely dependent of plants to survive since plants are the only source of carbon for the obligatory biotrophic VAM. Arbuscular mycorrhizae are a type of endo-mycorrhizal association, characterized by the formation of intracellular structures such as arbuscules formed as a result of joint plant–fungal by specialized hiphae formed in the germinating spore. The intraradical structures, such as arbuscules, vesicles (lipid storage structures), coils, and the hyphae growing within the root cortical tissue, are connected to an extrarradical mycelium (Smith & Read 1997). This soil mycelium has a variety of functions which includes the formation of spores (propagules of dispersion in time and space), formation of runner hyphae (exploration of soil and new roots to be colonized), and nutrient uptake (Rillig, 2004). VAM are present in more than 80% on the roots of a large diversity of host trees, which includes the angiosperm, gymnosperm and pteridophyts as *Salix* and *Populus* genera (Smith & Read, 1997), *Acer*, *Fraxinus* and *Ulmus* (Brundrett, 2004), *Thymus*, *Lavandula*, *Salvia*, *Genista*, *Rosa*, *Juniperus* (Barea & Honrubia, 2004).The woody *Eucalyptus globulus* have VA mycorrhiza and ECM and is one of the few woody plants with both type of symbionts present in the same root apex, but VA mycorrhizas seems to be more prevalent on young plantations (Boudarga *et al.*,1990).

The ECM fungus forms another trophic strategy, important in many habitats but restricted to certain plant families (Brundrett, 2004) with often symbiotic association in conifers and perennial foliage trees of temperate and boreal climate. The basic pattern of an ECM is a tissue and a network with some development of an extra radical mycelium not inside the cells. The hartig net develops on the long root of an established tree and then the laterals roots will be colonized, as in the case of *Pinus* sp., or from the inner mantle of a parent root as in *Eucalyptus* sp. In general the hyphal system penetrates the root cortex, forming the hartig net, where changes between the two symbionts occur. Under favourable conditions fungus reproduction occur forming mushrooms near the mycorrhized tree, which release spores contributing to dispersal and survival and also to the formation of a new mycorrhiza. When the parent root is uncolonized or a seed germinates to produce a new root system, the potential for root colonization will depend on the presence of fungal propagules in the soil and the existence of compatibility between fungal propagules, inoculum potential and the root system (Smith & Read, 1997). The ECM have some strategies to potentiate the symbioses in particular the incorporation of phosphorus and nitrogen from organic sources or the establishment of a mycelium network to mediate nutrient transference between different species. The mycorrhizal fungus protects the trees from pathogenic attack and help to obtain water and nutrients (Smith & Read, 1997).

Several studies were carried with VAM and ECM fungi and trees, by different authors: (Mason 1983; Molina *et al.*, 1992; Barea *et al.*, 1996; Smith & Read, 1997; Azul, 2002; Brundrett, 2002; Brundrett, 2004; Machado, 2004, Barea & Honrubia, 2004; Azul & Freitas, 2005; Román *et al.* 2005. This information could be used as support for construction of a database for fungus forming symbioses.

In non-native forests of Eucalyptus, Machado *et al.* (1995), found that the fungi forming mycorrhiza were *Pisolithus*, *Laccaria* and *Scleroderma* in young plantations and disturbed habitats and experimentally tested the inoculation of these fungi on the roots of *Eucalyptus globulus* noting that *Pisolithus tinctorius* and *Cenococcum geophilum* revealed a positive growth and that these ECM associations were very effective in the increase and survive of *E. globulus* and also in areas with dry summers. *C.geophillum* forms *sclerotium,* a mycelium with reserve substances which lies dormant in the soil for several years, has a very low host specificity, a wide geographical distribution, is very competitive, and is considered to have the broadest host range of all ECM fungi (Molina *et al.*, 1992) a good resistance to drought and adaptation to the Mediterranean ecosystems (Román *et al.*, 2005).

In montados/ shrubland areas Azul and Freitas (2005), reported symbioses in the rhizosphere between *Quercus* and *Cistus salvifolius* and the fungi *Russula, Tomentella*, *Lactarius* and *Cortinarius* whereas Capela (2009), described ECM fungal association for similar vegetation stands with *Scleroderma* sp. and *Xerocomus* sp. In three different stands of Quercus ilex on burned and non-burned plots Román *et al*. (2005), reported the dominance of *Sphaerosporella brunnea* (pioneer and severe contaminant fungus, Molina et al.1992) and *Pisolithus tinctorius* on the burned plots. McAfee & Fortin (1988), reported that this fungus have a slow growth when occurs with other ECM fungi, for example *Laccaria bicolor* and *Hebeloma cylindrosporum*. Other ECM fungi that are abundant in burned stands reported by Román *et al.* (2005) get water and nutrients on the soil from rhizomorphs with long emanating elements.

4.2. The importance of fungi for soil-plant systems in the forests

A manuscript published by Rillig (2004) described the importance of AM in the colonization of terrestrial environments and highlight the role of VAM in the carbon cycle. This author also described the presence of VAM in soil mycelium and its products that can be relevant at the ecosystem level. The VAM can alter primary production, due to their influence on plant respiration rates to obtain net primary production or due to the production of plant biomass. The AMF also influences plant community composition and individual plants and this will be of greatest importance to determine the net primary production (NPP).

Mycorrhizal symbioses are essential to stability, conservation and productivity of all terrestrial ecosystems. The benefits of mineral nutrition are reflected in plant vitality and in the maintenance of soil equilibrium and also prevent the colonization of parasitic opportunistic fungus, decreasing soil susceptibility to erosion and desertification (Azul & Freitas, 2005). According to Smith & Read, 1997, ECM, confers protection against root diseases, and resistance to hydric *stress*, resistance to extreme temperatures and to soil aggregate stability. Mycorrhizal fungus has also an important role in nutrient recycling and as potential bio indicators. The symbioses between ECM and the *Pinaceaeae* are efficient in nutrient uptake, protection against root pathogens and surrounding soil structure (Smith & Read, 1997; Smith *et al.*, 2005). The high diversity of ECM in the soil enhances the growth of *Fagus sp*., seedlings that regenerate after tree cutting or thinning (Goicoechea *et al.*, 2009). The presence of AMF in the soil (roots and /or soil mycelium) can influence ecosystem process and the soil biota. Future research should exam the processes beyond primary production providing the inclusion of AMF when studied the ecosystem process (Rillig, 2004). Soil mycelium has a variety of functions, including spore formation (propagules for dispersal in time and space) and the formation of runner hyphae (exploration of soil and new roots to be colonized) and nutrient uptake.

The establishment of plant-fungus mycorrhizal interactions are important to the forest ecosystem, especially after fire disturbance, timber or harvesting. Several fungi species are very important in the process of ecological succession especially when the root plant was damaged. The occurrence of pioneer fungus that occupy inhabited areas in conjunct with latter stage fungus are keys in the recovery process, ensuring a good tolerance to disturbance enabling the establishment of vegetation (Azul & Freitas, 2005). Mediterranean forests are particularly threatened by fire and further studies on the potential effects of fire on ECM communities in these ecosystems are urgently needed. Also Claridge and collaborators (2009), referred the lack of scientific information on the answer of fungi after disturbance and the importance to select ECM fungus present in natural conditions after fire that could be *inoculum* sources to guaranty the success in reforestation programs.

4.3. The function of fungi (and mycorrhiza) in forest structure (after fire)

The study of diversity and function of soil mycobiota in natural habitats is essential to understand the role of fungi in the process of nutrient cycling and in particular on their role in the maintenance of forest health and recovery after disturbance. In this topic we aim to compile information of different studies about:

a) The fungi (non mycorrhizal) and mycorrhizal symbiosis in forest ecosystem before and after natural and prescribed fire;

b) Changes in the structure, richness and diversity of fungi (particularly in mycorrhizal symbiosis) and evaluate if the study of fungi communities may indicate disturbance on forest soil.

c) Physiological and metabolic cellular processes in soil as indicators of forest disturbance;

d) Tools for fungi research after soil exposure to fire and techniques to induce mycorrhization.

4.3.1. The fungi (non mycorrhizal) and mycorrhizal symbiosis in forest ecosystem before and after natural and prescribed fire

According to Claridge *et al.* (2009) the term post fire fungus, refers to species that fruits after fire or an eruptive effect (heat and incineration) and are pivotal species to the system recovery which is different from survivals fungi to the passage of a fire that were active in the pre-fire habitat conditions. Some of the post-fire fungi are terrestrial and fruit in ashes or volcanic dust others are mycorrhizal or pathogenic in the tree roots. The majority of post fire fungi are A*scomycota* fungi but some are G*lomeromycota* and B*asydiomicota*. Goicoechea *et al.* (2009), define pioneer fungi as the existing in roots of seedlings regenerating after disturbance that are earlier colonisers in a disturbed habitat. In the study of Ingleby *et al.* (1998), logging treatment with minimal soil disturbance was done and many of the fungi that dominate in the growing seedlings are thought to be pioneer fungi. However, the functional role of fungi in forest ecosystem recovery after fire is poorly documented (Claridge *et al.*, 2009).The ECM fungi that normally prevail after disturbance are less specific species that quickly germinates from spores after disturbance and/or after root stimulation. In a forest ecosystem with different aged trees the persistence of pioneer fungi (non-selective) and more specific species of fungi (selective) has a great capacity of responsiveness mainly after disturbing situations (Azul & Freitas, 2005). After thinning and prescribed fire in ponderosa pine forests the ECM survives and rapidly re-establish after disturbance but ECM species richness were significantly reduced comparing to non-burned treatments. The ECM species present in the pine forest before and after treatment were *Cenococcum sp.*, *Piloderma sp*., *Rhizopogon salebrosus*, and *Wilcoxina rehhmii* (Smith *et al.*, 2005). Claridge *et al.* (2009) studied the variations in the top 5cm of soil in a burned *pinus* forest and the diversity of fungi and bacteria in this length. After a high severe fire the diversity of fungus increased in the top soil reinforcing the idea that fungus can be activated after fire and important in forest recovery. Claridge *et al.* (2009) reported an example about the function of mycorrhizal fungi in the recovery of forest structure after wildfire disturbance. The authors showed the occurrence of the genus *Anthracobia* after wildfire at two separated sites, one in Pacific Northwest United States and the other in southeastern Australia. This fungus belongs to the group of fungi known as "*post fire fungi*", their growth in burned areas is associated with the appearance of an extensive mycelial mass and to the high quantity of fruited bodies. Fungi belonging to the genus *Anthracobia* are known for their importance in the aggregation of soil matrix (aggregability), on the recovery of soil stability and helping to minimize the loss of soil in the absence of plant roots. Futhermore, the mycelial mass production of this fungi group contributes to the immobilization of soil granules and for nutrients supply during the process of plant re-colonization. Therefore, the post fire fungi hyphae initiates the capture and concentration of nutrients for further ecological supplies in the process of succession where the initial soil colonizers will be replaced by other fungi and by photosynthetic pioneer species (e.g. algae and bryophytes). In general, the fungal activity in the top soil strongly mitigates the post fire erodibility and also increases the resistance of introduced seedlings due to a lack of water (Claridge *et al.* 2009; Goicoechea *et al.* 2009; Azul & Freitas 2005; Machado 2004). The mycelial mats of fungus are also very important due to the functional role on-site, enabling the aggregation of soil particles in highly erodible landscapes. The mycelium network aggregates soil particles, promoting aeration and water infiltration. However, negative factors may occur once network mycelium is near the surface increasing water repellence in the soil by the production of hydrophobic substances and due to a dense network hiphae causing surface tension and preventing infiltration. The fungi inducing water repellence can promote runoff and increase erosion.

4.3.2. Fungus and mycorrhiza as indicators of ecosystem disturbance

Some disturbance as fire, thinning and/or root turnover influences the presence or absence of certain mycorrhizal types, the percentage of mycorrhizal plants, the emergence of pioneer fungi and the post fire fungus may be used as indicators of disturbance. The AMF that positively contribute to soil aggregate stability and against erosion may become limited through AMF species losses (Rillig, 2004).

The studies carried out in *Fagus sylvatica* forest in Navarre region north-western Spain showed that *Thelephora* spp. were absent from disturbed stands (Jones *et al.*, 2003, Goicoechea *et al.*, 2009). The authors concluded that this could be due to changes in forest soils by the mechanical removal of mature trees. *Thelephora* spp. requires associations with mature trees in order to survive. On the other hand, *Cenococcum geophilum* and some not identified ECM fungus, have high frequency and/or abundance in cutted stands while Hebeloma-Cortinarius ECM showed the greatest abundance and relative frequency in the disturbed area. These ECM have medium-distance exploring rhizomorphs. Smith and collaborators (2005), studied the response of ECM fungi after low intensity prescribed fire and mechanical thinning of small diameter trees, in dry pine dominated forests, of wood and cut after fire in 3 *pinaceae* species. In their study the combination of fire and thinning reduced by about 60% the number of occurring species more than burning treatment alone comparing the pre and post-fire conditions. Additional data evidences that the control and thinned stands had more live root biomass than burned stands. According to these results, Claridge *et al.* (2009) reported that high intensity burning is detrimental to the majority of fungus that contributes to soil aggregation but early post fire fungus whose importance to the reestablishment of vegetation after fire is high could indicate fire disturbance. The occurrence of the gender *Anthracobia* is associated with highly impacted microsites, after fire occurrence and after volcanic eruption contributing to minimize the movement of soil in the absence of plant roots. The same author reported that *Geopyxis carbonaria* occured after low intense fires.

Other variable indicating disturbance is the variation on the percentage of mycorrhizal roots by the ECM fungus symbionts that is also affected by seasonal variation (Goicoechea et al., 2009; Román et al., 2005) because the first author reported that in unmanaged forests no variation occurred in the percentage of mycorrhizal roots. The disturbance of thinning and seasonality influences the percentage of mycorrhizal roots. It decreases from spring (with the highest values) to summer and the lowest rates were reached in winter. In *Quercus alba* and *Quercus rubra* forests the maximum percentage of mycorrhization occurred in autumn (Román et al., 2005; Azul & Freitas, 2005).

After disturbance of cutting in *Fagus sylvatica* forests reported by Román et al., (2005) and Goicoechea et al. (2009), it was reported the occurrence of *Sphaerosporella brunnea*, a pioneer fungi and a contaminant that forms ectendomycorrhiza in the young seedlings of *Fagus* after cutting. Cenococcum geophillum is described in a later stage in Fagus sylvatica forests. This fungus was also described in mature forests of *Fagus* sp. and *Abies* sp., in Germany and after the occurrence of fire in *Quercus ilex* forests, due to a good adaptation to drought in the Mediterranean conditions.

In the revision article of Barea & Honrubia (2004), it is concluded that after forest fires there is a decrease of all fungal parameters (diversity and number of seedlings) in ecto and endomycorrhiza, and the effects on mycorrhizal population depends on fire severity.

4.3.3. Physiological and metabolic cellular processes as indicators of disturbance

Soil respiration is an important fraction of gross primary production (Edwards & Riggs, 2003). The organisms responsible for the primary productivity are known as primary producers or autotrophs and are the base of food chain (in terrestrial ecoregions they are mainly plants). Primary production is important to carbon reservoir in ecosystems. Some variables influence the rates of CO2 efflux from the soil, as the changes in soil temperature, the seasonal variation on root growth, the litter inputs and moisture content.

Edward & Riggs (2003) design an open-flow system non-invasive to soil microclimate and measured CO2 rates in forest soils. Their work showed that it is possible to compare the effects of disturbance in burned and non-burned, disturbed and non-disturbed areas through the parameter soil respiration.

In the past, Wood & Goodenough (1977) tried to analyse the activity of extracellular enzymes acting on a substratum. Testing different mycelium colonized compost they concluded that laccase concentration increased during mycelia growth and declined at the start of fruiting, while cellulose activity was detected during growth but increased at fruiting stage. No variability was detected with other extracellular enzymes (xylanase, laminarinase and alkaline protease) during the fruit body development. The activity of laccase in soils could be used as metabolic indicator of soil disturbance, since this enzyme is present in the majority of basidyomicota and in several ascomycota fungi. Laccase is a poliphenoloxidase enzyme that catalyzes the oxidation of phenolic substratum in the corresponding quinones and the molecular O_2 is also important to regenerate the activity form of the enzyme. This enzyme is also produced by plants and bacteria (Gil *et al.*, 2009). Baptista *et al.* (1999), also studied the production of phenolic compounds during the ectomycorrhizal infection of *Pisolithus tinctorius* in *Eucalyptus urophylla*. This study suggest a simple experiment to understand the influence of temperature in laccase activity to the system soil biota-plant after fire, throughout the assay of phenolic component products as a response of laccase activity.

4.3.4. Tools for fungus and mycorrhizal research after fire and techniques to plant mycorrhization

Several studies have proposed tools to monitor the effects of fire on soil microbiota -plant system and particularly on mycorrhizal symbiosis. Johnson and co-workers (2001) described a method called in-growth core system, which enable the study of natural communities of arbuscular mycorrhizal mycelia in the soil. The experiment consists on rotating a core containing the experimental soil while a non-rotated set of control cores permits hyphal re-colonization. The impacts of hyphal severance are achieved by periodic rotation of some of the cores, upon arbuscular mycorrhizal colonization of plants. Using cores of burned soil and non-burned soil from an adjacent area with similar characteristics of topography, vegetation community and slope it is possible to infer about the effects of fire on VAM - plant system and their capability of regeneration after fire. According to Johnson and co-workers (2001), the in-growth core system approaches the functioning of mycorrhizal mycelial networks under conditions close to those occurring in nature.

Other techniques can greatly improve the study of post fire fungus by quantifying the mycelim fungus content in the soil. Basically this is achieved by biochemical analysis of fungus-specific compounds, such as mannitol and arabitol (sugar alcohols) in mycorrhizal roots (Claridge *et al.*, 2009; Wingler *et al.*, 1993). According to Rillig (2004), the impacts of fire in soil microbial communities and direct mycelium effects, will be important in the processing of soil organic matter and in soil aggregability. The biochemical analysis of fungus-specific compounds is a methodology that allows investigating the response of fungus per se in the post fire conditions but doing not estimate fungus biomass in the whole ecosystem. This occurs due to the fact that fungi may inhabit two functionally distinct locations, the root and the soil.

There are also others techniques to describe the ECM post-fire fungus which are based on the presence of fruit bodies, although with limitations related to climacteric conditions (temperature, moisture, photoperiod), edaphic factors, physiology, predation and due to the fact that many ECM fungus do not have epigeous fruiting bodies or could not have fruited during the field survey.

Therefore, the frequency and relative abundance of fruit bodies only indicate the productivity of species according to specific conditions as the emergence of fruiting bodies and the pattern of distribution of the vegetative mycelium in the soil (Claridge *et al.*, 2009; Anderson & Cairney, 2007).This method is considered a qualitative approach to monitor the occurrence of ECM fungus after fire.

During the last years the evaluation of morphological characteristics of the developing resistance spores was the only technique available to characterize and follow fungus forming vesicular arbuscular mycorrhiza. However, recently developed molecular techniques enable us to study fungi communities inhabiting bulk soil, rhizosphere and plant simbiontes (mycorrhiza). The DNA has become the most used signature molecule for microbial community analyses. DNA-based fingerprinting methods (Muyzer et al., 1993; Gomes et al., 2003) have substantially increased our knowledge on environmental microbial diversity and the newly developed bar-coded pyrosequencing methods (Rousk et al., 2010) will virtually dissect microbial communities and will substantially increase the understanding of the role of fungi in the soil-plant system.

The genomic DNA can be also used for genotypic characterization of specific fungal population. Anderson & Cairney (2007), described a method consisting in the analysis of genomic DNA of the fungi fruiting body to map the mycorrhizal mycelial networks in a forest stand. Fruiting bodies with the same vegetative genotype are developed from a genetically identical mycelium growing in the soil. The information based on the genotype distribution of fruiting bodies allow to infer about mycelium distribution on the below-ground soil but no information on mycelium continuity, density or vertical distribution in the soil profile is available.

To test if the mycorrhizal infection was effective Malajczuk and collaborators (1990) suggested testing the symbiosis between eucalyptus seedlings and *Pisolithus tinctorius*, inoculating the seedlings with these fungi using the cellophane-over-agar method to reveal the formation of ECM fungus. To follow the changes on protein biosynthesis the method of polyacrylamide gel electrophoresis permits the comparison between the protein profiles of mycorrhizal roots with non-colonized roots and with the free living mycelium (Smith & Read, 2002). In the mycorrhizal roots there was a large decrease of polypeptides comparing to free living partners and the major changes of polypeptide synthesis occurred within hours of colonization of the seedlings of primary roots by the fungus. This methodology was described to non-burned conditions and to specific host specie but this protocol can be easily adapted in order to follow mycorrhizal roots after fire.

It is also important the production of fungal inoculums and the adequate selection of fungi to establish symbioses with plants that will improve the soil-plant establishment. It is known that VAM symbioses are formed by approximately 80% of vascular plants in all major terrestrial biomes in natural conditions (Smith *et al.*, 2010) however human activity with the consequently land uses changes or natural causes as fire, herbivorism, climacteric conditions could decrease or extinguish mycorrhizal potential and infection in plant roots. To overcome this problem the transplanting approaches based on utilization of plant seedlings, previously inoculated with beneficial mycorrhizal fungi, will promote plant growth and the re-inoculation of these fungi into the ecosystem. The most common methods of plant inoculation are based on the utilisation of a solid substrate to grow the host plant (soil or vermiculite/ sepiolita/ perlita) under different concentrations and different host plants. Azcón-Aguilar and collaborators (2000) published a protocol using a substrate with no soil, with only fertilizer and the host plants, which allows a good production of arbuscular mycorrhiza inoculum. The inoculums are a mixture of fungus spores, fragments of the colonized roots, hyphae and the substrate with a high density of fungus. However, before selection of the best fungus or fungi consortium from natural populations it is necessary to characterize the existent fungi communities, isolate and evaluate the cultures and then re-inoculate in the ecosystem following the process. The best fungus to the plant-soil system under the dominant soil-climatic conditions is designed ecotype.

According to Brundrett (2004), the adequate selection of mycorrhizal fungus (ecto, endo, asco or basidiomycete) forming symbioses allows the increase of vegetal biomass but it is important the knowledge about their biology and their ecological role. To Barea & Honrubia (2004) it is important to select primary and pioneer fungi capable of prospering in hostile situations of erodibility, transforming the edaphic conditions to an adequate microbiological and mycorrhizal succession. To estimate the mycorrhizal diversity on the field, morpho-anatomic and molecular techniques (PCR, fingerprints, sequencing and genotyping) could be studied on the carpophorus and on mycorrhizal structures. Then to collect, process, isolate fungus carpophorus and conserve mycorrhiza strains using techniques described by Barea & Honrubia (2004) and Brundrett (2008).

Basically the mycorrhizal inoculum (asco or basydiomicete) can be derived from three different sources: a) directly from the forest soil with many different mycorrhizal inoculum sources and possible introducing pathogenic organisms (bacteria, fungus); b) spores from fungus hymenophore tissues directly on seedlings; c) a mycelium *inoculum* produced by fermentation of a mixed substratum of "vermiculite and turba" in the seedlings growth medium. The inoculation of ecto-mycorrhizal fungus is useful in reforestation programs and to agricultural soils uses so in agroforestry systems the inoculation of *Pinus halepensis* with *Pisolithus* sp. and *Rhizopogon* sp. was done.

Mediterranean forests are particularly threatened by fire and further studies on the potential effects of fire on the role of fungi both pioneer, intermediate and post fire fungus in ecosystems is poorly documented (Claridge *et al.*, 2009). Rillig (2004) suggested the study of VAM to a proper study of ecosystem processes.

It is known that mycorrhizal fungi are important to soil productivity and to ecosystem recovery (Azul & Freitas, 2005) and the capacity of recovering of mycorrhizal populations is highly dependent on intensity, exposure and repetitiveness of a fire event and fungus diversity varies dramatically in the first months after fire but symbiotic associations are maintained by the most resistant species (Barea & Honrubia, 2004).

It is important the knowledge about plant symbiosis and the study of the corresponding fruit bodies of ECM fungi to evaluate their potential as indicators of ecosystem disturbance.

Some metabolic processes studied by Wood & Goodenough (1977), test the activity of extracellular enzymes and the corresponding organic compounds (polysaccharide) during mycelial growth and fruiting stage since there is variability on their content being possible to quantify the products of enzymatic activity although it is not possible to estimate the total fungi biomass after fire because the fungi inhabit the root and the soil.

Recently Rousk *et al*. (2010), Muyzer *et al*. (1993) and Gomes *et al*. (2003), develop molecular techniques to increase the knowledge about microbial communities in the bulk soil, rizosphere and in mycorrhiza associations.

In order to an adequate silviculture practice into reforestation programmes it is possible to inoculate plants according with methodologies described by several authors. Azcón-Aguilar & collaborators (2000) suggest to produce arbuscular mycorrhiza *inoculum* from natural populations and collect, process, isolate fungus carpophorus and conserve mycorrhiza strains following the techniques of Barea & Honrubia (2004) and Brundrett (2008) using as sources spores from fungus hymenophore tissues directly on seedlings. The mycorrhizal network in the soil is enhanced after disturbance if a forest system already had inoculum potential from trees with strong AM associations (Jeffris & Barea, 2001; Brundrett, 2004). The fungi are also important to organic matter decomposition to the release of woody debris and organic nutrients in the soil.

To take better uses from forests Barea & Honrubia (2004), inoculate species of truffles and *Lactarius* sp. on *Quercus* sp. and *Pinus* with good results. Some Mediterranean shrubs (*Lavandula lanata*, *Crataegus monogyna, Rosa canina, Olea europae, Pistacia lentiscus, Genista sp., Rosmarinus officinalis*) were tested with VAM infection by Jeffris & Barea (2001) and an extensive database of ectomycorrhiza association with trees is available (http://www.deemy.de). It will be important to select the best plant symbiosis to the post fire conditions according with specific climacteric conditions, soil properties, geology, vegetation cover and slope.

In order to future research about the effects of fire on soil biota, a possible methodology to collect soil samples is proposed to compare the differences on soil biota after fire with the pre-fire conditions or before fire or with a non-burned adjacent area. The fire could be an experimental fire, a prescribed fire or a natural wildfire. To do a prescribed or an experimental fire is important to control some variables as soil and atmospheric conditions, wind direction and intensity (http://www.fire.wur.nl).In a prescribed fire the surface soil temperature is low and burned under a controlled manner to control vegetation and biomass. A practical example is the removal of shrubs beneath trees in winter. Under an experimental fire in a plot it is important to burn the edge of plots creating a safety zone around an area or digging ditches or removing strips of vegetation and plan fire to fight against wind and let fire to move from the top to the bottom of the slope (http://www.fire.wur.nl). To study the effects of fire intensity on soil biota the comparison between burned and non-burned samples could be done after prescribed and experimental fires to compare the effects of different fire intensity on the soil biota diversity and on mycorrhiza.

A possible sampling methodology to study soil biodiversity can be done after fire in plots with specific measures placed randomly in a burnt area and the plots can be dug in successive soil layers (0-2cm), (2-5cm) and (5-10 cm) where each layer forms a sample (Ferrandis P.,1996). In the study of Zuzana Sýkorová (2007) to characterize the communities of arbuscular mycorrhizal fungi in the roots of selected plant species using molecular methods was done a sampling with cores of 20cm diameter in a depth of 15cm removed randomly from an area and the roots were washed and blotted dry to further studies. The mycorrhizal fungi diversity in the soil is assessed by spores found in the soil and either in the roots of the fungal mycelium. Another possibility to collect samples is along transects of 60-100m for example collecting samples every 5m. Each soil sample and each transects represent intra-site variation as different slope and other variables. The soil samples can be collected under different microhabitat types as beneath trees, shrubs or gaps of the remaining vegetation (Gidi Ne'eman & Ido Izhaki, 1999), and then brought to the laboratory for further study.

4.4. REFERENCES

Adjoud-Sadadou D., Hargas-Halli R. (2000). Occurrence of arbuscular mycorrhizas on aged *Eucalyptus*. Mycorrhiza, 9: 287-290.

Anderson I.C., Cairney J.W.G. (2007). Ectomycorrhizal fungi: Exploring the mycelial frontier. Microbiology Reviews, 31 (4): 388-406.

Azcón- Aguillar C., Palenzuela E.J. & Barea J.M. (2000). Substrato para la producción de inóculos de hongos formadores de micorrizas. Patente N. 9901814, España,CSIC.

Azul A.M. & Freitas H. (2005). Sobre a diversidade de micorrizas e o funcionamento dos ecossistemas florestais mediterrânicos. A Pantorra, 5: 45-48.

Azul A.M., Ramos V., Pato A., A M., Freitas H. (2008). Mycorrhizal types in the mediterranean basin: safety teaching and training.JBE, 42 (3): 130-137.

Baptista M.J., Gloria B. A., Pascholati S.F., Krugner T.L. (1999). Revta Brasil.Bot., 22 (2): 309-315.

Barea J.M., Requena N., Jimenez I. (1996). A revegetation strategy based on the management of arbuscular mycorrhizae, *Rhizobium* and rhizobacterias for the reclamation of desertified Mediterranean shrubland ecosystems. In: Seminaire du Groupe de Travail sur l´Utilisation des Mycorhizes pour la lutte contre la Desertification dans le Bassin Mediterraneen. (Zaragoza: CIHEAM-IAMZ),pp.75-86.

Barea J.M. & Honrubia M. (2004). La Micorrización dirigida de la planta forestal. In: Avances en el estudio de la gestión del monte Mediterráneo. Vallejo V.R. & Alloza J.A. (Eds.). Fundación CEAM, pp.215-260.

Boudarga K., Lapeyrie F., Dexheimer J. (1990). A technique for dual Vesicular-Arbuscular endomycorrhizal/ectomycorrhizal infection of Eucalyptus in vitro, 114 (1): 73-76.

Brundrett M.C. (2002).Coevolution of roots and mycorrhizas of land plants. New Phytologist, 154: 275-304.

Brundrett M.C. (2004). Diversity and classification of mycorrhizal associations. Biological Reviews, 79 (3): 473-495.

Brundrett M.C. (2008). Mycorrhizal Associations: The Web Resource.
Available on: mycorrhizas.info.

Capela, P. (2009). As micorrizas e a floresta autóctone in: Workshop “A importância das micorrizas na floresta autóctone”, Quercus Braga, Braga.

Carlile M.J., Watkinson S.C. (1994). The fungi. San Diego: Academic, 428p.

Claridge A.W., Trappe J.M., Hansen K. (2009). Do fungi have a role as soil stabilizers and remediators after forest fire? Forest Ecology and Management, 257 (3): 1063-1069.

Coelho COA, Shakesby RA, González del Tánago M, Ternan L, Walsh RPD, Williams AG (1995) IBERLIM: Land management and erosion limitation in the Iberian Peninsula. Final Report to the EC in fulfilment of Project EV5V-0041 `Land management practice and erosion limitation in contrasting wildfire and gullied locations in the Iberian Peninsula (unpublished), 246 pp.

Deemy: An information System for Characterization and Determination of Ectomycorrhizae (2004–2010 by Ludwig-Maximilians-Universität München, Dept. Biologie I – Systematische Mykologie) Available on: http://www.deemy.de

Deliberately on fire, Magazine (2009) by Cathelijne Stoof. Available on: http://www.fire.wur.nl

Dickie I.A. (2006). Mycorrhiza of forest ecosystems. In: Encyclopedia of soil science. Lal R. (Ed.). University of Minnesota, U.S.A.

Edwards N.T. & Riggs J.S. (2003).Automated Monitoring of soil respiration: a moving chamber design. Soil Science Society of America Journal, 67:1266-1271.

Ferrandis P., Herranz J.M., Martínez-Sánchez J.J. (1996). The role of soil seed bank in the early stages of plant recovery after fire in a *Pinus pinaster* forest in SE Spain. International Journal of Wildland fire, 6 (1): 31-35.

Gil E.S., Muller L., Santiago M.F., Garcia T.A. (2009). Biosensor based on brut extract from laccase (Pycnoporus sanguineus) for environmental analysis of phenolic compounds. Portugaliae Electrochimica Acta, 27 (3): 215-225.

Goicoechea N., Closa I., Miguel A.M. (2009). Ectomycorrhizal communities within beech (Fagus sylvatica) forests that naturally regenerate from clear-cutting in northen Spain. New Forests, 38 (2):157-175.

Gomes N. C. M., Fagbola O., Costa R. S., Rumjanek N. G., Buchner A., Mendonça-Hagler L. C S., Smalla K. (2003). Dynamics of fungal communities in bulk and maize rhizosphere soil in the tropics. Applied and Environmental Microbiology, 69: 3758-3766.

Hibbet D.S.,Gilbert L.B., Donoghue M.J.(2000).Evolutionary instability of ectomycorrhizal symbiosis in basidiomycetes. Nature, 407: 506-508.

Hibbett DS, Binder M,Bischoff JF, *et al.* (2007).A higher-level phylogenetic classification of the Fungi. Mycological Research, 111:509-547.

Honrubia M. 2009. Las micorrizas: una relación planta-hongo que dura más de 400 millones de años. Anales Jard. Bot. Madrid, 66S1: 133-144.

Ingleby K.,Munro R.C., Noor M., Mason P.A., Clearwater M.J. 1998. Ectomycorrhizal populations and growth of Shorea parvifolia (Dipterocarpaceae) seedlings regenerating under three different forest canopies following logging. Forest Ecology Management, 111: 171-179.

Jeffris P. & Barea J.M. (2001). Arbuscular mycorrhiza – a key component of sustainable plant-soil ecosystems. In: Hock B. (Ed.), The Mycota IX ,Fungal Associations. Springer-Verlag Berlin Heidelberg,pp.95-113.

Johnson D., Leake J.R. & Read D.J. (2001). Novel in growth core system enables functional studies of grassland mycorrhizal mycelial networks. New Phytologist, 152: 555–562.

Jones M.D., Durall D.M., Cairney J.W. (2003). Ectomycorrhizal Fungal Communities in Young Forest Stands Regenerating after Clearcut Logging. New Phytologist, 157 (3): 399-422.

Machado H. (2004). A review on *Eucalyptus globulus* symbiosis in Portugal. In: Eucalyptus in a changing world. Borralho N. et al (Ed.), (IUFRO conference, Aveiro).

Malajczuk N., Lapeyrie F., Garbaye J. (1990). Infectivity of pine and eucalypt isolates of Pisolithus tinctorius on roots of Eucalyptus urophylla in vitro. New Phytologist, 114: 627-631.

Mason P.A., Wilson J., Last F.T., Walker C. (1983). The concept of succession in relation to the spread of sheathing mycorrhizal fungi on inoculated tree seedlings growing in unsterile soils.Plant and Soil,71:247- 256.

McAFEE, B.J. & Fortin, J.A. (1988). Comparative effects of the soil microflora on ectomycorrhizal inoculation of conifer seedlings. New Phytologists, 108: 443 – 449.

Muyzer G., de Waal E.C. & Uitterlinden A. G. (1993). Profiling of complex microbial populations by denaturing gradient gel electrophoresis analysis of polymerase chain reaction-amplified genes encoding for 16S rRNA. Applied and Environemtal Microbiology, 59: 695–700.

Ne'eman Gidi, Izhaki I.(1999).The effect of stand age and microhabitat on soil seed banks in Mediterranean Aleppo pine forests after fire. Plant Ecology,144:115-125.

Rillig, M.C. (2002). Arbuscular mycorrhizae and terrestrial ecosystem processes. Ecology Letters, 7: 740-754.

Rousk J., Bååth E., Brookes P.C., Lauber C.L., Lozupone C., Caporaso J.G., Knight R. and Fierer N. 2010. Soil bacterial and fungal communities across a pH gradient in an arable soil. ISME J. in press.

Roman M., Claveria V. & Miguel A.M. (2005). A revision of the descriptions of ectomycorrhizas published since 1961. Mycological Researche, 10: 1063-1104.

Smith S.E. & Read D.J. (1997) *Mycorrhizal Symbiosis*, 2nd ed. Academic Press, San Diego, CA.

Smith F.A., Smith S.E. (1997). Tansley Review No.96.Structural diversity in (vesicular)-arbuscular mycorrhizal symbioses. New Phytologists, 137: 373-388.

Smith J.E., Mckay D., Brenner G., McIver J., Spatafora J. (2005). Early impacts of forest restoration treatments on the ectomycorrhizal community and fine root biomass in a mixed conifer forest. *Journal of applied ecology*, 42:526-535.

Wang B. & Qui Y.L.(2006). Phylogenetic distribution and evolution of mycorrhizas in land plants. Mycorrhiza, 16: 299-363.

Wilson G.W.T. & Harnett D.C. (1998). Interspecific variation in plant responses to mycorrhizal colonization in tallgrass prairie. American Journal of Botany, 85(12):1732-1738.

Wingler A., Guttenberger M., Hampp R. (1993). Determination of mannitol in ectomycorrhizal fungi and ectomycorrhizas by enzymatic micro-assays. *Mycorrhiza*,3: 69-73.

Wood D.A., Goodenough P.W. (1977). Fruiting of Agaricus bisporus. Changes in Extracellular Enzyme Activities during Growth and Fruting. Archives of Microbiology, 114:161-165.

Zuzana Sýkorová (2007). Molecular ecological analyses of specific interactions between symbionts in the arbuscular mycorrhizal symbiosis, Doktors der Philosophie, Tschechische Republik, Basel.

Universidade Federal Rio Grande do Norte. Atlas Virtual de Botânica. Available on: http://www.cb.ufrn.br/atlasvirtual/fungos.htm

Web resources:

http://ecosanto.files.wordpress.com/2008/02/relat_4ctv_belazaima.pdf

http://forestry-dev.org/cgi-bin/matchmaker/MatchMaker.asp http://www.deemy.de

http://www.fire.wur.nl

CHAPTER 5

Conclusions

CHAPTER 5 CONCLUSIONS

The aims of this thesis were i) to compile bibliography about the effects of fire on the soil-plant dynamics in forests; ii) to investigate areas of study to test the effects of fire on vegetation and on the fungus of the soil.

It is concluded that the loss of forested areas leads to the impoverishment of biodiversity of the biotic components of soil and changes in ecosystem function. According to Lloret & Zedler (2009), after the passage of fire and according to the regeneration strategies of plant populations the ecosystem function are potentially restore in the Mediterranean region. The recovery of vegetation depends on competition for sunlight, nutrient availability from ashes deposition and water losses (Naveh, 1975). The resilience of soil system depends on vegetation type, on soil characteristics and on the topography of the burnt area (Certini, 2005).

The importance of symbiotic interactions between plant roots and fungi are important to the regeneration of vegetation after fire (Azul & Freitas, 2005) and according to Doerr *et al*. (1984), Wilson & Harnett (1998), Barni & Siniscalco (2000), the mycorrhizal infection improve nutrient uptake, affects the demographic response and the relative abundance of species in a plant community.

Anderson & Cairney (2007) and Wood & Goodenough (1977) described recent techniques to study fungus–specific compounds and others Gomes *et al*. (2003), develop molecular techniques to increase the knowledge about microbial communities in the soil.

According to Goicoecha *et al.*(2009), Claridge *et al.* (2009) it is important the study of post fire fungi and the capture of nutrients through the mycorrhizal network system due to the importance to vegetation reestablishment.

It is important to study the effects of fire on mycorrhiza and the rates of mycorrhiza and soil biota destruction after the passage of fire in burned areas in the Portuguese territory.