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Wildfire and forest management effects on soil properties

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TESIS DOCTORAL

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WILDFIRE AND FOREST MANAGEMENT EFFECTS ON SOIL PROPERTIES

EFECTOS DE LOS INCENDIOS FORESTALES Y LA GESTIÓN FORESTAL EN LAS PROPIEDADES DEL SUELO

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Fdo.: Xavier Úbeda Cartañá

Fdo.: Paulo Alexandre da Silva Pereira

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"There will be obstacules

There will be doubters

There will be mistakes

But with hard work

There are not Limits"

Anonymous

"Me enseñaron que el camino del progreso

No es ni rápido ni fácil"

Marie Curie

"However difficult may seem,

there is always something you can do,

and succeed at.

It matters that you do not just give up."

Stephen Hawking

"Los conocimientos y

las habilidades son

muy importantes,

pero lo que diferencia

a los grandes de los

mediocres, es su

actitud"

Victor Kuppers

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PRESENTACIÓN

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Todos los artículos presentados en esta tesis doctoral, versan sobre el impacto de los incendios forestales, la gestión forestal y otros aspectos que afectan a las propiedades del suelo. Estos estudios se han ordenado en los diferentes capítulos que pomponen esta Tesis Doctoral según la crono-secuencia que se establece en un incendio forestal respetándose el formato de las referencias bibliográficas según la normativa de cada revista. Las referencias utilizadas en los apartados de introducción y material y métodos han sido referenciadas siguiendo el modelo APA. Primero de todo se ha emplazado una revisión bibliográfica donde se presentan los artículos más relevantes en materia de gestión forestal y sus impactos en el suelo, así como otros factores de carácter ambiental que determinan la evolución temporal de las propiedades edáficas. El segundo artículo que se presenta se centra en el análisis del efecto de la gestión forestal pre-incendio en las propiedades físicas, químicas y microbiológicas del suelo. El tercer estudio se centra en la evaluación del impacto de una lluvia torrencial y la gestión mediante el uso de maquinaria pesada en un suelo justo después de un incendio forestal. El cuarto manuscrito presenta la evolución a corto plazo de diferentes gestiones forestales post-incendio y su impacto en las propiedades del suelo. El quinto y el sexto estudio se centran en el análisis del impacto del fuego a diferentes severidades a largo plazo y la falta de gestión; y en la necesidad de gestión forestal para disminuir la densidad vegetal, con ello disminuir el riesgo de incendio forestal y mejorar las propiedades edáficas.

Todos estos trabajos se basan, como apuntaba anteriormente, en el impacto de los incendios forestales y la gestión forestal en las propiedades del suelo. Esta investigación se ha realizado gracias a la beca FPU concedida a partir de la cual pude comenzar a trabajar en el GRAM (Grup de Recerca Ambiental Mediterrània) dirigido por Xavier Úbeda en la Universitat de Barcelona (UB). Este grupo de investigación nació en 1992 y fue uno de los primeros grupos de investigación reconocidos formalmente en el ámbito de la Geografía. Este reconocimiento se mantuvo durante los primeros diez años de existencia, pero, posteriormente, un descenso en el número de miembros activos provocó un paréntesis en ese reconocimiento. Como consecuencia, el grupo tuvo que ser repensado y reestructurado, sobre la base de incorporar nuevos miembros e idear una estrategia para planificar acciones de futuro y proponer y conseguir nuevos proyectos de investigación. De esta manera se pudo postular nuevamente como grupo de investigación y se consiguió el reconocimiento como grupo emergente (2009SGR1515) para el periodo 2009-2013. Siguiendo en la misma línea obtuvimos a continuación, y para el periodo 2014-2016, el reconocimiento como grupo consolidado (2014SGR825). Un reconocimiento que, en la última convocatoria, hemos podido revalidar (2017SGR1344) para el período que empieza en 2017 y que, en principio, se termina a finales de 2019. El eje vertebrador de las diversas líneas temáticas del GRAM es el concepto de paisaje, como un totalizador geográfico e histórico. Se trata de un concepto genuinamente integrador, que nos permite conectar los hechos físicos con los factores humanos y sociales. Una conexión que establecemos a través del medio, esencialmente del entorno físico dentro del cual se desarrollan los procesos biológicos (también la vida humana), y desde el territorio, fijando la atención en la realidad humana y social, dentro de la cual, el entorno físico y natural juega un papel fundamental. Por ello, esta Tesis Doctoral, se ha centrado en cómo el ser humano gestiona el impacto de un fenómeno natural como son los incendios forestales en el medio ambiente, concretamente en el suelo que es el soporte de la vida.

Durante este tiempo de realización de la Tesis Doctoral he participado en dos proyectos convocados por el Ministerio de Economica y Competitividad de España, ambos de ellos dirigidos por Jorge Mataix Solera y Xavier Úbeda. En ambos proyectos, el autor de la presente Tesis Doctoral ha participado como equipo de trabajo y ha contribuido con sus investigaciones. Estos proyectos son POSTFIRE (CGL2013-47862-C2-1) "Calidad del suelo, control de erosión y recuperación de la cobertura vegetal en diferentes escenarios de manejo post incendio" y POSTFIRE_CARE (CGL2016-75178-C2-2-R) "Estrategias de manejo post-incendio orientadas a preservar y mejorar la calidad del suelo) como parte del grupo de investigación".

GENERAL INTRODUCTION

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1. Wildfires and territory

Fire was present as a natural phenomenon – provoked by electrical storms and volcanic eruptions – long before man trod the surface of this planet (Naveh, 1975). In fact, the fires of the Pliocene have been identified as one of the factors – together with the climate – that did most to contribute to the formation of the Mediterranean vegetation (Bond and Keeley, 2005). During the Carboniferous, Jurassic and Cretaceous Periods, and in the Quaternary with the appearance of man, countless wildfires broke out across the Earth (Rodriguez-Trejo, 2012). Subsequently, man was to harness this natural element as a tool to carry out a variety of activities in the territories they inhabited: scaring off predators, keeping warm, controlling pests, cooking and burning vegetation to make the landscape habitable (Figure 1). Indeed, the earliest evidence of the use of fire by humans dates back 400,000 years (Santin and Doerr, 2016). Moving closer to present times, during the twentieth century, because of the abandonment of rural areas and the human pressure in given areas, and owing to the shifting economic interests of forest management, the frequency and intensity of forest fires increased (see detailed explanation in Section 1.2) (Vélez, 2000), which explains why for much of the last 100 years fires were perceived as harmful. In short, fire is an element that has accompanied the existence of human beings throughout their history (Pausas and Keeley, 2009).

Figure 1. Stone bearing the inscription "Our ancestors were able to use and control fire at least 1 million years ago in the cradle of humankind" in the Maropeng and Sterkfontein caves, South Africa. Picture: Marcos Francos.

This negative perception of fire led to attempts to achieve complete “fire suppression”. Yet, this somewhat paradoxical conception meant that many areas of the United States, Australia and Europe have registered their most devastating fires in recent history. Fire suppression involves those activities undertaken to control and extinguish a wildfire (Pearce et al., 2000) and include such actions as eliminating the fuel to prevent the spread of combustion, controlling the temperature of burning fuels and excluding oxygen from the combustion area by smothering.

The “fire paradox”, as it has come to be known in many forums, identifies the fact that increasing levels of professionalism in the fighting of fires results over time in a marked increase in fuel loads that are highly conducive to the generation of large forest fires. Today, many authors, as well as administrators and managers, have learnt to assume that fire is natural and that it has played and continues to play an important ecological role, above all in geographical areas characterized by a Mediterranean climate (Plana, 2004; Mataix-Solera and Cerdà, 2009). In the 21st century, the conception of fire has thus changed and it is now seen by many as a useful, even necessary, tool. Indeed, forest fires are by no means always catastrophic and, in some cases, they only result in the slight burning of the aerial parts of vegetation, resulting in the fertilization of the soil thanks to the ash that is produced. Various authors have described how natural cycles of low-severity fires can help improve the productivity of ecosystems thanks to the deposition of this organic matter and ash, and to the subsequent release of nutrients thus generated (Cromer, 1967).

Although today, for example in Catalonia (explained in full in Section 1.2), work is being undertaken in this direction: assuming 1) the presence of recurring fires in our forests, 2) that fire plays a role in modelling the landscape, and 3) that climate change can exacerbate this trend, forests are still characterized by their huge fuel load and are at risk of generating the ‘Great Forest Fire’, a threat that began in the 20th century with the onset of rural abandonment (Plana, 2011). Forest fires are a global phenomenon and in recent decades they have evolved to constitute ‘high intensity’, even, ‘mega-fires’ (Ferreira-Leite et al., 2013). Controlling and fighting wildfires of these characteristics pose enormous challenges and, moreover, they can be especially destructive.

The causes of wildfires can be identified as follows:

1) Rural abandonment, resulting in the accumulation of fuel and consequent landscape homogeneity (Benayas et al., 2007) (Figure 2).

2) The effectiveness of modern firefighting techniques have resulted in the eradication of the ecological function of fire (i.e. the so-called ‘fire paradox’: the better we become at extinguishing fires, the more prepared the landscape is to burn at high intensities (Strauss et al., 1989).

3) Land use changes (Pauses and Keeley, 2009).

4) The substitution of autochthonous species with fast-growing, more productive species.

5) A global approach that is responsible for changing wildfire regimes in specific areas (Summers et al., 2011). This phenomenon is more evident in the north of the planet than in the south, but in both boreal and austral regions monocultures are especially vulnerable (Tsibart et al., 2015).

Figure 2. Rural abandonment in Avinyó (Barcelona). Picture: Marcos Francos.

The supply of forest management practices today is enormously limited and this has exacerbated the increased risk of wildfire, which result primarily from a) the abandonment of agricultural activity and b) rural depopulation. In this first case, this has facilitated an increase in horizontal fuel (via the natural reforestation of abandoned agricultural land) and in vertical fuel (via the abandonment of forest management and understory grazing). Such is the extent of these processes that in Catalonia we would have to go back to the beginning of the first millennium to find a forest area comparable to that found today (Peix, 1999). In the second case, rural depopulation has meant the loss of passive or indirect fire prevention practices – that is, the immediate detection and extinguishing of fires – and the knowledge of the territory together with the maintenance of rural roads that the farmers/inhabitants of rural environments used to ensure thanks to their daily activity. The reasons for rural abandonment are essentially: 1) the low level of economic profitability provided by agricultural activities (Martinez-Alier and Roca-Jusment, 2000); 2) the lack of services and infrastructure in rural areas as professional opportunities become concentrated in urban centers; and 3) cultural factors, including the predominant values of the leisure/consumer society that fail to satisfy the requirements of agricultural activity, and the lack of social recognition afforded the figure of the farmer (Plana et al., 2001).

The wildfires produced can be classified into three main categories (Carballas, 2007):

- Crown fires – episodes in which burning spreads across the tops of trees or shrubs.
- Surface fires – episodes in which only surface litter and duff burn.
- Sub-surface fires (sometimes known as undersoil or subsurface fires) – episodes that occur in deep accumulations of humus, peat and similar dead vegetation which on drying are at risk of burning.

Likewise, there are three factors that characterize the behavior of these fires:

- The fuel: Light (herbs, leaves and small branches), heavy (trunks, branches and roots) and green (living vegetation) (Fernandes and Botelho, 2003).

- The topography: This can help predict and explain the behavior of a fire based on such properties as orientation, roughness and slope.

- Weather conditions: Primarily temperature, humidity and wind.

Some areas of the world are especially prone to forest fires, due to their climate type; yet, this does not mean that fires are unique to one type of climate. Great forest fires or mega-fires can occur in Mediterranean, boreal, austral or tropical climates, and in all vegetation zones (Archibald et al., 2013). The tropical rainforests, the African savanna, southern Chile and the Siberian tundra burn every year and thousands of hectares of forests are destroyed (Pereira et al., 2015a). For details on studies conducted of the world's mega-fires see Ferreira-Leite et al. (2015a), Jones et al. (2016) and Pyne (2016). These studies make it clear that wildfires are a global phenomenon and not restricted to the Mediterranean ecosystem (Úbeda and Francos, 2018). Having said that, a disproportionate number of studies have been conducted in the Mediterranean countries of Portugal, Spain, Greece, Algeria and Israel (see, for example, Inbar et al., 1998; Mataix-Solera et al., 2002; Wittenberg et al., 2007; Úbeda et al., 2009; Ferrera-Leite et al., 2015b) and in countries with a Mediterranean ecosystem, above all Chile (see, for example, Castillo-Soto, 2015; Úbeda and Sarricolea, 2016). Other countries, including Russia (Achard et al., 2008; Tsibart et al., 2015), Lithuania (Pereira et al., 2013; Pereira et al., 2015a), Finland (Vanha-Majamaa et al., 2007), the United States of America (Marlon et al., 2003; Collins and Stephens, 2007), Indonesia (Abram et al., 2003), Hungary (Deak et al., 2014); and other environments, including grasslands (Pereira et al., 2018a), steppe

(Maksimova and Abakomov, 2013), rain forest (Pew et al., 2001), tundra (Hudspith et al., 2017) and taiga/boreal (Viereck, 1973; Kharuk et al., 2010; Valendik et al., 2014; Pereira et al., 2016) have experienced the impact of an increasing number of wildfires in recent decades.

A range of different forest management tools have been used to avoid wildfires and to manage the forest fuel (Marino et al., 2014) (Figure 3). These include, most notably:

a) Forest planning
b) Prescribed Fires
c) Controlled grazing
d) Mechanical thinning

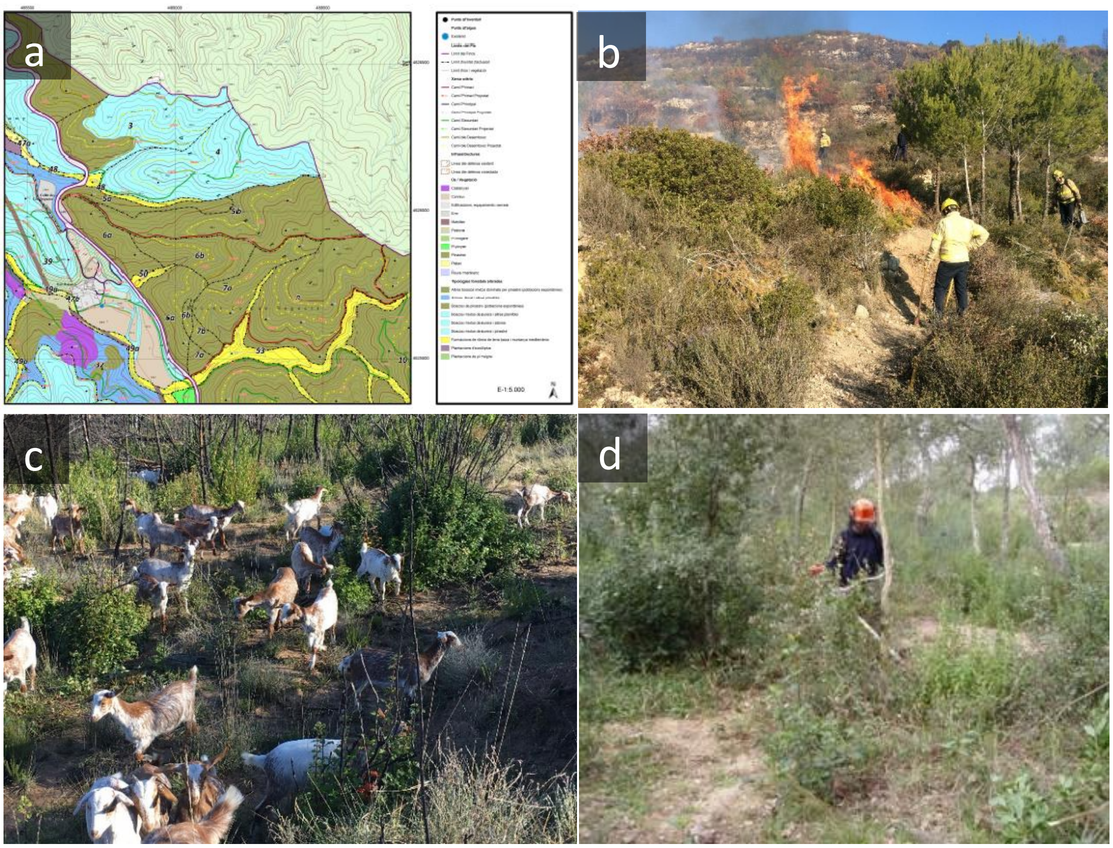

Figure 3. a) Forest Management and Improvement Technical Plan (PTGyMF) in Mas Bassets (Girona). Picture: Xavier Úbeda. b) Prescribed fire in Olesa (Barcelona). Picture: Marcos Francos. c) Controlled grazing to manage vegetal fuel loads in Ódena (Barcelona). Picture: Marcos Francos. d) Mechanical thinning to manage forest fuel in las Gavarres (Tarragona). Picture: Xavier Úbeda.

Based on the foregoing discussion, it can be seen that forest fires represent both a major environmental problem, resulting in the loss and degradation of the biosphere, and a major social problem, resulting in the loss of human lives, the contamination of aquifers, landslides, destruction of property in inhabited areas, and the degradation of crops. Yet, we should not forget that, in many parts of the world, forest fires have shaped the landscape, acting as ecosystem agent and, as such, we have learned to live with this phenomenon (Belcher, 2013).

2. Wildfires in Mediterranean environments

Fire is the main destructive force of forest areas in Mediterranean ecosystems (Vélez, 2000). The most important feature of this climate is the fact that the dry season coincides with the warmest temperatures. In the summer, the Mediterranean climate is under the influence of dry subtropical anticyclones, which retreat in winter towards lower latitudes, leaving behind stormy Atlantic conditions in temperate latitudes. This peculiarity means these ecosystems are especially susceptible to forest fires when dry summer vegetation becomes highly flammable, favoring the appearance and spread of fires (Arianoutsou et al., 1993). The rainy season that typically arrives after the dry season can erode large amounts of soil, a soil that is especially fragile if its organic matter has been exposed to combustion and its nutrients have been leached (Mataix-Solera and Cerdà, 2009).

In Spain in the 1960s, industrialization and rural exodus led to the abandonment of mountain areas and contributed to an increase in forest cover, a process which continues today. The change in land use combined with the increased risk of forest fires have also contributed to the reforestation of pine forests and the emergence of new residential areas (Pausas et al., 2008). The outcome has been an increase in the number of forest fires since the 1970s especially when climate conditions are favorable. Fires are now more frequent and, although Mediterranean ecosystems are able to coexist with this, the regime change can have adverse effects (Bodí et al., 2012). In the economic transition of the 60s and 70s, the risk of fire in rural areas increased greatly due to the progressive accumulation of fuel as the rural population stopped extracting this resource to exploit as a source of energy. Unwittingly, these people had previously been taking care of the forest and protecting it against fires. The consequences of these changes in use have been an increase in the fuel and biomass present in the mountains and, as a result, an increase in the number of fires (Mataix-Solera and Cerdà, 2009). The 80s and 90s in Spain were crucial as regards the outbreak of wildfires, as these were the decades of the great forest fires, the direct result of this accumulation of fuel and the prevailing fire suppression policies. The year 1994 was the worst on record in terms of hectares affected by fire (437,000 ha in Spain as a whole and 80,000 ha in Catalonia alone) (ADCIF, 2002). For this reason, this decade recorded an increase in the number of scientific studies dedicated to analyzing the effects of fire on soil properties and the impact of heightened levels of erosion. In Catalonia, the major fires of Central Catalonia of 1994 and 1998 caused considerable devastation and, in this autonomous community, forest fires are the main factor underpinning the loss of forest areas (Plana, 2004).

Numerous publications have examined the short-term effects of forest fires in Spain on soil properties (Úbeda, 1998; Mataix-Solera, 1999). Some conclude that, following the fire, a loss of nutrients from the system is recorded due to the episode of combustion, with various elements being volatilized; yet, it has also been documented that after a fire new, highly mineralized elements are incorporated together with the ash that has been generated (Alcañiz et al., 1996). These impacts of wildfire on soils are explained in depth in Section 1.3. Fundamental to this discussion is the incorporation of the variables fire “intensity” (defined as temperature and duration of residence of the fire in a specific point), and the consequences for the environment of this intensity, that is, fire “severity”. More specifically, fire intensity is the duration of maximum temperature recorded at a certain point, expressed in ºC/sec (Úbeda et al., 1998) – that is the energy released in the combustion of organic matter as a function of temperature (Keeley, 2009) – while fire severity is the result of fire intensity and the duration of ignition (Keeley, 2009). The severity captures the response of the ecosystem to fire and can be used, in turn, to analyze the response of the soil, flora and fauna to this kind of perturbation (Bento-Gonçalves et al., 2012).

Thus, the current lines of research being developed by both researchers and managers can be summarized as 1) understanding the balance between the negative and positive effects of a forest fire, and 2) studying the variables that intervene in our understanding of the behavior of fire. The successful isolation of these two objectives should facilitate the assimilation of the forest fire as a critical element to be taken into consideration in forest management and, more broadly, in the planning of the territory (Piqué et al., 2011). Management practices of this type designed to prevent forest fires should play a major preventive role (González and Pukkala, 2007; Corona et al., 2015).

In Catalonia, forest management is centered primarily on preventive forestry practices, the main objective of which is to obtain forestry structures characterized by masses of low combustibility, that is, that present greater resistance to the propagation of fire (Vélez, 2000) (Figure 4). Here, the sequence of treatments that make up preventive silviculture – that is measures aimed at reducing the vulnerability of a forest to fire – are, according to Fernandes and Rigolot (2007), the following:

- Reduction of surface fuel to limit the potential intensity of surface fire.

- Pruning and scaling of fuel to reduce the probability of the spread of vertical fire.

- Thinning to minimize the probability of fire transmission between crowns.

- Controlled replication of species that generate forest structures of low vulnerability to forest fires.

Figure 4. Medium- or long-term forest management after wildfire is essential to reduce risk of wildfire. Ódena (Barcelona). Picture: Xavier Úbeda.

These measures or treatments can be carried out using livestock or machinery or by employing fire as a tool to prevent fires, that is, by means of prescribed fires (Alcañiz et al., 2018). In the last two decades, social conceptions of forest fires and the non-management of the forest have changed. Although, on the whole, this phobia of forest fires remains, part of society is slowly becoming aware that fire has a role to play in the evolution of Mediterranean ecosystems (Figure 5). This gradual paradigm shift, together with the current fire regime, has created the need not only to prevent and combat fire, but also to integrate the phenomenon into the territory (Plana, 2011).

Figure 5. Environmental education is essential to explain the role of fire in Mediterranean ecosystems and the consequences of forest management. Las Gavarres (Girona). Picture: Xavier Úbeda.

In Catalonia, this change was triggered by the great wildfire of 1994. According to Maza et al. (1995), forest fires need to be taken into account in the planning and managing of the territory as well as in urban development. As a result of the fires that broke out in the 90s, forest fire risk management strategies were greatly expanded, with fire being introduced as a fuel management tool (Castellnou and Nebot, 2007) and common prevention and firefighting policies being incorporated in land use planning (Plana, 2011). The integration of fire risk in forest planning practices means taking into account such risks in forest management at the farm level (Piqué et al., 2011), including the modeling of vulnerability to ignition and propagation in forest masses, and the development of a vision of the territorial matrix and a perspective grounded in landscape ecology (González et al., 2006). In this way, everything related to livestock use, forest management and urban and territorial planning is perceived in such a way as to envisage the dangers and the vulnerability of the territory (Plana, 2011). Territorial planning seen in this way is directly related with the increase in recent years of the wildland-urban interface and the *deseasonalization* of forest fires. Badia et al. (2002) call for the recovery of the landscape mosaic that historically characterized Mediterranean areas as a means of countering the growing occurrence and impact of forest fires. These authors stress the importance of forest fire prevention measures and the need to combine traditional and alternative forest management practices that can take into account such relevant factors as agricultural abandonment, rural depopulation and lack of resources. Wildfires in areas subject to human disturbances can record catastrophic consequences many years before the actual outbreak of the fire (Pausas et al., 2008). In recent decades, many studies have highlighted the growth of the period during which forests are at risk from wildfires, a period no longer limited to the summer months (Versini et al., 2013). This expansion has, on occasions, led to the concurrence of wildfires and torrential rainfall (Girardin et al., 2013), a phenomenon that can have a great impact on soil properties (see Section 1.4 for a more detailed explanation) (Francos et al., 2016a). Catalonia's forest, in common with most forests in the Mediterranean basin, is typified by small, private properties (Cervera et al., 2015). With the passage of time, it has become increasingly evident that this privatization has failed and that the owners have been unable to preserve the forest to the same degree as they have succeeded in exploiting it. At the beginning of the 20th century, the owners of certain large estates (originating from *la Desamortización* or the land confiscations instigated by Mendizábal and Madoz in the 19th century) encountered a range of difficulties impeding their successful management and in many cases failed to exploit their many hectares of forest resources. At the end of the 20th century and the beginning of the 21st, any advantages to be gained from the exploitation of the forests were concentrated in a small number of areas yet the land dedicated to forestry continued to increase. The concomitant abandonment of the multiple and integrated uses of the forest, which served to maintain a mosaic of different land covers in forest areas, led to the appearance of dense, continuous forest stands. This in turn reduced the diversity of both habitats and species while providing the ideal conditions for large forest fires.

A number of the studies presented in this doctoral thesis have been carried out in Ódena (Barcelona province). These forests are a good example of the initiatives being undertaken by the Catalan government to promote sustainable forest management. Moreover, between 2014 and 2018, the area has been the setting for a *LIFEMontserrat* conservation project, which enjoys the financial support of *Natura 2000* and European Union funding. Given the threats posed to the territory, and the need to protect, conserve and manage the region, *LIFEMontserrat* seeks, above all else, to increase the resilience and stability of the forest in the face of large forest fires. It does so by implementing a set of measures of ecosystem management that contribute to the conservation and improvement of biodiversity in the Montserrat area and to the creation of a mosaic of scrub, natural meadows and forests aimed at strengthening ties between the *Natura 2000* spaces and the socioeconomic development of the territory. The *LIFE* project is promoted by the European Union as part of an initiative to finance environmental conservation projects and create community legislation in this field. In the case of *LIFEMontserrat*, the budget is 3.5 million euros, half of which is financed by the European Union, while the other half is covered by six partners: the *Diputación* (Provincial Council) de Barcelona; the Department of Agriculture, Livestock, Fisheries and Food; the

Firefighters of the *Generalitat* of Catalonia; the Board of the Montserrat Mountain; Catalonia-La Pedrera Foundation and the Association of Forest Owners.

In this area, the active forest fire preventive measures focus on forest fuel management, strengthening firewalls and the management of the wildland-urban interface. The passive measures focus on developing forest profitability, creating links between the urban and forest interface, promoting forestry associations and the inclusion of forest management practices in territorial plans. In social terms, the priority guidelines include the regulation of the use of fire in agricultural practices, boosting the involvement of rural society and providing environmental education (Romero and Senra, 2006). Fire prevention measures in combination with forest management practices prior to the occurrence of a wildfire can reduce fire severity and thus avoid soil sterilization. In this regard, Mataix-Solera and Cerdà (2009) stressed the need to promote forest management and to ensure that territorial management gives careful consideration to the role of wildfires in Mediterranean ecosystems.

3. Wildfire impact on soils

Oxygen, forest fuel and the heat resulting from ignition are the three basic components for the generation of fire and they are all found on the earth's surface (Mataix-Solera and Cerdà, 2009). As such, fire should be considered a natural factor of terrestrial ecosystems (Naveh, 1975). Some authors, including Certini (2005), consider fire a soil-forming factor due to the great influence it has on the soil environment. Indeed, changes in soil properties depend on fire intensity, post-fire weather conditions, forest management and the measurement terms themselves. Neary et al. (1999) identify the following soil disturbances with increasing degrees of fire intensity (see Table 1).

Table 1. Soil changes produced by fire intensity, according to temperature (ºC) (Based on Neary et al. 1999).

Temperature (ºC)	Disturbance
48-54	Dehydration of certain roots and death
70-90	Death of certain seeds
50-121	Soil microorganism death
180-300	Combustion of 85% of the organic horizon
200-250	Increments in soil water repellency
200-315	Complete destruction of organic matter
275-300	Destruction of soil hydrophobic compounds
200-400	Distillation of nutrients (mostly N)
>300	Complete destruction of soil organic horizons
= 450	Total consumption of soil organic matter
600	Maximum losses of soil phosphorus and potassium, and oxidation of metal bindings
800	Soil sulfur oxidation
1240	Soil calcium volatilization

Fire severity describes the response of the ecosystem to fire and can be used to describe the effects of fire on the soil, water system, flora, fauna, atmosphere and society (Bento-Gonçalves et al., 2012). In fact, the severity of the fire is a product of both the intensity and the duration of the fire. Keeley (2009) established different qualitative categories of fire severity taking into account the state of vegetation and organic matter after a forest fire:

- Not burned
    - The plants are unchanged. They present no observable effects as a result of heat.
- Little burned
    - Not burned. Plants have lost some leaves as a result of heat but there are no changes in soil properties and plants are unaffected.
- Low-severity
    - The treetops have green needles, while some branches are burned.
    - Leaf litter, mosses and other herbs are charred or consumed producing black ash (Figure 6).
    - The organic layer of the soil is almost intact. Only the first millimeter has been carbonized.
- Moderate/Medium-severity

- Part of the tree tops are dead, but the needles have not been consumed.
- All the understory plants have been consumed producing grey ash (Figure 6).
- Fine branches consumed and tree trunks charred.
- Organic horizon of soil widely consumed.

- High-severity
  - Cups of dead trees and needles consumed.
  - Organic horizon consumed.
  - White ash on the surface and charred material about one centimeter below the surface (Figure 6).

Figure 6. Black, grey and white ash produced during plant combustion during forest fire. Castellolí (Barcelona). Picture: Marcos Francos.

The impact of fire on soil can be direct or indirect. Direct impact is a consequence of the heating produced in the first few centimeters of the soil and the indirect impact is the result of the temporary elimination of part of the vegetation cover and the incorporation of ash into the soil (Mataix-Solera and Guerrero, 2007). Soil quality is recognized as the most dynamic and sensitive way of measuring the resilience of the soil to changes induced by natural or anthropic forces (Karlen et al., 2003). Soil thermal conductivity is very low and so the impact of fire on soil depth is not great and so most studies conduct samples to a depth of 2.5 cm. Indeed, according to Badía-Villas et al. (2014), the changes produced by fire are limited to the first 2 cm, while deeper soils are not modified significantly by fire (Badía et al., 2017). Some authors, including

Francos et al. (2016a), Pereira et al. (2017) and Francos et al. (2018a, b, c, d), sampled to a depth of 5 cm to identify the direct and indirect impacts of fire, management and plant regrowth. Neary et al. (1999) reported that high intensity wildfires can reach 675 ºC at soil surfaces (0-2.5 cm), and recorded temperatures of 190 ºC and 75 ºC at depths of 2.5 and 5 cm, respectively. However, Caon et al. (2014) report that soil temperature at 0–5 cm soil depth rarely exceeds 150 °C, though in some cases this is sufficient to change soil properties. According to Doran (1994), the main indicators used to determine the quality of the soil can be divided as follows:

- **Physical**: soil water repellency, aggregate stability and infiltration capacity.
- **Chemical**: organic matter, inorganic carbon, total nitrogen, C/N ratio, pH, electrical conductivity, major extractable elements and other elements including phosphorus.
- **Biological**: basal soil respiration and soil biomass carbon.

Forest fires are one of the natural agents that impact the quality of soils in Mediterranean forests, altering the physical (aggregate stability, porosity, distribution, hydrophobicity and runoff), chemical (availability of nutrients, mineralogy, pH and C/N ratio), and biological (biomass productivity, microbial composition and storage of carbon) quality of the soil (Doerr and Cerdà, 2005; Mataix-Solera and Guerrero, 2007).

Soil properties are often interrelated and to isolate the individual effect of fire in each soil property is complex. Pereira et al. (2018b) described attempts to synthetize the impact of fire in a brief and very complete review. According to different studies, the impacts on the quality of soil at the physical level can be summarized as follows:

Soil water repellency (SWR) (Figure 7) depends on the soil's pre-fire conditions (hydrophobic or hydrophilic soils) and on fire intensity (Mataix-Solera and Cerdà, 2009) and the plant fuel quantity and type (Arcenegui et al., 2007). Low-intensity fires can destroy part of the soil's organic matter (SOM) and the accumulation of ash after ignition can lead to an increase in of SWR. After high-intensity fires, hydrophobic compounds may be destroyed, thereby reducing SWR (Mataix-Solera and Guerrero, 2007). In some cases, there is an increase in hydrophobicity as ash clogs soil pores increasing the soil's hydrophobic properties (Pereira et al., 2015b) and reducing the soil's infiltration capacity and soil permeability (Imeson et al., 1992). The ash produced in low-intensity fires is more hydrophobic than that produced at high intensity (Bodí et al., 2011), although this is no guarantee that such fires increase the general hydrophobicity of the soil, given that in high- intensity fires the ash layer tends to be greater (Pereira et al., 2015). Likewise, plant species condition the hydrophobicity of each area with pine forest soils being more hydrophobic than kermes oak forest soils (Mataix-Solera et al., 2007). In the long term, hydrophobicity disappears due to factors of erosion and vegetation cover (Hubbert and Oriol, 2005). The increase in hydrophobicity can limit the increase in aggregate stability (AS) due to the creation of hydrophobic compounds with heating (Mataix-Solera and Doerr, 2004). AS is strongly related to a soil's infiltration capacity and the dynamics of AS after a fire is difficult to evaluate, being much more closely related to fire intensity (Mataix-Solera et al., 2007). At the onset of ignition in a severe fire, the SOM is destroyed (Oades, 1993). In some cases, an increase in AS has been recorded as a result of thermal fusion in soils with high quantities of iron and aluminum (Giovannini et al., 1988), but this increase serves no purpose in soil dynamics (Guerrero et al., 2001). The increase in AS can be produced by the generation of hydrophobic compounds during the fire that protect the aggregate from rainfall (Mataix-Solera and Doerr, 2004). The destruction of SOM during the fire (due to its high intensity) has been reported by many authors as the main factor reducing a soil's AS (Úbeda et al., 1990; DeBano et al., 1998; Badía and Martí, 2003). Recurrent wildfires can reduce a soil's infiltration capacity, thereby increasing SWR (Vallejo and Alloza, 1998; Caon et al., 2014). Mataix-Solera et al. (2013) concluded that high values of SWR are often accompanied by lower infiltration rates.

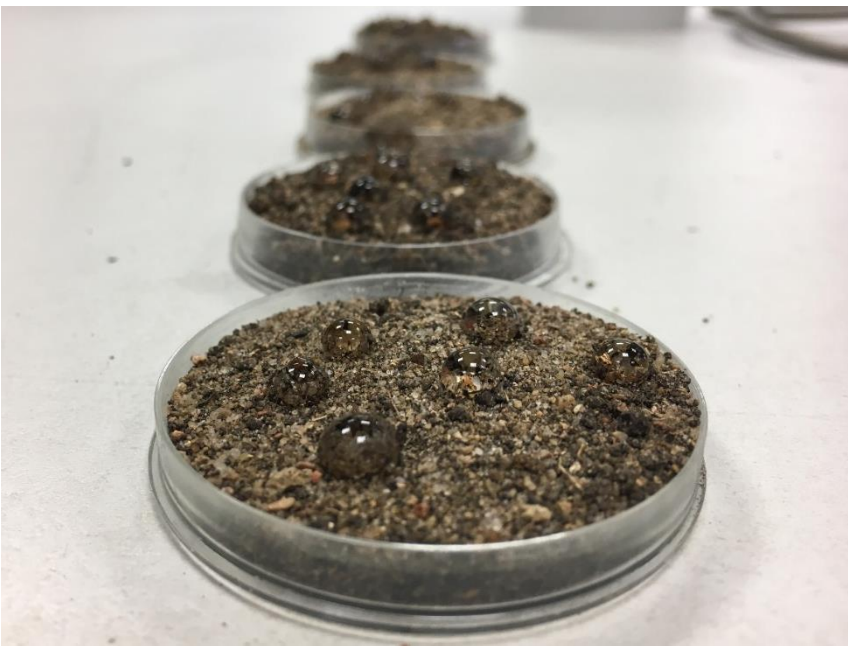

Figure 7. Analyzing soil water repellency with the water drop penetration time (WDPT) test under laboratory conditions. Picture: Marcos Francos.

The impacts on a soil's chemical properties can be described as follows:

Each type of fire, in combination with the specific characteristics of the ecosystem, impacts differently on a soil's organic matter and its chemical composition (Certini et al., 2011). Gonzalez-Pérez et al. (2004) identified the following as being the main effects of fire on SOM: a) removal of external oxygen producing materials of low solubility, b) reduction of alkyl compounds, c) aromatization of sugars and lipids, d) creation of heterocyclic N-compound, e) condensation of humic substances and f) production of black carbon that is almost unalterable. SOM consumption begins at 200 ºC and is complete at 460 ºC (Giovannini et al., 1988). Basically, the amount and quality of the SOM remaining after a fire depends on the intensity of the fire (de la Rosa et al., 2014), the extent of its spread, the amount of fuel and its combustibility and the humidity of the soil (Almendros et al., 2012). In high-intensity fires, SOM is reduced immediately, although in the long term, it usually exceeds pre-fire levels (Certini, 2005). Low- intensity fires can increase the amount of SOM due to lower combustion temperatures and the addition of partially carbonated plant material from the aerial parts of the plants that suffer from post-fire stress (Úbeda et al., 2005; de la Rosa et al., 2014; Pereira et al., 2015b). SOM can be reduced significantly in burned soils by heating (Badía-Villas et al., 2014; Ershad et al., 2013). This reduction persists from the short to the long term depending on the fire severity and the composition of the ash derived from ignition (Francos et al., 2018b). This SOM increases in the first post-fire period due to the incorporation of pyrolytic material into the soil and the incorporation of dead plants and microbial tissues (DeLuca and Zouhar, 2000). The quality of SOM changes notably. The most recalcitrant fraction increases. This occurs when fresh waste is burned and new aromatic compounds and highly polymerized compounds are formed. The charred material, produced by incomplete combustion, may reside for several years on the soil (Certini, 2005)

High fire temperatures and, therefore, high-severity fires contribute to the creation of inorganic compounds and carbonates (Francos et al., 2016a), commonly equated and labelled as inorganic carbon (IC). The ash produced after a severe wildfire is mainly white in color and rich in inorganic compounds (Goforth et al., 2005). Ash of this kind is easily eroded and wind transported (Úbeda et al., 2009) and its incorporation into the soil depends on post-fire meteorological conditions (explained in greater detail in Section 1.4) (Pereira et al., 2013; Francos et al., 2016a). The increase in IC after a fire can occur as a result of the volatilization of organic carbon during the wildfire (Pereira et al., 2014). The production of white ash is confined to the high temperatures reached during heating (Certini, 2005) and this depends mainly on the pre-fire forest fuel conditions (Francos et al., 2018c).

Soil total nitrogen (TN), an essential element for plants (Caon et al., 2014), decreases after fire (Caldwell et al., 2002). Losses occur in organic nitrogen while inorganic nitrogen increases (Giovannini et al., 1988; Mataix-Solera et al., 2009). The losses in organic N are due to volatilization after the fire (Fisher et al., 2000) and mineralization to ammonium, which is easily assimilated by plants. The ammonium is absorbed in the negative charges of the soil's organic particles and minerals, although over time they are transformed into nitrates, which are quickly released. Soil TN losses after severe fire can be attributed to leaching, soil erosion, runoff and volatilization (Marion et al., 1991; Ice et al., 2004) and TN levels fall significantly with fire intensity in burned areas (Ershad et al., 2013). The element is easily volatilized (Mataix-Solera and Cerdà, 2009). Despite this, low-severity fires often do not affect soil TN content (Santin and Doerr, 2016). High-severity fires increase inorganic nitrogen and reduce organic nitrogen (Raison, 1979). Although changes in TN are rarely dramatic, reductions in organic nitrogen affect plant growth (Mataix-Solera and

Guerrero, 2007). This might be attributed to the absorption of nutrients by plants and nitrification by inorganic nitrogen (decreasing over time), which reduces the amount of TN and the nitrogen available to plants (Durán-Humia et al., 2008). The C/N ratio also usually falls, probably as a result of the formation of recalcitrant nitrogenous heterocyclic forms (Certini, 2005). Other authors, including Mataix-Solera et al. (2009), report a rise in the C/N ratio with increasing fire temperatures. In those instances when ash is effectively incorporated into the soil, there is a reduction in the C/N ratio compared with that of unburned soils as a result of the mineralization of the SOM (Volkova et al., 2014).

A short-term increase in pH is reported following the combustion of OM during a fire and the release of soluble cations in the soil (Knoepp et al., 2005). The accumulation of ash and the solubility of the compounds present lead to an increase in pH, which varies according to the amount and the degree of combustion recorded (Certini, 2005; Úbeda et al., 2005; Mataix-Solera et al., 2009; Pereira et al., 2011). Marcos et al. (2007) observed marked increases in pH with rising temperature. However, pH does not increase at the same rate in all cases. In calcareous soils, the increase is not so great due to the high buffer capacity of the soil (Mataix-Solera et al., 2009). Likewise, in low-intensity fires, pH might not rise so much because the ash presents a lower pH (Úbeda et al., 2009; Pereira et al., 2015b). Severe wildfires produce an increment in soil pH as a result of the formation of oxides, and because of the carbonates present in ash (Ulery et al., 1993; Ershad et al., 2013) and the destruction of organic compounds (Mataix-Solera and Guerrero, 2007). Increases in pH can favor microorganism activity thereby increasing organic matter decomposition, but, in some cases, the variation in pH produces plant nutrition issues that hinder the assimilation of some nutrient compounds by the soil.

Typically, electrical conductivity (EC) increases after fire as a result of the incorporation and solubilization of ash (DeBano et al., 1977; Badía and Martí, 2003). DeByle (1976) and Ershad et al. (2013) observed increases in EC after fire due to the ignition of litter and the release of large amounts of base cations into the soil. Naidu and Srivasuki (1994) noted that the ephemeral increase in EC is produced by the release of inorganic ions from the combustion of OM. The concentration of soluble salts, measured by electrical conductivity, increases due to the solubilization of compounds from ash (Mataix-Solera et al., 2009). This increase depends on the intensity of the fire and the consequent mineralization of organic compounds (Certini, 2005; Pereira et al., 2011). In a low-intensity fire, changes in pH and EC are not significant (Marcos et al., 1998).

Ash production also produces an increase in released cations (Pereira et al., 2011). The ease with which these elements are solubilized is critical to vegetation, although the type and amount of nutrients released depend heavily on the severity and intensity of the fire (Úbeda et al., 2009). This increase in cations and their composition are closely related to the degree of mineralization and oxidation of organic compounds (black carbon) during the fire (Lima et al., 2002). At low intensity, the pH of the ash is low, facilitating the solubility of heavy metals. At moderate intensity, ash pH is 7-8 and most cations released are basic: Ca^{2+} and Mg^{2+} (Pereira et al., 2015b). Ash originating from burnt wood also provides large amounts of Ca^{2+}, Na^{+}, Mg^{2+} and K^{+} (Etiegni and Campbell, 1991; Demeyer et al., 2001; Mandre et al., 2004). A reduction in cation exchange capacity can be achieved by limiting reducing the amount of organic matter. Base saturation increases because of the release of basic cations on the combustion of organic matter (Certini, 2005). Soil extractable calcium (Ca), extractable magnesium (Mg) and extractable sodium (Na) increase ephemerally with fire temperatures between 250 and 500 ºC; higher temperatures produce a fall in levels of the extractable elements (Badía and Martí, 2003; Giovannini, 2012). These post-fire increases in extractable Ca, Mg, Na and K have been reported by several authors (Grove et al., 1986; Raison et al., 1990; Soto and Diaz-Fierros, 1993) and are more marked in fires of greater severity (Grove et al., 1980). Soil can both increase and decrease the cation exchange capacity (CEC), depending on fire severity and, consequently, on the consumption of organic matter. In high-severity fires organic matter is destroyed and there is a fall in the CEC due to erosion and lixiviation (Mataix-Solera and Guerrero, 2007). In low-severity fires, organic matter is not totally destroyed and the absorbent complex retains these cations (Alcañiz et al., 2018). The increase in soil nutrient levels depends on the burned tree species, soil properties and leaching processes (Certini, 2005). The increases in extractable Ca, Mg and K after fire have a duration that can rangs from months (Adams and Boyle, 1980) to years (Simard et al., 1991).

In general, there is an increase in available phosphorous (P) after fire due to the incorporation into the soil of the ash produced during the combustion of vegetation (Raison, 1979). Post-fire available P can result from the mineralization of organic P by fire (Cade-Menun et al., 2000). Soil extractable P losses by volatilization or leaching are very small (Certini, 2005). Soto et al. (1991) and Saá et al. (1998) observed decreases in extractable soil P due to post-fire erosion that prevents the incorporation of ashes into the soil. Available P also increases readily due to the accumulation of ash and its high solubility and mobility (Khanna et al., 1994). The higher the intensity of the fire, the greater the available phosphorus input (Pereira et al., 2015b). In the cases where ash was incorporated into the soil (having avoided erosion and the mineralization of P from organic forms), there was an increase in soil extractable P (Mataix-Solera, 1999). Badía-Villas et al. (2014) registered a short-term increase in extractable P due to the dissolution of P from ash beds and the mineralization of organic phosphorous due to heating. In cases when the ash is transported and not incorporated into the soil, there are no significant changes between burned and unburned soils (Johnson et al., 2007).

Soil microbiology is essential for the functioning of a soil and it is highly sensitive to high-severity wildfires (Pereira et al., 2018b). Soil microbial properties can be directly affected by fire, by heating, and, indirectly, by the modification of soil conditions that impact soil microbiology (for details see the review undertaken by Mataix-Solera et al. (2007). The impact of fire on a soil's biological properties are outlined in the paragraph below.

Fire can increase basal soil respiration (BSR) due to the solubilization of organic compounds by heat (Pietikäinen and Fritze, 1993; Fernández et al., 1997). After this ephemeral increase, BSR falls (Bisset and Parkinson, 1980; Almendros-Martín et al., 1990), at least for a few months or years (Hernández et al., 1997; Mataix-Solera et al., 2006). Soil biomass carbon (C_{mic}) falls as a result of the impact of fire, even in low-intensity episodes (Prieto-Fernández et al., 1998; Santín and Doerr, 2016). The fungal population is one of the microorganisms most affected by fire and represents one of the main contributors to the microbial biomass. The thermal shock produced by fire reduces the number of fungal propagules by the scattering of spores (Mataix-Solera et al., 2002), resulting in a decrease in C_{mic} (Díaz-Raviña et al., 1992; Hernández et al., 1997). When temperatures reach 400 ºC C_{mic} levels become undetectable (Guerrero et al., 2005).

4. Impact of post-fire weather conditions on soil

The impact of fire may be direct, attributable to heating, or indirect, attributable to the ash-bed effect, plant recovery, post-fire weather patterns, topography or post-fire management (Pereira et al., 2018b). Environmental factors are critical to the evolution of soil properties over time. The type of soil, the severity and frequency of fires and post-fire climatic conditions all influence the short-, medium- and long-term alteration of soil properties (Certini, 2005). After a high-severity fire a site's topography is essential in the evolution of the dynamics of the soil properties (Pereira et al., 2018b). Burned sites in highly sloping areas are more vulnerable to degradation and erosion as are those on south-facing slopes (Jones et al., 2014) (Figure 8) Other factors, such as an abundance of rock fragments covering the soil, can reduce soil erosion, while soil micro-depressions allow ash to be retained facilitating the incorporation of ash into the soil profile.

Figure 8. Slope erosion process and soil losses after wildfire. Ódena (Barcelona). Picture: Marcos Francos

Post-fire weather studies are essential for understanding soil property dynamics over time (Pereira et al., 2018b). Extreme wind after a wildfire can modify the distribution of ash, increase soil compaction and soil erosion, due to the elimination of tree cover by fire. Extreme precipitation episodes can modify the landscape and lead to catastrophic environmental hazards (Pereira et al., 2010) especially in fragile environments after a severe wildfire. Post-fire rainfall promotes the incorporation of ash into the soil in areas of low slope. In areas of high slope, and in those instances in which the intensity of precipitation is high, ash is more likely to be eliminated by runoff and erosion, thus, preventing the incorporation of ash into the soil and impoverishing and degrading the soil (Francos et al., 2016a). Low-intensity rainfall has less capacity to transport ash and sediment facilitating the incorporation of ash into the soil and, consequently, reducing SWR and increasing soil hydraulic conductivity. Rainfall intensity and duration must be taken into account after a wildfire, especially in high-slope areas to account for soil erosion processes and soil losses (Staley et al., 2016). Slope and rainfall intensity-duration are essential for facilitating (in low-slope areas) or preventing (in high-slope areas) the incorporation of ash (and the concomitant increase in available elements) into the soil, especially after the first rainfall event following a wildfire. In recent years, as various authors report (e.g. Bravo et al., 2010; Francos et al., 2016a; Jimenez-Ruano et al., 2017; Rodrigues et al., 2018 among others), the wildfire regime has been modified and torrential rainfall events are likely to occur immediately on the heels of a forest fire in the summer-fall transition, with devastating effects for on slope soils. Vegetation is essential to facilitate regrowth and to protect the soil before these torrential precipitation events (Outeiro et al., 2007). Moreover, if the storm event occurs before plant regrowth, the soil seedbank can be damaged, further conditioning revegetation (Moreno et al., 2011). The impact of rain drops on bare soil after a severe wildfire can prevent ash being incorporated into the soil, destroy soil aggregates (Outeiro et al., 2007) and reduction the infiltration rate (Ela et al., 1992). In instances of high intensity rainfall on slopes, nutrients are dragged away in the overland flow (Úbeda et al., 1998; Úbeda and Sala, 2001). Despite the absence of erosion, rainfall can modify soil nutrient availability after fire (Outeiro et al., 2008), leading to soil degradation and soil scars (Figure 9).

Figure 9. Soil scars after wildfire, torrential rainfall and forest management. Picture: Marcos Francos

5. Effects of forest management on soils

Forest management can affect long-term soil properties (Francos et al., 2018d) and vegetation dynamics (Strom and Fulé, 2007). Pre-wildfire forest management practices have yet to be analyzed in depth but they can be a major determinant of the impact of a wildfire and of the resilience of forest structures (Stevens-Rumann et al., 2013; Shive et al., 2014). Management practices of this type can modify both forest fuel and wildfire severity and so alter the effects of fire on soils (Cram et al., 2015). Francos et al. (2018c) stressed that forest areas exposed to management practices a few months before a wildfire avoid the accumulation of dead plant fuel. This contrasts with forest areas managed a few years before a fire episode and the considerable detrimental effects recorded on soil properties. The alterations noted by the authors after wildfire affected soil TN, SOM, IC, pH, EC and extractable Ca and Mg. The reduction in SOM can affect other soil properties (e.g., AS and SWR among others) and so forest structures that are liable to high-severity burning should be avoided (Varela et al., 2015). Rapp (2007), similarly, concluded that wildfire severity is dependent on prior fuel treatment and the effectiveness of these measures. He reports, in common with Francos et al. (2018c), that treatments undertaken less than 10 years prior to a fire were effective in reducing fire severity, resulting in fewer fire impacts and soil modifications. Finally, Rapp (2007) concludes that the treatments designed to prevent forest fires were no longer effective after a lapse of 10 years, as he found no differences with untreated areas.

Post-fire management can modify soil properties in the short (Ginzburg and Steinberger, 2012), medium (Wagenbrenner et al., 2015) and long term (Francos et al., 2018d; Johnson et al., 2005). Post-fire treatments employed in recent years include the application of mulch, pine needles and organic waste in burned forest areas, the strategic placement of burned trunks to stop erosion and salvage logging, consisting in the cutting and extraction of burned wood using machinery. The aims of these practices are to improve the quality of soil properties, avoid erosion and promote plant regrowth (Guerrero et al., 2000, 2003; Mataix-Solera et al., 2001; Olsen, 2016). Among the soil modifications recorded, we find most notably soil erosion (Robichaud et al., 2011) and compaction (Rab, 1996; McIver and Starr, 2000). Different post-fire treatments seek to improve soil properties and to restore them to their pre-fire values a) as rapidly as possible to ease post-fire plant recovery; and b) to ensure suitable long-term forest management. Marques and Mora (1998) did not observe any changes as regards erosion and vegetation in clear-cut areas after fire; despite this, the authors recommend that care be taken in implementing wood removal practices to minimize erosion hazards and soil losses. Clear-cutting in this last study involved using the burned branches to construct dikes, a practice that prevents soil losses due to runoff, but its use in large areas is limited. Gómez-Rey et al. (2013) evaluated the short-term impact of grass seeding and straw mulching on the chemical properties of soils and concluded that they limit erosion and increase TN, Ca, Mg and K. Bai et al. (2014)

compared the effects of mulching and hydroseeding on slopes affected by fire and concluded the latter to be more effective in improving fertility in disturbed soils. Mulching produced increases in carbon and nitrogen levels compared to those recorded after hydroseeding, because the latter proved less effective in retaining water, thus increasing SOM (see also Eck et al., 2010). Ginzburg and Steinberger (2012) reported the short-term impact of salvage logging on microbial soil properties and, thus, recommend postponing post-fire practices at least until one year after a wildfire. Francos et al. (2018a) described the benefits of extracting burned wood manually. This post-fire management practice can be useful in small, clearly defined areas (forest slopes or areas surrounding rural hotels) in order not to affect the landscape. Moreover, it appears to protect soil properties and avoids the damage associated with salvage logging. Differences between this type of management, no treatment and cut-and-leave techniques have been recorded in soil AS, TN, SOM, IC and EC. Spanos et al. (2005), in an evaluation of the use of mules to extract burned logs, found that such practices reduced damage to soil properties compared to the use of machinery (specifically skidders). Specifically, short-term differences were recorded in soil pH and SOM. Wagenbrenner et al. (2014) observed higher degrees of compaction, lower SWR and lower plant regrowth in skidder-managed areas than in non-treated areas after fire, the consequences of these dynamics being greater increases in erosion rates. Manual or animal wood removal have lower impacts on soil properties (Wagenbrenner et al., 2016) and seem to represent a good alternative in small areas, on critical sites and in exceptional cases. Jennings et al. (2012) compared the respective impacts of salvage logging in a burned area and in an area not treated mechanically. The authors recorded changes in available P, nitrogen and BSR attributable to the use of heavy machinery and the consequent soil compaction when undertaking the cutting and extracting of burned wood. Salvage logging using heavy machinery reduced short-term plant cover and soil quality after wildfire affecting SOM, TN, available P, C_{mic} and BSR. The extraction of burned wood, especially when using heavy machinery after wildfire, resulted in damage to the seedbank and soil degradation. In response, Garcia-Orenes et al. (2017) suggested postponing treatments until the medium-term. Serrasolses and Vallejo (1999) observed short-term increments in soil fertility in areas in which logs had been cut after a fire but not extracted.

Long-term soil management after wildfire can produce soil properties in managed areas that are more similar to those of a non-fire affected area than an unmanaged area (Francos et al. 2018d). Any differences are a consequence of high vegetation density and competition for water reserves and sunlight, which cause soil stress and inhibit the long-term recovery of pre-fire soil values. Fernandes et al. (2013) noted that forest management should aim to reduce the impact of fire on soil. The most important changes reported by Francos et al. (2018d) are as follows: unmanaged vs managed burned soils present long-term differences in soil TN, IC, SOM, pH, K, Al, Mn, Fe and the C/N ratio after wildfire and long-term post-fire management due to the high plant density (Figure 10). The studies noted the need to manage areas so as to improve soil quality and reduce the risk of wildfire. Management practices of this kind should be taken into account to avoid the recurrence of severe wildfires and to manage burned forest areas in the medium term (Francos et al., 2016b).

Figure 10. High vegetal density 30 years after wildfire without any management. Ódena, Barcelona. Picture: Marcos Francos.

INTRODUCCIÓN GENERAL

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1. Incendios forestales y territorio

El fuego ha estado presente como un fenómeno natural mucho antes de que el hombre existiera, a menudo causado por rayos durante tormentas eléctricas o por erupciones volcánicas (Naveh, 1975). De hecho, los incendios forestales durante el Plioceno fueron uno de los factores que contribuyeron a la formación como se hoy en día se conoce, junto con el clima, de la vegetación mediterránea (Bond y Keeley, 2005). Durante los períodos del Carbonífero, Jurásico y Cretácico y en el Cuaternario con la aparición del hombre, se produjo una gran cantidad de incendios forestales en la Tierra (Rodríguez-Trejo, 2012). Este elemento natural se ha utilizado como una herramienta para llevar a cabo actividades o acciones en el territorio, como ahuyentar a los depredadores, calentar el hogar, controlar plagas, cocinar o quemar espacios para hacerlos habitables (Figura 1). Las primeras muestras del uso del fuego por los humanos como herramienta fueron evidentes hace 400 000 años (Santin y Doerr, 2016). Durante el siglo XX, debido al abandono de las áreas rurales, la presión humana en ciertas áreas y el cambio en los intereses económicos del manejo forestal, la frecuencia e intensidad de los incendios forestales aumentaron (se explica en la sección 1.2) (Vélez, 2000) y esto fue la razón de percibir el fuego durante gran parte del siglo XX como algo dañino. Por lo tanto, podemos decir que el fuego es un elemento que ha acompañado la existencia de seres humanos a lo largo de la historia (Pausas y Keeley, 2009).

Figura 1. Roca con la inscripción "Nuestros antecesores fueron capaces de usar y controlar el fuego al menos desde hace 1 millón de años en la cuna de la humanidad" en las cuevas de Maropeng y Sterkfontein, Sudáfrica. Imagen: Marcos Francos.

Este pensamiento negativo del fuego dio lugar desde las últimas tres décadas del siglo XX lo que se llama "supresión del fuego". Esta concepción de la extinción llevó a muchas áreas de los Estados Unidos, Australia y Europa a registrar, debido a la paradoja de la extinción, los incendios más grandes en la historia reciente. La supresión de incendios involucra las actividades llevadas a cabo para controlar y extinguir un incendio forestal (Pearce et al., 2000). En estas actividades, encontramos acciones como eliminar los combustibles, reducir la temperatura de los combustibles que se queman y evitar que el oxígeno llegue al área de combustión por medio del ahogamiento.

La paradoja del fuego o "fire paradox", como suele llamarse en muchos foros, indica que cada vez se es más profesional en la extinción y esto con el tiempo hace que se alcancen niveles de cargas de combustible propicios para generar grandes incendios forestales. Hoy en día muchos autores, así como administradores y gerentes, han asumido que el fuego es algo natural y que ha tenido y continúa teniendo un papel ecológico, especialmente en áreas geográficas con clima mediterráneo (Plana, 2004; Mataix-Solera y Cerdà, 2009). Todo esto ha llevado a la concepción en el siglo XXI del fuego como una herramienta útil y positiva e incluso necesaria. Sin embargo, los incendios forestales no siempre son catastróficos. En algunos casos, solo queman levemente las partes aéreas de la vegetación, produciendo una fertilización del suelo debido a la incorporación de las cenizas. Algunos autores han visto que los ciclos naturales de los incendios de baja severidad ayudan a mejorar la productividad de los ecosistemas mediante el aporte de materia orgánica por las cenizas, y la posterior liberación de nutrientes que causa la combustión (Cromer, 1967).

Aunque hoy en día, por ejemplo, en Cataluña (explicado de manera detallada en la sección 1.2), se está trabajando en esta dirección: 1) asumiendo la existencia de incendios recurrentes en nuestros bosques, 2) que el fuego está modelando el paisaje y 3) que el cambio climático puede aumentar esta tendencia ascendente del número de incendios. Los bosques aún tienen la carga de combustible restante y el riesgo de gran incendio forestal que comenzó en el siglo XX con el abandono rural (Plana, 2011). En las últimas décadas, los incendios forestales en todo el mundo han evolucionado hasta convertirse en lo que se conoce como incendios de alta intensidad o incluso mega incendios (Ferreira-Leite et al., 2013). Este tipo de incendios forestales plantean enormes problemas en términos de control y extinción y son especialmente destructivos.

Las causas que se han podido identificar en estos incendios forestales son:

1) Abandono rural, que tiene como resultado la acumulación de combustible y la consiguiente homogeneidad del paisaje (Benayas et al., 2007) (Figura 2).

2) La extinción de pequeños incendios y la efectividad de las técnicas de extinción de incendios modernos, han conducido a la erradicación de la función ecológica del fuego (la llamada "paradoja del fuego") indica

que cuanto mejores sean en la extinción de incendios, más preparado está el paisaje para quemar a altas intensidades produciendo altas severidades (Strauss et al, 1989).

3) Cambios en el uso del suelo (Pausas y Keeley, 2009).

4) La sustitución de especies autóctonas por especies de rápido crecimiento con mayor productividad económica.

5) No podemos ignorar el enfoque global, que puede ser responsable de cambiar los regímenes de incendios forestales en áreas concretas (Summers et al., 2011). Este fenómeno es más evidente en el norte del planeta que en el sur, pero las regiones boreales y australes, son especialmente vulnerable ya que los monocultivos son muy importantes en grandes extensiones de territorio (Tsibart et al., 2015).

Figura 2. Abandono rural en Avinyó (Barcelona). Imagen: Marcos Francos.

Es decir, el apoyo de estas medidas de manejo forestal es enormemente limitado. El mayor riesgo de propagación es el resultado de a) el abandono de la actividad agrícola y b) la despoblación rural. En el primer caso, este hecho permite aumentar el combustible horizontal y, en la medida en que en Cataluña debamos volver al principio del primer milenio para encontrar un área forestal como en la actualidad (Peix, 1999). Por otro lado, la despoblación rural conlleva la pérdida de la prevención pasiva o indirecta, es decir, la detección y extinción inmediata, el conocimiento del territorio y el mantenimiento de los caminos que ofrecen los agricultores y habitantes del medio rural con su actividad diaria. Las razones del abandono rural son, principalmente; 1) la baja rentabilidad económica de las actividades agrícolas (Martínez-Alier y Roca-Jusment, 2000); 2) la falta de servicios e infraestructuras en las zonas rurales que concentran las oportunidades profesionales en los centros urbanos; y 3) aspectos culturales, como el predominio de los valores de la sociedad de ocio y consumo que apenas cumplen los requisitos de la actividad agrícola, o la falta de reconocimiento social de la figura del agricultor (Plana et al., 2001).

En base a los tipos de incendios se pueden agrupar en 3 grandes categorías (Carballas, 2007):

- Los incendios de copa que queman árboles en toda su longitud hasta la parte más alta.

- Los incendios superficiales que solo queman las partes inferiores y la superficie de la vegetación existente además de los restos vegetales que se encuentran sobre el suelo.

- Los incendios sub-superticiales (a veces llamados incendios subterráneos) ocurren en acumulaciones profundas de humus, turba y vegetación muerta similar que se secan lo suficiente como para quemar.

Con respecto al comportamiento de los incendios, se puede determinar que hay tres factores que los caracterizan:

- El combustible: Ligero (hierbas, hojas y ramas pequeñas), pesado (troncos, ramas y raíces) y verde (vegetación viva) (Fernandes y Botelho, 2003).

- La topografía: puede ayudar a predecir y explicar el comportamiento de un fuego de acuerdo con la orientación, la rugosidad y la pendiente.

- Condiciones climáticas: Especialmente temperatura, humedad y viento.

Como se apuntaba anteriormente, algunas áreas del mundo son más propensas a los incendios forestales, debido a su tipo de clima, pero eso no significa que los incendios sean exclusivos de un solo tipo de clima. Los grandes incendios forestales o mega incendios pueden ocurrir en climas mediterráneos, boreales, australes y tropicales, y en zonas de diferente vegetación (Archibald et al., 2013). Los bosques tropicales, la sabana africana, la zona sur de Chile y la tundra siberiana se queman cada año y se destruyen miles de hectáreas de bosques (Pereira et al., 2015a). Algunos estudios sobre los mega incendios en todo el mundo fueron realizados por Ferreira-Leite et al. (2015a), Jones et al. (2016) y Pyne (2016). Como se explicó anteriormente, los incendios forestales no solo forman parte del ecosistema mediterráneo, éstos son un fenómeno global (Úbeda y Francos, 2018). Muchos de dichos estudios se han realizado en países mediterráneos como Portugal, España, Grecia, Argelia e Israel (Inbar et al., 1998; Mataix-Solera et al., 2002; Wittenberg et al., 2007; Úbeda et al., 2009; Ferrera-Leite et al., 2015b) y en países con ecosistemas mediterráneos como Chile (Castillo-Soto, 2015; Úbeda y Sarricolea, 2016). Otros países como Rusia (Achard et al., 2008; Tsibart et al., 2015), Lituania (Pereira et al., 2013; Pereira et al., 2015a), Finlandia (Vanha-Majamaa et al., 2007) Estados Unidos de América (Marlon et al., 2003; Collins y Stephens, 2007), Indonesia (Abram et al., 2003), Hungría (Deak et al., 2014); y ambientes como Pastizales (Pereira et al., 2018a), estepa (Maksimova y Abakomov, 2013), selva tropical (Pew et al., 2001), tundra (Hudspith et al., 2017) y taiga / boreal (Viereck, 1973; Kharuk et al., 2010; Valendik et al., 2014; Pereira et al., 2016) han experimentado el impacto de un aumento en el número de incendios forestales en las últimas décadas.

Existen diferentes herramientas de manejo forestal utilizadas para evitar incendios forestales y administrar el combustible forestal (Marino et al., 2014) (Figura 3):

a) Planificación forestal

b) Quemas prescritas

c) Pastoreo controlado

d) Clareo mecánico

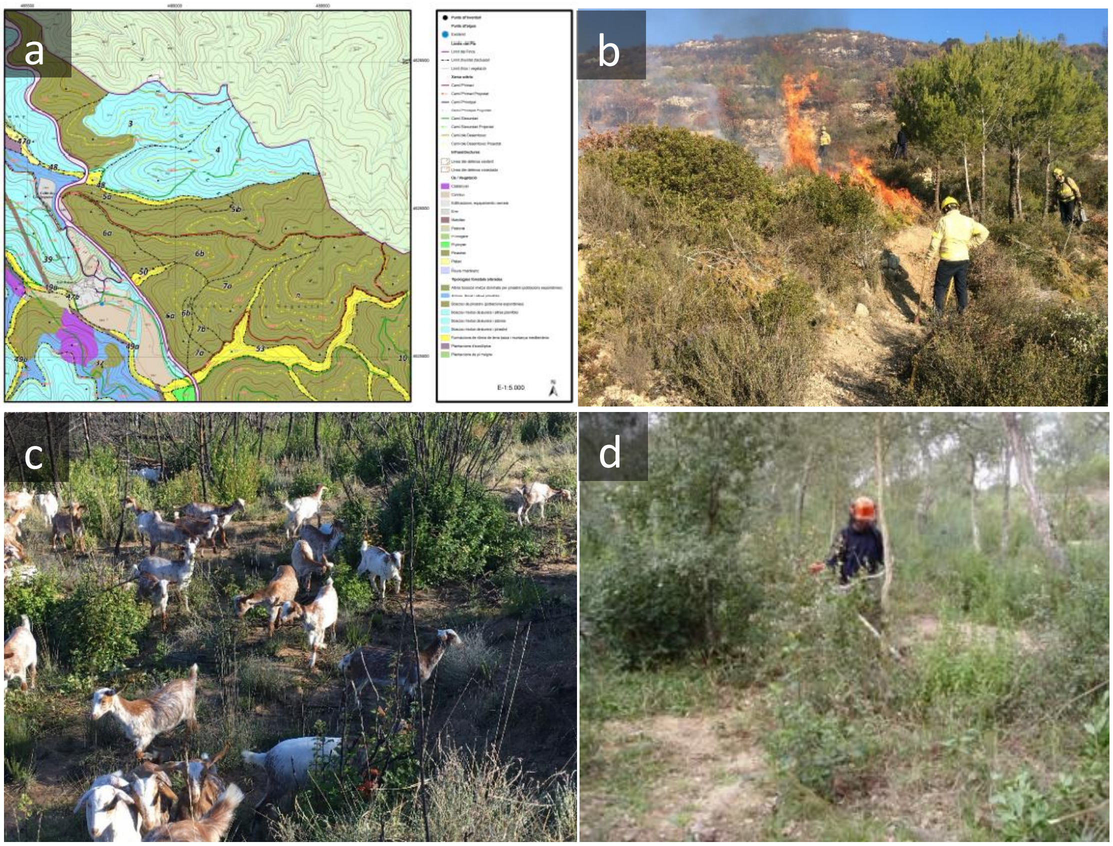

Figura 3. a) Plan Técnico de Gestión y Mejora Forestal (PTGyMF) en Mas Bassets (Girona). Imagen: Xavier Úbeda. b) Quema prescrita en Olesa (Barcelona). Imagen: Marcos Francos. c) Pastoreo controlado para gestionar la carga de combustible vegetal en Ódena (Barcelona). Imagen: Marcos Francos. d) Clareo mecánico para gestionar la carga de combustible en las Gavarres (Tarragona). Imagen: Xavier Úbeda.

Por todo ello, los incendios forestales son tanto un problema ambiental, que ocasiona la pérdida y degradación de la biosfera, como un problema social importante, que ocasiona la pérdida de vidas humanas, la contaminación de los acuíferos, desprendimientos de tierras y la degradación de los cultivos. No debemos olvidar que, en muchas partes del mundo, los incendios forestales han dado forma al paisaje, que actúa como un agente de los ecosistemas y, como tal, hemos aprendido a vivir con este fenómeno (Belcher, 2013).

2. Incendios forestales en los ecosistemas Mediterráneos

El fuego es el principal destructor de las áreas forestales en los ecosistemas mediterráneos (Vélez, 2000). La característica más importante de este clima es la coincidencia de la estación seca con la cálida. En el verano, el Mediterráneo está bajo la influencia de los anticiclones subtropicales secos, que se retiran en invierno hacia las latitudes más bajas, dejando atrás el tormentoso Atlántico en latitudes templadas. Esta peculiaridad hace que los ecosistemas sean susceptibles a incendios forestales con vegetación seca en verano, lo que, junto con su naturaleza inflamable, favorece la aparición y expansión de los incendios (Arianoutsou et al., 1993). La temporada de lluvias que generalmente llega después de la estación seca puede erosionar grandes cantidades de suelo, un suelo con una mayor fragilidad si ha sufrido la combustión de su materia orgánica, así como el lavado de nutrientes (Mataix-Solera y Cerdà, 2009).

En España, en la década de 1960, la industrialización y el éxodo rural llevaron al abandono de las zonas montañosas y contribuyeron a un aumento de la cubierta forestal, que continúa hasta nuestros días. El cambio en los usos del suelo, la reforestación de bosques de pino y el crecimiento de nuevas áreas residenciales han contribuido a incrementar el riesgo de incendios forestales (Pausas et al., 2008). Las consecuencias han sido un aumento de los incendios forestales desde la década de 1970, cuando las condiciones climáticas comenzaron a ser claramente favorables. El fuego ocurre con mayor frecuencia y, aunque los ecosistemas mediterráneos pueden coexistir con él, este cambio de régimen puede causar efectos adversos (Bodí et al., 2012). En la transición económica de los años sesenta y setenta, el riesgo de que se iniciara un incendio forestal en las áreas rurales aumentó considerablemente debido a la acumulación progresiva de combustible que la población rural dejó de extraer como fuente de energía. Inconscientemente, estas personas estaban cuidando el bosque y protegiéndolo contra incendios. La consecuencia de los cambios en los usos ha sido un aumento en el combustible y la biomasa presentes en las montañas, y por lo tanto un aumento en el número de incendios (Mataix-Solera y Cerdà, 2009). Las décadas de los años 80 y 90 en España fueron cruciales en términos de incendios, ya que los grandes incendios forestales se produjeron como resultado de esta acumulación de combustible y también de las políticas de extinción de incendios. El año 1994 fue el peor en registros de incendios y, lo que es más importante, en hectáreas quemadas (437,000 ha en toda España y 80,000 ha solo en Cataluña) (ADCIF, 2002). Por este motivo, a nivel científico en esta década de los años 90 aumentaron los estudios sobre el efecto de los incendios forestales en las propiedades del suelo y la generación de erosión. En Cataluña, los grandes incendios de la Cataluña central en 1994 y 1998 fueron muy importantes en cuanto a su extensión. En esta comunidad autónoma, los incendios forestales son la principal causa de pérdida de masa forestal (Plana, 2004).

Numerosas publicaciones trataron el tema de los efectos a corto plazo en las propiedades del suelo después de los incendios forestales en España (Úbeda, 1998; Mataix-Solera, 1999). Algunos de ellos concluyen que después del incendio hay una pérdida de nutrientes del sistema debido a la combustión, ya que hay elementos que se volatilizan; pero también se sabe que después de un incendio se incorporan nuevos elementos altamente mineralizados gracias a la incorporación de las cenizas en el suelo (Alcañiz et al., 1996). Estos impactos de incendios forestales en los suelos se explican ampliamente en la sección 1.3. Esta discusión fue fundamental para introducir en las variables el término "intensidad" del fuego, (definido como la temperatura y el tiempo de residencia del mismo en un punto específico) y la consecuencia en el entorno de esta intensidad, la "severidad". La intensidad del fuego es la duración de la temperatura máxima registrada en un cierto punto, expresada en ºC / seg (Úbeda et al., 1998). Es la energía liberada en la combustión de materia orgánica en función de la temperatura (Keeley, 2009). La severidad del incendio es el resultado de la intensidad del incendio y la duración de la ignición (Keeley, 2009). Esta severidad describe la respuesta del ecosistema al fuego y se puede utilizar para analizar la respuesta del suelo, la flora y la fauna a este tipo de perturbación (Bento-Gonçalves et al., 2012).

Por lo tanto, las líneas de investigación actuales llevadas a cabo por investigadores y administradores pueden resumirse en dos: 1) Comprender el equilibrio entre los efectos negativos y positivos que puede tener un incendio forestal y 2) Estudiar todas las variables que intervienen para conocer el comportamiento del fuego. El logro de estos dos objetivos nos permitirá asimilar los incendios forestales como un elemento a tener en cuenta en la gestión forestal e incluso en la planificación del territorio (Piqué et al., 2011). Esta gestión forestal puede desempeñar un gran papel en la prevención de incendios (González y Pukkala, 2007; Corona et al., 2015).

En Cataluña, el manejo forestal principal es la silvicultura preventiva cuyo objetivo primordial es lograr estructuras forestales con masas de menor grado de combustibilidad, es decir, con una mayor resistencia a la propagación del fuego (Vélez, 2000) (Figura 4). La secuencia de tratamientos en silvicultura preventiva para reducir la vulnerabilidad de un bosque al fuego sería la siguiente, según Fernandes y Rigolot (2007):

- Reducción del combustible de superficie para limitar la intensidad potencial del fuego.

- Poda de combustible para reducir la probabilidad de desarrollo vertical del fuego.

- Realizar clareos para minimizar la probabilidad de transmisión de incendios entre bloques.

- Replicación controlada de especies que generan estructuras forestales de baja vulnerabilidad a los incendios.

Figura 4. Gestión forestal medio- o largo-plazo después de un incendio forestal es esencial para disminuir el riesgo de incendio. Ódena (Barcelona). Imagen: Xavier Úbeda.

Estas medidas o tratamientos pueden llevarse a cabo con ganado, con maquinaria o, utilizando el fuego como herramienta para prevenir incendios mediante quemas prescritas (Alcañiz et al., 2018). En las últimas dos décadas, la concepción social sobre los incendios y la falta de manejo del bosque está cambiando. Aunque, en mayor medida, esta fobia aún existe, una parte de la sociedad se está dando cuenta de que el fuego es un proceso natural en la evolución de los ecosistemas mediterráneos (Figura 5). Este cambio de paradigma, junto con el actual régimen de fuego, ha creado la necesidad no solo de prevenir y combatir, sino también de integrar el fuego en el territorio y en la planificación (Plana, 2011).

Figura 5. La educación ambiental es esencial para explicar el rol del fuego en el ecosistema Mediterráneo y las consecuencias de la gestión forestal. Las Gavarres (Girona). Imagen: Xavier Úbeda.

En Cataluña, este cambio se determinó después de los grandes incendios de 1994. Según Maza et al. (1995), los incendios forestales deben tenerse en cuenta al planificar y gestionar las decisiones sobre el territorio y también en el desarrollo urbano. A partir de estos incendios, durante los años 90, se amplíaron las estrategias de gestión del riesgo de incendios forestales: el uso del fuego se utiliza casi desde ese momento como herramienta de extinción y gestión del combustible forestal (Castellnou y Nebot, 2007) y la planificación de los usos del suelo se incorporaron a las políticas comunes de prevención y extinción de incendios (Plana, 2011). La integración del riesgo de incendios en la planificación forestal incluye tener en cuenta el riesgo de incendio en los tratamientos forestales de manejo forestal a nivel de finca (Piqué et al., 2011), así como la vulnerabilidad a la ignición y la propagación del fuego en las masas forestales; además de esto, se ha de tener en cuenta la matriz territorial y la perspectiva de la ecología del paisaje (González et al., 2006). De esta manera, todo lo relacionado con los usos ganaderos, la gestión forestal y la planificación urbana y territorial afectaría tanto el peligro de incendio como la vulnerabilidad del territorio (Plana, 2011). Esta planificación territorial tiene una relación directa con el aumento en los últimos años de la interfaz urbana de áreas silvestres y la desestacionalización de incendios forestales. Badia et al. (2002) reclaman la recuperación del mosaico del paisaje que caracterizó las áreas históricamente mediterráneas para oponerse a la creciente incidencia e impacto de los incendios forestales. Estos autores señalaron la importancia extrema de las gestiones de prevención contra incendios forestales y la necesidad de la ordenación forestal tradicional y alternativa, teniendo en cuenta factores importantes como el abandono agrícola, la despoblación rural y la falta de recursos. Los incendios forestales en estas áreas donde las perturbaciones humanas se llevaron a cabo muchos años antes de los incendios, pueden producir consecuencias catastróficas (Pausas et al., 2008). En las últimas décadas, muchos estudios observaron la ampliación del período de riesgo de incendios forestales, dejando de estar limitada a los meses de verano (Versini et al., 2013). Esta extensión en el período de incendios forestales produjo ocasionalmente la confluencia de incendios forestales y lluvias (Girardin et al., 2013). Dicha confluencia puede afectar fuertemente a las propiedades del suelo (explicado en profundidad en la sección 1.4) (Francos et al., 2016a). La finca en el bosque catalán, y generalmente en la cuenca mediterránea, se caracteriza por ser propiedades privadas y pequeñas (Cervera et al., 2015). Con el paso de los años fue más evidente que la privatización había fracasado y que los propietarios no preservaron el bosque tanto como cuando lo explotaban eonomicamente. A principios del siglo XX, los propietarios de las grandes propiedades (procedentes de las desamortizaciones de Mendizábal y Madoz en el siglo XIX) tuvieron problemas para gestionarlas y, en muchos casos, no aprovechan todas las hectáreas. A finales del siglo XX y principios del siglo XXI, las ventajas de la madera forestal se concentraron en menos bosques que en años anteriores y las áreas forestales continuaron aumentando. El abandono del uso múltiple e integrado del bosque, que mantuvo un mosaico de diferentes coberturas en las áreas forestales, provoca el crecimiento descontrolado de estas masas forestales densas y continuas, lo que reduce la diversidad de hábitats y especies al tiempo que facilita la existencia de grandes incendios forestales.

Algunos de los estudios presentados en esta Tesis Doctoral se han realizado en Ódena, provincia de Barcelona. Esta zona es un buen ejemplo de lo que está tratando de hacer el gobierno catalán en áreas forestales impusando medidas que conllevan a una gestión forestal sostenible. En esta zona, uno de los factores más importantes porque seleccionamos Ódena como área de estudio, desde 2014 hasta 2018 se está llevando a cabo el proyecto “Life Montserrat” con el apoyo económico de la Red Natura 2000 y la Unión Europea. Por lo tanto, con todas las amenazas que sufre el territorio, la necesidad de proteger, conservar y administrar la región se crea a través de un proyecto importante, el Life Montserrat, que busca, como uno de sus principales resultados, aumentar la resiliencia y estabilidad del bosque frente a grandes incendios forestales a través de un conjunto de medidas de manejo de ecosistemas, la contribución a la conservación y mejora de la biodiversidad en el área de Montserrat y la creación de un mosaico de matorrales, prados naturales y bosques que mejorarán la conectividad entre los espacios Natura 2000, además del desarrollo socioeconómico del territorio. El proyecto LIFE está siendo promovido por la Unión Europea para financiar proyectos de conservación ambiental y para crear legislación comunitaria en esta área. En el caso del LIFE

Montserrat, el presupuesto es de 3,5 millones de euros, financiado por la Unión Europea en un 50%, mientras que el resto se divide en 6 socios: la Diputación de Barcelona, el Departamento de Agricultura, Ganadería, Pesca y Alimentación, Bomberos de la Generalitat de Catalunya, Junta de la montaña de Montserrat, Fundación Cataluña-La Pedrera y la Asociación de Entornos Forestales.

En cuanto a las prevenciones activas de incendios forestales, se centran en la gestión de combustibles vegetales, aumentar la eficiencia de los cortafuegos y la gestión forestal de las interfaces urbano-forestales. Las prevenciones pasivas se centran en la promoción de la rentabilidad forestal, la creación de vínculos entre lo urbano y lo forestal, la promoción de asociaciones en áreas forestales y la contemplación de la gestión forestal en la planificación territorial. En términos sociales, las pautas prioritarias son la regulación del uso del fuego en las prácticas agrícolas, incrementando la participación de la sociedad rural y la educación ambiental (Romero y Senra, 2006). La prevención de incendios y el manejo forestal antes de que se produzcan incendios forestales pueden disminuir la severidad del incendio y evitar la esterilización del suelo. Mataix-Solera y Cerdà (2009) señalaron la necesidad de incorporar la gestión forestal y la gestión territorial teniendo en cuenta los incendios forestales en los ecosistemas mediterráneos.

3. Impacto de los incendios en los suelos

El oxígeno, el combustible y el calor son los tres componentes básicos para que se genere el fuego, y todos ellos se encuentran en la superficie de la Tierra (Mataix-Solera y Cerdà, 2009). Por lo tanto, el fuego debe considerarse un factor natural de los ecosistemas terrestres (Naveh, 1975). Algunos autores como Certini (2005) consideran que el fuego es un factor formador del suelo debido a la gran influencia que tiene en éste. Los cambios en las propiedades del suelo dependen de la intensidad del fuego, las condiciones climáticas posteriores al incendio, el manejo forestal y el momento de la medición. Según la temperatura alcanzada, Neary et al. (1999) realizó una tabla con los efectos que produce en el suelo (ver Tabla 1).

Tabla 1. Cambios en el suelo producidos por la intensidad del fuego de acuerdo con la temperatura (ºC).

Temperatura (ºC)	Perturbación
48-54	Deshidratación y muerte de algunas raíces
70-90	Muerte de ciertas semillas
50-121	Muerte de los microorganismos del suelo
180-300	Combustión de 85% del horizonte orgánico
200-250	Incremento de la hidrofobicidad
200-315	Completa destrucción de la materia orgánica
275-300	Destrucción de los compuestos hidrofóbicos del suelo
200-400	Volatilización de algunos nutrientes (principalmente N)
>300	Completa destrucción de los horizontes orgánicos del suelo
= 450	Consumo total de la material orgánica del suelo
600	Máximas pérdidas del fósforo y el potasio del suelo y oxidación de los enlaces metálicos
800	Oxidación del azufre del suelo
1240	Volatilización del calcio del suelo

La severidad del fuego describe la respuesta del ecosistema al fuego y se puede usar para describir los efectos del fuego en el suelo, el sistema de agua, la flora, la fauna, la atmósfera y la sociedad (Bento-Gonçalves et al., 2012). De hecho, la severidad del fuego es un producto de la intensidad del fuego y la duración del mismo. Keeley (2009) estableció diferentes categorías cualitativas de severidad del fuego teniendo en cuenta el estado de la vegetación y la materia orgánica después de un incendio forestal:

• No quemado

- Las plantas no han cambiado. No tienen efectos observables como resultado del calor.

• Poco quemado

- No quemado. Las plantas han perdido algunas hojas como resultado del calor, pero no hay cambios en las propiedades del suelo y las plantas no se ven afectadas.

• Baja severidad

- Las copas de los árboles tienen agujas verdes, mientras que algunas ramas se queman.

- La hojarasca, musgos y otras hierbas se carbonizan o consumen produciendo cenizas negras (Figura 6).
- La capa orgánica del suelo está casi intacta. Sólo se carbonizan los mm superficiales del suelo.

• Severidad moderada / media

- Parte de las copas de los árboles mueren, pero las agujas no se consumen.
- Se han consumido todas las plantas de la superficie produciendo cenizas grises (Figura 6).
- Las ramas finas son consumidas y carbonizadas.
- El horizonte orgánico del suelo resulta ampliamente consumido.

• Alta severidad

- Copas de árboles muertas y agujas completamente consumidas.
- Horizonte orgánico totalmente consumido.
- Ceniza blanca en la superficie y material carbonizado que alcanza unos cm dentro del suelo (Figura 6).

Figura 6. Cenizas negras, grises y blancas producidas durante la combustión vegetal durante el incendio. Castellolí (Barcelona). Imagen: Marcos Francos.

El impacto del fuego en el suelo puede ser directo o indirecto. Los impactos directos son consecuencia del calentamiento producido en los primeros centímetros del suelo y los indirectos son el resultado de la eliminación temporal de parte de la cubierta vegetal, la incorporación de cenizas en el suelo, etc. (Mataix-Solera y Guerrero, 2007). La calidad del suelo es reconocida como la forma más dinámica y sensible de medir la resistencia del suelo frente a los cambios realizados por fuerzas naturales o antrópicas (Karlen et al., 2003). La conductividad térmica del suelo es muy baja y el impacto del fuego en la profundidad del suelo no es muy alto; por esta razón muchos estudios muestrean a 2,5 cm de profundidad. Según Badía-Villas et al. (2014), los cambios producidos por el fuego se reducen solo hasta 2 cm de profundidad; los suelos más profundos no cambian significativamente con el fuego (Badía et al., 2017). Autores como Francos et al. (2016a), Pereira et al. (2017) y Francos et al. (2018a, b, c, d) entre otros, han realizado muestreos a 5 cm de profundidad para identificar los impactos directos e indirectos del fuego, del manejo forestal y del crecimiento vegetal. Neary et al. (1999) informaron que los incendios forestales de alta intensidad pueden alcanzar los 675 ºC en la capa superficial del suelo (0-2,5 cm), siendo la temperatura de 190 ºC a 2,5 y 75 ºC a 5 cm de profundidad. La profundidad del suelo (5 cm) rara vez supera los 15 ºC, lo que en algunos casos es suficiente para cambiar las propiedades del suelo (Caon et al., 2014). Según Doran (1994), los principales indicadores utilizados para determinar la calidad del suelo se dividen en:

- Físico: repelencia del suelo al agua, estabilidad estructural de los agregados y capacidad de infiltración.

- Química: materia orgánica, carbono inorgánico, nitrógeno total, relación C/N, pH, conductividad eléctrica, elementos mayoritarios y minoritarios extraíbles y otros elementos como el fósforo.

- Biológico: respiración basal edáfica y carbono de la biomasa microbiana.

Los incendios forestales forman parte de los agentes naturales que generan impactos en la calidad de los suelos en los bosques mediterráneos. Estos incendios pueden alterar la calidad física del suelo (estabilidad de agregados, porosidad, distribución, hidrofobicidad y escorrentia), química (disponibilidad de nutrientes, mineralogía, pH y relación C/N) y biológica (productividad de biomasa, composición microbiana y almacenamiento de carbono) (Doerr y Cerdà, 2005; Mataix-Solera y Guerrero, 2007).

Las propiedades del suelo a menudo se relacionan unas con otras y es muy complejo individualizar el efecto del fuego en cada propiedad del suelo. Autores como Pereira et al. (2018b) trató de sintetizar los impactos del fuego en una breve y completa revisión. Según diferentes estudios, los impactos en la calidad del suelo a nivel físico son:

La hidrofobicidad o repelencia del suelo al agua (SWR) (Figura 7) depende de las condiciones del suelo previas al fuego (suelos hidrófobos o hidrófilos), la intensidad del fuego (Mataix-Solera y Cerdà, 2009) y la cantidad y tipo de combustible vegetal (Arcenegui et al., 2007). Los incendios de baja severidad pueden destruir una parte de materia orgánica del suelo y la acumulación de cenizas después de la ignición puede producir un aumento de la repelencia. Después de incendios de alta intensidad y consecuentemente de alta severidad, los compuestos hidrófobos pueden destruirse disminuyendo la SWR (Mataix-Solera y Guerrero, 2007). En algunos casos, hay un aumento en la hidrofobicidad, causada por las cenizas que obstruyen los poros del suelo y potencian las propiedades hidrofóbicas del suelo (Pereira et al., 2015b). Dichas cenizas generalmente producen una disminución en la capacidad de infiltración y permeabilidad del suelo (Imeson et al., 1992). La ceniza producida a baja intensidad es más hidrófoba que la producida a alta intensidad (Bodí et al., 2011), aunque no se puede asegurar que este tipo de incendios aumente la hidrofobicidad general del suelo con respecto a los incendios de alta intensidad, ya que en estos últimos la capa de ceniza es mayor (Pereira et al., 2015b). Las especies vegetales condicionan la hidrofobicidad de cada área siendo más hidrófobos los suelos de pinares que los suelos de robledales (Mataix-Solera et al., 2007). A largo plazo, la hidrofobicidad está desapareciendo debido a factores como la erosión y la cobertura vegetal (Hubbert y Oriol, 2005). El aumento en la hidrofobicidad puede conllevar un aumento en la estabilidad de agregados por la creación de compuestos hidrófobos debido al calentamiento (Mataix-Solera y Doerr, 2004). La estabilidad de los agregados (AS) está fuertemente relacionada con la capacidad de infiltración del suelo. La dinámica del AS después del incendio es difícil de evaluar y está estrechamente relacionada con la intensidad del fuego (Mataix-Solera et al., 2007). Al comienzo de la ignición en un incendio severo, hay una destrucción de la materia orgánica (Oades, 1993). En algunos casos hay un incremento de la AS por una fusión térmica en suelos con alta cantidad de hierro y aluminio (Giovannini et al., 1988), pero este incremento de AS no tiene un papel útil en la dinámica del suelo (Guerrero et al., 2001). El incremento de AS puede producirse por la generación durante el incendio de compuestos hidrófobos que protegen el agregado de la lluvia (Mataix-Solera y Doerr, 2004). Muchos autores (Úbeda et al., 1990; DeBano et al., 1998; Badía y Martí, 2003) han estudiado la destrucción de la materia orgánica durante el incendio debido a la alta severidad, como el principal factor que disminuye la AS. Los incendios forestales recurrentes pueden reducir la capacidad de infiltración y aumentar la repelencia del suelo al agua (Vallejo y Alloza, 1998; Caon et al., 2014). Mataix-Solera et al. (2013) concluyeron que los valores altos de SWR a menudo se acompañan de tasas de infiltración más bajas.

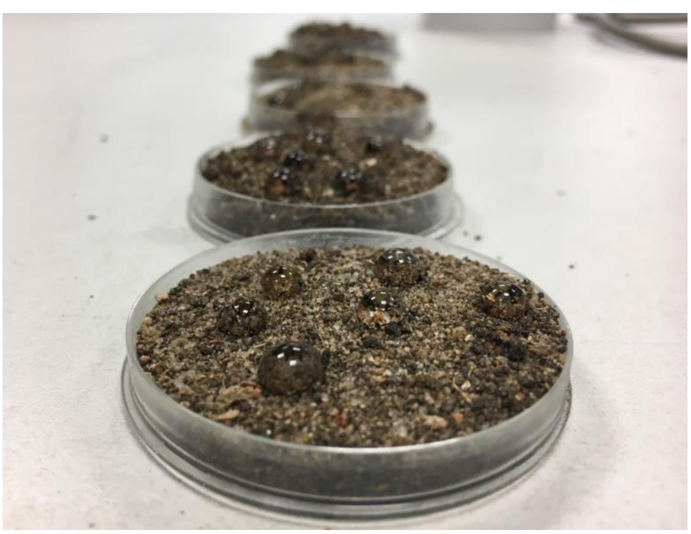

Figura 7. Analizando la repelencia al agua mediante el test WDPT en condiciones de laboratorio. Imagen: Marcos Francos.

Los impactos en las propiedades químicas del suelo son:

Cada tipo de incendio y las características de cada ecosistema influyen de manera diferente en la reserva de materia orgánica y su composición química (Certini et al., 2011). González-Pérez et al. (2004) designaron como efectos principales del fuego en la materia orgánica: a) la eliminación de oxígeno externo que produce materiales con baja solubilidad, b) la reducción de compuestos alquílicos, c) la aromatización de azúcares y lípidos, d) creación de compuestos heterocíclicos de nitrógeno, e) condensación de sustancias húmicas y f) producción de carbón negro que es casi inalterable. El consumo de la materia orgánica en el suelo comienza a 200 ºC y es total a 460 ºC (Giovannini et al., 1988). Básicamente, la cantidad y calidad de la materia orgánica remanente después de un incendio dependerá de factores como la intensidad del incendio (de la Rosa et al., 2014), la propagación del mismo, la cantidad de combustible y su combustibilidad y la humedad del suelo (Almendros et al., 2012). En incendios de alta intensidad, se reduce la OM inmediatamente, aunque a largo plazo, generalmente supera el nivel previo al fuego (Certini, 2005). Los incendios de baja intensidad pueden aumentar la cantidad de materia orgánica debido a una menor combustión y una contribución del material vegetal parcialmente carbonizado de las partes aéreas de las plantas (Úbeda et al., 2005; de la Rosa et al., 2014; Pereira et al., 2015b). La materia orgánica del suelo disminuye significativamente en los suelos quemados a causa del calentamiento (Ershad et al., 2013; Badía-Villas et al., 2014). Esta disminución se mantiene de corto a largo plazo de acuerdo con la severidad del fuego y la composición de las cenizas derivadas de la ignición (Francos et al., 2018b). Esta materia orgánica del suelo aumenta en los primeros años de incendio causados por la incorporación de material pirolizado en el suelo y la incorporación de plantas muertas y tejidos microbianos (DeLuca y Zouhar, 2000). La calidad de la materia orgánica cambia notablemente cuando la fracción más recalcitrante aumenta. Esto ocurre cuando se queman residuos frescos y se forman nuevos compuestos aromáticos y compuestos altamente polimerizados. El material carbonizado, producido por una incompleta combustión, puede permaneces durante varios años en el suelo (Certini, 2005).

Las altas temperaturas del fuego y, en consecuencia, la alta severidad del fuego contribuye a la creación de compuestos inorgánicos y carbonatos (Francos et al., 2016a), que suele equipararse en dinámica y denominarse de manera genérica como Carbono Inorgánico (IC). Las cenizas producidas después de incendios forestales severos son principalmente de color blanco y son ricas en compuestos inorgánicos (Goforth et al., 2005). Este tipo de cenizas son fácilmente erosionadas y transportadas por el viento (Úbeda et al., 2009) y la incorporación al suelo depende de las condiciones meteorológicas posteriores al incendio (explicadas profusamente en la sección 1.4) (Pereira et al., 2013; Francos et al., 2016a). El aumento de carbono inorgánico (IC) después del incendio se puede producir por la volatilización del carbono orgánico durante el incendio forestal (Pereira et al., 2014). La producción de cenizas blancas se limita a las altas temperaturas alcanzadas durante el calentamiento (Certini, 2005) que depende principalmente de las condiciones del combustible vegetal antes del incendio (Francos et al., 2018c).

La disminución del Nitrógeno Total (TN) del suelo después del fuego depende de la severidad del incendio (Caldwell et al., 2002). Las pérdidas ocurren en el nitrógeno orgánico y aumentan el nitrógeno inorgánico (Giovannini et al., 1988; Mataix-Solera et al., 2009). Las pérdidas de N orgánico se deben a la volatilización después del incendio (Fisher et al., 2000) y a una mineralización de amonio, fácilmente asimilable a en las plantas. Este amonio se absorbe en las cargas negativas de las partículas orgánicas y minerales del suelo, aunque con el tiempo se transforman en nitratos, que se liberan rápidamente. Las pérdidas de TN en el suelo después de un incendio severo pueden atribuirse a la lixiviación, erosión del suelo, escorrentía y volatilización por el fuego (Marion et al., 1991; Ice et al., 2004). El nitrógeno total del suelo es un elemento esencial para las plantas (Caon et al., 2014). Este nitrógeno total del suelo disminuye significativamente a medida que aumenta la intensidad del fuego en las áreas quemadas (Ershad et al., 2013). Este elemento se volatiliza fácilmente como observaron Mataix-Solera y Cerdà (2009). A pesar de esto, los incendios de baja severidad a menudo no afectan el contenido de TN en el suelo (Santin y Doerr, 2016). El fuego produce aumentos de nitrógeno inorgánico y descensos de nitrógeno orgánico en incendios de alta severidad (Raison, 1979) siendo los cambios de TN no demasiado dramáticos, pero reduciendo el nitrógeno orgánico que las plantas utilizan para crecer (Mataix-Solera y Guerrero, 2007). La razón podría deberse a la absorción de nutrientes por parte de las plantas que se regeneran y la nitrificación por nitrógeno inorgánico (disminución con el tiempo) disminuyendo la cantidad de TN y disminuyendo por lo tanto el nitrógeno disponible para las plantas (Durán-Humia et al., 2008). La relación C/N también suele disminuir, probablemente debido a la formación de formas heterocíclicas nitrogenadas recalcitrantes (Certini, 2005). Otros autores como Mataix-Solera et al. (2009) observaron un incremento de la relación C/N a medida que la temperatura del incendio se volvía más alta. En el caso en que las cenizas se incorporan en el suelo, hay una reducción de la relación C/N en comparación con los suelos no quemados por la mineralización de la materia orgánica del suelo (Volkova et al., 2014).

Algunos estudios reportaron un aumento a corto plazo del pH por la combustión de la OM durante el incendio y la liberación de cationes solubles en el suelo (Knoepp et al., 2005). La contribución de las cenizas y la solubilidad de los compuestos presentes provoca un aumento en el pH que varía según la cantidad y el grado de combustión de los mismos (Certini, 2005; Úbeda et al., 2005; Mataix-Solera et al., 2009; Pereira et al., 2011). Marcos et al. (2007) observaron fuertes aumentos de pH con altas temperaturas de fuego. No en todos los casos el pH del suelo aumenta significativamente. En suelos calcáreos, este aumento es menor debido a la alta capacidad amortiguadora del suelo (Mataix-Solera et al., 2009). Otro factor que puede no alterar el pH o afectar de manera muy sucinta es el fuego de baja intensidad, ya que las cenizas producidas tienen un pH más bajo (Úbeda et al., 2009; Pereira et al., 2015b). Los incendios forestales muy severos producen un incremento del pH del suelo por la formación de óxidos y carbonatos presentes en las cenizas (Ulery et al., 1993; Ershad et al., 2013) y la destrucción de compuestos orgánicos (Mataix-Solera y Guerrero, 2007). Los aumentos en el pH pueden favorecer la actividad de los microorganismos, lo que aumenta la descomposición de la materia orgánica, pero en algunos casos, la variación del pH produce problemas vegetales de nutrición que evitan la asimilación de algunos compuestos de nutrientes por parte del suelo.

Comúnmente, la Conductividad Eléctrica (EC) aumenta después del incendio por la incorporación y solubilización de las cenizas (DeBano et al., 1977; Badía y Martí, 2003). DeByle (1976) y Ershad et al. (2013) observaron aumentos de EC después del incendio debido a la ignición de la hojarasca y la liberación de grandes cantidades de cationes en el suelo. Autores como Naidu y Srivasuki (1994) observaron que el aumento efímero de la EC se debe a la liberación de iones inorgánicos por la combustión de la materia orgánica. La concentración de sales solubles, medida por la conductividad eléctrica, aumenta debido a la solubilización de los compuestos que contienen las cenizas (Mataix-Solera et al., 2009). Este aumento depende de la intensidad del fuego y la consiguiente mineralización de los compuestos orgánicos (Certini, 2005; Pereira et al., 2011). En muchos casos, el fuego de baja intensidad, produce cambios de pH y EC pero no significativos (Marcos et al., 1998).

La producción de ceniza también produce un aumento en los cationes liberados (Pereira et al., 2011). La facilidad de solubilización que tiene estos elementos es fundamental para la vegetación, aunque el tipo y la cantidad de nutrientes liberados dependen en gran medida de la severidad del incendio y del tipo de incendio (Úbeda et al., 2009). Este aumento y su composición están relacionados con el grado de mineralización y oxidación de los compuestos orgánicos (carbono negro) durante el incendio (Lima et al., 2002). A baja intensidad, el pH de las cenizas es bajo y facilita la solubilidad de los metales pesados. A intensidad moderada, el pH de la ceniza está entorno a 7-8 y la mayoría de los cationes liberados son básicos: Ca^{2+} y Mg^{2+} (Pereira et al., 2015b). Las cenizas que provienen de la madera quemada también proporcionan grandes cantidades de Ca^{2+}, Na^{+}, Mg^{2+} y K^{+} (Etiegni y Campbell, 1991; Demeyer et al., 2001; Mandre et al., 2004). Una reducción en la capacidad de intercambio catiónico puede producirse debido a un descenso en la cantidad de materia orgánica. La saturación de bases aumenta debido a la liberación de cationes básicos al quemar materia orgánica (Certini, 2005). El Calcio (Ca) extraíble en el suelo, el Magnesio (Mg) extraíble y el Sodio (Na) extraíble aumentaron de manera efímera con temperaturas de incendio entre 250º y 500º C; temperaturas más altas producen la disminución de los elementos extraíbles (Badía y Martí, 2003; Giovannini, 2012). Varios autores (Grove et al., 1986; Raison et al., 1990; Soto y Diaz-Fierros, 1993) informaron de estos incrementos posteriores al fuego de Ca, Mg, Na y K extraíbles; y son mayores cuanto mayor es la severidad del fuego (Grove et al., 1980). El suelo puede aumentar o disminuir la capacidad de intercambio de cationes (CEC) de acuerdo con la severidad del fuego y, en consecuencia, el consumo de materia orgánica. En los casos en los que existe pérdida de suelo por la alta severidad alcanzada, la materia orgánica y la CEC pueden disminuir por erosión y lixiviación (Mataix-Solera y Guerrero, 2007). En incendios donde la severidad fue baja, no hay una destrucción total de la materia orgánica y el complejo absorbente retiene estos cationes (Alcañiz et al., 2018). El aumento de los nutrientes del suelo depende de las especies de arbóreas que se ven afectadas por el fuego, las propiedades del suelo y los procesos de lixiviación (Certini, 2005). El tiempo que permanecen estos incrementos de Calcio (Ca), Magnesio (Mg) y Potasio (K) extraíbles después del incendio varía de meses (Adams y Boyle, 1980) a años (Simard et al., 1991).

Generalmente, hay un incremento del fósforo (P) disponible en el suelo después del incendio por la incorporación de las cenizas producidas durante la combustión de la vegetación (Raison, 1979). El fósforo disponible después del incendio puede ser el resultado de la fuerte mineralización del P orgánico por el fuego (Cade-Menun et al., 2000). Las pérdidas de P extraíbles en el suelo por volatilización o lixiviación son muy pequeñas generalmente (Certini, 2005). Soto et al. (1991) y Saá et al. (1998) observaron disminuciones en el fósforo disponible en el suelo extraíble por la erosión posterior al incendio que evita la incorporación de cenizas en el suelo. El fósforo disponible también aumenta fácilmente debido a la contribución de las cenizas en el suelo y a su alta solubilidad y movilidad (Khanna et al., 1994). Cuanto mayor sea la intensidad del fuego, mayor será la entrada de fósforo disponible (Pereira et al., 2015b). En los casos en que las cenizas se incorporaron al suelo evitandose los procesos de erosión y la mineralización de P a partir de formas orgánicas, hubo un incremento de P extraíble en el suelo (Mataix-Solera, 1999). Badía-Villas et al. (2014) registraron un aumento a corto plazo del fósforo disponible por la disolución de P de los lechos de ceniza y la mineralización del fósforo orgánico debido al calentamiento. En los casos en que las cenizas se transportan y no se incorporan al suelo, no hay cambios significativos entre los suelos quemados y no quemados (Johnson et al., 2007).

La microbiología del suelo es esencial para el funcionamiento del suelo y es muy sensible a los incendios forestales de alta severidad (Pereira et al., 2018b). Las propiedades microbianas del suelo pueden verse afectadas, en primer lugar, directamente por el calentamiento y, en segundo lugar, por la modificación de las condiciones del suelo que afecta a la microbiología del suelo como indican detalladamente en su revisión Mataix-Solera et al. (2007).

Los impactos en las propiedades microbiológicas del suelo son:

El fuego puede producir un incremento de la Respiración Basal del Suelo (BSR) por la solubilización de compuestos orgánicos por calor (Pietikäinen y Fritze, 1993; Fernández et al., 1997). Después de este aumento efímero, se produce la disminución de la BSR (Bisset y Parkinson, 1980; Almendros-Martín et al., 1990) al menos durante algunos meses o años (Hernández et al., 1997; Mataix-Solera et al., 2006). El carbón de la biomasa microbiana del suelo (C_{mic}) disminuye por el impacto del fuego incluso a baja intensidad (Prieto-Fernández et al., 1998; Santín y Doerr, 2016). La población de hongos es uno de los microorganismos más afectados por el fuego y es uno de los contribuyentes más populares de la biomasa microbiana. El choque térmico producido por el fuego disminuye los propágulos de hongos por la dispersión de esporas (Mataix-Solera et al., 2002) causando la consecuente disminución del C_{mic} (Díaz-Raviña et al., 1992; Hernández et al., 1997). Cuando las temperaturas alcanzan los 400 ºC, los niveles de C_{mic} se vuelven indetectables (Guerrero et al., 2005).

4. Efecto de las condiciones meteorológicas post-incendio en los suelos

Los impactos de un incendio en el suelo pueden ser causados directamente por el calentamiento e indirectamente por el efecto de las cenizas, la recuperación vegetal, los patrones climáticos posteriores al incendio y la topografía y el manejo posterior al incendio (Pereira et al., 2018b). Los factores ambientales son trascendentales en la evolución de las propiedades del suelo a lo largo del tiempo. El tipo de suelo, la severidad y la frecuencia de los incendios y las condiciones climáticas posteriores al incendio influirán en la dinámica a corto, medio y largo plazo de las propiedades del suelo (Certini, 2005). Además de la severidad, la topografía es esencial en las dinámicas de propiedades del suelo (Pereira et al., 2018b). Las áreas quemadas con alta pendiente son más vulnerables a la degradación y la erosión, así como a las pendientes orientadas al sur (Jones et al., 2014) (Figura 8). Otros elementos, como la gran cantidad de rocas fragmentadas que cubren el suelo, reducen la erosión y las micro depresiones del suelo permiten la retención de cenizas y hacen posible la incorporación de cenizas en el perfil del suelo.

Figura 8. Proceso de erosión en laderas y pérdida de suelo después de un incendio forestal. Ódena (Barcelona). Imagen: Marcos Francos.

Los estudios climáticos posteriores al incendio son totalmente necesarios para comprender la dinámica de las propiedades del suelo a lo largo del tiempo (Pereira et al., 2018b). El viento extremo justo después de un incendio forestal puede modificar la distribución de las cenizas, aumentar la compactación del suelo y las cenizas y la erosión debido a la eliminación de la cubierta arbórea a causa del fuego. Las precipitaciones extremas pueden modificar también el paisaje y provocar peligros ambientales catastróficos (Pereira et al., 2010), sobretodo, en entornos frágiles, como es después de un incendio forestal de alta severidad. La lluvia después del incendio conduce a la incorporación de cenizas en el suelo en áreas con poca pendiente; en áreas con una pendiente alta y en los casos en que la intensidad de la precipitación fue alta, las cenizas se pueden eliminar por escorrentía y erosión, impidiéndose la incorporación de las cenizas en el suelo empobreciéndolo y degradándolo (Francos et al., 2016a). Las precipitaciones de baja intensidad tienen menos capacidad para transportar cenizas y sedimentos, lo que permite la incorporación de cenizas en el suelo y, en consecuencia, disminuye la repelencia al agua (SWR) y aumenta la conductividad hidráulica. La intensidad y la duración de las precipitaciones deben tenerse en cuenta después de un incendio forestal, además en áreas con pendientes altas para predecir y evitar los procesos de erosión y las pérdidas del suelo (Staley et al., 2016). La pendiente y la intensidad de la lluvia son esenciales para permitir (en áreas de poca pendiente) o evitar (en áreas de alta pendiente) la incorporación de cenizas (y, en consecuencia, el incremento de elementos disponibles en el suelo) después de la primera precipitación tras un incendio forestal. En realidad, y como apuntaron varios autores (por ejemplo, Bravo et al., 2010; Francos et al., 2016a; Jiménez-Ruano et al., 2017; Rodrígues et al., 2018 entre otros), el régimen de incendios forestales se ha modificado y en los períodos de transición verano-otoño puede ocurrir la confluencia de una lluvia torrencial justo después de un incendio forestal que causa efectos devastadores en los suelos de zonas con elevada pendiente. El crecimiento vegetal es esencial después de un incendio forestal para proteger el suelo antes de estas lluvias torrenciales (Outeiro et al., 2007). Esta lluvia, si se produce antes del rebrote vegetal, puede dañar el banco de semillas del suelo, lo que condiciona la revegetación de la zona (Moreno et al., 2011). El impacto de las gotas de lluvia en el suelo descubierto después de un incendio forestal severo puede evitar la incorporación de cenizas y destruir los agregados del suelo (Outeiro et al., 2007) y además provocar la reducción de la velocidad de infiltración (Ela et al., 1992). En los casos de lluvias de alta intensidad en las zonas de pendiente, hay un arrastre de nutrientes por el flujo terrestre (Úbeda et al., 1998; Úbeda y Sala, 2001). A pesar de la ausencia de erosión, la lluvia puede modificar la disponibilidad de nutrientes del suelo después del impacto del incendio (Outeiro et al., 2008), lo que produce que un excesivo lavado del suelo, la consecuente degradación y cicatrices del suelo (Figura 9).

Figura 9. Cicatrices del suelo después de un incendio forestal, una lluvia torrencial y la gestión post-incendio. Colomers (Girona). Imagen: Marcos Francos.

5. Efecto de la gestión forestal en los suelos

El manejo forestal puede afectar las propiedades del suelo (Francos et al., 2018d) y la dinámica de la vegetación (Strom y Fulé, 2007) incluso a largo plazo. La gestión forestal previa a incendios forestales es un tema que no se ha analizado de manera detallada pero que puede ser determinante en cuanto a los impactos de los incendios forestales y la creación de estructuras forestales resistentes y resilientes a éstos (Keeley, 2009; Stevens-Rumann et al., 2013; Shive et al., 2014). Sólo unos pocos estudios han analizado este tema. Este tipo de manejo puede modificar el combustible vegetal tanto la cantidad como la calidad y por lo tanto influir en la severidad de los incendios forestales y con ello cambiar el efecto del fuego en los suelos (Cram et al., 2015; Francos et al., 2018c). Francos et al. (2018c) observaron que las zonas no gestionadas y los bosques gestionados pocos meses o pocos años antes de un incendio forestal producen diferencias en el efecto que tiene el fuego en las propiedades del suelo. Los cambios registrados en el suelo por los autores después de incendios forestales fueron en las propiedades edáficas TN, SOM, IC, pH, EC y Ca y Mg extraíbles. La reducción de SOM puede afectar otras propiedades del suelo (por ejemplo, AS y SWR, entre otras). Además, se deben evitar las estructuras forestales que podrían quemarse alcanzando una alta severidad (Varela et al., 2015) como son los bosques densos y con continuidad horizontal y vertical del combustible vegetal. Rapp (2007) concluyó que la severidad de los incendios forestales depende del tratamiento del combustible y su efectividad. El mismo autor comentó que los tratamientos con menos de 10 años de edad fueron efectivos para disminuir la severidad y producir menos modificaciones en el suelo, según señalaron también Francos et al. (2018c). Rapp (2007) llegó a la conclusión de que los tratamientos contra los incendios forestales solo fueron efectivos hasta 10 años, los tratamientos más antiguos no mostraron diferencias con las áreas no tratadas.

La gestión posterior al incendio puede modificar las propiedades del suelo a corto (Ginzburg y Steinberger, 2012), medio (Wagenbrenner et al., 2015) y a largo plazo (Johnson et al., 2005; Francos et al., 2018d). Algunos tratamientos posteriores al fuego utilizados en las últimas décadas han sido la aplicación de paja que se produzca el efecto *mulch*, las agujas de pino o desechos orgánicos en el área quemada del bosque, la colocación de troncos quemados para detener la erosión y el corte y extracción de madera quemada utilizando maquinaria. El objetivo de estos tratamientos es mejorar la calidad de las propiedades del suelo, evitar la erosión y favorecer el recrecimiento vegetal (Guerrero et al., 2000, 2003; Mataix-Solera et al., 2001; Olsen, 2016). Entre los efectos que dichas gestiones producen en el suelo se encuentran la erosión (Robichaud et al., 2011) y la compactación (Rab, 1996; McIver y Starr, 2000). Existen diferentes tratamientos posteriores al incendio con el objetivo de mejorar las propiedades del suelo y alcanzar los valores previos al incendio a) tan pronto como sea posible para facilitar la recuperación vegetal después del incendio; y b) buscando ser un manejo forestal adecuado a largo plazo. Marques y Mora (1998) no observaron cambios en la respuesta a la erosión y la vegetación en las áreas donde se extrajo la madera quemada después del incendio; A pesar de esto, los autores recomendaron ser cuidadosos con las prácticas de remoción de madera para minimizar los peligros de erosión y las pérdidas de suelo. La gestión explicada en el estudio anterior consiste en la construcción de diques utilizando las ramas quemadas. Estas prácticas evitan las pérdidas de suelo por escorrentía, pero el uso en grandes áreas es muy limitado. Gómez-Rey et al. (2013) evaluaron el uso de la aplicación de semillas y *mulch* de paja en las propiedades químicas del suelo a corto plazo después del incendio y concluyeron que éstos evitan la erosión y aumentan el TN, el Ca, el Mg y el K extraíbles en comparación con el área no tratada. Bai et al. (2014) analizaron el efecto del *mulching* en las áreas afectadas por el fuego en zonas de pendiente en comparación con la hidrosiembra y concluyeron que el primer tipo de manejo es el más efectivo para mejorar la fertilidad del suelo en suelos perturbados. El acolchado de paja o *mulch* produjo incrementos de carbono y nitrógeno en el suelo por encima de los resultados obtenidos en las zonas donde se llevó a cabo la hidrosiembra debido a que el último fue menos efectivo para retener el agua incrementándose con ello la materia orgánica del suelo, como concluyeron también Eck et al. (2010). Ginzburg y Steinberger (2012) observaron el impacto a corto plazo de la tala y extracción de madera en las propiedades microbianas del suelo y, por lo tanto, recomendaron posponer las prácticas post-incendio al menos un año después del incendio forestal. Francos et al. (2018a) señalaron como alternativa útil la gestión basada en la extracción de la madera quemada manualmente. Este tipo de manejo posterior al incendio puede ser útil para zonas concretas y de tamaño reducido como áreas de pendiente o zonas entorno a albergues muy turísticos (para evitar afectar el paisaje); esta gestión muestra ser útil en la protección de las propiedades del suelo evitando el daño causado por la tala y extracción mecánica de la madera. Las diferencias entre este manejo y zonas donde no se realizó ningún tratamiento o la gestión se basó en la tala y no extracción de la madera fueron registradas en diferentes propiedades del suelo como AS, TN, SOM, IC y EC. Spanos et al. (2005) evaluaron el uso de mulas para extraer los troncos quemados. En este caso, las mulas disminuyeron el daño en las propiedades del suelo en comparación con el uso de maquinaria pesada, concretamente con *skidders*. Las diferencias se registraron a corto plazo en el pH del suelo y la SOM. Wagenbrenner et al. (2014) observaron mayor compactación, menor SWR y menor crecimiento vegetal en áreas gestionadas usando *skidder* en comparación con áreas no gestionadas después del incendio. Este último tipo de gestión usando maquinaria pesada produjo grandes incrementos de las tasas de erosión. La extracción manual o animal de madera muestra menores impactos en las propiedades del suelo (Wagenbrenner et al., 2016) y parece ser una buena alternativa para usar en áreas pequeñas, sitios concretos y casos excepcionales. Jennings et al. (2012) observaron y compararon el impacto de la tala y la extracción mecánica de la madera en un área quemada con un área no tratada mecánicamente. En este estudio, los autores observaron cambios en el P, el nitrógeno y el BSR disponibles causados por el uso de maquinaria pesada y la consiguiente compactación del suelo durante la tala y la extracción de madera quemada. Este tipo de gestión con maquinaria pesada reduce la cobertura vegetal y la calidad del suelo a corto plazo después de un incendio forestal viéndose también afectadas la SOM, TN, P, C_{mic} y BSR. La extracción de madera quemada y, especialmente, el uso de maquinaria pesada después del incendio forestal daña el banco de semillas y degrada el suelo, por lo que García-Orenes et al. (2017) sugirió posponer los tratamientos hasta el medio plazo después del incendio. Serrasolses y Vallejo (1999) observaron incrementos de la fertilidad del suelo a corto plazo en las áreas donde se cortaron troncos después del incendio, pero esos troncos no se extrajeron no afectando de manera tan profusa a las propiedades del suelo.

La gestión forestal a largo plazo después de un incendio puede producir que las propiedades del suelo en el área gestionada sean más similares a las áreas no afectadas por el fuego que un área no gestionada (Francos et al., 2018d). Estas diferencias son consecuencia de la alta densidad vegetal y la competitividad para el reservorio de agua y la luz solar que producen cierto estrés edáfico, evitando la recuperación de estas propiedades a valores pre-incendio incluso a largo plazo. Fernandes et al. (2013) señalaron la necesidad de manejo forestal para reducir los impactos del fuego en el suelo. Los cambios más importantes reportados por Francos et al. (2018d) son los siguientes: El suelo quemado no gestionado, a diferencia del suelo quemado gestionado mostró diferencias con la zona no quemada en las propiedades TN, IC, SOM, pH, K, Al, Mn, Fe extraíbles y la relación C/N a largo plazo después del incendio y a largo plazo después de la gestión post-incendio debido a la alta densidad vegetal (Figura 10). Los autores señalaron la necesidad de gestionar áreas forestales para mejorar la calidad del suelo y disminuir el riesgo de que ocurran grandes incendios forestales. Este tipo de manejo se debe tener en cuenta para evitar incendios forestales de alta severidad (una vez más) gestionando el área quemada de bosque a medio plazo después de un incendio (Francos et al., 2016b).

Figura 10. Alta densidad vegetal 30 años después de un incendio sin ninguna gestión forestal. Ódena (Barcelona). Imagen: Marcos Francos.

MAIN OBJECTIVES

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The origins of this study lie in the pressing need to expand our knowledge of forest management practices and of the natural dynamics that govern forests in Mediterranean ecosystems, characterized by the recurring nature of the forest fire phenomenon. The post-fire period in these forests is transcendental in determining the dynamics of their soil properties in the ensuing months and even years. The disruption caused by the fire leaves the soil system in a very fragile state and means that extreme caution must be exercised in identifying the optimum procedures to be implemented. Thus, the general objective of the present doctoral thesis is to evaluate the impact of forest fires in the short, medium and long term and to evaluate the forest management practices carried out in forested areas. The methodology employed in the studies making up this thesis is that of the chronosequence of wildfire events.

The main objectives of each of these studies is as follows:

1. In the first study, a literature review is undertaken of the impact of fire on soils, the factors that influence this impact, the recovery of the ecosystem after the fire, and the methods used for the recovery of forest areas and, in turn, the impact of these management practices.
2. The second study analyses the impact of pre-fire management on soil properties, in an attempt at determining the effectiveness of such practices in areas prone to fire. It begins by examining one of the main management practices – that of clear cutting – and its impact on soil AS, TN, SOM, inorganic carbon (IC), the C/N ratio, pH, EC, extractable calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), microbial biomass carbon (C_{mic}), basal soil respiration (BSR), and the C_{mic}/C, BSR/C and BSR/C_{mic} ratios. These areas managed prior to the passage of fire can result in different fire severities. Secondly, the study evaluates the influence of the time lapse between the site management and the wildfire event itself to determine if this time-lapse affected post-fire soil properties.
3. The third study evaluates the effect of torrential rainfall immediately following a wildfire on soil properties. Torrential rainfall after a fire is one of the main determinants of the recovery of pre-fire soil values and of the soil dynamics that allows or prevents the incorporation of ash and the generation of erosion and soil losses. The study monitors the significance of changes in soil properties immediately after the fire and their evolution over time. Thus, the main objective is to examine the impact of an intense rainfall event on the soil properties of a recently burned area and to compare these results with data collected one year later.
4. The fourth study evaluates different short-term post-fire management practices following a fire. Recently the government of Catalonia opted to invest heavily in post-wildfire forest management, as wildfires are a recurrent problem with severe repercussions for the environment, society and the economy. As such, it is essential to understand the effectiveness of post-fire management practices and to identify

which have the most beneficial effects on soils while protecting the soil system. The aim of this study, therefore, is specifically to examine the short-term effects of different post-wildfire management practices on the soil's physico-chemical properties. Its main goals are to observe if the post-wildfire management a) has an impact on soil properties 2 and 10 months after the wildfire and b) if there are notable differences in these properties depending on the practices implemented. The study also undertakes a comparison of the impacts of the various post-wildfire manual practices employed and those of a no-intervention scenario.

5. The fifth study seeks to determine the long-term recovery of soil properties in areas affected by fires of different severity. To the best of our knowledge, no previous studies have examined the effects of different fire severity regimes from a long-term perspective, although a number of studies have examined the short- and medium-term effects of low-, moderate- and high-severity fires. The importance of long-term studies of the effects of varying fire severities is that they should shed light on the capacity of soils to respond to different levels of disturbance. This is critical for determining just how long low- and high-fire severity regimes are likely to modify a soil ecosystem and delay the recovery of the soil's pre-fire values. Thus, the aim of this study is to monitor the long-term impact of a wildland fire of different severities on the chemical properties of soil.

6. The sixth and final study seeks to determine the long-term effects of wildfire in managed and non-managed areas and, consequently, in areas of different plant density and to monitor the effects on their respective physico-chemical soil properties, including the minor elements. However, we do not dispose of information about the effects of variations in plant density on the minor elements in the soil and their ratios. The aim is to monitor soil status in areas subject to different long-term post-fire management practices after a wildfire event. The specific objectives are: a) to analyze differences in soil properties associated with post-fire management practices; b) to study how plant density affects these soil properties; and, c) to identify long-term sustainable post-fire management practices as revealed by soil properties.

OBJETIVOS PRINCIPALES

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Este estudio nace de la necesidad de ampliar el conocimiento sobre el impacto de la gestión forestal y las variaciones debidas a causas naturales en las propiedades del suelo después de un incendio forestal en un ecosistema Mediterráneo. Este ecosistema se caracteriza por la recurrencia de los incendios forestales. Los momentos posteriores al incendio son trascendentales en la dinámica de que las propiedades del suelo en los siguientes meses e incluso años. Estos momentos en los que el suelo acaba de recibir la perturbación del fuego, hacen que el sistema suelo sea muy frágil y se debe tener extrema precaución con respecto a los procedimientos que se llevan a cabo. Por lo tanto, el objetivo general de la presente Tesis Doctoral es evaluar el impacto de los incendios forestales a corto, medio y largo plazo y evaluar el manejo forestal realizado en estas áreas forestales. La metodología en la que se presentan y compilan los estudios en esta tesis se estableció de acuerdo con la Crono secuencia establecida por estos eventos en los incendios forestales.

Los principales objetivos de cada uno de los estudios incluidos en la tesis doctoral han sido los siguientes:

1. En el primer estudio, se ha realizado una revisión de la literatura sobre el impacto del fuego en el suelo, los factores que influyen en este impacto y la recuperación del ecosistema después del incendio, además de los métodos utilizados para la restauración de áreas forestales y el impacto de estas prácticas.
2. El segundo estudio se enfoca en el análisis del impacto del manejo forestal previo al incendio en diferentes propiedades del suelo, tratando de entender la efectividad del manejo forestal en aquellas áreas que son propensas a los incendios. Por lo tanto, el objetivo principal ha sido examinar el impacto del clareo en propiedades del suelo como la estabilidad de agregados (AS), nitrógeno total (TN), materia orgánica del suelo (SOM), carbono inorgánico (IC), relación C/N, pH, CE, calcio extraíble (Ca), magnesio (Mg), sodio (Na), potasio (K), carbono de biomasa microbiana (C_{mic}), respiración basal del suelo (BSR) y las relaciones C_{mic}/C, BSR/C y BSR/C_{mic}. Estas áreas gestionadas antes del paso del fuego pueden tener como resultado diferentes severidades de fuego. El segundo objetivo fue evaluar la influencia del lapso de tiempo entre el manejo del cada sitio y el incendio forestal y determinar si este lapso de tiempo resultó en diferencias en las propiedades del suelo posteriores al incendio.
3. El tercer estudio tiene como objetivo evaluar el efecto sobre las propiedades del suelo de las lluvias torrenciales justo después de un incendio forestal. La lluvia torrencial después de un incendio es uno de los principales factores que influyen en la recuperación de los valores de las propiedades edáficas previos al incendio y en la dinámica del suelo, permitiendo o evitando la incorporación de cenizas y la generación de erosión y pérdida de suelo. En este sentido, es muy importante determinar su efecto sobre las propiedades del suelo. Es interesante monitorizar la importancia de los cambios justo después del incendio y la evolución

en el tiempo. Además, es esencial comparar el impacto inmediato de la lluvia y analizar como el suelo se ha recuperado o no un año después con el tercer muestreo.

4. El cuarto estudio pretende evaluar el manejo forestal posterior al incendio a corto plazo después del incendio. Como se señaló en el estudio, recientemente el gobierno de Cataluña ha optado por invertir fuertemente en la gestión de bosques después de incendios forestales, ya que los incendios forestales son una perturbación recurrente con graves impactos para el medio ambiente, la sociedad y la economía. Entonces, es esencial comprender la efectividad de los tratamientos de manejo post incendio e identificar qué prácticas tienen un efecto más positivo en los suelos evitando el daño del sistema suelo. Con todo ello, el objetivo de este estudio es examinar los efectos a corto plazo de diferentes tratamientos de manejo post-incendio en las propiedades físico-químicas del suelo. Los objetivos específicos son observar si el manejo posterior a los incendios forestales a) tiene impactos en las propiedades del suelo 2 y 10 meses después de los incendios forestales y b) si difieren de manera importante entre los tratamientos utilizados. Queríamos comparar los impactos de las operaciones manuales posteriores a incendios forestales con un escenario de no intervención y con otro donde la madera no se ha extraido y por ello las parcelas experimentales se delimitaron en la misma vertiente de una ladera donde las condiciones topográficsa fueran exactamente iguales y que así dichas condiciones no afectaran en los resultados.

5. El quinto estudio tiene como objetivo determinar la recuperación de las propiedades del suelo en áreas de diferentes severidades a largo plazo. En nuestro conocimiento, ningún estudio ha examinado los efectos de diferentes regímenes de severidad del fuego desde una perspectiva a largo plazo. Sin embargo, cuenta que varios estudios han examinado los efectos a corto y mediano plazo de los incendios de severidad baja, moderada y alta. La importancia de los estudios a largo plazo de los efectos de las diferentes severidades del fuego debería arrojar luz sobre la capacidad de los suelos para responder a diferentes niveles de perturbación. Esto es crítico para determinar por cuánto tiempo los regímenes de severidad del fuego bajo y alto pueden modificar un ecosistema del suelo y posponer la recuperación a valores anteriores al fuego. Por lo tanto, el objetivo de este estudio es monitorizar el impacto a largo plazo de un incendio forestal de diferentes severidades en las propiedades químicas del suelo.

6. El sexto y último estudio quiere determinar el efecto a largo plazo de los incendios forestales en áreas gestionadas y no gestionadas y, en consecuencia, con diferentes densidades vegetales y cómo se han visto afectadas las propiedades del suelo. Para este estudio seleccionamos las propiedades físico-químicas del suelo. Dentro de las propiedades químicas también se han analizado los elementos minoritarios. Sin embargo, carecemos de información sobre los efectos de las variaciones en la densidad de la vegetación sobre los elementos minoritarios en el suelo ya que muy pocos estudios los analizan. El objetivo es determinar el estado del suelo en áreas sujetas a diferente gestión posterior al incendio a largo plazo después de un incendio forestal. Los objetivos específicos son: a) analizar las diferencias en las propiedades del

suelo asociadas con las prácticas de manejo posteriores al incendio; b) estudiar cómo la densidad de las plantas afecta a las propiedades del suelo; y, c) identificar las prácticas sostenibles a largo plazo para el manejo del incendio según lo revelado por las propiedades del suelo.

MATERIAL AND METHODS

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1. Fieldwork

Before carrying out fieldwork, it is necessary to perform different actions. First of all, it is necessary to contact the fire service (in this case, the Fire Department of the Generalitat de Catalunya-GRAF) responsible for the area where the wildfire occurs. The information they can provide to us regarding the wildfire perimeter, the cartography, the wildfire characteristics, the extension of fire-affected area... is very important. On the field, it is necessary to make a round of the perimeter of the wildfire in order to observe the extension of the burned areas. In this round, the wildfire dynamics should be checked and identified, as well as the areas where the fire severity has been greater and the areas where previously, or after wildfire, forest actions have been carried out and could be of interest. In the wildfire area, the information that corresponds to the geology, geomorphology, vegetation, burned forest structures and other important aspects, is created and subsequently detailed in each sampling area. All this work is necessary to find suitable and representative areas for carrying out the samplings. In addition, it is necessary to carry out a bibliographic search that will be repeated in an extended way during office work. This search task aims to find information about manuscripts within the same study area, studies with similar soil or plant characteristics or studies where similar forest management practices have been described. Doing all this work, it is possible to decide which zones are suitable for the analysis in our study.

The collected soil samples try to represent the soil conditions in a specific location, this is why sampling points must be selected carefully to be representative of the soil type. Once in the sampling selected area, it is necessary to select a valid experimental design. In this Doctoral Thesis we have used several experimental designs. The selection of the experimental design is based on the needs of each study area, the temporality of each of the studies and the analysis that can be carried out in each study. The first experimental design is composed of a grid of thirty points, each point separated by two meters, forming a rectangular piece of soil of eighteen meters long by four broad (72 m2) (Francos et al., 2016a). This type of experimental design provides great reliability to the results of the laboratory work and allows subsequent studies about the spatial distribution of the data thanks to the creation of the plot or sampling grid. The weakness of this type of experimental design is the large amount of economic and human resources needed. The second experimental design is based on the use of a transect of 10 sampling points. This type of experimental design allows us to make comparisons between different areas without excessive work load or a waste of money. However, this experimental design does not allow a study of the spatial distribution of the data (Francos et al., 2018b). The last experimental design is very similar to the previous one and is based on the sampling of nine samples, separated by two meters each and one forming a shape of three contiguous triangles. This design shows the same strengths and the same limitations as the previous experimental design, but can be used in areas of smaller than 20 meters hillsides that the previous experimental design will not have fit (Francos et al., 2018a; Francos et al., 2018c; Francos et al., 2018d). From a theoretical point of view all the experimental designs are valid due to the fact that in field experiments 7 samples are considered a valid “n”, representative and statistically reliable. In addition, from the point of view not only theoretical but practical, it has been possible to observe with the use of the different experimental designs that all of them are valid and reliable and their use will depend on the choice of the researchers.

The experimental design used is very important in order to avoid subsequent problems in sampling soil fieldwork, such as pseudo-replicate or the edge effect. The pseudo-replicates in our case are not performed. They consist in sampling areas with the same conditions as those found in the sampled areas and carrying out sampling according to the same sampling methodology and timing. Its purpose is to verify the reliability of the data obtained in the analysis of samples collected from the study area. Another problem to avoid in the sampling areas is the edge effect. This effect is characterized by the attenuation of the results between two areas with different management, different severities ... due to the proximity between them in the collecting samples moment. To avoid this edge effect, samples are taken in the central area of each of the plots that we want to analyse, avoiding the proximity to the extremes and therefore the interference of other factors in the results obtained from the samples extracted from each zone.

During the collection of samples to ensure that the studied areas were comparable and therefore representative, it is necessary to select areas with the same topographic, soil characteristics, with the same vegetation and with similar fire severity. Before the sampling, annotations and pictures were taken to consult later on aspects such as orientation, slope, location coordinates, soil and vegetal description and characterization prior to wildfire of the sampled areas. In addition, notes were taken about fire severity according to ashes colour (Pereira et al., 2012; Úbeda et al., 2009), and the diameter of the branches that remain in the trees (García-Jiménez et al., 2017; Moreno and Oechel, 1989; Úbeda et al., 2006) and in the case of pines also for the pineapples found in each tree. In addition, a control zone is located, unburned, close to the burned and sampled areas, which has the same characteristics to be able to compare and evaluate the impact of fire and understand the forest pre-fire conditions. In areas with different pre- or post-fire management, another burned but unmanaged area is also sampled to assess the effects of forest management on soil properties. In these areas, the experimental design, the procedures, and the number of samples should be the same as in the areas that are intended to be analysed in order to be comparable.

During the sampling, different materials are needed, such as shovel, geological hammer, plastic bags, tape measure, metal ring with a maximum width of 5 cm, etc. The depth of the collection of samples is very important since it will determine the reliability of the obtained data. Some authors (Badía et al., 2017;

Badía-Villas et al., 2014) have studied which is the appropriate depth to sample. As in the selection of experimental design there is no absolute truth about which is better; this depth will vary based on what you intend to study. These samplings are made of 0-2.5 cm, 0-5 cm, 0-10 cm, even 0-15 cm, among others. Many authors point out the soil low thermal conductivity (0-2 cm) which is a determining factor if the analysis only aims the direct impact of fire on the soil and therefore the maximum sampling depth should be 0-2.5 cm. In our case, it is intended to analyse the direct and indirect effects of fire on soil and therefore in the different studies the depth of the sampling has been 0-5 cm (to see the extended information consult section 1.3 of this Thesis). In addition to the direct and indirect impact, the objective of the Thesis has been to analyse the impact of forest management. This management and post-fire plant regrowth (tree roots, shrubs and herbaceous plants), has a great influence at a greater depth of 2.5 cm, for this reason it has been chosen to collect samples 0-5 cm, in order to include the direct impact of the fire, the indirect and analyse the forest management carried out at each moment.

2. Laboratory work

Once the samples are carried to the laboratory, they are opened and remain at room temperature and humidity for seven days (20ºC and 50% relative humidity) (Doerr et al., 2002). Subsequently, the task of manual sieving of the samples was carried out. To do this, different sieves were used as < 2 mm sieve (with the soil that we will subsequently carry out many of our analyses), the 4 mm sieve and the 4.8 mm sieve, from which the aggregates were collected, which we use later to carry out the aggregate stability analysis. To have all the available soil sieved to less than 2 mm, the aggregates were broken using a roller. This task requires care to not break the gravels and account them erroneously in the sand fraction. Part of the sample <2 mm will be crushed using a ball mill at 40 Hz during 2 minutes. The purpose of this process is to pulverize and homogenize the samples in a size smaller than 0.2 mm to prepare it for the subsequent analyses of total nitrogen, inorganic carbon and organic carbon.

A. Analysis of soil physical properties

To measure the soil infiltration capacity, a Decagon mini-disk infiltrometer was used. This procedure is performed as designed by Zhang (1997) under laboratory conditions. In this case, the time required to infiltrate 80 ml of deionized water in the soil is measured and expressed as the infiltration ratio. The infiltrometer is composed of two chambers. The upper chamber controls the suction speed and the lower chamber contains the volume of water that remains to be infiltrated (80 ml at the beginning of the experiment). This second chamber has graduation marks on the cylinder to control the volume of water that remains to be infiltrated at any time. In the lower part of the infiltrometer there is a metallic porous mesh that prevents water leakage. Once that part is in contact with the soil sample, the experiment begins. The infiltrometer is made of a polycarbonate tube with a semi-permeable metal disc in the base. The method measures the time required to infiltrate 80 ml of deionized water. The measurements are recorded every 5 ml and the results are expressed in mm / h.

Water repellency or hydrophobicity is a property of the soil that describes its affinity to water. To determine the degree of repellency (SWR), the test called WDPT (Water Drop Penetration Time) was used (Wessel, 1988). The samples were placed in Petri dishes and introduced at constant temperature and humidity (21ºC and 50% humidity) in a methacrylate cabinet for 7 days. After that time, begin the experiment that consists in depositing on the surface of each of the samples 6 drops of deionized water (0.05 ml) and measuring the time each drop takes to fully infiltrate. With this, what is intended is to check the repellency of water samples in seconds and classify following Doerr (1998) into four classes according to the severity of the repellency: no repellent (<5 seconds), low repellency (6-60 seconds), strong repellence (61-600 seconds) and severe repellency (601-3600 seconds).

To determine the structural stability of the soil, aggregates of 4 to 4.8 mm of each of the samples were used. To determine the stability of the aggregates (AS), the TDI (Ten Drop Impact) method has been used, which consists of subjecting 10 aggregates of each sample to a "bombing" of 10 drops in each of them and measuring the amount of material disaggregated (Low, 1954). The objective of this experiment is to test the dispersion capacity of soil aggregates. The 10 aggregates are placed on top of a 2.8 mm sieve and the material that passes through the sieve was counted as a percentage (%) of disaggregated material (Úbeda et al., 1990). Therefore, the higher the percentage of disaggregated material, the lower the stability of the soil aggregates.

B. Analysis of soil chemical properties

The total nitrogen was analysed using a machine called Flash 11 Series (Thermo-Fisher, Milan) and the calculations were made with the Eafer 300 software. The procedure of this machine focuses on the introduction of a few milligrams (between 2000 and 5000) of soil of each of our samples and the addition of between 700 and 1000 milligrams vanadium pentoxide V2O5 that acts as a catalyst for the process (Olsen et al., 1954).

To determine the content of organic matter (SOM) and inorganic soil carbon, the LOI (Loss on ignition) method described by Heiri et al. (2001) were used. This method is based on the following procedure: we weigh the porcelain glass that later was burned, introduce one gram of soil and place it in the oven at 60ºC for 24 hours. Subsequently, it is weighed again and reintroduced in the oven, but this time at 105 ° C for 12 hours. This procedure is performed to extract the moisture content of the sample. Once that time has passed, they are reweighed and placed in the furnace at 550ºC for 4 hours. This step determines the % of organic matter existing in the soil since the variation in weight will be due to the combustion and loss of organic matter. Finally, the sample is introduced into the furnace at 950 ° C for 2 hours, after that time it is removed and when it has cooled it is weighed. This last step is performed to check the amount of inorganic carbon in the sample. Both organic matter and inorganic carbon are expressed in %.

For the soil pH and the soil electrical conductivity determination, they have been analysed at the same time following the potentiometric method. For this, 20 grams of soil of the fine fraction (<2 mm) of each sample have been used and 50 ml of deionized water has been added. Subsequently, it is stirred for half an hour on a magnetic stirrer and left to stand for half an hour before the measurement. The measurements were made with a pH-meter and a conducti-meter both with a glass electrode. The electrical conductivity is expressed in μS / cm.

Regarding the amount of extractable elements, an extraction with ammonium acetate was used following Knudsen et al. (1982). In this experiment, we tried to determine the soil extractable amounts of calcium, magnesium, sodium, potassium, aluminum, iron, zinc, manganese, boron, silicon and sulfur. For this experiment, 5 grams of soil of each sample were weighed and 100 ml of ammonium acetate were added, it is stirred manually and left in a turner for 24 hours. Once that time has passed, the samples were filtered and measured by Induction Plasma (ICP) taking into account that the sample must be diluted 1/10 with deionized water. The results were subsequently expressed in ppm or its equivalent mg / kg of soil.

The determination of the available phosphorus in each sample was carried out following the Osmond-Bray method. This consists in the extraction of phosphorus from the soil using a solution of ammonium fluoride (A) and hydrochloric acid (B). The procedure is as follows: one gram of soil of the sample (<2 mm) was weighed, placed in a 100 ml Erlenmeyer flask, and 20 ml of extractant solution was added, stirred for 40 seconds and filtered. Subsequently, 2 ml of the filtered sample, 10 ml of deionized distilled water, 10 drops of solution A prepared previously and 10 drops of solution B, also prepared beforehand, were added in a test tube. The results were expressed in ppm or its equivalent mg / kg of soil.

C. Analysis of soil microbiological properties

Just after sieving the samples, part of the samples (<2 mm) are kept in a cold room at a constant temperature between 2 and 4 ºC. Regarding the carbon analysis of the microbial biomass (C_{mic}), the analysis was based on the method described by Vance et al. (1987). The procedure is as follows: 5 grams of each sample were placed in open laboratory canisters in a bell and in the middle of all the cans an Erlenmeyer with 50 ml of liquid chloroform. Little by little, a vacuum pump was extracted in existing oxygen creating a chloroform atmosphere and left covered with a black plastic for 24 hours. After that period, the cans were removed from the bell and 20 ml of potassium sulphate were added to each one and stirred in a mechanical turner for one hour. Subsequently, the samples were filtered and 3 ml of the filtered sample were extracted and placed in a test tube and 1 ml of potassium dichromate and 1.5 ml of sulphuric acid were added. The measurement was carried out by ultraviolet which gives the results in glucose ppm and later it was necessary to convert them to mg (of glucose) / kg (of soil).

To perform the measurement of basal soil respiration (BSR) the method proposed by Anderson (1982) has been followed. To this end, 20 grams of each sample were placed in laboratory glass bottles and were wetted to 55%. Subsequently it was submerged at 25 ° C for 4 days covered with a black plastic. These bottles were connected to a computer that manages the air that traspasses the sand and that makes CO2 measurements every 6 hours. The results were obtained with a multiple sensor respirometer (Micro-Oxymax, Columbus, OH, USA). The results of the graph are the CO2 released and the CO2 accumulated in each sample and the final results were given in CO2 / kg of soil.

3. Office work

Once all the analyses have been made and transcribed to the Excel spreadsheet, the last phase of methodology with the data begins. With the data created it is very interesting to calculate ratios and indexes to check the soil conditions and understand the impacts that the soil of each zone has received. Within these ratios and indexes, we find: C/N (it is a ratio that is made with the organic carbon obtained in each sample and the total nitrogen), SPAR (for the calculation of this index, the SAR sodium absorption index has been used and the PAR potassium absorption and it is very interesting to check the soil degradation), Ca: Al (it is a ratio in which calcium and aluminum were used) as well as Ca: Mg (uses calcium and magnesium) that are two ratios commonly used in burnt areas to check the soil recovery capacity. In addition, different indexes have been done mixing chemical and microbiological properties such as: C_{mic} / C (microbial biomass carbon and organic carbon were used), BSR / C (basal soil respiration and soil organic carbon were used) and the BSR / C_{mic}, better known as qCO2 (it uses basal soil respiration and the microbial biomass carbon and it is commonly used to detect soil stress, environmental impacts such as a forest fire, the efficient use of coal and the ecological forest succession). These indexes have been used in many scientific studies previously to the use in ours (Anderson, 2003; Fritze et al., 1993; Insam and Domsch, 1988; Sarah, 2004; Morugan-Coronado et al., 2015).

In the office work phase, based on the annotations taken during fieldwork, the type of soil is identified and its concordance with the classification of the Soil Survey Staff or the WRB of the FAO. Another task is to find the nearest weather station to the sampling area to be able to use the data from that station as a reference to know the post-fire weather conditions in the study area.

Regarding the statistical treatments carried out, these have been the following: first, once the data are in the database, the tests called Shapiro-Wilk and Levene's were carried out with the purpose of checking the normality and homogeneity of the data. In cases where data have shown to follow a Gaussian distribution and fits to normality, an ANOVA was carried out; and in cases where the data did not follow a normal distribution nor were they homogeneous, a test called Kruskal-Wallis were carried out. To detect significant differences with a confidence level of 95%, the Tukey post-hoc test was applied. In the studies presented in this Doctoral Thesis, in some cases a one-way ANOVA has been carried out to compare the same area over time or different zones in a single sampling moment. In other cases, a two-way ANOVA has been done in order to identify differences to compare different areas and different moments of sampling or management and to be able to establish which of these conditions produces the differences in soil properties. All of them have been carried out using different programs to obtain both the graphs and the statistical tables. These programs have been Excel, SPSS 23.0, CANOCO 4.5, XLSTAT and GraphPad PRISM 5.

MATERIAL Y MÉTODOS

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1. Trabajo de Campo

Antes de la realización del trabajo de campo, es necesario realizar diferentes tareas. Primero de todo es necesario contactar con el servicio de bomberos (en este caso Bomberos de la Generalitat de Cataluña-GRAF) responsable de la zona donde se ha producido el incendio. La información que nos pueden aportar a cerca del perímetro de incendio, la cartografía, las características del incendio, extensión del mismo… es de suma importancia. Ya en el campo es necesario realizar un recorrido por el perímetro del incendio para poder apreciar la extensión y las zonas quemadas. En este recorrido por el perímetro se comprueban y se identifican las dinámicas de incendio, las zonas donde la severidad ha sido mayor y menor y las zonas donde de manera previa o posterior al incendio se han realizado actuaciones forestales y que pueden ser interesantes para ser estudiadas. En la zona del incendio se realiza y anota la información relevante a la geología, geomorfología, vegetación, estructuras forestales quemadas y otros aspectos importantes, que a posteriori, se detallan específicamente en las zonas de muestreo. Todo este trabajo es necesario para encontrar zonas adecuadas y representativas para la realización de los muestreos. Además, es necesario llevar a cabo una búsqueda bibliográfica que se repetirá de manera ampliada durante la etapa del trabajo de gabinete donde se tratará de encontrar información de los artículos existentes de esa zona de estudio a analizar, los artículos donde se muestren características edáficas o forestales similares y en los que se han realizado prácticas forestales similares a las que se quieren evaluar en cada estudio. A partir de este trabajo es posible decidir qué zonas se ajustan adecuadamente a lo que queremos analizar en nuestro estudio.

La muestra recogida intenta representar condiciones del suelo en un lugar concreto, por ello los puntos de muestreo deben ser seleccionados cuidadosamente para que sean verdaderamente representativos del tipo de suelo. Una vez en la zona seleccionada para realizar el muestreo, es necesario seleccionar un diseño experimental válido. En esta tesis doctoral se han utilizado varios diseños experimentales. La selección del diseño experimental utilizado se basa en las necesidades de cada zona de estudio, la temporalidad de cada uno de los estudios y los análisis que se quieran realizar en cada estudio. El primero de los diseños experimentales está compuesto por una malla de treinta puntos, separados por dos metros cada punto formando una parcela rectangular de dieciocho metros de largo por cuatro de ancho (72 m^2) (Francos et al., 2016a). Este tipo de diseño experimental aporta gran fiabilidad a los resultados obtenidos en el trabajo de laboratorio y permite realizar estudios posteriores sobre la distribución espacial de los datos gracias a la creación de la malla o retícula de muestreo. La debilidad de este tipo de muestreo es la gran cantidad de recursos tanto económicos como humanos necesarios para trabajar cada una de las parcelas muestreadas ya que supone un ingente trabajo de análisis de laboratorio. El segundo diseño experimental utilizado se basa en el uso de un transecto de 10 puntos de muestreo separados cada punto por dos metros de distancia. Este tipo de diseño experimental nos permite hacer comparaciones entre diferentes zonas sin una excesiva carga de trabajo ni un ingente gasto de dinero. Este diseño experimental no permite realizar un estudio sobre la espacialización de los datos obtenidos ya que no se utiliza una malla de muestreo (Francos et al., 2018b). El último diseño experimental es muy similar al anterior y se basa en la recogida de nueve muestras, separadas por dos metros cada muestra formando tres triángulos contiguos. Este diseño muestra los mismos puntos fuertes y las mismas limitaciones que el diseño experimental previo, pero puede ser utilizado en zonas de laderas pequeñas inferiores a 20 metros en los que el anterior diseño experimental no tendría cabida (Francos et al., 2018a; Francos et al., 2018c; Francos et al., 2018d). Todos los diseños experimentales desde el punto teórico son válidos ya que en experimentos en campo se considera necesario un mínimo de 7 muestras para que sea válido, representativo y estadísticamente fiable. Además, desde el punto de vista no solo teórico sino práctico, se ha podido comprobar con la utilización de los diferentes diseños experimentales que todos ellos son válidos y fiables y su uso dependerá de la elección por parte de los investigadores según los análisis que se pretendan realizar.

El diseño experimental utilizado es muy importante para poder evitar problemas subsecuentes al muestreo de suelo en el campo, como pueden ser las pseudo-réplicas o el efecto borde. Las pseudo-réplicas en nuestro caso no se llevan a cabo. Éstas consisten en el muestreo de zonas con las mismas condiciones a las encontradas en las zonas muestreadas y realizando los muestreos siguiendo la misma metodología y temporalidad de muestreo. Su finalidad es comprobar la fiabilidad de los datos obtenidos en el análisis de las muestras recogidas de la zona de estudio. Otro de los problemas a evitar en las zonas de muestreo es el efecto borde. Este efecto se caracteriza por la atenuación de los resultados entre dos zonas con diferentes manejos, diferentes severidades… por la cercanía entre ellas a la hora de recoger las muestras. Para evitar este efecto borde se muestrea en las zonas centrales de cada una de las parcelas que queremos analizar evitando la cercanía a los extremos y por lo tanto la interferencia de otros factores en los resultados obtenidos de las muestras extraídas de cada zona.

Durante la recogida de las muestras para asegurarse que las zonas estudiadas eran comparables y por lo tanto representativas se han tratado de seleccionar zonas con las mismas características topográficas, edáficas, con la misma vegetación y donde la severidad del fuego haya sido lo más similar posible. Previamente al muestreo se realizaron anotaciones y fotografías para poder consultar posteriormente sobre aspectos como la orientación, la pendiente, coordenadas de localización descripción y caracterización de suelo y de la vegetación previa al incendio… de las zonas muestreadas. Además, se realizan anotaciones sobre la severidad del fuego en cuanto al color de las cenizas producidas (Pereira et al., 2012; Úbeda et al., 2009), y el diámetro de las ramas que quedan en los árboles (García-Jiménez et al., 2017; Moreno y Oechel, 1989; Úbeda et al., 2006) y en caso de pinos también por las piñas encontradas en cada árbol. Además, se localiza una zona control, no quemada, cercana a las zonas quemadas y muestreadas, que tenga las mismas características para poder comparar y evaluar el impacto del fuego y comprender las condiciones pre-incendio. En zonas con diferente gestión pre- o post-incendio, también se realiza el muestreo de otra zona quemada pero no gestionada para poder evaluar los efectos de la gestión forestal en las propiedades del suelo. En estas zonas tanto el diseño experimental como los procedimientos y el número de muestras serán las mismas que en las zonas que se pretenden analizar para con ello poder ser comparables.

A la hora de muestrear son necesarios diferentes materiales como pala, martillo geológico, bolsas de plástico, cinta métrica, anilla metálica de 5 cm de ancho como máximo, etc. La profundidad de la recogida de muestras es muy importante ya que determinará la fiabilidad de los datos obtenidos. Algunos autores (Badía et al., 2017; Badía-Villas et al., 2014) han estudiado cual es la profundidad adecuada para muestrear. Así como en la selección del diseño experimental no hay una verdad absoluta sobre cual es mejor; esta profundidad variara en base a lo que se pretenda estudiar. Estos muestreos se realizan de 0-2.5 cm, 0-5 cm, 0-10 cm incluso 0-15 cm entre otros. Muchos autores apuntan la baja conductividad térmica del suelo (0-2 cm) lo cual es un factor determinante si únicamente se quiere analizar el impacto directo del fuego en el suelo y por lo tanto la profundidad máxima de muestreo debe ser 0-2.5 cm. En nuestro caso, se pretende analizar los efectos directos e indirectos del fuego en el suelo y por lo tanto en los diferentes estudios la profundidad del muestreo ha sido 0-5 cm (para ver la información ampliada consultar la sección 1.3 de esta Tesis). Además del impacto directo y el indirecto, el objetivo de la tesis ha sido analizar el impacto de la gestión forestal. Esta gestión y la regeneración vegetal post-incendio (raíces de árboles, arbustos y las herbáceas), tiene gran influencia a mayor profundidad de 2.5 cm por ello se ha optado por recoger las muestras hasta los 5 cm, para con ello englobar el impacto directo del incendio, el indirecto y analizar la gestión forestal realizada en cada momento.

2. Trabajo de Laboratorio

Una vez las muestras son trasladadas al laboratorio son abiertas y permanecen a temperatura y humedad ambiente durante siete días (20 ºC y 50 % de humedad relativa) (Doerr et al., 2002). Posteriormente se realiza la tarea de tamizado manual de las muestras. Para ello se utilizarán los tamices de < 2 mm (con el suelo que posteriormente realizaremos gran cantidad de nuestros análisis), el tamiz de 4 mm y el de 4.8 mm entre los cuales se recogerán los agregados que posteriormente utilizaremos para realizar el análisis de estabilidad de agregados. Para tener todo el suelo disponible menor de 2 mm se romperán los agregados utilizando un rodillo. Esta tarea requiere cuidado para no romper las gravas y contabilizarlas erróneamente en la fracción arenas. Parte de la muestra < 2 mm será machacada o triturada usando un molinillo de bolas a 40 Hz durante 2 minutos. La finalidad de este proceso es pulverizar y homogeneizar las muestras en un tamaño inferior a 0.2 mm para prepararla para los posteriores análisis de nitrógeno total, carbono inorgánico y carbono orgánico.

A. Análisis de las propiedades físicas del suelo

Para medir la capacidad de infiltración del suelo se usa un infiltrómetro mini-disk Decagon. Este procedimiento se realiza según diseñó Zhang (1997) en condiciones de laboratorio. En este caso se trata de medir el tiempo necesario para infiltrar 80 ml de agua des-ionizada en el suelo y se expresa como ratio de infiltración. El infiltrómetro está compuesto por dos cámaras. La cámara superior controla la velocidad de succión y la cámara inferior contiene el volumen de agua que falta por infiltrarse (80 ml al inicio del experimento). Esta segunda cámara tiene marcas de graduación en el cilindro para controlar el volumen de agua que falta por infiltrarse en cada momento. En la parte inferior del infiltrómetro hay una malla porosa metálica que evita las fugas de agua. Una vez esa parte está en contacto con la muestra de suelo, comienza el experimento. El infiltrómetro está construido de un tubo de policarbonato con un disco metálico semi-permeable en la base. El método mide el tiempo necesario en infiltrarse 80 ml de agua desionizada. Las mediciones se anotan cada 5 ml y los resultados se expresan en mm/h.

La repelencia del agua o hidrofobicidad es una propiedad del suelo que describe su afinidad por el agua. Para la determinación del grado de repelencia (SWR) se ha utilizado el test denominado WDPT (Water Drop Penetration Time) (Wessel, 1988). Las muestras fueron colocadas en placas Petri y se introdujeron a humedad y temperatura constante (21 ºC y 50 % de humedad) en un armario de metacrilato durante 7 días. Pasado ese tiempo comenzó el experimento que consiste en depositar en la superficie de cada una de las muestras 6 gotas de agua desionizada (0.05 ml) y medir el tiempo que tarda en infiltrarse completamente cada gota. Con esto lo que se pretende es ver la repelencia de las muestras al agua en segundos y se clasificará según Doerr (1998) en cuatro clases según la severidad de la repelencia: nada repelente (< 5 segundos), repelencia baja (6-60 segundos), repelencia fuerte (61-600 segundos) y repelencia severa (601-3600 segundos).

Para determinar la estabilidad estructural del suelo, se utilizarán los agregados de 4 a 4.8 mm de cada una de las muestras. Para determinar la estabilidad de los agregados (AS) se ha utilizado el método TDI (Ten Drop Impact) que consiste en someter a 10 agregados de cada muestra a un “bombardeo” de 10 gotas en cada uno de ellos y medir la cantidad de material desagregado (Low, 1954). El objetivo de este experimento es comprobar la capacidad de dispersión de los agregados del suelo. Los 10 agregados se colocan encima de un tamiz de 2.8 mm y el material que traspase el tamiz se contabilizará como porcentaje (%) de material disgregado (Úbeda et al., 1990). Por lo tanto, cuanto mayor sea el porcentaje de material disgregado, menor será la estabilidad de los agregados del suelo.

B. Análisis de las propiedades químicas del suelo

El nitrógeno total es analizado utilizando una máquina denominada Flash 11 Series (Thermo-Fisher, Milan) y los cálculos son realizados con el software Eafer 300 (Pereira et al., 2012). El procedimiento de esta máquina se centra en la introducción de unos miligramos (entre 2000 y 5000) de suelo de cada una de nuestras muestras y la adición de entre 700 y 1000 miligramos de pentóxido de vanadio V2O5 que actúa como catalizador del proceso (Olsen et al., 1954).

Para determinar el contenido de material orgánica (SOM) y de carbón inorgánico del suelo se ha seguido el método LOI (*Loss on ignition*) descrito por Heiri et al. (2001). Este método se basa en el siguiente procedimiento: se pesa el vaso que posteriormente será quemado, se introduce un gramo de suelo y se coloca en la estufa a 60 ºC durante 24 horas. Posteriormente se pesa de nuevo y se vuelve a introducir en la estufa, pero esta vez a 105 ºC durante 12 horas. Este procedimiento se realiza para extraer el contenido de humedad de la muestra. Una vez pasa ese tiempo se vuelven a pesar y se introducen en la mufla a 550 ºC durante 4 horas. Este paso determinará el % de materia orgánica existente en el suelo ya que la variación de peso será debido a la combustión y pérdida de la materia orgánica. Por último, se introduce la muestra en la mufla a 950 ºC durante 2 horas, una vez pasado ese tiempo se saca y cuando se haya enfriado se pesa. Este último paso se realiza para comprobar la cantidad de carbono inorgánico existente en la muestra. Tanto materia orgánica como carbono inorgánico se expresan en %.

En cuanto a la determinación del pH y la conductividad eléctrica del suelo han sido analizados de manera conjunta siguiendo el método potenciométrico. Para ello se han utilizado 20 gramos de suelo de la fracción fina (< 2 mm) de cada muestra y se ha añadido 50 ml de agua desionizada. Posteriormente se agita durante media hora en un agitador magnético y se deja reposar media hora antes de la medición. Las mediciones se realizan con un pH-metro y un conductímetro ambos con electrodo de vidrio. La conductividad eléctrica se expresa en µS/cm.

Respecto a la cantidad de elementos extraíbles se ha utilizado una extracción con acetato amónico siguiendo a Knudsen et al. (1982). En este experimento se trata de determinar las cantidades de calcio, magnesio, sodio, potasio, aluminio, hierro, zinc, manganeso, boro, silicio y azufre extraíbles en el suelo. Para ello se pesan 5 gramos de suelo de cada muestra y se le añaden 100 ml de acetato amónico, se agita manualmente y se deja en un volteador durante 24 horas. Una vez pasado ese tiempo se filtran las muestras y se mide por Plasma de Inducción (ICP) teniendo en cuenta que la muestra debe ser diluida a 1/10 con agua desionizada. Los resultados posteriormente se expresarán en ppm o su equivalente mg/kg de suelo.

La determinación del fósforo asimilable en cada muestra se realiza siguiendo el método Osmond-Bray. Éste consiste en la extracción del fósforo del suelo mediante una solución de floruro amónico (A) y ácido clorhídrico (B). El procedimiento es el siguiente: se pesa un gramo de suelo de la muestra (< 2 mm), se coloca en un Erlenmeyer de 100 ml, y se añaden 20 ml de solución extractante, se agita durante 40 segundos y se filtra. Posteriormente se añaden en un tubo de ensayo 2 ml de la muestra filtrada, 10 ml de agua destilada desionizada, 10 gotas del reactivo A preparado anteriormente y 10 gotas del reactivo B, también preparado anteriormente. Los resultados se expresan en ppm o su equivalente mg/kg de suelo.

C. Análisis de las propiedades microbiológicas del suelo

Justo después del tamizado, parte de las muestras (< 2 mm) se conservan en una cámara frigorífica a una temperatura constante entre 2 y 4 ºC. Respecto al análisis del carbono de la biomasa microbiana (C_{mic}) el análisis se basa en el método descrito por Vance et al. (1987). El procedimiento es el que sigue: Se colocan 5 gramos de cada una de las muestras en botes de laboratorio abiertos en una campana y en medio de todos los botes un Erlenmeyer con 50 ml de cloroformo líquido. Poco a poco con una bomba de vacío se extrae en oxigeno existente creando una atmósfera de cloroformo y se deja tapado con un plástico negro durante 24 horas. Pasado ese periodo de tiempo se extraen los botes de la campana y se le añade a cada uno 20 ml de sulfato potásico y se agita en un volteador durante una hora. Posteriormente se filtran las muestras, se extraen 3 ml de la muestra filtrada y se ponen en un tubo de ensayo y se le añaden 1 ml de dicromato potásico y 1.5 ml de ácido sulfúrico. La medición se realiza mediante ultravioleta que da los resultados en ppm de glucosa y posteriormente hay que convertirlos a mg (de glucosa) / kg (de suelo).

Para realizar la medición de la respiración basal edáfica (BSR) se ha seguido el método propuesto por Anderson (1982). Para ello se colocaron 20 gramos de cada muestra en botellas de cristal de laboratorio y se humedeció al 55 %. Posteriormente se sumergió a 25 ºC durante 4 días tapado con un plástico negro. Dichas botellas se encuentran conectadas a un ordenador que gestiona el aire que pasa por la dierita y que realiza mediciones de CO_2 cada 6 horas. Los resultados obtenidos se obtuvieron con un respirómetro de sensor múltiple (Micro-Oxymax, Columbus, OH, USA). Los resultados de la gráfica son el CO_2 liberado y el CO_2 acumulado en cada muestra y los resultados finales se dan en CO_2/kg de suelo.

3. Trabajo de Gabinete

Una vez todos los análisis han sido realizados y transcritos a la hoja de cálculo Excel comienza la última fase de metodología con los datos. Con los datos creados es muy interesante realizar ratios e índices para comprobar el estado del suelo y entender los impactos que ha recibido el suelo de cada zona. Dentro de estos ratios e índices realizados encontramos: C/N (es una ratio que se realiza con el carbono orgánico obtenido en cada muestra y el nitrógeno total), SPAR (para el cálculo de este índice se ha usado el índice SAR de absorción de sodio y el índice PAR de absorción de potasio y es muy interesante para comprobar la degradación de los suelos), Ca:Al (es un ratio en el que usan el calcio y el aluminio) así como Ca:Mg (utiliza calcio y magnesio) son dos ratios utilizados en zonas quemadas para ver la capacidad de recuperación del suelo. Además, se han realizado diferentes índices mezclando propiedades químicas y microbiológicas como son: C_{mic}/C (se utiliza el carbono de la biomasa microbiana y el carbono orgánico), BSR/C (se utiliza la respiración basal edáfica y el carbono orgánico del suelo) y el BSR/C_{mic}, más conocido como qCO2 (utiliza la respiración basal edáfica y el carbono de la biomasa microbiana y es comúnmente utilizado para detectar estrés edáfico, impactos ambientales como un incendio forestal, el uso eficiente el carbón y la sucesión ecológica del bosque). Estos índices han sido utilizados en diferentes artículos científicos previamente al uso en los nuestros (Anderson, 2003; Fritze et al., 1993; Insam and Domsch, 1988; Sarah, 2004; Morugan-Coronado et al., 2015).

En la fase de trabajo de gabinete, a partir de las anotaciones tomadas en el campo, de identifica el tipo de suelo y su concordancia con la clasificación del *Soil Survey Staff* o del *WRB* de la FAO. También se trata de encontrar la estación meteorológica más cercana a la zona del muestreo para poder utilizar los datos de dicha estación como referencia para conocer las condiciones meteorológicas post-incendio.

Respecto a los tratamientos estadísticos realizados estos han sido los siguientes: primero de todo, una vez los datos están en la base de datos, se realizan los test denominados Shapiro-Wilk y Levene's con la finalidad de comprobar la normalidad y homogeneidad de los datos. En los casos en que los datos han mostrado seguir una distribución gaussiana y se ajustan a la normalidad, se lleva a cabo una ANOVA, y en los casos en los que los datos no siguen una distribución normal ni eran homogéneos, se realiza un test llamado Kruskal-Wallis. Para detectar diferencias significativas con un nivel de confianza del 95 % se aplica el test Tukey post-hoc. En los estudios presentados en esta Tesis Doctoral, en algunos casos se ha llevado a cabo una ANOVA de una dirección para comparar una misma zona a lo largo del tiempo, o diferentes zonas en un solo momento de muestreo. En otros casos, se ha aplicado una ANOVA de doble dirección para poder identificar diferencias comparando diferentes zonas y diferentes momentos de muestreo o gestiones y poder establecer cuál de estos condicionantes produce las diferencias en las propiedades del suelo. Todos ellos se han realizado combinando diferentes programas para la realización tanto de los gráficos como de las tablas estadísticas. Estos programas han sido Excel, SPSS 23.0, CANOCO 4.5, XLSTAT y GraphPad PRISM 5.

CHAPTER 1

CHAPTER 1. POST-FIRE SOIL MANAGEMENT

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Post-fire soil management

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Abstract

Soils are an important natural capital and can be harshly affected by high severity wildfires. The capacity of soil to recover to the degradation imposed by fire disturbance depends upon fire history, ash properties, topography, post-fire weather, vegetation recuperation and post-fire management. These factors are interdependent, and can increase or decrease the effects of high severity fires in soil degradation. Normally, ecosystems are resilient to fire disturbance and a scenario of no intervention should be considered. Post-fire management should be carried out in specific areas more sensible to degradation. Post-fire interventions such as mulching are important to decrease soil degradation, whereas salvage logging increase it. Overall, the management options that we choose can trigger, or reduce the positive and negative impacts on soil.

Keywords: Soil degradation, High fire severity, post-fire management.

1. Introduction

Soils are the base of life on earth and affect directly or indirectly all forms of life. They provide an important number of ecosystem services essential to human life and fundamental to address future global challenges such as climate change, water scarcity, loss of biodiversity, air pollution land degradation and food security. Soil resource is not renewable at human scale, is the major carbon stock in the world, act as water filters and reservoir and is the basis for vegetation growth, food production and human activities development (Brevik et al. 2015; Keesstra et al. 2016).

Fire is a global phenomenon that shape the ecosystems, as we know today and depend very much of vegetation growth that would not be possible without soils. With exception of Artic and Antarctica environments, fire visited with more or less frequency the other world ecosystems. Throughout history, the ecosystems adapted to this disturbance and created mechanisms of protection to this disturbance. Fossil evidences showed that fire appeared in the Siluric period (420 million years ago) with the appearance of vascular plants. Fire was used by hominins in Africa back to 1.6 million years ago, and became part of their technological repertoires around 300.000 -400.000 years ago. The homo manipulation of fire induced unprecedented changes in the earth ecosystems, including in soils. Fire is considered the seventh soil-formation factor and was an important tool to create agriculture and grazing fields. Some ecosystems (e.g Mediterranean) cannot be understood without the presence of fire. Despite the importance of fire for several ecosystems, since the beginning of the 20th century, and especially since the 1950-1960, strong and effective suppression policies were established to protect ecosystems from fire. These measures coupled with the rural exodus contributed to an increase in the accumulation of fuel in wildland areas. Land abandonment in countryside, lead to investment in industrial monospecific plantations with extreme flammable species (e.g Pinus and Eucalyptus) and the increase of the frequency, duration and intensity of summer drought periods increased the vulnerability of wildlands to fire. All these factors coupled are responsible for the high frequency of mega-fires with high severity, responsible for the loss of human lives and millions of euros of direct and indirect losses (Certini 2014; Pereira et al. 2016; Santin and Doerr 2016).

Ecosystems developed a strong adaptation to fire disturbance, however the increase of fire recurrence and the high severity of these fires reduced strongly the capacity of vegetation to recover. One of the most important factors that hamper this recuperation is the soil status that after fire may be strongly affected, especially in areas where high severity fires are frequent (Tessler et al. 2016; Taboada et al. 2017). In this work, we will make an overview about the impacts of fire on soil, the factors that influence post-fire ecosystems recuperation, the methods used for restoration and the impacts of these practices.

2. Fire impacts on soils

Fire impacts on soil properties can be direct and indirect. Direct impacts are related to the heat impact. With the exception of smoldering fires and burn logs and piles, the direct impact of fire on soil properties normally is short. Since soils are a bad conductor of energy, the heating induced by fire is restricted to the first centimeters of soil (Badia et al. 2014). This effect depends on the type of soil affected, texture, conditions pre-fire (e.g moisture content), type of vegetation and structure (e.g. density and connectivity), ecosystem affected, fire intensity, severity, recurrence, meteorological conditions during the fire and topography (e.g. slope and aspect) that influences fire behavior (Caon et al. 2014; Francos et al. 2018). The indirect impacts of fire are related to the ash-bed effects, degree of vegetation recuperation, post-fire weather patterns, topography and post-fire management (developed in the next section).

Low severity fires can have beneficial impacts on soil properties, since the temperatures reached are not high and the loss of nutrients by volatilization and with smoke are reduced. One good example of these type of fires are grassland fires (Pereira et al. 2017) or prescribed fires carried out during autumn and winter season for landscape management (Alcaniz et al. 2018). The impact on soil cover is minimal and the soil heating is negligent, therefore, overland flow and soil erosion are reduced compared to high severity wildfires (Shakesby et al. 2015). As consequence of the production of ash rich in carbon, there is an increase of soil organic matter, pH, electrical conductivity and extractable cations such as calcium, magnesium, sodium and some forms of nitrogen such as ammonia, important for vegetation recuperation (Alcaniz et al. 2018). In sandy soils, the incorporation of ash is facilitated and the impacts on deeper layers is high (Bodi et al. 2014). A potential negative of low severity fires is the production of hydrophobic ash that can increase temporarily soil water repellency until the hydrophobic compounds be leached (Bodi et al. 2011).

Contrary to low-severity fires, high-severity fires, combust a large amount of fuel, have extremely negative impacts on soil. One of the most important impacts of these types of fire is the strong reduction of soil cover (Pereira et al. 2015). The high temperatures at soil surface reduce importantly the quantity, and induce important transformations of organic matter composition (Merino et al. 2018). The major changes on soil organic matter composition occur at temperatures between 250-450 °C (Araya et al. 2018). The consumption of soil organic matter is also responsible for soil aggregates destruction. However, high temperatures may increase aggregation as consequence of recrystallization of some iron and aluminum oxyhydroxides, if present in soil. (Mataix-Solera et al. 2011) (Figure 1). The oxidation of iron minerals at high temperatures increase soil redness and increase the proportion of sand (Ulery and Graham, 1993; Cancelo-Gonzalez et al. 2014). Fire can increase soil hydrophobicity as consequence of the volatilization of hydrophobic compounds from organic matter that are condensed onto soil particulates, increasing water repellency. Burning process decreases aliphatic series of alkanes and alkenes and increases the relative content of aromatic compounds and fatty acids, increasing the soil hydrophobicity after heating (Attassanova and Doerr, 2011; Jimenez-Morillo et al. 2017). However, at high temperatures, hydrophobicity disappears. Bellow 175 °C, no changes are observed in water repellency, increasing at temperatures around 200 °C. At temperatures between 280-400 °C hydrophobicity is destroyed (Doerr et al. 2005). High severity fires can destroy hydrophobicity in previous water repellent soils (Varela et al. 2015) (Figure 1).

Severe wildfires volatilize high amounts of carbon and nitrogen since these nutrients start to vaporize at reduced temperatures (~200 °C). Carbon and nitrogen are totally lost in soils burned at temperatures higher than 550 °C (Gray and Dighton, 2006) (Figure 1). Wildfire temperatures can go up to 1100 °C, thus nutrients such as calcium, magnesium, aluminum or manganese that need high temperatures to be volatilized, can only be lost by evacuation with ash with smoke. Other elements with low temperatures of volatilization such as Potassium (>774°C), Sodium (>880°C) and Phosphorous (>700°C), may be lost by direct volatilization (Wotton et al. 2011; Pereira et al. 2011, 2015).

High severity fires increase importantly soil pH, because of organic acid denaturation and the increase of sodium and potassium oxides, carbonates and hydroxides (Ulery et al. 1993) and electrical conductivity as consequence of the mineralization induced by fire on organic matter (Pereira et al. 2011). Soil extractable anions, that are highly soluble from burned material. The increase of pH, favour the solubility of some cations such as calcium, magnesium, sodium and potassium, and reduces others such as cooper and zinc (Escuday et al. 2015) (Figure 1). Phosphorous solubility is limited in high severity wildfires as consequence of the pH increase above 8, which limit the solubility of this element, and if in solution, is easily precipitated with Calcium Aluminium or Iron (Caon et al. 2014). Soil microbes are very sensitive to high severity wildfires, especially in the topsoil, where soil heating is more intense. The thermal shock induced by fire and changes the activity, size and composition of the microbial biomass. At temperatures higher than 70-80 °C, the majority of soil microbes are destroyed and at temperatures between 115 and 150 °C they complete disappear (Hart et al. 2005; Mataix-Solera et al. 2009) (Figure 1).

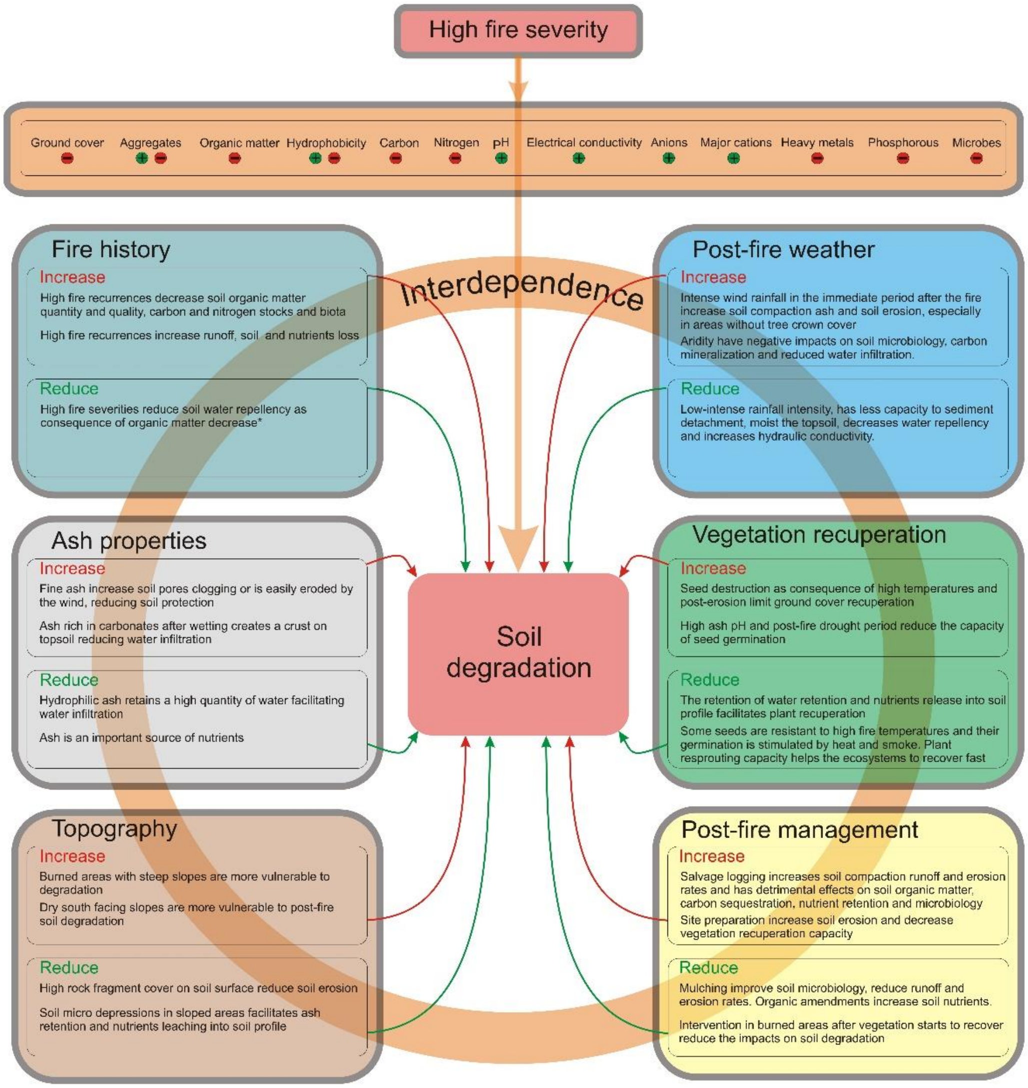

Figure 1. High severity fire effects and factors that control post-fire recuperation.

3. Post fire soil management: natural vs human factors

Pre-fire land use and management is a crucial aspect to understand the degree of the impacts of a wildfire on soil. The highest wildfire severities and the highest environmental, economic and social damage occurs in areas where there is a lack of management or exist industrial plantations of Pinus and Eucalyptus. In these areas is where the soil degradation is high because of the impacts of the high temperatures on the volatilization of nutrients (that will limit the capacity of ecosystem restoration) and the impacts of soil heating. Ecosystems with high biodiversity are less vulnerable to wildfire impacts, recover faster, and act as a barrier to fire spreading (Granados et al. 2016; Gomez-Gonzalez et al. 2018)

Post-fire impact on soil degradation extension and magnitude depends on the fire history, environmental conditions of the fire affected area and the human management. These interactions are very difficult to generalize because they are interdependent (Figure 1). Fire history have important implications on soil properties. For example, high fire frequencies increase the flammability of vegetation (Pausas, 2015). This increases the vulnerability of those areas to wildfire occurrence creating a fire recurrence cycle that can be disastrous for soils in these ecosystems and induce a negative trend in soil fertility (Mayor et al. 2016). The areas affected by high fire recurrences have a lower quantity and quality of soil organic matter and store less nutrients such as nitrogen and carbon (**Pellegrini et al. 2018). It was also reported that decrease importantly soil biota (Buscardo et al. 2015) and increase total runoff, the amount of nutrients present on overland flow, organic matter losses and sediment transport (Hosseini et al. 2016; 2017). On the other hand, fire recurrence may decrease soil water repellency, responsible for the increase of overland flow and soil erosion (Keesstra et al. 2017). However, this may not be considered a positive aspect, because the loss of organic matter is extremely negative for soil quality. Hydrophobicity occurs naturally in several types of soil, and to some degree, is not considered a negative aspect since can increase key soil aspects such as aggregate stability (Zheng et al. 2016). In addition, the study of (Keesstra et al. 2017) was carried out in the Mediterranean environment and may not be applied to other ecosystems (Figure 1).

After a fire, ash covers soil surface providing a temporary protection against erosion processes. However, the capacity to protect the soil depends on the severity it is produced. Normally the ash created at high temperatures is very fine, can be incorporated into soil profile, and clog the soil pores reducing water infiltration. Since the ash produced at high fire severities is light, is also easy to be removed and increase the bare soil surface. In order words, does not provide an efficient protection. In addition to this, the ash produced at high temperatures is rich in carbonates that after wetting creates a crust on topsoil, increasing overland flow. These factors contribute to soil degradation. On the other hand, the ash produced at high severity is very hydrophilic and can retain water and leaching an important amount of key nutrients into soil profile. In post-fire environments, the increase of soil nutrients is often attributed to ash, which have a positive impact on soil nutrient status soil cover reestablishment (Bodi et al. 2014; Pereira et al. 2015) (Figure 1).

Soil is more vulnerable to degradation in steep slopes, especially after high-severity fires. Steep slopes a the most vulnerable to the effects of the flames, especially if the fire line heads upslope, since the fire heat convection has the capacity of pre-heat the fuel and reduce the vegetation moisture before the combustion. These effects are especially important in dry south faced slopes that are also the most vulnerable to soil erosion. This is also aggravated by the slow vegetation recuperation (developed further) (Viedema et al. 2015; Perrault et al. 2017). On the other hand, high rock fragments cover on soil surface on soil surface mitigate soil erosion and soil micro depressions retain ash, facilitating the incorporation of nutrients into soil profile (Pereira et al. 2015; Zhang et al. 2016) (Figure 1).

Post-fire wind and rainfall intensity influence the degree of soil degradation in fire-affected areas. Ash and soil erosion and nutrient losses are high, when intense rainfall periods (normally accompanied by strong winds) occur in the immediate period immediate after the fire. During this time, soils are very sensible to any kind of disturbance, especially because vegetation did not start to recover. The lack of soil protection and the sparse ash cover increases the impacts of raindrops on soil compaction and facilitates sediment detachment. These conditions are very common in Mediterranean environments because of the occurrence of high intensity and short duration summer thunderstorms and autumn intense rainfall periods. The impact of wind and rainfall drops on topsoil is intensified if tree crown is totally combusted (Francos et al. 2016; Esposito et al. 2017). High intensity precipitation events are also responsible for large mass movements such as debris flow and geomorphic changes (Kampf et al. 2016; *Staley et al. 2017). On the other hand, aridity changes post-fire microbial activity, reduce carbon mineralization rate, the availability of nutrients (Hinojosa et al. 2016) and increases the probability of debris flow occurrence, since is associated to low infiltration rates (as consequence of soil water hydrophobicity increase) and high runoff (*Sheridan et al. 2016). In this context, the excess and the lack of rainfall may increase soil degradation in soils affected by a high fire severity. Low intensity rainfall events can be more beneficial to reduce soil degradation since the capacity to compact the soil and sediment detachment is reduced, the soil moisture increases fast, and soil water repellency is destroyed, facilitating water infiltration and the input of nutrients (Figure 1).

High fire severities affect seed abundance in soil and this is especially observed when high temperatures are combined with prolonged periods of contact. Normally, there is a reduction of seeding germination rate with increasing temperature, and at 300 °C most of the seeds are killed (Emery et al. 2011; Reyes et al. 2015). The existence of large bare soil areas, increases the vulnerability of seed to erosion (Yu et al. 2017). In addition to seed destruction by soil heating and erosion, post-fire high soil pH as consequence of ash leachates, decreases the rate of seed germination capacity (Reyes et al. 2016). This is intensified by the occurrence of drought periods (Harvey et al. 2016). High fire recurrences decrease the capacity of vegetation to recover and change dramatically plant community, diversity and composition (Tessler et al. 2016). All these aspects contribute importantly to soil degradation.

Despite the reduction of seeds in the soil, the seeds of some species can resist to high temperatures and can be stimulated by heat and smoke, especially shrubland and heatland vegetation (Hall et al. 2017). This can be enhanced by ash nutrients released (after pH decrease from high alkaline levels) into soil profile. The impacts of soil heating is complex and can be different according to the ecosystem affected. Mediterranean species are more adapted than other ecosystems, very likely due to the long-presence of fire in these ecosystems (Fichino et al. 2016). Other important aspect that reduce soil degradation after high severity wildfires is the capacity of some plants to resprout. This is a key adaptation of plants to fire disturbance, allowing the rapid recover of vegetation, which is an evidence of the high resilience of the ecosystems to this disturbance (*Pausas and Keelley, 2017) (Figure 1).

Ecosystems developed strategies to respond to high fire severity fires and mitigate the effects of fire on soil degradation. This disturbance is temporal and the recuperation depends upon the resilience of the ecosystems. For this reason, in great part of the cases, scenario of no intervention in burned areas is the most appropriated to reduce soil degradation in fire affected areas. This does not mean that in specific areas where the there is a high risk of soil erosion, debris flow or potential environmental, economic and social losses downstream. Several types of intervention are applied in fire-affected areas such as salvage logging, site preparation, mulching (e.g. straw, wood chips, hydromulching), seeding, erosion barriers, channel treatments. The table S1a and b (supplementary material) summarize the impacts of post fire managements on soil properties in experimental and wildland fires during the first year after the fire, when the soil is more vulnerable to disturbance. The restoration techniques carried out in areas where experimental fires where applied showed in the majority of the cases no improvement. Only some major cations increased in mulched plots. This shows that the investment in these techniques in areas affected by low or moderate fire severity had little or no effect on soil properties.

If some intervention is needed, this should be carried out in areas affected by high fire severity. However, some can be considered disastrous for soil and increase their degradation. Salvage logging in many cases is carried out in the immediate period after the fire to provide some economic benefit to the owner, since wood value decreases with the time. However, these economic benefits depend upon on the market conditions and the investments need to remove the wood (Vallejo et al. 2009). These apparent benefits can be overwhelmed by the subsequent soil degradation (e.g. soil compaction, aggregate stability and organic matter loss, reduction of carbon sequestration, increase, soil erosion) and the negative impacts in vegetation recuperation, richness and diversity (Figure 1, Table S1b and S2 Supplementary material) (Leverkus et al. 2014:*Garcia-Orenes et al. 2017; Waggenbrenner et al. 2017). Other practice that is detrimental for soil is site preparation (post- fire mounding, ripping and tree plantation) increase substantially soil erosion rates and reduce vegetation recuperation capacity (Slesak et al. 2015). These practices represent always a negative disturbance to soil.

On the other hand, mulching practices are a sustainable practice to reduce high fire severities impacts on soil degradation. The impacts of mulching are not so evident in soil chemical properties, but some improvements were observed in soil microbiology (especially with a low dose of wheat straw). Organic ammenedments are more appropriated to restore soil nutrients. Mulching as a high capacity to reduce overland flow and soil erosion. Post-fire management practices such as wood removal, should be carried out when vegetation show signs to recuperate. If possible one or two years after the fire (Figure 1, Table S1b and S2).

4. Final remarks and conclusion

Soils can be importantly affected by fire. Low fire severities can be beneficial to soil properties, while high fire can produce long-term negative impacts on soil degradation. The degree of impact is very dependent on the pre-fire land use and is most visible, where there is a lack of management and in areas where there are large areas of monospecific plantations. In these areas, high severity fires have the highest negative impacts on soil. The effect of high severity fires on soil degradation depends upon fire history of the site, the type of ash produced, topography, post fire weather, degree of vegetation recuperation. These factors are interdependent and can increase or mitigate the effects of fire. However, the fact is that many ecosystems are resilient to fire and recover fast raises the question that a scenario of no intervention can be considered. This does not mean that intervention is not need in specific situations. The human intervention and the way that we manage areas affected by high severity fires is crucial for the increase or decrease of soil degradation. Sustainable measures such as mulching are important contrary to salvage logging techniques that causes an important soil degradation. Overall, Post-fire management options, can trigger or reduce soil degradation in areas affected by high severity fire.

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References

*Brevik EC, Cerda A, Mataix-Solera J, Pereg L, Quinton JN, Six J, Van Oost K: **The interdisciplinary nature of soil**. Soil 2015,1:117-129.

Keesstra S, Bouma J, Wallinga J, Tittonell P, Smith P, Cerda A, Montanarella L, Quinton JN, Pachepsky Y, van der Putten WH, Bardgett RD, Moolenaar S, Mol G, Jansen B, Fresco LO: **The signficance of soils and soil science towars realization of the United Nations Sustainable Development Goals**. Soil 2016,2:111-128.

Certini G: Fire as a soil-forming factor. Ambio 2014,43:191-195.

Pereira P, Rein G, Martin D: **Editorial: Past and present post-fire environments**. Sci Total Environ 2016,573:1275-1277.

Santin C, Doerr S: **Fire effects on soils: the human dimension. Phil Trans R Soc B, 371: 20150171

Tessler N, Wittenberg L, Greenbaum N: **Vegetation cover and species richness after recurrent fires in the Easthern Mediterranean ecosystem of mount Carmel, Israel**. Sci Total Environ 2016,572:1395-1402.

Taboada A, Tarrega R, Marcos E, Valbuena L, Suarez-Seoane S, Calvo L: **Fire recurrence and emergency post-fire management influence seedling recruitment and growth by altering plant interactions in fire-prone ecosystems.** Forest Ecol Manag 2017,402:63-75.

Badia D, Marti C, Aguirre AJ, Aznar JM, Gonzalez-Perez J, De la Rosa JM, Leon J, Ibarra P, Etcheverria T: **Wildfire effects on nutrients and organic carbon of a Rendzic Phaeozem in NE Spain: Changes at cm-scale topsoil.** Catena 2014,113:267-275.

Caon L, Vallejo VR, Coen RJ, Geissen V: **Effects of wildfire on soil nutrients in Mediterranean ecosystems**. Earth Sci Rev 2014, 139:47-58.

Francos M, Úbeda X, Pereira P, Alcaniz M: **Long-term impacts of wildfire on soils exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula)**. Sci Total Environ 2018, 615:664-671.

Pereira P, Cerdà A, Úbeda X, Mataix-Solera J, Arcenegui, V, Zavala L: **Modelling the impacts of wildfire on ash thickness in a short-term period**. Land Degrad Develop 2015, 26:180-192.

Bodi M, Martin DA, Santin C, Balfour V, Doerr SH, **Pereira P,** Cerdà A, Mataix-Solera J: **Wildland fire ash: production, composition and eco-hydro-geomorphic effects**. Earth Sci Rev 2014, 130,103-127.

Pereira P, Francos M, Úbeda X, Brevik E: **Fire impacts in European grasslands.** In Wildfires: Perspectives, Issues and Challenges of the 21st Century. Edited by Bento-Goncalves A, Vieira A, Rosario Costa M, Aranha J. Nova Publishers:2017:1-27.

Alcaniz M, Outeiro L, Francos M, Úbeda X: **Effects of prescribed fires on soil properties: A review**. Sci Total Environ 2018, 613-614:944-957.

Shakesby RA, Bento CPM, Ferreira CSS, Ferreira AJD, Stoof CR, Urbanek E, Walsh RPD: **Impacts of prescribed fire on soil loss and soil quality: An assessment based on experimentally-burned catchment in Portugal.** Catena 2015,128:278-293.

Bodi M, Mataix-Solera J, Doerr S, Cerdà A: **The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content**. Geoderma 2011,160:599-607.

Merino A, Fonturbel MT, Fernandez C, Chavez-Vergara B, Garcia-Oliva F, Vega JA: **Inferring changes in soil organic matter in post-wildfire soil burn severity levels in a temperate climate**. Sci Total Environ 627:622-632.

Araya SN, Fogel ML, Berhe AA: **Thermal alteration of soil organic matter properties: A systematic study to infer response of Sierra Nevada climosequence soils to forest fires**. Soil 2017,3,31-44.

Mataix-Solera J, Cerdà A, Arcenegui V, Jordan A, Zavala LM: **Fire effects on soil aggregation: A review**. Earth Sci Rev 2011,109:44-60.

Ulery AL, Graham RC: **Forest fire effects on soil color and texture.** Soil *Sci Soc Am* J 1993,57:135-140.

Cancelo-Rodriguez J, Cachaldora C, Diaz-Fierros F, Prieto B: **Colorimetric variations in burnt granitic forest soils in relation to fire severity**. Ecol Ind 2014,46,92-100.

Atassanova I, Doerr S: **Changes in soil organic compond associated with heat-induced increases in soil water repellency**. Eur J Soil Sci 2011,62:519-532.

Jimenez-Morillo NT, Spangenberg JE, Miller AZ, Jordan A, Zavala LM, Gonzalez-Vila FJ, Gonzalez-Perez JA: **Wildfire effects on lipid composition and hydrophobicity of bulk soiland soil fractions under Quercus suber cover (SW-Spain)**. Environ Res 2017,159:394-405.

Doerr SH, Llewellyn CT, Douglas P, Morley CP, Mainwaring KA, Haskings C, Johnsey L, Ritsema C, Stagnitti F, Allinson C, Ferreira AJD, Keizer JJ, Ziogas AK, Diamantis J: **Extraction of compounbds associated with water repellency in sandy soils of different origin**. Aust J Soil Res 2005,43:225-237.

Varela ME, Benito E, Keizer JJ: **Influence of wildfire severity on soil physical degradation in two pine forest stands of NW Spain**. Catena 2015,133:342-348.

Gray DM, Dighton J: **Mineralization of forest litter nutrients by heat and combustion**. Soil Biol Biochem 2006,38:1469-1477.

Wotton BM, Gould JS, McCaw WL, Cheney NP, Taylor SW **Flame temperature and residence time of fires in dry eucalypt forest**. Int J Wildland Fire 2011, 21: 270-281.

Ulery A, Graham RC, Amrhein C: **Wood-ash composition and soil pH following intense burning**. Soil Sci 1993,156:358-364.

Escudey M, Arancibia-Miranda N, Pizarro C, Antilen M: **Effects of ash from forest fires on leaching in volcanic soils**. Catena 2015,135:383-392.

Hart SC, DeLuca T, Newman GS, Mackenzie MD, Boyle SI: **Post-fire vegetative dynamcs as drivers of microbial community structure and function in forest soils**. For Ecol Manage 2005,220:166-184.

Mataix-Solera J, Guerrero C, Garcia-Orenes F, Barcenas GM, Torres MP: Forest fire effects on soil microbiology. Edited by Cerda A, Robichaud P. Science Publishers;2008:134-175.

Granados ME, Vilagrosa A, Chirino E, Vallejo VR: Reforestation with resprouter species to increase diversity and resilience in Mediterranean pine forests. For Ecol Manage 2016,326:231-240.

Gomez-Gonzalez S, Ojeda F, Fernandes PM: **Portugal and Chile: Longing for sustainable forestry while rising from the ashes**. Environ Sci Policy 2018, 81:104-107.

Pellegrini AFA, Ahlstrom A, Hobbie SE, Reich PB, Nieradzik LP, Staver AC, Scharenbroch BC, Jumpponen A, Anderegg WRL, Randerson JT, Jackson RB: **Fire frequency drives decadal changes in soil carbon and nitrogen and ecosystems productivity. Nature 2018,553,194-198.

Pausas JG: **Alternative fire-driven vegetation states**. J Veg Sci 2015,26:4-6.

Mayor AG, Valdecantos A, Vallejo VR, Keizer JJ, Bloem J, Baeza J, Ginzalez-Pelayo O, Macado AI, de Ruiter PC: **Fire-induced pine woodland to shrubland transitions in Southern Europe may promote shifts in soil fertility**. Sci Tot Environ 2016,573:1232-1241.

Buscardo E, Rodriguez-Echeverria S, Freitas H, De Angelis P, Pereira JS, Muller LA: **Contrasting soil fungal communities in Mediterranean pine forests subjected to different wildfire frequencies**. Fungal Divers 2015, 70:85-99.

Hosseini M, Keizer JJ, Gonzalez-Pelayo O, Prats SA, Ritsema C, Geissen V: **Effects of fire frequency on runoff, soil erosion, and loss of organic matter at the micro-plot scale in north-central Portugal**. Geoderma 2016,269:126-137.

Hosseini M, Geissen V, Gonzalez-Pelayo O, Serpa D, Machado AI, Ritsema C, Keizer JJ: **Effects of fire occurence and recurrence on nitrogen pools and phosphorous losses by overland flow in maritime pine plantations in north-central Portugal**. Geoderma 2017,289:97-106.

Keesstra S, Wittenberg L, Maroulis J, Sambalino F, Malkinson D, Cerda A, Pereira P: **The influence of fire history, plant species and post-fire management on soil water repellency in a Mediterranean catchment: The Mount Carmel range**. Catena 2017,149:827-866.

Zheng W, Morris KE, Lehmann A, Rillig M: **Interplay of soil water repellency, soil aggregation and organic carbon. A meta analysis.** Geoderma 2016,283:39-47.

Viedema O, Quesada J, Torres I, De Santis A, Moreno JM: **Fire severity in a large fire in a Pinus Pinaster forest is highly predictable from burning conditions, stand surface, and topography**. Ecosystems 2015,18:237-250.

Perrault LM, Yager EM Aalto R: **Effects of gradient, distance, curvature and aspect on steep burned and unburned hillslope soil erosionand deposition**. Earth Surf Proc Land 2017,42:1033-1048.

Zhang Y, Zhang M, Niu J, Li H, Xiao R, Zheng H, Bech J: **Rock fragments and soil hydrological process: Significance and progress**. Catena 2016,147:153-166.

Francos M, Pereira P, Alcaniz M, Mataix-Solera J, Ubeda X: **Intense rainfall impact on soil properties in fire affected areas**. Sci Tot Environ 2016,572:1353-1362.

Esposito G, Matano F, Molisso F, Ruoppolo G, Di Benedetto A, Sacchi M: **Post-fire erosion response in a watersheed mantled by volcaniclastic deposits, Sarno Mountains, Southern Italy**. Catena 2017, 152:227-241.

*Staley DM, Negri JA, Kean JW, Laber JL, Tillery AC, Youberg AM: **Prediction of spatially explicit rainfall-intensity-duration thresholds for post-fire debris-flow generation in the western United States**. Geomorphology 2017,278:149-162.

Kampf SK, Brogan DJ, Schmeer S, MacDonald LH, Nelson PA: **How do geomorphic effects of rainfall vary with storm type and spatial scale in a post-fire landscape**. Geomorphology 2016,273:39-51.

Hinojosa MB, Parra A, Laudicina VA, Moreno JM: **Post-fire soil functionality and microbial community structure in a Mediterranean shrubland subjected to experimental drought**. Sci Tot Environ 2016,573:1178-1189.

*Sheridan GJ, Nyman P, Langhans C, Cawson J, Noske PJ, Oono A, Van Der Sant R, Lane PNJ: **Is aridity a high-order control on the hydro-geomorphic response of burned landscapes**. Int J Wildland Fire 2016,25:262-267.

Reyes O, Garcia-Duro J, Salgado J: **Fire affects soil organic matter and emergence of *Pinus radiata* seedlings**. Ann For Sci 2015,72:267-275.

Emery SM, Uwimbabazi J, Flory SL: **Fire intensity effects o seed germination of native and invasive eastern deciduous forest understory plants**. For Ecol Manage 2011,261:1401-1408.

Yu WJ, Jiao JY, Chen Y, Wang DL, Wang N, Zhao HK: **Seed removal due to overland flow on abandoned slopes in the Chinese hilly guillied loess plateau region**. Land Degrad Develop 2017,28:274-282.

Reyes O, Kaal J, Gago R, Bernal J, Garcia-Duro J, Basanta M: **The effects of ash and black carbon (biochar) on germination of different tree species**. Fire Ecol 2015, 11: doi: 10.4996/fireecology.1101119

Harvey BJ, Donato DC, Turner MG: **High and dry: post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches**. Global Ecol Biogeogr 2016,25:655-669.

Tessler NT, Sapir Y, Wittenberg L, Greenbaum N: Recovery of Mediterranean vegetation after recurrent fires: Insight from the 2010 forest fires on Mount Carmel, Israel. Land Degrad Develop 2016,27:1424-1431.

Hall SA, Newton RJ, Holmes PM, Gaertner M, Esler KJ: **Heat and smoke pre-treatment of seeds to improve restoration of an endangered Mediterranean climate type**. Austral Ecol 2017,42:354-366.

Fichino BS, Dombroski JR, Pivello VR, Fidelis A: **Does fire trigger seed germination in the Neotropical Savannas? Experimental tests with six Cerrado species**. Biotropica 2016,48:181-187.

*Pausas J, Keeley JE: **Epicormic resproting in fire-prone ecosystems**. Trends Plant Sci 2017,22:1008-1015.

Supplementary Material

Table S1. Post-fire managements impact on soil properties according to different managements in the immediate period after (0-1/1.5 year). A) Experimental fires and B) Wildland fires. Comparisons (impact) related to no intervention (control plot). Significant decrease (red), no changes (yellow) and significant increase (green). Only studies after 2010 were considered.

Ecosystem	Severity	Burned species	Treatment	Soil property	Impact	Reference
Mediterranean Scrubland	Moderate	*Ulex europaeus* L., *Pteridium aquilinum* (L.) Kuhn., *Ulex* gallii Planch., *Daboecia cantabrica* (Huds.) K. Koch and *Pseudoarrenhaterum longifolium* Rouy	Seeding	pH		Barreiro et al. (2015)
				Electric conductivity		
				Water retention		
				Total carbon		
				Soluble carbon		
				Total nitrogen		
				Inorganic nitrogen		
				Total biomass PLFA		
				Fungal PLFA		
				Bacterial PLFA		
				Bacterial PLFA		
				Bacterial PLFA G+		
				Fungal/bacterial PLFA		
				Bacterial G−/G+ PLFA		
				Cyclopropyl/monoenoic precursors		
				Saturated/monounsaturated		
			Straw mulch	pH		
				Electric conductivity		
				Water retention		
				Total carbon		
				Soluble carbon		
				Total nitrogen		
				Inorganic nitrogen		
				Total biomass PLFA		
				Fungal PLFA		
				Bacterial PLFA		
				Bacterial PLFA		
				Bacterial PLFA G+		
				Fungal/bacterial PLFA		
				Bacterial G−/G+ PLFA		
				Cyclopropyl/monoenoic precursors		
				Saturated/monoun saturated		
Mediterranean Scrubland	Moderate	*Ulex europaeus* L., *Pteridium aquilinum* (L.) Kuhn., *Ulex* gallii Planch., *Daboecia cantabrica* (Huds.) K. Koch and *Pseudoarrenhaterum longifolium* Rouy	Mulching	pH water		Gomez-Rey et al. (2013)
				pH KCl		
				Total carbon		
				δ13carbon		
				Total nitrogen		
				δ15Nitrogen		
				Ammonium		
				Nitrate		
				Extractable Aluminium		
				Extractable Sodium		
				Extractable Potassium		
				Extractable Magnesium		
				Extractable Calcium		
				Extractable Phosphorous		
				Extractable Iron		
				Extractable Managanese		
				Extractable Copper		
				Extractable Zinc		
				Extractable Barium		
				Extractable Cobalt		
			Seeding	pH water		
				pH KCl		
				Total carbon		
				δ13carbon		

				Total nitrogen		
				δ15N		
				Ammonium		
				Nitrate		
				Extractable Aluminium		
				Extractable Sodium		
				Extractable Potassium		
				Extractable Magnesium		
				Extractable Calcium		
				Extractable Phosphorous		
				Extractable Iron		
				Extractable Managanese		
				Extractable Copper		
				Extractable Zinc		
				Extractable Barium		
				Extractable Cobalt		
Temperate	Low	*Ulex europaeus* L., *Pteridium aquilinum* (L.) Kuhn., *Ulex* gallii Planch., *Daboecia cantabrica* (Huds.) K. Koch and *Pseudoarrenhaterum longifolium* Rouy	Seeding	Moisture		Fonturbel et al. (2012)
				pH water		
				Electric conductivity		
				Water retention at field capacity		
				Total carbon		
				Microbial carbon[1]		
				Microbial Carbon[2]		
				Basal respiration		
				Glucosidase		
				Urease		
				Phosphatase		
				AWCD[3]		
				MR[4]		
				H'[5]		
			Straw mulch	Moisture		
				pH water		
				Electric conductivity		
				Water retention at field capacity		
				Total carbon		
				Microbial carbon[6]		
				Microbial Carbon[7]		
				Basal respiration		
				Glucosidase		
				Urease		
				Phosphatase		
				AWCD[8]		
				MR[9]		

[1] Fummigation extraction

[2] Substrate induced respiration

[3] Average well color development

[4] Microbial richness

[5] Shannon–Weaver diversity index

[6] Fummigation extraction

[7] Substrate induced respiration

[8] Average well color development

[9] Microbial richness

Ecosystem	Severity	Burned species	Treatment	Soil property	Impact	Reference
Mediterranean	Moderate to High	*Pinus sylvestris* L. plantation and shrubs (*Erica* spp., *Vaccinium myrtillus* L., *Pterospartum tridentatum* Willk. and *Cistus* spp)	Mulching	pH water		Gomez-Rey et al. (2014)
				pH KCl		
				Ammonium		
				Nitrate		
				Extractable Sodium		
				Extractable Potassium		
				Extractable Magnesium		
				Extractable Calcium		
				Extractable Phosphorous		
				Extractable Aluminium		
				Extractable Iron		
				Extractable Managanese		
				Extractable Copper		
				Extractable Zinc		
				Extractable Cobalt		
			Seeding	pH water		
				pH KCl		
				Ammonium		
				Nitrate		
				Extractable Sodium		
				Extractable Potassium		
				Extractable Magnesium		
				Extractable Calcium		
				Extractable Phosphorous		
				Extractable Aluminium		
				Extractable Iron		
				Extractable Managanese		
				Extractable Copper		
				Extractable Zinc		
				Extractable Cobalt		
Mediterranean	High	*Pinus sylvestris* L. plantation and shrubs (*Erica* spp., *Vaccinium myrtillus* L., *Pterospartum tridentatum* Willk. and *Cistus* spp)	Straw mulch	Moisture		Diaz-Ravina et al. (2012)[10]
				Aggregate stability		
				Water repellency		
				Water retention at field capacity		
				pH water		
				pH KCl		
				Electric conductivity		
				Organic carbon		
				Total nitrogen		
				Microbial carbon		
				Respiration		
				qCO2		
				Bacterial activity		
				Urease activity		
				Glucosidase activity		
			Rye seeding	Moisture		
				Aggregate stability		
				Water repellency		
				Water retention at field capacity		
				pH water		
				pH KCl		
				Electric conductivity		
				Organic carbon		
				Total nitrogen		
				Microbial carbon		
				Respiration		
				qCO2		
				Bacterial activity		
				Urease activity		
				Glucosidase activity		
Mediterranean	High Severity	Shrubland (dominated by *Ulex* spp.)	Mulching. Wheat straw (low-dose)	Soil respiration		Barreiro et al. (2016)
				Fungal growth		
				Bacterial growth		
				Fungal biomass		

				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Wheat straw (low-dose)	Soil respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Coconut fibre (low-dose)	Soil respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
Mediterranean	High Severity	Shrubland (dominated by *Ulex* spp.)	Mulching. Coconut fibre (low-dose)	Respiration		Barreiro et al. (2016)
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Coconut fibre (high-dose)	Respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Eucalyptus bar strands (low-dose)	Respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Eucalyptus bar strands (high-dose)	Respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
			Mulching. Eucalyptus wood chips (low-dose)	Respiration		
				Fungal growth		
				Bacterial growth		
				Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
				Respiration		
				Fungal growth		
				Bacterial growth		

[10] Results from 16 weeks after the fire

			Mulching. Eucalyptus wood chips (high-dose)	Fungal biomass		
				Total PLFA		
				Fung PLFA		
				Fungal/bacterial PLFA index		
				G^-/G^+		
Mediterranean	High	Herbaceous (*Bituminaria bituminosa* L. and *Lotus* species), woody species (*Cistus monspeliensis* L., *Calycotome spinosa* L. and *Erica arborea* L.) and *Quercus suber* L.	Young compost	Total organic carbon		Guenon and Gros (2015)[11]
				pH water		
				Total nitrogen		
				Total phosphorous		
				C/N ratio		
				C/P ratio		
				Ammonium		
				Nitrate		
				Phosphate		
				Microbial biomass		
				Respiration		
			Intermediate compost	Total organic carbon		
				pH water		
				Total nitrogen		
				Total phosphorous		
				C/N ratio		
				C/P ratio		
				Ammonium		
				Nitrate		
				Phosphate		
				Microbial biomass		
				Respiration		
			Old compost	Total organic carbon		
				pH water		
				Total nitrogen		
				Total phosphorous		
				C/N ratio		
				C/P ratio		
				Ammonium		
				Nitrate		
				Phosphate		
				Microbial biomass		
				Respiration		
			Young compost	Catabolic diversity		Guenon and Gros (2016)[12]
				AWCD		
			Intermediate compost	Catabolic diversity		
				AWCD		
			Old compost	Catabolic diversity		
				AWCD		
Mediterranean	High	*Pinus Pinaster* and *Pinus nigra*	Salvage logging	Respiration		Marañón-Jiménez et al. (2011)
			Cut and plus lopping			
		Pinus sylvestris	Salvage logging			
			Cut and plus lopping			
Mediterranean	High	*Pinus sylvestris*	Salvage logging	Carbon sequestration		Serrano-Ortiz et al. (2011)

[11] Results from 10 months after compost amendment

Mediterranean	Moderate	*Pinus halepensis* and shrubs (*Quercus coccifera*, *Rosmarinus officinalis*, *Thymus vulgaris* and *Brachypodium retusum*)	Salvage logging	Aggregate stability		Garcia Orenes et al. (2017)
				Field capacity		
				Organic matter		
				Nitrogen		
				Phosphorous		
				Microbial biomass carbon		
				Respiration		
				Total DNA		Pereg et al. (2018)
				16S rRNA gene		
				nifH gene		
				nosZ gene		
				nirK gene		
				nirS gene		
				amoA-Arch gene		
				amoA-B		
Temperate	Moderate to High	*Pinus pinaster* Ait., *Eucalyptus globulus* Labill. and *Pinus radiata* D. Don. Understory (*Ulex* sp., *Erica* sp. and *Pterospartum tridentatum* (L.) Willk.).	No Mulching[12] +Savage logging	Shear strenght		Fernandez and Vega (2016)
			Mulching[2] + Savage logging	Penetration resistence		
Mediterranean (Eucalyptus plantation)	Moderate	Eucalyptus	Salvage logging (skid low)[13]	Bulk Density	0-5 cm	Malvar et al. (2017)
					5-10 cm	
				Penetration resistence		
				Shear strenght		
			Salvage logging (skid low)[14]	Bulk Density	0-5 cm	
					5-10 cm	
				Penetration resistence		
				Shear strenght		

[12] Eucalypt bark strands
[13] *"skid trails by the forwarder tractor with two passes"* (Malvar et al. 2017)
[14] *"with skid trails resulting from several passes by both the forwarder tractor and the feller-buncher"* (Malvar et al. 2017)

Table S2. Impacts of wildfire on overland flow and soil erosion according to different post-fire managements in the immediate period after (0-1/1.5 year). Comparisons (impact) related to no intervention (control plot). Significant decrease (green), no changes (yellow) and significant increase (red). Comparisons related to no intervention. Studies form 2010.

Type of fire	**Ecosystem**	**Fire severity**	**Plot /catchment size**	**Treatment**	**Overland flow**	**Soil erosion**	**Reference**
Experimental fire	Temperate	Moderate	300 m^2	Straw mulching	-		Vega et al. (2014)
				Seeding	-		
Wildfire	Mediterranean Eucalyptus plantation)	Moderate	0.28 m^2(control) and 0.27 m^2 (mulched)	Mulching		-	Prats et al. (2016)
			83 m^2 (control) and 128 m^2 (mulched)	Mulching		-	
Wildfire	Mediterranean (Eucalyptus plantation)	Moderate	0.28 m^2	Mulching			Prats et al. (2016)
				Polyacrylamide			
Wildfire	Mediterranean	High	500 m^2	Straw mulching	-		Fernandez et al.(2011)
				Wood-chip mulch	-		
				Cut-shrub barriers	-		
Wildfire	Mediterranean	High	80 m^2	Straw mulching	-		Diaz-Ravina et al. (2012)[15]
				Rye seeding	-		
Wildfire	Temperate	Moderate to high	80 m^2	No Mulching[16] + Savage logging	-		Fernandez and Vega (2016a)
				Mulching[4] +Savage logging	-		
Wildfire	Temperate	High	500 m^2	Straw mulching	-		Fernandez and Vega (2014)
				Eucalyptus bark strands	-		
Wildfire	Temperate	High	80 m^2	Straw mulching	-		Fernandez and Vega (2016b)
				Erosion barriers	-		
Wildfire	Mediterranean (Eucalyptus plantation)	Moderate	18 m^2	Salvage logging (skid low)[17]			Malvar et al. (2017)
			17 m^2	Salvage logging (skid high)[18]			
Wildfire	Hayman (Ponderosa pine and Douglas-fir), Hot Creek (Douglas-fir and Subalpine fir), Myrtle Creek (Douglas-fir and Ponderosa Pine) and School (Douglas-fir and Grand fir)	High	22-282 m$_2$ (average range)	Straw mulch	-		Robichaud et al. (2013a)[19]
				Wood	-		
				Hydromulch	-		
Wildfire	Hayman	High	Straw mulch (3.3 ha)	Straw mulch			Robichaud et al. (2013b)
			Hydromulch (5.2 ha)	Hydromulch			
	Cedar (Mediterranean)		Fully treated (2.1 ha)	Fully treated[20]			
			Partially treated (2.6 ha)	Partially treated[21]			

[15] Results from 16 weeks after the fire

[16] Eucalypt bark strands
[17] *"skid trails by the forwarder tractor with two passes"* (Malvar et al. 2017)
[18] *"with skid trails resulting from several passes by both the forwarder tractor and the feller-buncher"* (Malvar et al. 2017)

[19] This work was carried out in different environments and the results presented correspond to the average of the different treatments in all study sites.

[20] Hydromulch of wood and paper fiber with non-water soluble binder

[21] Hydromulch of wood and paper fiber with non-water soluble binder applied in 30 m contour strips

CHAPTER 2

CHAPTER 2. HOW CLEAR-CUTTING AFFECTS FIRE SEVERITY AND SOIL PROPERTIES IN A MEDITERRANEAN ECOSYSTEM.

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How clear-cutting affects fire severity and soil properties in a Mediterranean ecosystem

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Highlights

Pre-fire management, such as clear cutting, can influence in soil properties

The vegetation cut and left on the soil affects the severity of the fire

The time between management action and wildfire affect post-fire soil properties

Abstract

Forest management practices in Mediterranean ecosystems are frequently employed to reduce both the risk and severity of wildfires. However, these pre-fire treatments may influence the effects of wildfire events on soil properties. The aim of this study is to examine the short-term effects of a wildfire that broke out in 2015 on the soil properties of three sites: two exposed to management practices in different years – 2005 (site M05B) and 2015 (site M15B) – and one that did not undergo any management (NMB) and to compare their properties with those recorded in a plot (Control) unaffected by the 2015 wildfire. We analyzed aggregate stability (AS), soil organic matter (SOM) content, total nitrogen (TN), carbon/nitrogen ratio (C/N), inorganic carbon (IC), pH, electrical conductivity (EC), extractable calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K), microbial biomass carbon (C_{mic}) and basal soil respiration (BSR). In the managed plots, a clear-cutting operation was conducted, whereby part of the vegetation was cut and left covering the soil surface. The AS values recorded at the Control site were significantly higher than those recorded at M05B, whereas the TN and SOM values at NMB were significantly higher than those recorded at M05B. IC was significantly higher at M05B than at the other plots. There were no significant differences in C/N ratio between the analyzed sites. Soil pH at M05B was significantly higher than the value recorded at the Control plot. Extractable Ca was significantly higher at NMB than at both M05B and the Control, while extractable Mg was significantly lower at M05B than at NMB. Extractable K was significantly lower at the Control than at the three fire-affected plots. C_{mic} was significantly higher at NMB than at the Control. BSR, BSR/C and BSR/C_{mic} values at the fire-affected sites were significantly lower than those recorded at the Control. No significant differences were identified in C_{mic}/C. Overall, a comparison of the pre-fire treatments showed that NMB was the practice that had the least negative effects on the soil properties studied, followed by M15B, and that fire severity was highest at M05B due to the accumulation of dead plant fuel.

Keywords: pre-fire management; soil properties; fire severity; wildfire.

1. Introduction

Mediterranean ecosystems are characterized by the frequent occurrence and recurrence of fire (Tessler et al., 2016a, 2016b), which has, however, made these environments highly resilient to such events (Francos et al., 2016a; Keeley et al., 2012). Indeed, Mediterranean wildfires have been identified as one of the main disturbances suffered by these ecosystems (Bond and Van Wilgen, 1996), exacerbated by centuries of human impact and the use of fire for land management. More recently, land abandonment, fire suppression policies and climate change have increased the vulnerability of Mediterranean forests to wildfires (Pereira et al., 2016). In response, clear-cutting and prescribed fires are practices that seek to reduce the amount of fuel and, hence, the fire risk (Alcañiz et al., 2016; Cochrane et al., 2012; Corona et al., 2015). These forest management practices also reduce fire severity in Mediterranean environments (Corona et al., 2015; Keeley, 2009; Simard, 1991), an effect that can be maintained for up to 10 years. Yet, despite the medium- and long-term advantages of these practices, restrictions in the budget assigned to forestry management limit their implementation and, therefore, the capacity to mitigate the impact of wildfires (Martín-Alcon and Coll, 2016).

Recently, fire has been identified as a soil forming factor (Certini, 2014), and it has been widely reported as a temporary modifier of soil properties. The degree of this impact depends on pre-fire conditions, the type of soils affected and the severity of the fire. For example, low-to-moderate severity fires do not have a significant direct impact (e.g. heating) on a soil's physical and chemical properties (Francos et al., 2018). The changes recorded are attributable primarily to the ash produced by the fire in conjunction with the post-environmental conditions, including local topography and rainfall (Arcenegui et al., 2008; Certini, 2005; Francos et al., 2016a; Gordillo-Rivero et al., 2014; Pereira et al., 2014a, 2017). Thus, for example, low-to-moderate severity fires have been reported to increase soil aggregate stability (AS) (Mataix-Solera et al., 2011); soil organic matter (SOM) (Inbar et al., 2014; Pereira et al., 2014b; Úbeda et al., 2005); total carbon (TC) and total nitrogen (TN) (De Marco et al., 2005; Gonzalez-Perez et al., 2004); pH; electrical conductivity (EC) and the major extractable elements (Alcaniz et al., 2016; Liu et al., 2017), as a consequence of ash incorporation into the soil profile (Bodi et al., 2014). However, fires can also have various direct impacts on soil microbiology. Indeed, even low temperature fires can reduce soil microbial biomass and respiration (Certini, 2005; Frize et al., 1994), given the increase in temperature at the soil surface. The duration of the fire episode is one characteristic that can have the greatest impact on soil biology: the longer the duration of the wildfire, the greater the impact on soils (Mataix-Solera et al., 2009). However, low-to-moderate severity fires normally move rapidly and most of their heat is released in an upwards direction, which means their impact on soils is not great. Having said that, though, the heat generated in such fires might be enough to modify soil microbiology activity, since the threshold is low at between 40-121 ºC (Mataix-Solera et al., 2009), a temperature that is readily reached in the top few centimeters of the soil profile (Augustine et al., 2014; Kennard and Gholz, 2001). After the fire line is passed, the ash deposited on the soil surface, in conjunction with the post-fire environmental conditions, dictate the soil microbiology. Low-to-moderate fire severities produce reddish, black ashes (Pereira et al., 2012; 2014a), which can raise soil temperatures due to increased albedo, and provoke microbiological activity that contributes to a more rapid landscape recovery (Andersson et al., 2004; Bodí et al., 2014; Mataix-Solera et al., 2009).

High severity wildfires, in contrast, can consume the entire litter layer and the SOM. This greatly undermines soil protection (Pereira et al., 2015a) and leads to major changes in the top layer of the soil. Such fires have been reported as both reducing (Badia-Villas et al., 2014) and increasing (Jimenez-Pinilla et al., 2016) soil AS, depending on the temperature of combustion. Indeed, AS may fall as a result of organic matter consumption or increase due to mineralogical changes induced by high temperatures. Several studies have reported a fall in SOM (Holden et al., 2015), TC and TN (Badia et al., 2014; Ulery et al., 2017), because of high temperatures. As in low to moderate severity fires, pH and EC normally increase as a result of high organic matter mineralization, which increases the availability of the major cations (Lombao et al., 2015; Úbeda and Outeiro, 2009). The ash layer is grey and/or white in color, as a consequence of the total combustion of organic matter, which increases the amount of inorganic cations in soil solution (Pereira et al., 2012). High severity wildfires reduce microbial activity significantly in the immediate post-fire period due to the high temperatures recorded in the soil surface; however, normal activity is usually recovered a few months after the fire as a result of an increase in soil moisture, nutrients and vegetation recuperation (Fultz et al., 2016; Munoz-Rojas et al., 2016).

While a considerable amount of information has been gathered about the impact of pre-fire treatment on fire severity and the short-, medium- and long-term implications of such events, little research has been conducted into their respective effects on soil properties, especially in the immediate post-fire period, when the impact is most evident, and it is possible to evaluate the impact of pre-fire management and its effectiveness in reducing the impact of fire on soils. Moreover, the degree to which fire affects soil and soil conditions in the period immediately following a fire is crucial for plant recuperation (Pereira et al., 2016). This study of pre-fire management impact on soil properties is essential to understand the effectiveness of land management in sites, such as those in areas of Catalonia, that are prone to fire.

The objectives of this study are easily stated:

a) To examine the impact of clear cutting on soil AS, TN, SOM, inorganic carbon (IC), C/N ratio, pH, EC, extractable calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), microbial biomass carbon (Cmic), basal soil respiration (BSR), and the Cmic/C, BSR/C and BSR/Cmic ratios

b) To examine the influence of the time lapse between the management of the site and the wildfire and to determine if the treatment results in differences in the soil's post-fire properties.

2. Materials and methods

2.1 Study site

The study site is located in Ódena, Barcelona (41°38'42''N - 1°44'21E 420 m a.s.l.) in North-East Spain (Figure 1). A fire broke out in El Bruc, on 26th July 2015 and affected a total of 1,274 ha of *Pinus halepensis* Miller, *Pinus nigra* Arnold and *Quercus ilex* L. Understory vegetation was mainly composed of *Pistacea lentiscus* L. and *Genista scopius* L. The forest had last been affected by wildfire in 1986. The geological substrate is composed mainly of sediments originated from Paleocene shale (Panareda-Clopés and Nuet-Badia, 1993). Soil is classified as a Fluventic Haploxerept (Soil Survey Staff, 2014). The mean annual temperature of the study site is 14.2 °C and the mean annual rainfall ranges between 500 and 600 mm.

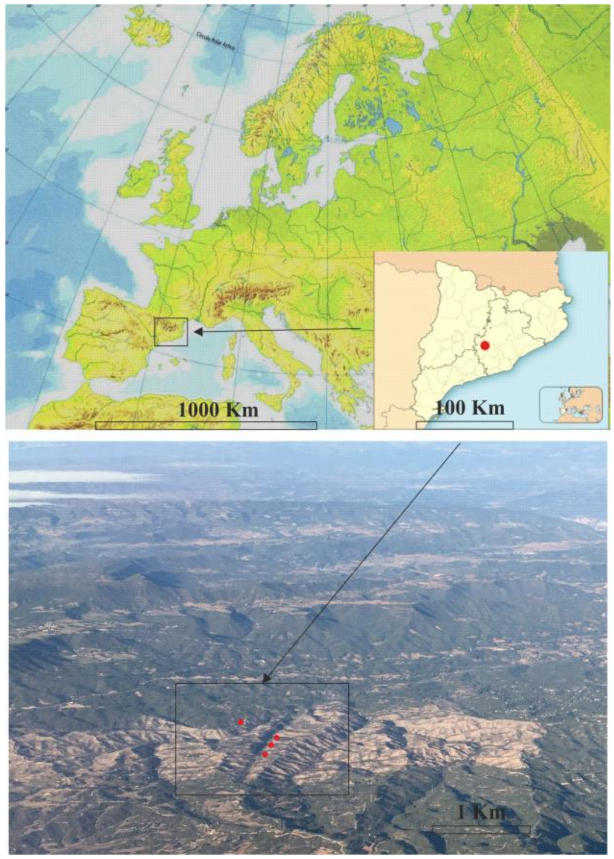

Figure 1. Location of study sites. Red dots show the places where soil samples were collected.

2.2 Experimental design and sampling

In this study, four different sites were selected within the site and sampled three months after the 2015 wildfire: one was not exposed to any management (NMB), one underwent management in 2005 (M05B), another was the target of a management program in 2015 – just two months before the wildfire event (M15B), and as a reference, a control plot was selected (Control), which had not been exposed to any management after the fire of 1986 and was unaffected by the wildfire in 2015 (Table 1). The management treatment involved a clear-cutting operation, leaving 1,000 trees per ha and leaving the cut vegetation over the soil surface in stems no taller than 1 m. In the case of the trees that were not felled, up to a third of their branches were removed. The wood was cut to a height of 1 meter, leaving waste of fine-to-medium thickness. At each site, we collected nine topsoil samples (0-5 cm), giving a total of 36. The plots were designed in sites of similar soil type, vegetation composition, and topographical characteristics (slope <10% and north-east aspect). The severity of the wildfire of 2015 was classified as high in accordance with Úbeda et al. (2006), given that 100% of tree crowns were combusted.

Table 1. Study sites, year of fire and year of management of studied sites.

Study sites	Years of fire/s	Date of management
M05B	1986 and 2015	2005
M15B	1986 and 2015	2015 (2 months before fire)
NMB	1986 and 2015	Not managed
Control	1986	Not managed

2.3 Laboratory methods

Soil samples were dried to constant weight at room temperature (approx. 23 °C) for 7 days. To analyze aggregate stability (AS), the ten-drop impact (TDI) method (Low, 1954) was used. The samples were sieved using a 4.8-mm mesh sieve. For each sample, ten air-dried aggregates (between 4-4.8 mm) were collected. A 2.8-mm sieve was placed over the aggregates and they were subjected to the impact of ten drops of deionized water, from a droplet placed at a height of 1 m. Each drop had a weight of 0.1 ± 0.001 g and a diameter of 2.8 mm. Aggregates were weighed before the test and the disaggregated material, which passed through the sieve during the test, was also weighed after drying at 105 ºC for 24 h. The results are expressed as a percentage of the aggregate weight that remains stable after TDI.

To determine the other soil properties, the samples were sieved using a 2-mm mesh sieve. Total nitrogen content was analyzed with a Flash EA 112 Series (Thermo-Fisher Scientific, Milan). Data acquisition and calculations were carried out using Eafer 300 software (Thermo-Fisher Scientific, Milan) (Pereira et al., 2012). SOM and inorganic carbon were measured using the loss-on-ignition method described in Heiri et al. (2001). For each sample, 1 g of soil was pulverized and dried in a muffle furnace at 105 ºC for 24 hours. To determine SOM content, the dried samples were exposed to a temperature of 550 ºC for 4 hours. Finally, to calculate soil inorganic carbon, the samples were subjected to a temperature of 950 ºC for 2 hours. The results were expressed as percentages. Soil pH and EC were analyzed with deionized water. The amounts of extractable elements (Ca, Mg, Na and K) were analyzed using the method described by Knudsen et al. (1982).

Each soil sample was kept under 4 ºC to measure its microbiological parameters. Microbial biomass carbon (C_{mic}) was extracted using the chloroform fumigation and extraction procedure (Vance et al., 1987). Basal soil respiration (BSR) was measured using a multiple sensor respirometer Micro-Oxymax (Anderson, 1982). Three indexes were calculated: C_{mic}/C ratio, carbon mineralization coefficient (BSR/C) and metabolic quotient (BSR/C_{mic}) (Morugán-Coronado et al., 2015).

2.4 Statistical analysis

Prior to conducting any statistical analyses, data normality and homogeneity were assessed using the Shapiro-Wilk and Levene's tests. If data followed the Gaussian distribution and respected the homogeneity of the variances, we applied a one-way ANOVA test; in those instances where the data did not satisfy normality and homogeneity requirements, we applied the non-parametric Kruskal-Wallis ANOVA test. If significant differences were identified at $p<0.05$, a Tukey post-hoc test was applied in order to identify differences within treatments. A redundancy analysis (RDA) was carried out to identify the extent to which the variation in one set of variables accounts for the variation in another. The soil properties used in the RDA were AS, SOM content, TN, C/N, IC, pH, EC, extractable Ca, Mg, Na, and K, C_{mic}, BSR and the three indexes: C_{mic}/C, BSR/C, and BSR/C_{mic}. Statistical analyses were carried out using SPSS 23.0 and CANOCO for Windows 4.5 software.

3. Results and discussion

3.1 Soil physical and chemical properties

Soil AS presented high values at all the sites ranging from 91.1 (M05B) to 95.9% (Control) (Table 2). Significant differences were found between the study sites, with a slightly lower AS at M05B. However, the differences between their respective mean values were very low and cannot be considered relevant. Different patterns of AS behavior have been reported in burned soils in relation to different degrees of fire severity and soil type (Mataix-Solera et al., 2011). Here, the slightly lower values found at M05B could be due to the higher fire severity suffered at this site. The impact of this severity is apparent in other soil properties recorded at this site (see discussion below). Previous studies in the laboratory (Badia-Villas et al., 2014;) and the field (Varela et al., 2015) have pointed to a decrease in AS in soils affected by high severity fires, which can be primarily attributed to SOM consumption. In other cases where inorganic constituents play an important role in AS, if high temperatures are registered in a soil, an increase in AS has been observed and attributed to mineralogical and micro-structural changes in the surface of soil aggregates (Jimenez-Pinilla et al., 2016). In our study, the differences observed were relatively very small.

Table 2. Descriptive statistics of physic-chemical characteristics. Different letters represent significant differences at a $p<0.05$. N=9.

Soil property	Study site	Mean	SD	p value
Aggregate Stability (%)	M05B	91.10^{b}	3.80	$*^{1}$
	M15B	$92.75^{a,b}$	3.98	
	NMB	$94.58^{a,b}$	3.60	
	Control	95.85^{a}	2.01	
Total Nitrogen (%)	M05B	0.24^{b}	0.04	$**^{1}$
	M15B	$0.28^{a,b}$	0.07	
	NMB	0.34^{a}	0.04	
	Control	0.25^{b}	0.06	
Soil Organic Matter (%)	M05B	4.18^{b}	0.98	$*^{1}$
	M15B	$5.07^{a,b}$	1.68	
	NMB	5.87^{a}	0.66	
	Control	4.15^{b}	0.77	
Inorganic Carbon (%)	M05B	5.77^{a}	0.14	$***^{2}$
	M15B	4.19^{b}	0.20	
	NMB	4.59^{b}	0.63	
	Control	4.61^{b}	0.28	
C/N ratio	M05B	17.70	1.87	n.s.^{1}
	M15B	17.67	1.75	
	NMB	17.46	0.93	
	Control	16.75	1.22	
pH	M05B	8.28^{a}	0.25	$*^{1}$
	M15B	$8.18^{a,b}$	0.17	
	NMB	$8.06^{b,c}$	0.04	
	Control	7.88^{c}	0.11	
EC (µS/cm)	M05B	$187^{b,c}$	60.57	$**^{2}$
	M15B	$273^{a,b}$	119.49	
	NMB	320^{a}	58.92	
	Control	149^{c}	38.46	
Extractable Ca (ppm)	M05B	17699^{c}	1738.62	$*^{1}$
	M15B	$19650^{a,b}$	1027.90	

	NMB	20630[a]	1762.87	
	Control	18613[b,c]	1366.75	
Extractable Mg (ppm)	M05B	825[b]	186.88	**[2]
	M15B	1157[a,b]	486.69	
	NMB	1356[a]	261.93	
	Control	1093[a,b]	317.25	
Extractable Na (ppm)	M05B	263	195.79	n.s.[1]
	M15B	278	95.41	
	NMB	204	52.24	
	Control	288	83.68	
Extractable K (ppm)	M05B	445[a]	101.71	***[2]
	M15B	467[a]	69.24	
	NMB	541[a]	89.54	
	Control	277[b]	29.07	

[1]One-way ANOVA, [2]Kruskal-Wallis. Four different sampling plots. * $p<0.05$, ** $p<0.01$, *** $p<0.001$.

TN and SOM were significantly higher in the NMB plot compared to the values recorded at M05B and the Control. Fire can reduce SOM content when high severity burning occurs (Mataix-Solera et al., 2002). In other cases, SOM has been found to increase as a result of the post-fire incorporation of partially burned material from charred vegetation into the soil (Caon et al., 2014; Knicker, 2007). Typically, high severity fires reduce TN and SOM in the period immediately after a fire because of the high temperatures reached in combustion (Certini et al., 2011; Mikita-Barbato et al., 2015; Vergnoux et al., 2011). However, if ash is deposited, then erosion is reduced; but, if ash is transported from other sites (Pereira et al., 2015b), an increase in TN and SOM can be expected. The gentle slope and the consequent accumulation of ash in the immediate post-fire period may have facilitated this incorporation. If we compare the sites burned in 2015, TN and SOM content values suggest M05B was exposed to higher fire severity.

In the case of IC, the values observed at M05B were significantly higher than those recorded at the other sites. This indicates that fire severity at this plot was slightly higher than that observed at NMB and M15B. Inorganic carbon or carbonate content in the soil or ash is a good indicator of fire severity. Typically, high severity fires produce ash with a high amount of carbonates (Pereira et al., 2012). Here, the high severity at the M05B plot can be attributed to the accumulation of dry biomass (for further discussion, see analysis of implications for forest management).

No significant differences were observed in the C/N ratio between sites. Santín et al. (2016) observed high C/N ratios in some burned plots attributable to high severity fires, but in our study there were no differences between the burned and unburned plots. Typically, wildfires mineralize the organic matter, reducing the soil C/N ratio in comparison to that found at unburned sites (Volkova et al. 2014) as a consequence of ash being incorporated into the soil profile (Bodi et al., 2014). Here, this did not occur presumably because of the high fire severity observed, which produced ash of similar C/N ratios to that at unburned sites (De Baets et al., 2016).

Soil pH values were significantly higher at M05B than they were at NMB and the Control. No significant differences were observed in these values between M05B and M15B, but the values were slightly higher at M05B, suggesting a higher fire severity at M05B. It is widely reported that after a high severity fire, soil pH increases in relation to the values found at unburned sites (Ulery et al., 1993). Jimenez-Esquilin et al. (2007) observed an increase in soil pH after a slash pile burning. This is attributed to the ash oxides and hydroxides released in solution (Bodi et al. 2014). pH levels increase with fire temperature/severity (Pereira et al., 2012), a finding corroborated by our study. The increase in pH is inversely related to the fall in TN and SOM. Soils affected by a high severity wildfire, followed by the incorporation of white ash, increase their pH levels. However, TN and SOM fall as a consequence of the high temperatures experienced (Certini et al., 2011; Mataix-Solera and Guerrero, 2007). EC was significantly higher in the burned plots, which is due to the release of anions and major cations in solution from the mineralized organic matter (Knicker, 2007). Significant differences were also identified between pre-fire treatments. The NMB plot released a significantly higher amount of salts than were released by M05B, which again may be due to the high fire severity recorded at M05B. Previous studies report that in soils and litter heated at very high temperatures, there is a decrease in EC (Badia and Marti, 2003; Úbeda et al., 2009), which is attributable to the reduced solubility of soils richer in carbonates that regulate the amount and type of nutrients in solution. The solubility of carbonates decreases with increasing pH. In addition, the carbonate surfaces with a pH between 7 and 10 are negatively charged, thus increasing their capacity to attract cations by sorption onto their surfaces (Pereira et al., 2012). For these reasons, EC was significantly lower at M05B than it was at the NMB site.

Extractable Ca was significantly higher at NMB and M15B than it was at the Control site (Table 2), which again can be attributed to ash incorporation, as observed in previous studies carried out in soils affected by high severity fires (Badia et al., 2014; Dzwonko et al., 2015; Rhoades et al., 2004). No significant differences in extractable Na were observed across the sites. However, the concentration of extractable K was significantly higher in the burned plots compared to levels reported at the Control. The increase in extractable K associated with high severity fires has been reported elsewhere (Bates and Davies, 2017; Murphy et al., 2006; Simard et al., 2001).

3.2 Soil microbiological properties

MBC was significantly higher at NMB than at the Control site (Table 3). Previous studies reported that C_{mic} levels normally fall in soils affected by wildfires compared to those observed in unburned soils. This can be attributed to the direct impact of soil heating, which reduces the soil microbe population (Barcenas-Moreno et al., 2016; Garcia-Orenes et al., 2017; Lombao et al., 2015). Some studies failed to identify any differences between unburned soils and those affected by high severity fires (Fultz et al., 2016; Holden et al., 2015), while others observed significant differences (Kumar Singh et al., 2017). In our case, we do not measure the direct impact of fire, rather we examine direct and indirect effects after the event. The significantly high C_{mic} at NMB compared to that at the Control can be attributed to the higher SOM content, which is known to have a positive effect on microbial populations (Blonska et al., 2017; Goberna et al., 2006; Pérez-Bejarano et al., 2010; Wolinska et al., 2015; Zornoza et al., 2007). In many cases, in the immediate post-fire period, there is an ephemeral increase in the amount of soluble carbon attributable to the cells of dead microorganisms (Mataix-Solera et al., 2009). This can lead to a short-term increase in microbial activity, but one that falls soon after the consumption of this fraction (Muñoz-Rojas et al., 2016).

Table 3. Descriptive statistics of microbiological characteristics. Different letters represent significant differences at a $p<0.05$. n=9.

Soil property	Study site	Mean	SD	p value
C_{mic}	M05B	983.89[a,b]	352.87	*[1]
	M15B	925.13[a,b]	221.65	
	NMB	1240.30[a]	238.02	
	Control	827.70[b]	371.02	
BSR	M05B	2.15[b]	0.86	*[1]
	M15B	1.78[b]	0.83	
	NMB	2.10[b]	0.99	
	Control	3.34[a]	0,73	
Cmic/C	M05B	239.47	77.38	n.s.[1]
	M15B	187.52	28.81	

	NMB	211.78	33.35	
	Control	195.22	57.74	
BSR/C	M05B	0.5056[b]	0.1076	***[1]
	M15B	0.3668[b]	0.1554	
	NMB	0.3615[b]	0.1740	
	Control	0.8087[a]	0.1418	
BSR/C_{mic}	M05B	0.0023[b]	0.0008	***[1]
	M15B	0.0020[b]	0.0010	
	NMB	0.0018[b]	0.0009	
	Control	0.0044[a]	0.0013	

[1]One-way ANOVA, [2]Kruskal-Wallis. Four different sampling plots. * $p<0.05$, ** $p<0.01$, *** $p<0.001$.

BSR and the BSR/C and BSR/C_{mic} ratios were significantly higher at the Control than they were at the other sites. No significant differences were identified in the C_{mic}/C ratio (Table 3). This is in agreement with previous studies that identified a marked reduction in BSR (Lopez-Serrano et al., 2016; Vega et al., 2013), BSR/C (Vega et al., 2013) and BSR/C_{mic} (Hernandez et al., 1997) at sites affected by high severity fires in comparison to the values reported at unburned sites. Guo et al. (2010) observed significantly higher values of BSR at their control site than at the clear-cut and slash-burned sites. Despite the higher values of C_{mic} at the fire-affected sites, we identified a significantly higher BSR at the Control plot, which shows that soil respiration is not always positively correlated with microbial biomass (Guo et al., 2015). The reduction in microbial activity at the burned sites may be attributed to the highly recalcitrant charcoal that is more resistant to microbial decomposition (Knicker et al., 2013). Microbial activity may also be perturbed by the release of organic pollutants and heavy metals (Vega et al., 2013). The significantly lower metabolic quotient (BSR/C_{mic}) at the burned sites confirms the conclusion drawn above that microbial activity was lower and failed to recover from the fire disturbance (Hernandez et al., 1997).

3.3 Multivariable analysis

The RDA enables us to determine whether in general the soil properties of the treated sites differ because of the effects of wildfire. Factor-1 in the RDA explains 32.4% of the variance while Factor-2 explains 29.7%, that is, a total of 62.1%. The variables with the highest explanatory capacity are IC, pH and extractable K, while those with the lowest are extractable Na, C/N ratio and C_{mic}. Thus, the RDA clearly separates the pre-fire management sites from the control plot and highlights the different behavior of the variables. BSR, AS, BSR/C and BSR/C_{mic} were closely associated with the Control site; SOM, TN, EC major cations, K and Cmic were closely associated with M15B and NMB; and, IC and C_{mic}/C were associated with M05B (Figure 2).

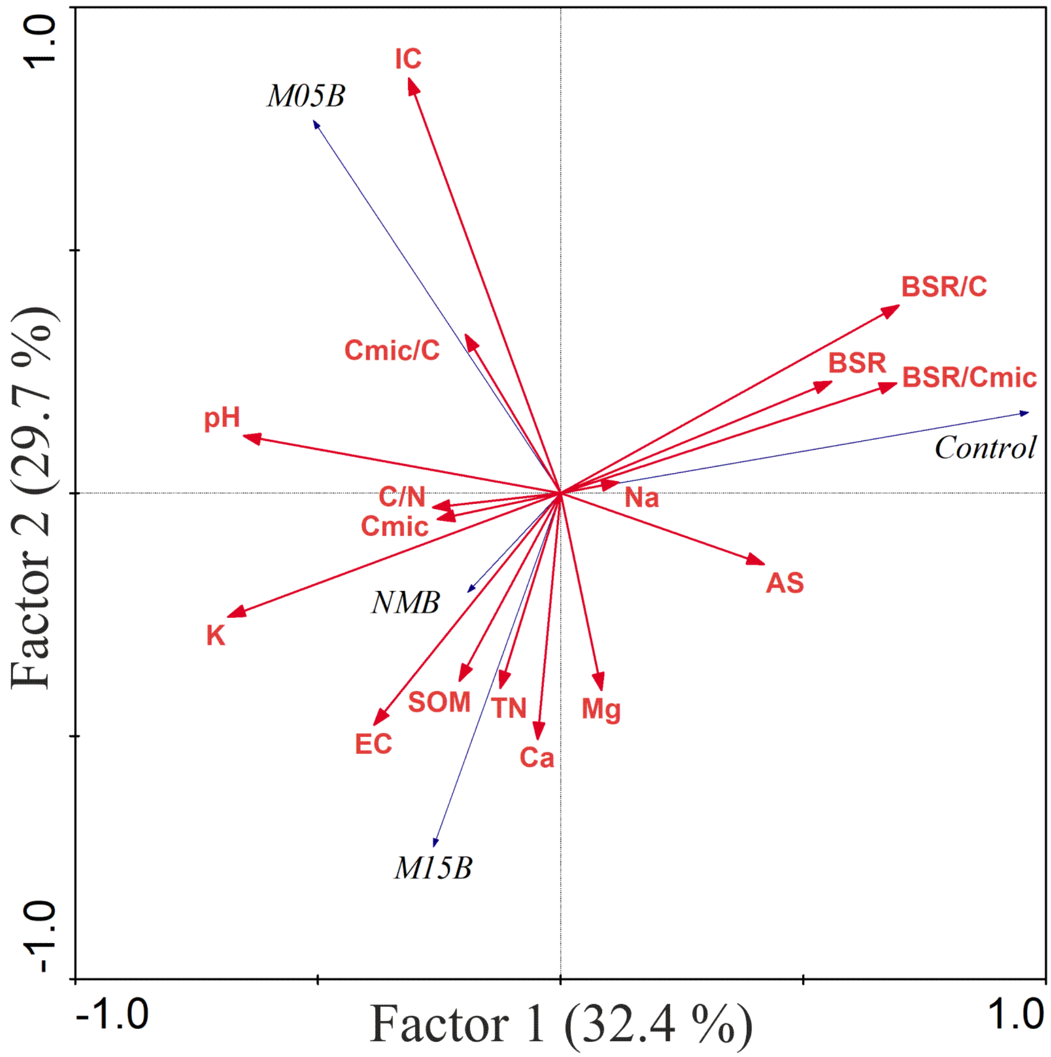

Figure 2. RDA about the relation between Factor 1 and 2. Aggregate Stability (AS), Soil Organic Matter (SOM) content, Total Nitrogen (TN), Carbon/Nitrogen ratio (C/N), Inorganic Carbon (IC), pH, Electrical Conductivity (EC), extractable Calcium (Ca), Magnesium (Mg), Sodium (Na), Potassium (K), Microbial Biomass Carbon (C_{mic}) and Basal Soil Respiration (BSR) and three different index C_{mic}/C ratio, carbon mineralization coefficient (BSR/C) and metabolic quotient (BSR/C_{mic}).

3.4. Implications for forest management - a discussion

Post-fire management, especially salvage logging, is controversial owing to its potentially negative effects, being implemented when major efforts are required to remedy the impact of wildfires. Previous studies observed that salvage logging has major negative impacts on soil properties in the short- (Donato et al., 2006; Garcia-Orenes et al., 2017; Wagenberger et al., 2016), medium- (Wagenberger et al., 2015) and long-term (Johnson et al., 2005). Indeed, this management practice has also been reported as having a marked effect on plant recovery and succession (Boucher et al., 2014; Hernández-Hernández et al., 2017). However, very little attention has been paid to preventive methods aimed at reducing forest fuel accumulation, but it is critical that they be taken into consideration in forest management plans, especially in fire-prone sites, such as the Mediterranean (Francos et al., 2016b; Leverkus et al., 2014; Marañon-Jimenez et al., 2013; Pique et al., 2011). The 2015 wildfire studied here had more negative impacts on the soil properties of the plot managed in 2005 than it did on the plot not exposed to any management and the plot managed in 2015. Despite the high severity of the fire in all treatments, the M05B plot presented the highest soil pH and IC, indicating that the impact here was more detrimental than elsewhere. The high TN, SOM, EC, extractable Ca and Mg values recorded at the NMB plot in relation to the values observed at M05B were evidence that, three months after the fire, the plot that had not undergone any management was in a better position to begin regeneration than was the plot managed in 2005. According to the impacts on the soil properties studied here, the three sites exposed to the effects of this high severity fire can be ranked in the following descending order: M05B>M15B>NMB. The plot managed in 2005 was the site where the effects were greatest because of the greater accumulation of dead fuel. Some authors argue that forest management is essential for reducing the impact of fire on soils (Fernandes, 2013; Tempel et al., 2014); however, here, the fact that the vegetation had been cut 10 years previously meant that prior to the fire the wood was dry and so the fire burned with greater intensity. Fuel type has been shown to have implications for flammability, the response to fire temperatures and, ultimately, fire severity (Dimitrakopoulos and Panov, 2001; Úbeda et al., 2009). Given our results, we hypothesize that the combustible fuel present at the NMB plot before the fire was less flammable. But, further research is needed on this point. Overall, for the ecosystem studied here, the pre-fire NMB option was the one with least impact and, in some instances, it had positive impacts on certain soil properties. Further studies are underway in these sites aimed at assessing medium- and long-term effects and should dissipate any doubts caused by these initial results. It is clearly critical to have better information about the impact of wildfires in relation to different pre-fire management practices in the immediate post-fire period, since this is when fire brings about the most significant changes in soil properties (Caon et al., 2014; Francos et al., 2016a; Wang et al., 2014). It is our contention that pre-fire management represents a good option for preventing fires, but here the fact that after clear-cutting the dead vegetation was left to accumulate over the soil surface created a high fuel density resulting in a higher fire intensity when the wildfire broke out.

4. Conclusions

Forest management practices, such as clear cutting a site some years prior to or only a few months before a wildfire event fire (as reported herein), affect soil properties. Not clearing the cut vegetation from the soil surface affects the severity of the fire and induces more changes in soil properties. However, the differences observed across our study sites were not sufficiently significant to conclude that this is a detrimental treatment. On balance, we recommend this treatment as a way of preventing the outbreak of new forest fires, but large accumulations of cut vegetation covering the soil surface should be avoided so as to reduce fire severity in potential medium- to long-term episodes of wildfire. Clearly, further studies are needed to analyze the effect of clear-cutting management practices on soil properties to ensure the implementation of appropriate forest management and to determine if differences between treated and untreated sites increase or disappear over time.

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References

Alcañiz, M., Outeiro, L., Francos, M., Farguell, J., Úbeda, X., 2016. Long-term dynamics of soil chemical properties after a prescribed fire in a Mediterranean forest (Montgrí Massif, Catalonia, Spain). Sci. Total Environ. 572, 1329-1335.

Anderson, J.P.E., 1982. Soil respiration, in: Dinauer, R.C. (Ed.), Methods of soil analysis. Part 2. Chemical and microbiological properties. ASA, SSSA Madison, Wisconsin (USA), pp. 831-871.

Andersson, M., Michelsen, A., Jensen, M., Kjoller, A., 2004. Tropical savannah woodland: effects of experimental fire on soil microorganisms and soil emissions of carbon dioxide. Soil Biol. Biochem. 36, 849–858.

Andreu, V., Imeson, A. C., Rubio, J. L., 2001. Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. Catena 44, 69-84.

Arcenegui, V., Mataix-Solera, V., Guerrero, C., Zornoza, R., Mataix-Beneyto, J., García-Orenes, F., 2008. Immediate effects of wildfire on water repellency and aggregate stability in Mediterranean calcareous soils. Catena 74, 219-226.

Augustine, D.J., Brewer, P., Blumenthal, D.M., Derner, J.D., von Fisher, J.C., 2014. Prescribed fire, soil inorganic nitrogen dynamics, and plant responses in a semiarid grassland. J. Arid. Environ. 104, 59–66.

Badia, D., Marti, C., 2003. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Res. Manag. 17, 23–41.

Badia, D., Marti, C., Aguirre, T., Aznar, J.M., Gonzalez-Perez, J.A., De la Rosa, J.M., Leon, J., Ibarra, P., Echeverria, T., 2014. Wildfire effects on nutrients and organic carbon of a Rendzic Phaeozem in NE Spain: Changes at cm-scale topsoil. Catena 113, 267–275.

Badia-Villas, D., Gonzalez-Perez, J.A., Aznar, J.M., Arjona-Gracia, B., Marti-Dalmau, C., 2014. Changes in water repellency, aggregation and organic matter of a mollic horizon burned in laboratory: Soil depth affected by fire. Geoderma 213, 400–407.

Barcenas-Moreno, G., Garcia-Orenes, F., Mataix-Solera, J., Mataix-Beneyto, J., 2016. Plant community influence on soil microbial response after wildfire in Sierra Nevada National Park. Sci. Total Environ. 573, 1265–1274.

Bates, J.D., Davies, K.W., 2017. Effect of conifer treatments on soil nutrients availability and plant composition in sagebrush steppe. For. Ecol. Manage. 400, 631-644.

Blonska E., Lasota J., Gruba P., 2016. Effect of temperate forest tree species on soil dehydrogenase and urease activities in relation to other properties of soil derived from loess and glaciofluvial sand. Ecol. Res. 31, 655-664.

Blonska E., Lasota J., Gruba P., 2017. Enzymatic activity and stabilization of organic matter in soil with different detritus inputs. Soil Sci. Plant Nutr. 63, 242-247.

Bodí, M., Martin, D., Santin, C., Balfour, V., Doerr, S.H., Pereira, P., Cerdà, A., Mataix-Solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth-Sci. Rev. 130, 103–127.

Bond, M.M., Van Wilgen, B.W., 1996. Fire and plants. Population and Community Biology Series 14, Chapman & Hall.

Boucher, D., Gauthier, S., Noel, J., Greene, D.F., Bergeron, Y., 2014. Salvage logging affects early-post fire tree composition in Canadian boreal forest. For. Ecol. Manage. 325, 118-127.

Caon, L., Vallejo, R., Coen, R.J., Geissen, V., 2014. Effects of wildfire on soil nutrients Mediterranean ecosystems. Earth-Sci. Rev. 139, 47–58.

Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143, 1–10.

Certini, G., 2014. Fire as a soil forming factor. Ambio 43, 191–195.

Certini, G., Nocentini, C., Knicker, H., Arfaioli, P., Rumpel, C., 2011. Wildfire effects on soil organic matter quantity and quality in two fire-prone Mediterranean pine forests. Geoderma 167-168, 148–155.

Cochrane, M.A., Moran, C.J., Wimberly, M.C., Baer, A.D., Finney, M.A., Beckendorf, K.L., Eidenshink, J., Zhu, Z., 2012. Estimation of wildfire size and risk changes due to fuels treatments. Int. J. Wildland Fire 21, 357-367.

Corona, P., Ascoli, D., Barbati, A., Bovio, G., Colangelo, G., Elia, M., Garfi, V., Iovino, F., Lafortezza, R., Leone, V., Lovreglio, R., Marchetti, M., Marchi, E., Menguzzato, G., Nocentini, S., Picchio, R., Portoghesi, L., Puletti, N., Sanesi, G., Chianucci, F., 2015. Integrated forest management to prevent wildfires under Mediterranean environments. Annals of Silvicultural Research 39, 1-22.

De Baets, S., van de Weg, M.J., Lewis, R., Steinberg, N., Meersmans, J., Quine, T.A., Shaver, G.R., Hartley, I.P., 2016. Investigating the controls on soil organic matter decomposition in tussock tundra soil and permafrost after fire. Soil Bio. Biochem. 99, 108–116.

De Marcos, 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. Int. J. Wildland Fire 14, 365-377.

Dimitrakopoulos, A.P., Panov, P.I., 2001. Pyric properties of some dominant Mediterranenan vegetation species. Int. J. Wildland Fire 10, 23–27.

Donato, D.C., Fontaine, J.B., Campbell, J.L., Robinson, W.D., Kauffman, J.B., Law, B.E., 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science 311, 352.

Dzwonko, Z., Loster, S., Gawronski, S., 2015. Impact of fire severity on soil properties and the development of tree and shrub species in a Scots pine moist forest site in Southern Poland. For. Ecol. Manage. 342, 56–63.

Faria, S.R., De La Rosa, J., Knicker, H., Gonzalez-Perez, J.A., Villaverde, J., Keizer, J.J., 2015. Wildfire-induced alterations of topsoil organic matter and their recovery in Mediterranean eucalypt stands detected with biogeochemical markers. Eur. J. Soil Sci. 66, 699–713.

Fernandes, P.M., Davies, G.M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., Stoof, C.R., Vega, J.A., Molina, D., 2013. Prescribed burning in southern Europe: developing fire management in a dynamic landscape. Front. Ecol. Environ. 11, 4-14.

Francos, M., Pereira, P., Alcañiz, M., Mataix-Solera, J., Úbeda, X., 2016a. Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain). Sci. Total Environ. 572, 1353-1362.

Francos, M., Úbeda, X., Pereira, P., Alcañiz, M., 2018. Long-term impact of wildfire on soils exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula). Sci. Total Environ. 615, 664-671.

Francos, M., Úbeda, X., Tort, J., Panareda, J.M., Cerdà, A., 2016b. The role of forest fire severity on vegetation recovery after 18 years. Implications for forest management of *Quercus suber* L. in Iberian Peninsula. Glob. Planet. Chang. 145, 11-16.

Fultz, L.M., Moore-Kucera, J., Dathe, J., Davinic, M., Perry, G., Wester, D., Schwink, D.W., Rideout-Hanzak, S., 2016. Forest wildfire and grassland prescribed fire effects on soil biogeochemical process and microbial communities: Two case studies in the semi-arid Southwest. App. Soil Ecol. 99, 118–128.

Garcia-Orenes, F., Arcenegui, V., Chrenkova, K., Mataix-Solera, J., Jara-Navarro, A.B., Torres, M.P., 2017. Effects of salvage logging on soil properties and vegetation recovery in a fire affected Mediterranean forest: A two year monitoring research. Sci. Total Environ. 586, 1057-1065.

Guo, J., Yank, Y., Chen, G., Xie, J., Gao, R., Qian, W., 2010. Effects of clear-cutting and slash burning on soil respiration in Chinese fir and evergreen broadleaved forests in mid-subtropical China. Plant Soil 333, 249-261.

Goberna, M., Sánchez, J., Pascual, A.J., García, C., 2006. Surface and subsurface organic carbon, microbial biomass and activity in a forest soil sequence. Soil Biol. Biochem. 38, 2233-2243.

González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fire on soil organic matter-a review. Environ. Int. 30, 855-870.

Gordillo-Rivero, A.J., García-Moreno, J., Jordán, A., Zavala, L., Granja-Martins, F.M., 2014. Fire severity and Surface rock fragments cause patchy distribution of soil water repellency and infiltration rates after burning. Hydrol. Process. 28, 5832–5843.

Guo, J., Chen, G., Xie, J., Yang, Z., Yang, Y., 2015. Effect of heat disturbance on microbial respiration in Chinese fire (*Cunninghamia lancelota*) forest soils. J. For. Res. 26, 933–939.

Heiri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J. Paleolimnol. 5, 101-110.

Hernández-Hernández, R., Castro, J., Del Arco Aguilar, M., Fernández-López, A.B., González-Mancebo, M., 2017. Post-fire salvage logging imposes a new disturbance that retards succession: The case of Bryophite communities in Macaronesian Laurel Forest. Forests 8, 252 doi:10.3390/f8070252

Hernández, T., García, C., Reinhardt, I., 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils 25,109-116.

Holden, S.R., Berhe, A.A., Treseder, K.K., 2015. Decreases in soil moisture and organic matter quality suppress decomposition following a boreal forest fire. Soil Biol. Biochem. 87, 1–9.

Inbar, A., Lado, M., Sternberg, M., Tenau, H., Ben-Hur, M., 2014. Forest fire effects on soil chemical and physicochemical properties, infiltration, runoff, and erosion in a semiarid Mediterranean region. Geoderma 221-222, 131–138.

Jimenez-Esquilin, A.E., Stromberger, M.E., Massman, W.J.,Frank, J.M., Shepperd, W.D., 2007. Microbial community structure and activity in a Colorado Rocky Mountain forest soil scarred by slash pile burning. Soil Biol. Biochem. 39, 1111-1120.

Jimenez-Gonzalez, M.A., De La Rosa, J.M., Jimenez-Morillo, N.T., Almendros, G., Gonzalez-Perez, J.A., Knicker, H., 2016. Post-fire recovery of soil organic matter in a Cambisol from typical Mediterranen forest in Southwesthern Spain. Sci. Total Environ. 572, 1414–1421.

Jiménez-Pinilla, P., Mataix-Solera, J., Arcenegui, V., Delgado, R., Martin-García, J.M., Lozano, E., Martínez-Zavala, L., Jordán, A., 2016. Advances in the knowledge of how heating can affect aggregate stability in Mediterranean soils: a XDR and SEM-EDX approach. Catena 147, 315–324.

Johnson, D.W., Murphy, J.F., Susfalk, R.B., Caldwell, T.G., Miller, W.W., Walker, R.F., Powers, R.F., 2005. The effects of wildfire, salvage logging, and post-fire N-fixation on the nutrient budgets of a Sierran forest. For. Ecol. Manag. 220, 155-165.

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. Int. J. Wildland Fire 18, 116-126.

Keeley, J.E., Bond, W.J., Bradstock, R.A., Pausas, J.G., Rundel, P.W., 2012. Fire in Mediterranean Ecosystems. Cambridge University Press. 515 p.

Kennard, D.K., Gholz, H.L., 2001. Effects of a high-and low-intensity fires on soil properties and plant growth in a Bolivian dry forest. Plant Soil 234, 119–129.

Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 85, 91–118.

Knicker, H., Gonzalez-Vila, F.J., Gonzalez-Vasquez, R., 2013. Biodegradability of organic matter in fire-affected mineral soils of Southern Spain. Soil Biol. Biochem. 56, 31–39.

Knudsen, D., Petersen, G.A., Pratt, P.F., 1982. Lithium, sodium and potassium, in: Dinauer, R.C. (Ed.), Methods of soil analysis. Part 2. Chemical and microbiological properties. ASA, SSSA Madison, Wisconsin (USA), pp. 225-246.

Kumar Singh, A., Kushwaha, M., Rai, A., Singh, N., 2017. Changes in soil microbial response across year following a wildfire in a tropical dry forest. For. Ecol. Manage. 391, 458–468.

Leverkus, A.B., Lorite, J., Navarro, F.B., Sánchez-Cañete, E.P., Castro, J., 2014. Post-fire salvage logging alters species composition and reduces cover richness and diversity in Mediterranean plant communities. J. Environ. Manage. 133, 323–331.

Liu, X., Chen, C.R., Hughe, J.M., Wang, W.J., Lewis, T., 2017. Temporal changes rather than long-term repeated burning predominately control the shift in the abundance of soil denitrifying community in an Australian Sclerophyl Forest. Microb. Ecol. doi:10.1007/s00248-016-0894-6

Lombao, A., Barreiro, A., Carballas, T., Fonturbel, M.T., Martin, A., Vega, J.A., Fernandez, C., Diaz-Ravina, M., 2015. Changes in soil properties after a wildfire in Fragas do Eume Natural Park (Galicia, NW Spain). Catena 135, 409–418.

Lopez-Serrano, F.R., Rubio, E., Dadi, T., Moya, D., Andres-Abellan, M., Garcia-Morote, F.A., Miettinen, H., Martinez-Garcia, E., 2016. Influence of recovery from wildfire and thinning on soil respiration of a Mediterranean mixed forest. Sci. Total Environ. 573, 1217–1231.

Low, A.J., 1954. The study of soil structure in the field and in the laboratory. J. Soil Sci. 5, 57-74.

Marañon-Jimenez, S., Castro, J., Querejeta, J.I., Fernandez-Ondono, E., Allen, C.D., 2013. Post-fire wood management alters water stress, growth, and performance of pine regeneration in a Mediterranean Ecosystem. For. Ecol. Manage. 308, 231–239.

Martín-Alcón, S., Coll, L., 2016. Unraveling the relative importance of factor driving post-fire regeneration trajectories in non-serotinous *Pinus nigra* forest. For. Ecol. Manage. 361, 13-22.

Mastrolonardo, G., Francioso, O., Di Foggia, M., Bonora, S., Rumpel, C., Certini, G., 2014. Application of thermal and spectroscopic techniques to assess fire-induced changes on soil organic matter in a Mediterranean forest. J. Geochem. Explor. 143, 174–182.

Mataix-Solera, J., Guerrero, C., 2007. Efectos de los incendios forestales sobre las propiedades edáficas, in: Mataix-Solera, J. (Ed.), Incendios forestales, suelo y erosión hídrica, Caja de Ahorros del Mediterráneo CEMACAM, Alcoi, pp. 5-40.

Mataix Solera, J., Guerrero, C., Garcia-Orenes, F., Barcenas, G.M., Torres, M.P., 2009. Forest fire effects on soil microbiology, in: Cerda, A., Robichaud, P.R. (Eds.), Fire effects on soils and restoration strategies. Science Publishers, Enfield (NH), pp. 133–175.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordan, A., Zavala, L.M., 2011. Fire effects on soil aggregation. Earth-Sci. Rev. 109, 44–60.

Mikita-Barbato, R.A., Kelly, J.J., Tate III, R.L., 2015. Wildfire effects on the properties and microbial community structure of organic horizon soils in the New Jersey Pinelands. Soil Biol. Biochem. 86, 67–76.

Morugán-Coronado, A., García-Orenes, F., Cerdà, A., 2015. Changes in soil microbial activity and physicochemical properties in agricultural soils in Eastern Spain. J. Soil Sci. 5-3. DOI: 10.3232/SJSS.2015.V5.N3.02

Muñoz-Rojas, M.., Erickson, T.E., Martini, D., Dixon, K.W., Merrit, D.J., 2016. Soil physicochemical and microbiological indicators of short, medium and long term post-fire recovery in semi-arid ecosystems. Ecol. Indic. 63, 14–22.

Murphy, J.D., Johnson, D.W., Miller, W.W., Walker, R.F., Carroll, E.F., Blank, R.R., 2006. Wildfire effects on soil nutrients and leaching in a Tahoe Basin Watersheed. J. Environ. Qual. 35, 479–489.

Neary, D., Klopatek, C.C., DeBano, L.F., Folliot, P.F., 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manag. 122, 51-71.

Panareda-Clopés, J.M., Nuet-Badia, J., 1993. Tipología y cartografía corológica de las plantas vasculares de Montserrat (Cordillera Prelitoral Catalana). *Re. Geogr*. 27, 33-58.

Pereira, P., Cerdà, A., Martin, D., Úbeda, X., Depellegrin, D., Novara, A., Martínez-Murillo, J.F., Brevik, E.C., Menshov, O., Comino, J.R., Miesel, J. 2017. Short-term low-severity spring grassland fire impacts on soil extractable elements and soil ratios in Lithuania. Sci. Total Environ. 578, 469–475.

Pereira, P., Jordan, A., Martin, D., Cerdà, A. 2015a. The role of ash in fire-affected ecosystems. Catena, 135, 337–339.

Pereira, P., Cerdà, A., Úbeda, X., Mataix-Solera, J., Arcenegui, V., Zavala, L. 2015b. Modelling the impacts of wildfire on ash thickness in a short-term period. Land Degrad. Dev. 26, 180–192.

Pereira, P., Rein, G., Martin, D., 2016. Past and Present Post-Fire Environments. Sci. Total Environ. 573, 1275 - 1277.

Pereira, P., Úbeda, X., Martin, D., 2012. Fire severity effects on ash chemical composition and water-extractable elements. Geoderma, 191, 105-114.

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Cerdà, A., Burguet, M., 2014a. Wildfire affect on extractable in ash from *Pinus pinaster* forest in Portugal. Hydrol. Process. 28, 2681-2690.

Pereira, P., Úbeda, X., Martin, D.A., Mataix-Solera, J., Oliva, M., Novara, A., 2014b. Short-term spatio-temporal spring grassland fire effects on soil colour, organic matter and water repellency in Lithuania. Solid Earth 5, 209–225.

Pérez-Bejarano, A., Mataix-Solera, J., Zornoza, R., Guerrero, C., Arcenegui, V., Mataix-Beneyto, J., Cano-Amat, S., 2010. Influence of plant species on physcial, chemical and biological soil properties in Mediterranean forest soil. Eur. J. For. Res. 129, 15-24.

Piqué, M., Castellnou, M., Valor, T., Pagés, J., Larrañaga, A., Miralles, M., Cervera, T., 2011. Integració del risc de grans incendis forestals (GIF) en la gestió forestal: Incendis tipus i vulnerabilitat de les estructures forestals al foc de capçades. Sèrie: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST). Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya

Rhoades, C.C., Meier, A.J., Rebertus, A.J., 2004. Soil properties in fire consumed log burnout openings in a Missouri oak Savanna. For. Ecol. Manag. 192, 277–284.

Santin, C., Doerr, S.H., Merino, A., Bryant, R., Loader, N.J., 2016. Forest floor chemical transformations in a boreal forest fire and their correlations with temperature and heating duration. Geoderma 264, 71–80.

Simard, A., 1991 Fire severity, changing scales, and how things hang together. Int. J. Wildland Fire 1, 23-34.

Simard, D.G., Fyles, J.W., Pare, D., Nguyen, T., 2001. Impacts of clearcut harvesting and wildfire on soil nutrient status in Quebec boreal forest. Can. J. Soil Sci. 81, 229–237.

Soil Survey Staff., 2014. Keys to soil taxonomy. 12th ed. USDA-Natural Resources Conservation Service, Washington, DC.

Tempel, D.J., Gutierrez, R.J., Whitmore, S.A., Reetz, M.J., Stoelting, R.E., Berigan, W.J., Seamans, M.E., Peery, M.Z., 2014. Effects of forest management on California Spotted Owls: implications for reducing wildfire risk in fire-prone areas. Ecol. Appl. 24, 2089–2106.

Tessler, N., Sapir, Y., Wittenberg, L., Greenbaum, N., 2016a. Recovery of Mediterranean Vegetation after Recurrent Forest Fires: Insight from the 2010 Forest Fire on Mount Carmel, Israel. Land Degrad. Dev. 27, 1424 - 1431.

Tessler, N., Wittenberg, L., Greenbaum, N., 2016b. Vegetation cover nd species richness after recurrent forest fires in Eastern Mediterranean ecosystem of Mount Carmel, Israel. Sci. Total Environ. 572, 1395 - 1402.

Úbeda, X., Lorca, M., Outeiro, L., Bernia, S., Castellnou, M., 2005. Effects of prescribed fire on soil quality in Mediterranean grassland (Prades Mountains, north-east Spain). Int. J. Wildland Fire 14, 379–384.

Úbeda, X., Outeiro, L.R., 2009. Physical and chemical effects of fire in soil, in: Cerdà, A., Robichaud, P.R. (Eds.). Fire Efects on Soils and Restoration Strategies. Science Publishers, Enfield, NH. pp. 105-133.

Úbeda, X., Outeiro, L.R., Sala, M., 2006. Vegetation regrowth after a differential severity forest fire in a Mediterranean environment, Northeast Spain. Land Degrad. Dev. 17, 429-440.

Úbeda, X., Pereira, P., Outeiro, L., Martin, D., 2009. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (*Quercus suber*). Land Degrad. Dev. 20, 589–608.

Úbeda, X., Sala, M., Imeson, A., 1990. Variaciones en la estabilidad y consistencia de un suelo forestal antes y después de ser sometido a un incendio. I Reunión Nacional de Geomorfología, Teruel. pp. 677-685.

Ulery, A., Graham, R.C., Amrhein, C., 1993. Wood-ash composition and soil pH following intense burning. Soil Sci. 156, 358–364.

Ulery, A.L., Graham, R.C., Goforth, B.R., Hubberth, K.R., 2017. Fire effects on cation exchange capacity of California forest and woodland soils. Geoderma 286, 125–130.

Vance, E.D., Brookes, P.C., Jenkinson, D.S., 1987. An extraction method for measuring soil microbial biomass C. Soil Biol. Biochem. 19, 703-707.

Varela, M.E., Benito, E., Keiser, J.J., 2015. Influence of fire severity on soil physical degradation in two pine forest stands of NW Spain. Catena 133, 342–348.

Vega, J.A., Fonturbel, T., Merino, A., Fernández, C., Ferreiro, C., Jiménez, E., 2013. Testing the ability of visual indicators of burn soil severity to reflect changes in soil chemical and microbial properties in pine forests and shrubland. Plant Soil 396, 73–91.

Vergnoux, A., Di Rocco, R., Domeizel, M., Guiliano, M., Doumenq, P., Theraulaz, F., 2011. Effects of forest fires on water extractable organic matter and humic substances from Mediterranean soils: UV-vis and fluorescence spectroscopy approaches. Geoderma 160, 434–443.

Volkova, L., Meyer, C.P., Murphy, S., Fairman, T., Reisen, F., Weston, C., 2014. Fuel reduction burning mitigates wildfire effects on forest carbon and greenhouse gas emission. Int. J. Wildland Fire 23, 771–780.

Wang, Y., Xu, Z., Zhou, Q., 2014. Impact of fire on soil gross nitrogen transformations in forest ecosystems. J. Soils Sediments 14, 1030–1040.

Wagenberger, J.W., MacDonald, L., Coates, R.N., Robichaud, P.R., Brown, R.E., 2015. Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. For. Ecol. Manag. 335, 176-193.

Wagenberger, J.W., Robichaud, P., Brown, R.E., 2016. Rill erosion in burned and salvage logged western montane forests: Effects of logging equipment type, traffic level, and slash treatment. J. Hydrol. 541, 889-901.

Wolinska, A., Stepniewska, Z., Pytlak, A., 2015. The effect of environmental factors on total soil DNA content and dehydrogenase activity. Arch. Biol. Sci. 67, 493-501.

Zornoza, R., Mataix-Solera, J., Guerrero, C., Mayoral, A.M., Morales, J., 2007. Soil properties under natural forest in the Alicante Province of Spain. Geoderma, 142, 334-341.

CHAPTER 3

CHAPTER 3. IMPACT OF AN INTENSE RAINFALL EVENT ON SOIL PROPERTIES FOLLOWING A WILDFIRE IN A MEDITERRANEAN ENVIRONMENT (NORTH-EAST SPAIN).

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Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain).

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Highlights

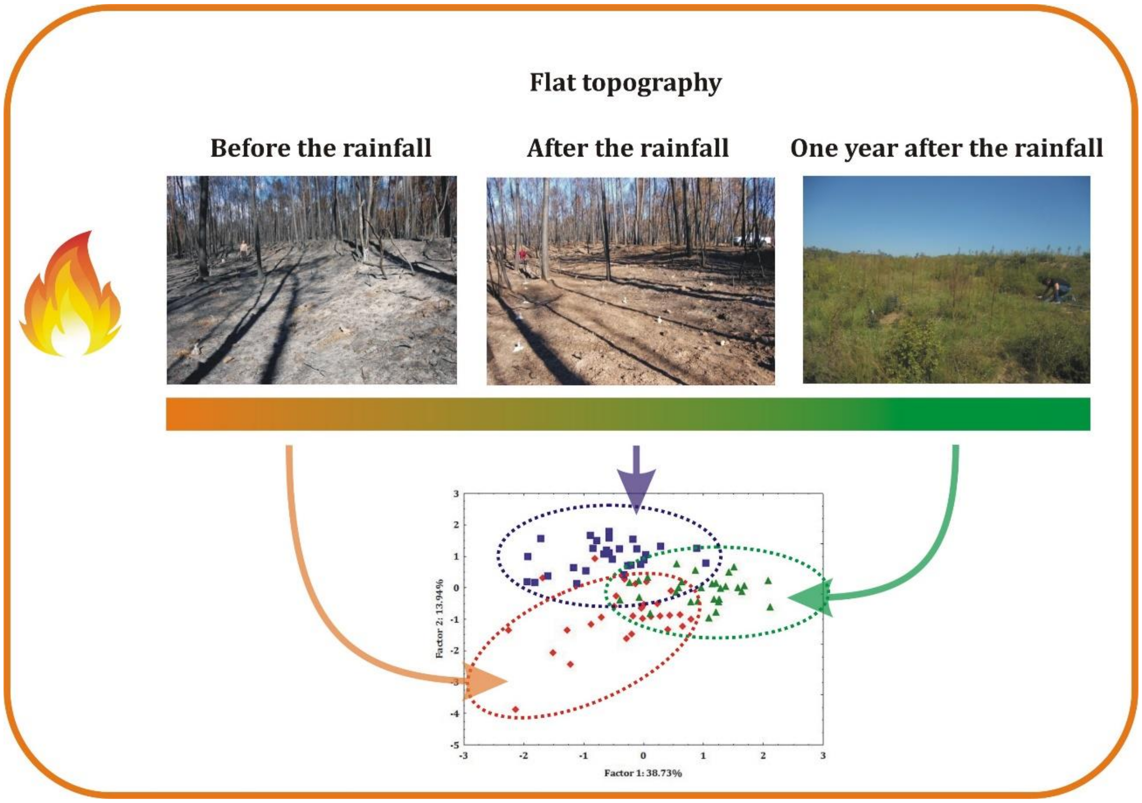
Ash layer resulting from severe fire episode reduces raindrop impact on soil aggregates.

Post-fire management, ash and soil erosion, recovery of vegetation and soil chemical composition may be responsible for the decrease in soil aggregate stability.

Intense rainfall on fire-affected soils does not affect soil total nitrogen and total carbon levels significantly.

Intense rainfall significantly increases the amount of extractable cations, available phosphorous and SPAR in soil solution.

Graphical Abstract



Abstract

Intense rainfall events after severe wildfires can have an impact on soil properties, above all in the Mediterranean environment. This study seeks to examine the immediate impact and the effect after a year of an intense rainfall event on a Mediterranean forest affected by a high severity wildfire. The work analyses the following soil properties: soil aggregate stability, total nitrogen, total carbon, organic and inorganic carbon, the C/N ratio, carbonates, pH, electrical conductivity, extractable calcium, magnesium, sodium, potassium, available phosphorous and the sodium and potassium adsorption ratio (SPAR). We sampled soils in the burned area before, immediately after and one year after the rainfall event. The results showed that the intense rainfall event did not have an immediate impact on soil aggregate stability, but a significant difference was recorded one year after. The intense precipitation did not result in any significant changes in soil total nitrogen, total carbon, inorganic carbon, the C/N ratio and carbonates during the study period. Differences were only registered in soil organic carbon. The soil organic carbon content was significantly higher after the rainfall than in the other sampling dates. The rainfall event did increase soil pH, electrical conductivity, major cations, available phosphorous and the SPAR. One year after the fire, a significant decrease in soil aggregate stability was observed that can be attributed to high SPAR levels and human

intervention, while the reduction in extractable elements can be attributed to soil leaching and vegetation consumption. Overall, the intense rainfall event, other post-fire rainfall events and human intervention did not have a detrimental impact on soil properties in all probability owing to the flat plot topography.

Keywords: Intense Rainfall, Severe Wildfire, C/N ratio, Aggregate Stability, Major Cations, SPAR.

1. Introduction

Intense rainfall episodes are common in the Mediterranean area (Pereira et al., 2015a), especially during the autumn following warm, dry summers that are responsible for severe wildfires (Versini et al., 2013). Extreme autumn rainfall is expected to increase in the future due to climate change in the Mediterranean basin (Kysely et al., 2012), generating flash floods, with a highly detrimental impact for fire-affected areas (Shakesby, 2011; Ferreira et al., 2015).

Fire is a natural phenomenon in many ecosystems and has obviously been fundamental to human evolution (Vélez, 2000). However, in recent times, as a consequence of land abandonment and climate change, fire regimes have changed and fire intensities and severities have increased, all of which has had major social, economic and environmental implications (Dale et al., 2001; Clark et al., 2014). Indeed, fire has even been described as a global 'herbivore' (Bond and Keeley, 2005). Since the 1950s, Mediterranean environments have become especially vulnerable to summer wildfires for three main reasons: first, the rural exodus, the gradual abandonment of forest management practices and, consequently, an increase of fuel accumulation; second, the industrial plantation of monospecific, flammable tree species; and, third, more frequent and more intense summer heat-waves and droughts, which have increased the meteorological and climate risks of the occurrence of forest fires (Mataix-Solera and Cerdà, 2009; Shakesby, 2011; San-Miguel-Ayanz et al., 2013). These high severity wildfires bring about the destruction of vegetation cover (Úbeda et al., 2006; Tessler et al., 2015), the volatilization and evacuation of nutrients by ash (Bodí et al., 2014; Pereira et al., 2015b,c), and an increase in soil and water loss through erosion and runoff (Cerdà and Lasanta, 2005; Cerdà and Doerr, 2008; Gimeno-García et al., 2008), which can have a negative impact on rivers and other water bodies (Smith et al., 2011; Brown et al., 2015; Estrany et al., 2015).

Previous studies of Mediterranean ecosystems observed that severe wildfires can make major changes to a soil's physical and chemical properties (Kutiel and Naveh, 1987; Knicker et al., 2005; Arcenegui et al., 2008; Certini et al., 2013; Badía et al., 2014; Inbar et al., 2014). Soil aggregate stability can be severely affected by fire (Cerdà, 1993; Cerdà et al., 1995). According to Mataix-Solera et al. (2011), the impact of fire on soil aggregates depends on how the fire affects other soil properties (including its mineralogy, water repellency, organic matter content and microbiology). Additionally, the impact will depend on associated geographical variables, such as aspect (Andreu et al., 2001), on the type of fire (Mataix-Solera et al., 2002) and on the soil type. The studies reported to date fail to provide a consensus, with some identifying an increase (Campo et al., 2006; Arcenegui et al., 2008), a decrease (Mataix-Solera et al., 2002; Kavdir et al., 2005) or no impact (Varela et al., 2010) on soil aggregate stability after severe wildfires. Indeed, Mataix-Solera et al. (2011) report that depending on many factors, fires can result in a disaggregation, owing to the consumption of organic matter, or in an aggregation, owing to the recrystallization of some iron and aluminum oxyhydroxides.

Wildfires have also been shown to change other essential soil chemical properties, albeit temporarily (Bento-Goncalves et al., 2012). Typically, nitrogen and carbon levels decrease after a severe wildfire due to low-temperature volatilization at around 200 ºC (Hernandez et al., 1997; Grogan et al., 2000; Johnson et al., 2004; Badía et al., 2014). The nitrogen and carbon fractions that are not volatilized, but present in the ash, are transformed by the heat from the fire (e.g., black carbon, black nitrogen and carbonates) (Knicker, 2007; Gonzalez-Perez et al., 2004), and later incorporated into the soil profile (Knicker et al., 2013; Boot et al., 2015). The pyrolyzed material incorporated into the soil profile has a low C/N ratio, given that nitrogen is mineralized more rapidly than carbon (Baird et al., 1999). This process of mineralization increases the short-term availability of soluble forms of nitrogen and carbon (Prieto-Fernández et al., 2004), soil pH, electrical conductivity (EC) and the major extractable elements (Kutiel and Inbar, 1993; Murphy et al., 2006). The ease of solubility of these elements is critical for the regeneration of vegetation. However, at the same time, the type and amount of nutrients released depend heavily on the severity of the fire and the type of forest affected (Raison and McGarity, 1980; Úbeda et al., 2009; Pereira et al., 2012; 2014).

All in all, the disturbance induced by fire in both the type and the amount of available nutrients, with some being more soluble than others, affects the ratios between these nutrients. Typically, in the immediate post-fire period, an increase is recorded in the major extractable nutrients and a reduction is seen in the minor elements as a consequence of the rise in soil pH (Pereira et al., 2011). Studies conducted elsewhere have shown that solutions richer in extractable sodium and potassium in relation to the levels of calcium and magnesium can increase clay dispersion and reduce aggregate stability (Sarah, 2004; 2005; Igwe, 2005; Bagarello et al., 2006), which in turn may make soils more vulnerable to erosion after a fire. Normally, soil extracts from fireaffected areas and ash slurries are rich in sodium and potassium (Khanna et al., 1994; Liodakis et al., 2009 Pereira et al., 2014). The increase in the amount of monovalent cations in burned soils can increase the potential effect of soil solute chemistry in the soil structure. This is especially relevant in recently burned areas, because it is when the soils are most vulnerable to erosion, owing to litter combustion and vegetation removal (White et al., 1996; Duguy et al., 2012). The presence of a layer of ash, rich in sodium and potassium, can release solutions that act as dispersants of soil particles after the first rainfalls, or when incorporated into the soil profile. This additional disturbance may increase the vulnerability of soils to erosion. The study of the sodium and potassium adsorption ratio (SPAR) is important to understand this impact. Very few studies, to date, have assessed the impact of fire on SPAR. To the best of our knowledge, the only study examining the effects of fire on the SPAR was conducted by Pereira et al. (2014) in ash extracts. Here, the authors identified a significant increase in the SPAR in relation to litter solutions in ash slurries. Other studies in fire-affected areas have examined the sodium adsorption ratio (SAR) in soils, but ignoring the contribution of extractable potassium (Blank et al., 2003; Inbar et al., 2014). However, it is important to consider extractable potassium as a potential dispersant, especially in fire-affected areas, where ash and burned soil release considerable amounts of this element in solution (Blank and Zamudio, 1998; Badía and Marti, 2003; Murphy et al., 2006).

The recovery of the burned areas depends on both the topography (e.g., aspect and slope) of the fire-affected area and the meteorological conditions, especially the wind, and the amount and intensity of rainfall. In the immediate post-fire period, wind and rain can (re)distribute ash and the nutrients available for plant recovery across the burned area (Moody et al., 2013; Pereira et al., 2013; 2015b). Meteorological variables are a critical aspect in post-fire environments. The interaction between extreme rainfall episodes and wildland fires is complex, with the fire severity variability being a key factor at the microplot scale (De Luis et al., 2003). Indeed, extreme rainfall events after wildland fires are the cause of high rates of soil erosion, and water and nutrient losses in Mediterranean environments (Inbar et al., 1998; Shakesby, 2011).

Given the impact of torrential rainfall events on areas affected by wildfires, it is clearly important to determine their effects on soil properties. In particular, it is interesting to monitor the significance of the changes attributable to a single event and the soil conditions in the medium-term (one year). The objective of this study is to examine the impact of an intense rainfall event on the soil properties of a recently burned area and to compare these results with data collected one year later.

2. Materials and Methods

2.1 Study Area

The study area is located in Colomers, Girona (42º05'17.6" N - 2º59'36.6" E 38 masl) in North-East Spain. The fire started in Camallera, on 12 November 2013 and burned a total of 550 ha of *Pinus halepensis* Miller, *Quercus ilex* L. and *Eucaliptus globulus* Labill. The geological substratum of the burned area is composed mainly of sedimentary rocks from the lower Bartonian. Soils were classified as Fluventic Haploxerept (Soil Survey Staff, 2014). The main soil properties of the studied area are shown in Table 1. The mean annual temperature of the studied area is 15 °C and the average annual rainfall ranges between 600 and 800 mm (Pacheco, 2010).

Table 1. Characteristics of the soils (0-5 cm depth) of the studied area.

	Value
Aggregate stability (% of disaggregation)	2.42
Total Nitrogen (%)	0.46
Organic Carbon (%)	8.47
Inorganic Carbon (%)	2.2
C/N ration	18.14
Carbonates (%)	10.71
pH	7.19
EC (µS/cm)	337
Extractable calcium (ppm)	32122
Extractable magnesium (ppm)	733
Extractable sodium (ppm)	194
Extractable potassium (ppm)	313
Avalable phosphorous (ppm)	34.84
SPAR	0.991

2.2. Experimental design and field sampling

Three days after the fire, a 72 m^2 experimental plot (18 x 4 m, with a grid with 2 m spacing between sampling points) in a flat area (slope <1%) was designed inside the burned area. Fire severity in this experimental plot was classified as high according to Úbeda et al. (2009) and Pereira et al. (2012), given that three crown had combusted and the soil was covered with a layer of grey and white ash (Figure 1). A total of 30 soil samples (0-5 cm) were collected in the aforementioned grid, 3 days after the fire and before the rainfall event. The second sampling campaign was carried out 4 days after the start of the intense rainfall event (163.7 mm during 2 days), that is, 7 days after the fire. Rainfall data were registered at the meteorological station of Bisbal d'Empordà (41º97' N - 03º04' E. 29 masl). The last sampling period was one year after the fire. The total rainfall between the second and third sampling dates was 538.6 mm. The samples were stored in plastic bags and taken to laboratory for analysis.

Figure 1. Ash layer covering the studied plot before the rainfall event.

2.3 Laboratory methods

Soil samples were dried to constant weight at room temperature (approx. 23 °C) during 7 days. To analyze pH, electrical conductivity (EC) and amounts of extractable elements (calcium, magnesium, sodium and potassium) the samples were sieved using a 2-mm mesh sieve. To analyze aggregate stability, the samples were sieved using a 4.8-mm mesh sieve. For each sample, ten air-dried aggregates (between 4-4.8 mm) were collected. The ten drop impact (TDI) method (Low, 1954) was used to study aggregate stability. The method involves placing a 2.8-mm sieve over the aggregates and subjecting them to the impact of ten drops of deionized water, from a droplet placed at a height of 1 m. Each drop had a weight of 0.1 ± 0.001 g and a diameter of 2.8 mm. Aggregates were weighed before the test and the disaggregated material, which passed through the sieve during the test, was also weighed after drying at 105 ºC for 24 h. The results are expressed as a percentage of the disaggregated weight.

Soil organic carbon, inorganic carbon and carbonates were measured using the loss-on-ignition (LOI) method described in Heiri et al. (2001). For each sample, 1 gr of soil was pulverized and dried in a muffle furnace at 105 ºC during 24 hours. To estimate soil organic carbon, the dried samples were exposed to a temperature of 550 ºC during 4 hours. Finally, to calculate soil inorganic carbon, the samples were subjected to a temperature of 950 ºC during 2 hours. The results were expressed as percentages. Total carbon and total nitrogen contents were analyzed in the pulverized samples using gas chromatography combustion-reduction with a thermal conductivity detector Flash EA 112 Series (Thermo-Fisher Scientific, Milan). Data acquisition and calculations were carried out using Eafer 300 software (Thermo-Fisher Scientific, Milan) (Pereira et al., 2012).

Soil pH [1:2.5] and EC [1:2.5] were analyzed following extraction with deionized water. Available phosphorus was analyzed using the Olsen Gray method (Olsen et al., 1954). Soil cations (calcium, magnesium, sodium, and potassium) were extracted from samples using ammonium acetate (Knudsen et al., 1986). The extractable cation content was analyzed by inductively coupled plasma mass spectrometry (ICP-MS), using a PerkinElmer Elan-6000 Spectrometer, and by optical emission spectrometry (OES), using a PerkinElmer Optima-3200 RL Spectrometer. Soil SPAR was calculated according to Sarah (2004) (extractable sodium+extractable potassium)/(extractable calcium+extractable magnesium)$^{1/2}$.

2.4 Statistical analysis

Prior to data comparison, data normality and homogeneity of variances were tested using the Shapiro-Wilk and the Levene tests, respectively. None of the data respected the Gaussian distribution or heteroscedasticity. Aggregate stability, total nitrogen, organic carbon, exchangeable magnesium and exchangeable potassium met assumptions of normality and homogeneity of variances after a logarithmic transformation. Available phosphorous only met the assumptions of data normality and homogeneity after a square root transformation. The remaining variables did not meet these requirements, even after a Box-Cox transformation. For those variables that satisfied assumptions of normality and heteroscedasticity, statistical comparisons among sampling dates were conducted using the parametric one-way ANOVA on the transformed data. If significant differences were found, a Tukey HSD test was applied. For the variables that did present normality after all the transformations, the non-parametric Kruskal-Wallis ANOVA (KW) was used. If significant differences were identified, the multiple comparison post-hoc test was applied. Principal component analysis (PCA) was conducted with a varimax rotation based on the correlation matrix (using log-transformed data) to identify the relations among the variables studied and their association according to the different sampling periods. Statistical analyses were conducted using Statistica 10.0.

3. Results

3.1 Aggregate stability

Significant differences were identified in the soil aggregate stability on the three sampling dates ($F=208.93$, $p<0.001$). A decrease in soil aggregate stability (i.e., *an increase in the percentage of the disaggregated sample*) was observed with time. Soil aggregate stability was significantly lower one year after the fire (Figure 2).

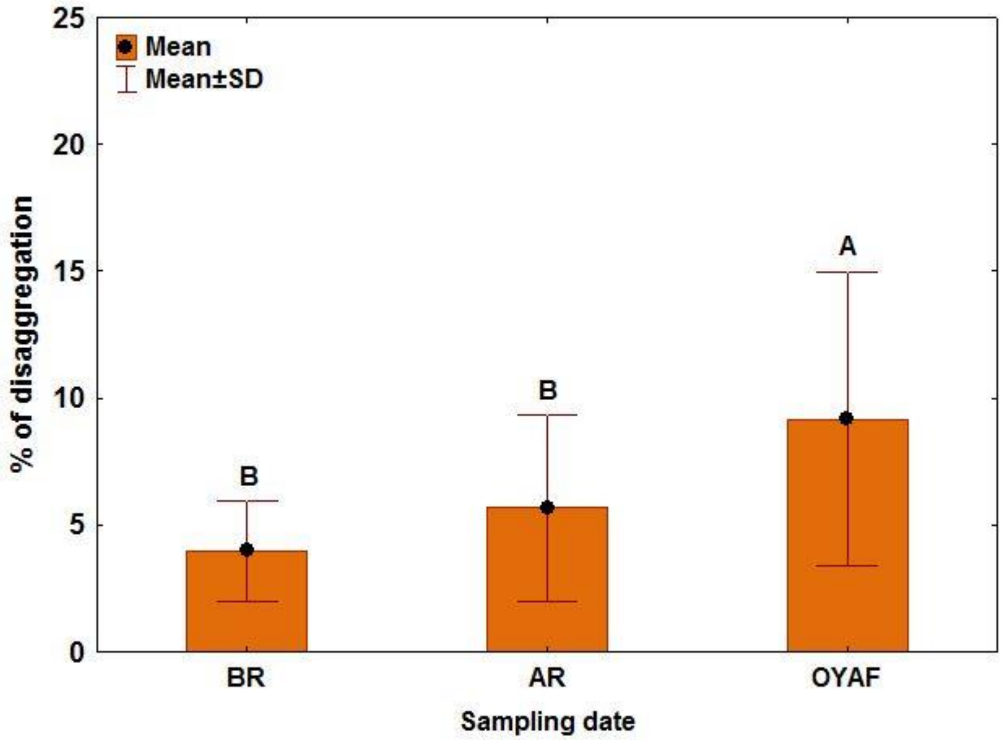


Figure 2. Mean and Standard deviation of the % of soil disaggregation. Different letters represent significant differences at a p<0.05. BR (Before the rainfall), AR (After the rainfall) and OYAF (One year after the fire). N=90

3.2 Total nitrogen, total carbon, organic carbon and inorganic carbon, C/N ratio and carbonates

No significant differences were observed in total nitrogen (F=1.57, p>0.05), total carbon (KW=3.80, p>0.05), inorganic carbon (KW=4.96, p>0.05), C/N ratio (KW=1.73, p>0.05) and carbonate content (KW=3.98, p>0.05) between the three sampling dates (Table 2). Significant differences were only observed in organic carbon content (F=4178, p>0.001), the amount being significantly higher immediately after the rainfall event than before the event and one year after the fire (Table 2).

Table 2. Descriptive statistics of total nitrogen, total carbon, organic carbon, inorganic carbon, C/N ratio, and carbonates. Different letters represent significant differences at a p<0.05. N=90.

		Mean	SD	Min	Max	p value
Total Nitrogen (%)	Before rainfall	0.32	0.08	0.17	0.52	n.s[2]
	After rainfall	0.34	0.10	0.15	0.55	

	One year after fire	0.30	0.08	0.17	0.47	
	All	0.32	0.09	0.15	0.55	
Total Carbon (%)	Before rainfall	7.58	2.14	4.85	17.21	
	After rainfall	7.69	1.78	3.50	11.73	n.s[1]
	One year after fire	7.08	1.59	4.90	10.94	
	All	7.45	1.85	3.50	17.21	
Organic Carbon (%)	Before rainfall	5.20b	1.28	2.86	8.23	
	After rainfall	6.03a	1.48	3.77	9.72	***[2]
	One year after fire	4.58b	1.10	2.91	7.44	
	All	5.27	1.42	2.86	9.72	
Inorganic Carbon (%)	Before rainfall	3.49	0.40	2.21	4.42	
	After rainfall	3.43	0.22	2.87	3.97	n.s[1]
	One year after fire	3.55	0.20	3.12	3.93	
	All	3.49	0.29	2.21	4.42	
C/N ratio	Before rainfall	24.20	4.50	18.76	44.13	
	After rainfall	23.33	2.53	19.55	30.19	n.s[1]
	One year after fire	24.43	3.40	20.05	37.45	
	All	23.99	3.56	18.76	44.13	
Carbonates (%)	Before rainfall	16.61	3.02	4.33	21.71	
	After rainfall	16.73	1.12	13.69	19.41	n.s[1]
	One year after fire	17.33	1.11	15.12	19.35	
	All	16.89	1.97	4.33	21.71	

[1]original, [2]log transformed and [3]square-root transformed data

3.3 pH, electrical conductivity, major cations and available phosphorous and SPAR.

Significant differences were observed in soil pH (KW=43.89, p<0.001) and EC (KW=64.52, p<0.001) between the sampling dates. Soil pH and EC values were significantly higher before the rainfall event than on both occasions after the intense rainfall episode. In the case of EC, a decrease was observed with time (Table 3).

Extractable calcium (KW=70.72, p<0.001), extractable magnesium (F=42802.06, p<0.001) and available phosphorous (F=790.18, p<0.001) also differed significantly between the different sampling dates. The amounts of these three elements were significantly higher immediately after the rainfall event than on the other two sampling dates. Significant differences were identified in extractable sodium (KW= 30.62, $p<0.001$) and extractable potassium (F=27283.18, $p<0.001$) levels, with both being significantly higher before the rainfall event. Finally, significant differences were also observed in the SPAR (KW= 43.05, $p<0.001$), with values being significantly higher both immediately after the rainfall event and one year after the fire (Table 3).

Table 3. Descriptive statistics of pH, electrical conductivity (EC), extractable calcium, extractable magnesium, extractable sodium, extractable potassium, available phosphorous and SPAR. Different letters represent significant differences at a $p<0.05$. N=90.

		Mean	SD	Min	Max	p value
pH	Before rainfall	8.52a	0.28	7.90	9.05	***[1]
	After rainfall	7.99b	0.09	7.84	8.21	
	One year after fire	8.19[a]	0.27	7.69	8.73	
	All	8.24	0.32	7.69	9.05	
EC (µS/cm)	Before rainfall	533.95a	390.40	155.40	2280.00	***[1]
	After rainfall	272.08b	123.93	146.80	687.00	
	One year after fire	113.71c	37.00	71.40	265.00	
	All	306.58	293.65	71.40	2280.00	
Extractable calcium (ppm)	Before rainfall	26064.15b	6825.92	17649.47	45245.4	***[1]
	After rainfall	47483.94a	11145.48	31279.83	70433.5	
	One year after fire	17142.14c	2775.97	13742.49	24454.3	
	All	30230.08	14903.23	13742.49	70433.5	
Extractable agnesium (ppm	Before rainfall	711.47b	231.17	411.12	1157.09	***[2]
	After rainfall	967.37a	304.23	424.90	1762.25	
	One year after fire	346.20c	78.86	214.27	526.19	
	All	675.02	339.56	214.27	1762.25	
Extractable sodium (ppm)	Before rainfall	367.17b	170.00	77.50	925.22	***[1]
	After rainfall	508.36a	222.15	168.09	1036.97	
	One year after fire	328.16b	746.68	62.40	4230.07	
	All	401.23	401.23	62.40	4230.07	
Extractable otassium (ppm)	Before rainfall	431.56a	251.76	251.76	1550.88	***[2]
	After rainfall	408.90a	408.90	115.54	733.44	

	One year after fire	174.27b	174.27	44.10	261.00	
	All	338.24	338.24	198.29	1550.88	
Available phosphorous (ppm)	Before rainfall	45.11b	30.80	2.28	140.61	***[3]
	After rainfall	65.54a	37.21	5.37	161.92	
	One year after fire	19.78c	6.29	2.90	29.48	
	All	43.47	33.58	2.28	161.92	
SPAR	Before rainfall	0.984b	0.007	0.952	0.990	***[1]
	After rainfall	0.991a	0.002	0.985	0.994	
	One year after fire	0.990a	0.002	0.986	0.993	
	All	0.988	0.005	0.952	0.994	

[1]original, [2]log transformed and [3]square-root transformed data

3.4 Multivariate Analysis

The PCA carried out identified five factors that at least account for one variable. These first five factors explained a total of 79.21% of the total variance. The first two factors explained 38.73 and 15.24% of the variance, respectively. The intersection of these factors allowed us to identify four different groups of factors: the first comprised EC, total nitrogen, total carbon, organic carbon, extractable calcium, magnesium, sodium, potassium and available phosphorous; the second was integrated solely by the SPAR; the third comprised the aggregate stability, inorganic carbon, carbonates and C/N ratio; and finally, the fourth was integrated by just the soil pH (Figure 3a). The variables in group 1 were negatively correlated with the variables in groups 2, 3 and 4. The variable in group 2 was negatively correlated with the sole variable in group 4. The variables in group 3 were positively correlated with the variable in group 2 and uncorrelated with the variable in group 4. The analysis of the cases showed that when considering all the variables there were no major differences between the sampling periods (Figure 3b).

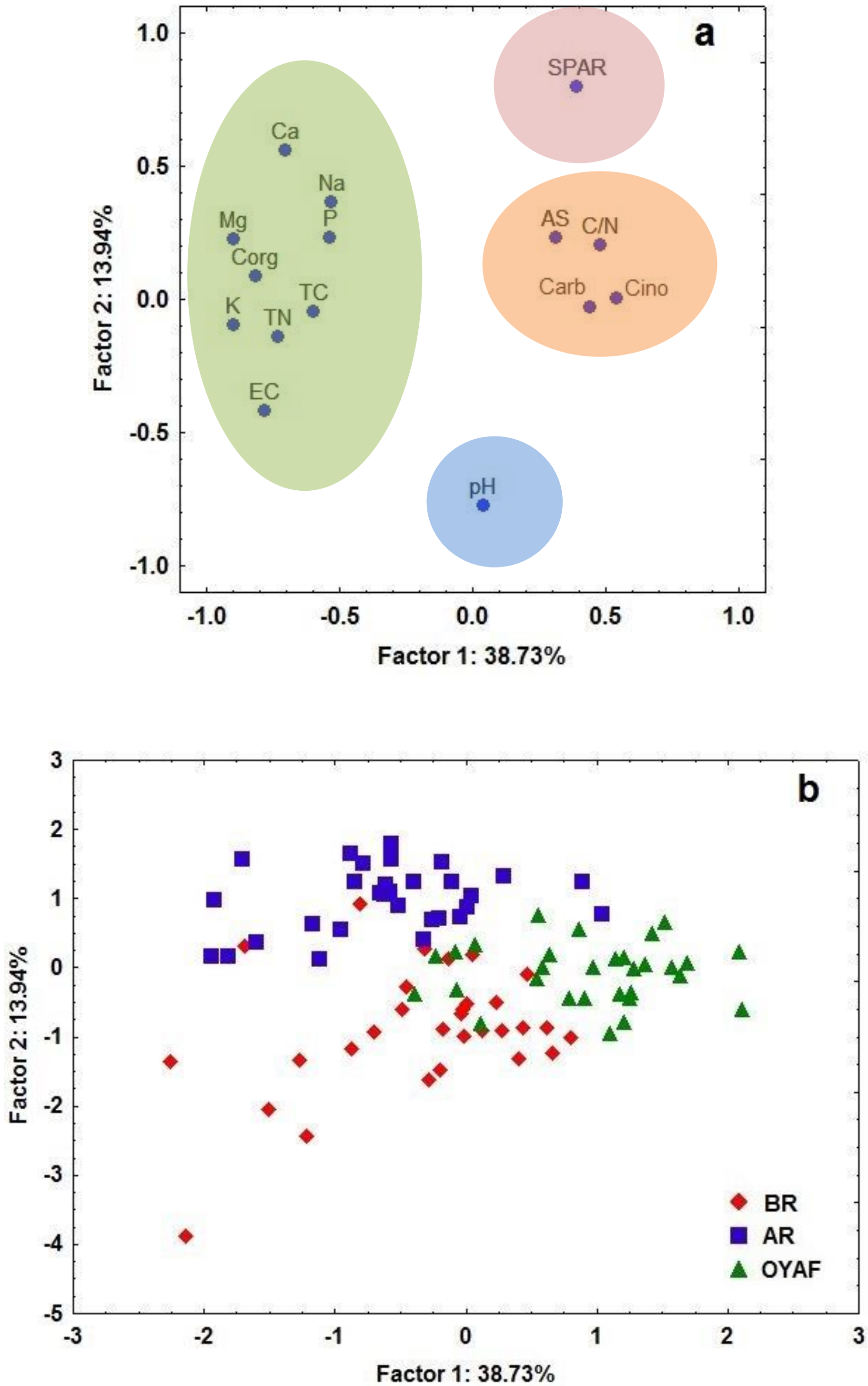


Figure 3. Relation between Factor 1 and 2, a) variables and b) cases. BR (Before the rainfall), AR (After the rainfall) and OYAF (One year after the fire). Aggregate stability (AS), total nitrogen (TN), total carbon (TC), organic carbon (Corg), inorganic carbon (Cino), carbon/nitrogen ratio (C/N), carbonates (Carb), electrical conductivity (EC), extractable calcium (Ca), extractable magnesium (Mg), extractable sodium (Na), extractable potassium (K) and sodium and potassium adsorption ratio (SPAR)

4. Discussion

4.1. Aggregate stability

The intense rainfall episode reduced soil aggregate stability immediately after the event, but not significantly; significant differences in soil disaggregation were only observed one year after the fire. Thus, despite the high severity of the fire, which consumed three crown layers, it seems likely that the ash layer reduced raindrop impact on the soil surface and protected soil aggregates (Figure 4a). The ash layer has been shown to be the first and most effective soil protector in the immediate post-fire period (Pereira et al., 2015b), acting as natural mulch, reducing runoff and soil erosion, and increasing water storage and infiltration (Cerdà and Doerr, 2008; Woods and Balfour, 2008; Bodí et al., 2012; León et al., 2013). Given the vulnerability of burned soils to raindrop impact and particulate detachment, the protection afforded by the ash is of major importance (Gabet, 2014).

This means that the significant differences noted a year after the fire could well be related to the reduction in this source of protection (i.e., as a result of ash erosion, Figure 4b), as well as to the post-fire management program, in which six months after the fire trees were cleared (Figure 4c), and to vegetation recovery. According to Velasco and Úbeda (2014), the length of time that the soil is exposed to raindrop impacts is more important than the temperature reached by the fire. Post-fire management represents an additional disturbance in fire-affected soils, in some instances more severe than the fire itself. For example, Mataix-Solera et al. (2015) report that salvage logging reduces soil aggregate stability in burned areas. Likewise, it has been observed that the recuperation of the vegetation can reduce aggregate stability, despite the fact that soil infiltration is high (Cerdà et al., 1995; Velasco and Úbeda, 2014). Fattet et al. (2011) reported that grass and shrub regrowth induces an initial soil disaggregation in the first centimeters of the soil due to root development. Here, it is highly likely that the type of vegetation cover influenced soil aggregate stability. Cerdà (1998) observed, for example, that soil aggregates under shrub vegetation (*Rosmarinus officinalis*, *Thymus* v*ulgaris*, *Ulex parviflorus* and *Anthyllis cystisoides*) were less resistant than those under mature forests (*Quercus ilex*). Soil aggregate stability response to fire is complex and depends on the type of soil affected, the temperature and severity of the fire, the topography of the burned area, the post-fire meteorological conditions, and the degree of vegetation recovery and management (Mataix-Solera et al., 2011). In the present study, it seems likely that the reduction in soil aggregate stability results from the interaction of all these variables and the soil's chemical properties, as we explain below.

a

b

c

Figure 4. Study area a) immediately after the fire, b) after the intense rainfall and c) one year after the fire

4.2 Total nitrogen, total carbon, organic carbon and inorganic carbon, C/N ratio and carbonates

The intense rainfall event did not have any significant impact on the soil's total nitrogen, total carbon, inorganic carbon, C/N ratio and carbonate levels. In the period immediately after the event this can be attributed to two factors. First, the total elimination of ash from the plot (see Figure 4b) reduced the amount of ash nutrients that could be incorporated into the soil profile. Indeed, previous studies have attributed the increase in nitrogen and carbon pools in fire-affected soils to ash (Gonzalez-Perez et al., 2004; De Marco et al., 2005). Badía et al. (2014) observed that after a severe wildfire the topsoil (0-1 cm) organic carbon was reduced significantly. One year after, the soil organic carbon of the burned soil was similar to that identified in the unburned plot. The authors attributed this to the incorporation of ash and new inputs derived from reblooming, which begins after the fire. However, ash is a highly mobile material, especially in high severity fires, because of its low density and, therefore, its light specific weight. It has been shown that ash produced at high temperatures loses most of its mass (Úbeda et al., 2009) and that it is easily transported by wind and water (Pereira et al., 2015c). The second factor was the severity of the fire. Nitrogen and carbon are volatilized at low temperatures (Fisher and Binkley, 2000), and so it seems highly likely that the ash deposited on the surface of the study area was poor in these elements. Light colored ash (see Figure 1) is an indication of high severity fires. Pereira et al. (2012) observed that the total nitrogen and total carbon of ash fell significantly with increasing fire severity. Overall, the intensity of the precipitation, wind and the severity of the fire may have limited the contribution of ash to the soil's total nitrogen and carbon content.

Ash produced at high temperatures contributes primarily to the soil's inorganic compounds and carbonates (Bodí et al., 2014). However, here, no differences were recorded in their respective levels in the burned soils before and after the rainfall event. As discussed, ash produced at high temperatures is light weight and easily transported. Moreover, white ash, rich in inorganic compounds and carbonates (Goforth et al., 2005; Pereira et al., 2012), is the first to be transported or incorporated into the soil profile (Pereira et al., 2013). Pereira et al. (2015c) observed that after a severe wildfire the white ash (rich in carbonates and other inorganic material) was easily transported by wind. They reported that 15 days after the fire, in a period without rainfall, the % area covered in light grey and white ash was reduced. Similar results were noted by Pereira et al. (2013) after a grassland fire where the ash cover reduction was especially important at points where the ash was light grey and white in color. In this study, it seems probable that most of the inorganic compounds and carbonates present in the ash produced at high severity were eroded by wind, because as Úbeda et al. (2009) observed they are more readily transported (light weight). Water may have had some small influence in ash erosion as the study plot was flat. Nevertheless, we did observe a significant increase in the amount of organic carbon after the rainfall episode, indicating that some material not completely combusted in situ was incorporated into the first few centimeters of the soil. Despite the severity of the fire, not all the organic matter was combusted, and some small particles of charcoal were visible (see Figure 1). Unlike white ash, black ash is rich in organic material (Dlapa et al., 2013) and more resistant to transport (being heavier than white ash), and may remain in the soil surface for several weeks after a fire (Pereira et al., 2013). This, and the fact that the studied area was flat, may have favored the maintenance of charcoal fragments on the soil surface, which were incorporated after the rainfall period. However, in areas affected by high severity fires, ash is extremely mobile, while it is also possible that such particles are deposited from other areas (Pereira et al., 2015c). Here, both processes are likely to have coincided during the rainfall event, but we are unable to identify the main contributor to the increase in soil organic carbon.

A year after the fire, the soil's organic carbon decreased and showed similar values to those observed before the rainfall event. This decrease can be attributed to the incorporation of this material in deeper soil profiles (Woods and Balfour, 2010), microbiological decomposition (Knicker et al., 2013), erosion in subsequent precipitation events (Novara et al., 2011) and post-fire management.

4.3 pH, electrical conductivity, major cations, available phosphorous and SPAR

Soil pH and EC decreased significantly after the intense rainfall event, a reduction that can be attributed to the transport of elements in overland flow and/or leaching into the soil profile. These results are in line with previous studies that report a decrease in soil pH and EC after periods of rain (Kutiel and Naveh, 1987). Overall, a decrease was identified with time, so that a year after the fire, soil pH and EC were lower than on the previous two sampling dates. All the extractable cations (with the exception of potassium), available phosphorous and the SPAR were significantly higher in the period after the rainfall event. This increase can be attributed to the leaching of ash nutrients (Raison and McGarity, 1980; Pereira et al., 2014). Badia et al. (2014) also observed that extractable calcium, magnesium and phosphorous increased one week after the fire. The changes observed were attributed to the heat released during the fire, which decreased the soil organic carbon, but mineralized the organic matter, increasing the solubility of calcium, magnesium and phosphorous. Despite the fact that soil pH and EC presented their highest levels prior to the rainfall event, these elements were released in greater amounts in the sampling period after the precipitation event. We hypothesize that this may be a consequence of the fact that:

1. Soil pH was too high for the optimal solubility of some of these elements. The optimal solubility of extractable calcium and magnesium is around pH 7-8 (Neary et al., 2005), while available phosphorous is extractable at a pH ranging from 6.5 to 7.5 (Varennes, 2003). In the period before the rainfall event, soil pH presented an average value of 8.52, thus limiting the solubility of these elements. After the intense period of precipitation, soil pH fell to 7.99, thus increasing the probability that these elements could be

extracted. Extractable sodium and potassium are highly soluble at a pH above 7.5 (Troeh and Thompson, 2005), a level that was observed both before and after the rainfall period. Despite this, we observed that levels of extractable sodium were significantly higher in the period after the rainfall event, while no differences were observed in those of extractable potassium. It seems probable that the ash, which was incorporated into the soil profile, significantly increased the amount of extractable sodium in the soil. These results are in line with previous reports that observed an increase in the major extractable elements and in available phosphorous after a fire (Murphy et al., 2006; Galang et al., 2010).

2. Soil EC was significantly higher before the rainfall event than it was after, despite the fact that the major cations presented the opposite dynamic. This high level of conductivity may reflect the contribution of other chemical elements not considered in this study, such as anions (including, nitrites, nitrates, sulfides, chlorides). Anion leaching makes a major contribution to water ionic composition (Kitamura, 2009); indeed, previous studies have shown that large amounts of anions can be leached from soils affected by wildfires (Kutiel and Inbar, 1993; Gimeno-García et al., 2008; Thomas et al., 2000; Ferreira et al., 2005; Lasanta and Cerdà, 2005; Murphy et al., 2006).

A year after the fire, soil pH had increased significantly (compared to the value recorded after the rainfall event), while EC, and the amounts of all the extractable elements, had fallen in relation to the results recorded in the previous sampling period. This can be attributed to further losses in the overland flow generated by subsequent rainfall events, soil leaching, post-fire management and plant consumption (Neary et al., 2005; Murphy et al., 2006; Badia et al., 2014). The latter can be verified in Figure 4c, which shows that vegetation covered 100% of the burned plot.

The SPAR index was significantly higher after the rainfall event and one year after the fire than in the period before the intense precipitation. Soil solutions with a high SPAR in these last two sampling periods may have increased the vulnerability of soils to erosion because of their high clay dispersion capacity (Sarah, 2004; 2005). This increase in the index after the rainfall event is probably the consequence of the incorporation of some elements of ash into the soil profile (Pereira et al., 2014). To our knowledge no previous studies have examined SPAR levels in fire-affected soils, and further studies need to be conducted in order to understand the impact of soil solution chemistry on soil structure. Soil solutions with a high SPAR can have a negative impact on soil structure (Sarah, 2005). The decrease in soil aggregate stability was significantly higher one year after the fire, when the SPAR values were high. In combination with the factors discussed above, the increase in SPAR values may well have had a negative impact on soil aggregate stability.

4.4. Overall discussion and implications for post-fire management

Intense rainfall events can have a negative impact on recently burned soils, given that they are greatly exposed to erosion agents. This is especially significant in mountainous areas where steep slopes facilitate soil erosion. In such instances it is recommended that some post-fire restoration strategies be applied in order to reduce the vulnerability of soils to degradation, as has been observed in other areas of the Mediterranean (Castro et al., 2011; Shakesby et al., 2011; Badía et al., 2015). In the present case, the torrential rainfall did not have an extremely negative impact on the burned area, since much of the vegetation recovered, as can be observed in Figure 4c. This can be attributed to the fact that the study area was flat, which reduced the impact of this precipitation event and other post-fire meteorological factors. Mediterranean ecosystems are highly resilient to fire and can recover rapidly without any need for restoration measures (Cerdà and Doerr, 2005; López-Poma et al., 2014). This was the case here, despite the disturbances induced by tree removal.

The intense rainfall event had different impacts on some soil properties and it seems probable that these can be connected to the ash layer that had formed after the fire. Thus, the ash layer reduced raindrop impact and soil aggregate degradation, but because it resulted from a high severity combustion event it did not modify total nitrogen and total carbon levels significantly (given that these elements appear in very low amounts in white ash). However, despite its severity, the fire did not consume all the organic material and these charred remains increased the soil's organic carbon content following the rainfall event. Ash is rich in cations and phosphorous, which volatize at higher temperatures than carbon and nitrogen (Bodí et al., 2014). These elements are easily soluble, especially at their optimum pH.

Post-fire management is well documented as having a marked impact on soil properties (Gomez-Rey et al., 2013; Powers et al., 2013; Wagenbrenner et al., 2015); however, here, the degradation of soil structure cannot be attributed solely to human intervention, as other variables – including ash and soil erosion, vegetation recuperation and soil chemical properties – are also influential. Further studies are needed to identify the direct impact of each of these variables. The initial increase in nutrients in soil solution had been reversed one year after the fire, as a result of erosion, leaching and plant consumption. Taking all the elements studied into consideration – the intense rainfall episode, other natural post-fire events and human management, no major differences were recorded in the soils in relation to their initial values (i.e., before the rainfall) and we did not identify any clear groups in the PCA analysis. The location of the sampling points in the graph does not point to any clear differences between them (Figure 3b). This can be attributed to the lack of significant differences in many of the variables analyzed, including total nitrogen, total carbon, inorganic carbon, the C/N ratio and carbonate content, which may have had an influence on this result. The effects of wildfires on the soil properties of burned areas have been well described (Murphy et al., 2006;

Caon et al., 2014), and, given the values recorded in Table 1, it is apparent that our study area experienced a similar impact; however, here, the overall effects were not significant.

Ash is a valuable protector of soils in post-fire environments, as highlighted in many studies (Onda et al., 2008; Bodí et al., 2012; 2014; Cerdà and Doerr, 2008; Jordán et al., 2015; Pereira et al., 2013; 2015a), and it is also a major source of soil nutrients (Pereira et al., 2014). However, to fulfill these roles, it is essential that the ash remains in the soil layer undisturbed by human intervention. Disturbance exposes soils to erosion agents and increases the mobility of ash, especially in areas affected by high severity fires (Pereira et al., 2015b) as the one experienced here. Intense rainfalls, though, are a normal occurrence in Mediterranean environments and so the protection against erosion agents is frequently destroyed (Cerdà and Doerr, 2008). The erosion of ash reduces the capacity of the vegetation to recover after a fire, and so it is in these environments in particular that salvage logging should be avoided, given the additional disturbance it represents to the soil ecosystem. Clearly, therefore, the fire management program (including tree removal) carried out in this area was not the most appropriate for an area affected by a high severity fire, since it almost certainly constituted an additional disturbance to the soil. In this study, we have not been able to identify the specific impact of tree removal and to isolate its effects from the other variables; however, it may have contributed to soil structure deterioration one year after the fire. Overall, despite this impact and the incorrect management, the topographic characteristics of the study plot may have facilitated the rapid recuperation of the ecosystem and mitigated the effects of human intervention. However, these practices on slopes, and especially with certain soil types, can lead to soil erosion and degradation (Mataix-Solera et al., 2015).

5. Conclusions

Significant impacts were only observed one year after the fire as a consequence of ash and soil erosion, tree removal, vegetation recuperation and changes in soil chemical properties. Overall, this study highlighted that:

a) This intense rainfall period did not have a significant impact on soil aggregate stability in this area affected by a high severity fire. Significant impacts were only observed one year after the fire as a consequence of ash and soil erosion, tree removal, vegetation recuperation and changes in soil chemical properties such as SPAR.

b) The fire did not have an impact on total nitrogen, total carbon, inorganic carbon, the C/N ratio or carbonate levels. However, particles of charred material, either transported from other areas or which were more resistant to erosion, may have contributed to a significant increase in organic carbon.

c) Extractable cations and available phosphorous increased after the rainfall event as a consequence of a favorable pH. SPAR values were also significantly higher after the rainfall event, indicating that it may have contributed to the decrease in soil aggregate stability.

d) One year after the fire, a significant decrease was observed in aggregate stability and extractable elements. This was probably due to the high SPAR levels (in the case of the decrease in soil aggregate stability), human intervention and soil leaching provoked by other rainfall events and vegetation consumption.

Thus, based on the results of the analyses conducted in this study, the intense rainfall event and other post-fire processes (attributable to natural or human causes) did not greatly modify the soil properties as recorded before the intense rainfall episode.

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References

Andreu V, Imeson AC, Rubio JL. Temporal changes in aggregates and water erosion after a wildfire in a Mediterranean forest. Catena 2001; 44: 69–84.

Arcenegui V, Mataix-Solera J, Guerrero C, Zornoza R, Mataix-Beneyto J, García-Orenes F. Immediate effects of wildfires on water repellency and aggregate stability in Mediterranean calcareous soils. Catena 2008; 74: 219–226.

Badía D, Martí C. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Res. Manag. 2003; 17: 23–41.

Badía D, Martí C, Aguirre AJ, Aznar JM, González-Pérez JA, De la Rosa JM, León J, Ibarra P, Echeverria T. 2014. Wildfire effects on nutrients and organic carbon of a Rendzic Phaeozem in NE Spain: Changes at cm-scale topsoil. Catena 2014; 113: 267–265.

Badía D, Sanchez C, Aznar JM, Martí C. Post-fire hillslope log debris dams for runoff and erosion mitigation in the semiarid Ebro Basin. Geoderma 2015; 237-238: 298–307.

Bagarello V, Iovino M, Palazzolo E, Panno M, Reynolds WD. Field and laboratory approaches for determining sodicity effects on saturated soil hydraulic conductivity. Geoderma 2006; 130: 1–13.

Baird M, Zabowski D, Everett RL. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant Soil 1999; 209: 233–243.

Bento-Goncalves A, Vieira A, Úbeda X, Martin D. Fire and soils: Key concepts and recent advances. Geoderma 2012; 191: 3–13.

Blank RR, Chambers JC, Zamudio D. Restoring riparian corridors with fire: Effects on soil and vegetation. J. Range Manag. 2003; 56: 388–396.

Blank RR, Zamudio DC. The influence of wildfire on aqueous extractable soil solutes in forested and wet meadow ecosystems along Eastern front of the Sierra-Nevada range, California. Int.J. Wildland Fire 1998; 8: 79–85.

Bodí M, Martin D, Santin C, Balfour V, Doerr SH, Pereira P, Cerdà A, Mataix-Solera J. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 2014; 130: 103–127.

Bodí MB, Doerr SH, Cerdà A, Mataix-Solera J. Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil. Geoderma 2012; 191: 14–23.

Bond WE, Keeley JE. Fire as a global herbivore: the ecology and evolution of flammable ecosystems. Trends Ecol. Evol. 2005; 20: 387–394.

Boot CM, Haddix M, Paustian K, Cotrufo MF. Distribution of black carbon in ponderosa pine forest floor and soils following High Park wildfire. Biogeosciences 2015; 12: 3029–3039.

Brown LE, Holden J, Palmer SM, Johnston K, Ramchunder SJ, Grayson R. Effects of fire on the hydrology, biogeochemistry, and ecology of peatland river systems.Freshwater Science 2015:http://www.jstor.org/stable/10.1086/683426

Campo J, Andreu V, Gimeno-Garcia E, Gonzalez-O., Rubio JL. Occurrence of soil erosion after repeated experimental fires in a Mediterranean environment. Geomorphology 2006; 82: 376–387.

Caon L, Vallejo VR, Ritsema CJ, Geissen V. Effects of wildfire on soil nutrients in Mediterranean ecosystems. Earth Sci. Rev. 2014; 139: 47–58.

Castro J, Allen DC, Molina-Morales M, Maranon-Gimenez S, Sanchez-Miranda A, Zamora R. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. 2011. Restor. Ecol., 19; 537–544.

Cerdà A, Doerr SH. The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. Catena 2008; 74: 256–263.

Cerdà A, Doerr SH. Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. Int. J. Wildland fire 2005; 14: 423–427.

Cerdà A, Imeson AC, Calvo A. Fire and aspect induced changes on the erodibility and hydrology of soils at La Costera, southeast Spain. Catena 1995; 24: 289–304.

Cerdà A, Lasanta T. Long-term erosional responses after fire in the Central Spanish Pyrenees: 1. Water and sediment yield. Catena 2005; 60: 59–80.

Cerdà A. Changes in overland flow and infiltration after a rangeland fire in a Mediterranean scrubland. Hydrol. Process. 1998; 12: 1031–1042.

Cerdà A. Incendios forestales y estabilidad de los agregados. Cuadernos de Geografía 1993; 53: 1–16.

Certini G, Nocentini C, Knicker H, Arfaioli P, Rumpel C. Wildfire effects on soil organic matter quantity and quality in two fire-phrone Mediterranean pine forests. Geoderma 2013; 167-168: 148–155.

Clark KL, Skowronski N, Renninger H, Scheller RS.Climate change and fire management in the middle Atlantic. Forest Ecol. Manag. 2014; 327: 306–315.

Dale VH, Joyce L, McNulty S, Neilson RP, Ayres MP, Flannigan MD, Hanson PJ, Ireland LC, Lugo A, Peterson CJ, Simberloff D, Swanson FJ, Stocks BJ, Wotton BM. Climate change and forest disturbances. BioScience 2011; 51: 723–734.

De Luis M, Gonzalez-Hidalgo JC, Raventos J. Effects of fire and torrential rainfall on erosion in a Mediterranean gorse community. Land Degrad. Develop. 2003; 14: 203–213.

De Marco A, Gentile AE, Arena C, De Santo AV. 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. 2005; 365-377.

Dlapa P, Bodí MB, Mataix-Solera J, Cerdà A, Doerr SH. FT-IR sprectoscopy reveals that ash water repellency is highly dependent on ash chemical composition. Catena 2013; 108: 35–43.

Duguy B, Alloza JA, Baeza MJ, De La Riva J, Echeverria M, Ibarra P, Llovet J, Cabello FB, Rovira P, Vallejo RV. Modelling the ecological vulnerability to forest fires in Mediterranean ecosystems using geographic information technologies. Environ. Manage. 2012; 50: 1012–1026.

Estrany J, Lopez-Tarazon JA, Smith H. Wildfire effects on suspended sediment delivery quantified using fallout radionuclide tracers in a Mediterranean catchment. Land Degrad. Develop. 2015 doi: 10.1002/ldr.2462

Fattet M, Ghestem YFM, Ma W, Foulonneau M, Nespoulous J, Bissonnais YLe, Stokes A. Effects of vegetation type on soil resistance to erosion: Relationship between aggregate stability and shear strength. Catena 2011; 87: 60–69.

Ferreira AJD, Coelho COA, Boulet AK, Lopes FP. Temporal patterns of solute loss following wildfires in Central Portugal. Int. J. Wildland Fire 2005; 14: 401–412.

Ferreira AJD, Prats Alegre S, Alves Coelho CO, Shakesby RA, Páscoa FM, Ferreira CSS, Keizer JJ, Ritsema C. Strategies to prevent forest fires and techniques to reverse degradation processes in burned areas. Catena 2015; 128: 224–237.

Fisher RF, Binkley D. Ecology and management of forest soils. 3rd ed. New York: Wiley; 2000.

Gabet EJ. Fire increases dust production from chaparral soils. Geomorphology 2014; 217: 182–192.

Galang MA, Markewitz D, Morris LA. Soil phosphorous transformations under forest burning and laboratory heat treatments. Geoderma 2010; 155: 401–408.

Gimeno-García E, Andreu V, Rubio JL. Changes in organic matter, nitrogen, phosphorous and cations in soil as a result of fire and fire and water erosion in a Mediterranean landscape. Eur. J. Soil Sci. 2008; 51: 201–210.

Goforth BR, Graham RC, Hubbert KR, Zanner CW, Minnich RA. Spatial distribution and properties of ash of thermally altered soils after high-severity forest fire, southern California. Int. J. Wildland Fire. 2005; 14: 343–354.

Gomez-Rey MX, Couto-Vasquez A, Garcia-Marco S, Gonzalez-Prieto SJ. Impact of fire and post-fire management on soil chemical properties. Geoderma 2013; 195-196: 155–164.

Gonzalez-Perez JA, Gonzalez-Vila FJ, Almendros G, Knicker H.The effect of fire on soil organic matter– a review. Environ. Int. 2004; 30: 855–870.

Grogan P, Burns TD, Chapin III, FS. Fire effects on ecosystem nitrogen cycling in a Californian bishop pine forest. Oecologia 2000; 122: 537–544.

Heiri O, Lotter AF, Lemcke G. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J. Paleolimnol. 2001; 5: 101–110

Hernandez T, Garcia C, Reinhardt I. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils 1997; 25: 109–116.

Igwe CA. Erodibility in relation to water-dispersable clay for some soils of Eastern Nigeria. Land Degrad. Develop. 2005; 16: 87–96.

Inbar A, Lado M, Sternberg M, Tenau H, Ben-Hur M. Forest fire effects on soil chemical and physicochemical properties, infiltration, runoff and erosion in a semiarid Mediterranean region. Geoderma 2014; 221-222: 131–138.

Inbar M, Tamir M, Wittenberg L. Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 1998; 24: 17–33.

Johnson DW, Susfalk RB, Caldwell TG, Murphy JD, Miller WW, Walker RF. Fire effects on carbon and nitrogen budgets in forests. Water Air Soil Pollut. 2004; 4: 263–275.

Jordán A, Zavala LM, Granjed AJP, Gordillo-Rivero AJ, García-Moreno J, Pereira P, Bárcenas-Moreno G, Celis R, Jimenez-Compan E, Alanis N. Wettability of ash conditions splash erosion and runoff rates in the post-fire. Sci. Total Environ. 2015:http://dx.doi.org/10.1016/j.scitotenv.2015.09.140

Kavdir Y, Ekinci H, Yüksel O, Mermut AR. Soil aggregate stability and ^{13}C CP/MASNMR assessment of organic matter in soils influenced by forest wildfires in Çanakkale, Turkey. Geoderma 2005; 129: 219–229.

Kitamura H. Leaching characteristics of anions and cations from evergreen leaves supplied to the stream bed and influences on stream water composition in the Southern Kyusyu Mountains. Bull. Minamikyushu Univ. 2009; 39A: 57–56.

Khanna PK, Raison RJ, Falkiner RA., 1994. Chemical properties of ash derived from *Eucalyptus* litter and its effects on forest soils. Forest Ecol. Manag., 1994; 66: 107–125.

Knicker H, Gonzalez-Vila F, Gonzalez-Vasquez R. Biodegradability of organic matter in fire affected soils of Southern Spain. Soil Biol. Biochem. 2013; 56: 31–39.

Knicker H, Gonzalez-Vila F, Polvillo O, Gonzalez JA, Almendros G. Fire-induced transformation of C- and N- forms in different organic soil fractions froma Dystric Cambisol under a Mediterranean pine forest (Pinus pinaster). Soil Biol. Biochem. 2005; 37: 701–718.

Knicker H. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 2007; 85: 91–118.

Knudsen D, Petersen GA, Pratt PF.. Lithium.Sodium and potassium. In: Soil Science Society of America. editor. Methods of soil analysis, Vol.2. ASA-SSSA: Madison, WI. 1986. p. 225-246.

Kutiel P, Inbar M. Fire impacts on soil nutrients and soil erosion in a Mediterranean pine forest plantation. Catena 1993; 20: 129–139.

Kutiel P, Naveh Z. The effect of fire on nutrients in a pine forest soil.Plant and Soil 1987; 104: 269–274.

Kysely J, Begueria S, Beranova R, Gaal L, Lopez-Moreno JI. Different patterns of climate change scenarios for short-term and multi-day precipitation extremes in the Mediterranean. Global Planet. Change 2012; 98-99: 63–72.

Lasanta T, Cerdà A. Long-term erosional responses after fire in Central Spanish Pyrenees 2. Solute release. Catena 2005; 60: 81–100.

León J, Bodí MB, Cerdà A, Badía D. Effects of ash type and thickness on the temporal variation of runoff from calcareous soil from SE Spain. Geoderma 2013; 209-2010: 143–152.

Liodakis S, Tsoukala M, Katsigiannis G. Laboratory study of leaching properties of Mediterranean forest species. Water Air Soil Pollut. 2009; 203: 99–107.

López-Poma R, Orr BJ, Bautista S. Successional stage after land abandonment modulates fire severity and post-fire recovery in a Mediterranean mountain landscape, Int. J. Wildland fire 2014; 23: 1005–1015.

Low AJ. The study of soil structure in the field and in the laboratory. Journal of Soil Science 1954; 5: 57–74.

Mataix-Solera J, Cerdà A, Arcenegui V, Jordan A, Zavala LM. Fire effects on soil aggregation. Earth Sci. Rev. 2011; 109: 44–60.

Mataix-Solera J, Cerdà A. Incendios forestales en España. Ecosistemas terrestres y suelos. In: Cerdà A, Mataix-Solera J, editors. Efectos de los incendios forestales sobre los suelos en España. Cátedra divulgación de la Ciencia. Universitat de València; 2009. p. 13-53.

Mataix-Solera J, Gómez I, Navarro-Pedreno J, Guerrero C, Moral R. Soil organic matter and aggregates affected by wildfire in a *Pinus halepensis* forest in a Mediterranean environment. Int. J. Wildland Fire. 2002; 11: 107–114.

Mataix-Solera J, Molto J, Arcenegui V, García-Orenes F, Chrenková K, Torres P, Jara-Navarro AB, Diaz A, Izquierdo E. Salvage logging effect on soil properties in fire-affected Mediterranean forest: a two years monitoring research. Geophysical Research Abstracts, 17, EGU2015-2460-2, 2015.

Moody JA, Shakesby RA, Robichaud PR, Cannon SH, Martin DA. Current research issues related to post-wildfire runoff and erosion processes. Earth Sci. Rev. 2013; 122: 10–37.

Murphy JD, Johnson DW, Miller WW, Walker RF, Carroll EF, Blank RR. Wildfire effects on soil nutrients and leaching in Tahoe basin watershed. J. Environ. Qual. 2006; 35: 479–489.

Neary DG, Ryan KC, DeBano LF. Wildland fire in ecosystems: effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-vol.4.U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT, 2005.

Novara A, Gristina L, Bodí MB, Cerdà A. The impact of fire in the redistribution of soil organic matter on a Mediterranean hillslope under under maquia vegetation type. Land Degrad. Develop. 2011; 530–536.

Olsen SR, Cole CV, Frank SW, Dean LA. Estimation of Available Phosphorus in soils by Extraction With Sodium Bicarbonate. USDA 1954. Circular No. 939. US Government Printing Office: Washington, DC.

Onda Y, Dietrich WE, Booker F. Evolution of overland flow after a severe forest fire, Point Reyes, California. Catena 2008; 72: 13–20.

Pacheco E. Dinámicas hidrológicas en la cuenca mediterránea litoral del río Daró. Años 2004 - 2010. Trabajo Fin de Máster. 2010. Universitat de Barcelona. 77 pp.

Pereira P, Cerdà A, Úbeda X, Mataix-Solera J, Arcenegui V, Zavala L. Modelling the impacts of wildfire on ash thickness in a short-term period. Land Degrad. Develop. 2015c; 26: 180–192.

Pereira P, Cerdà A, Úbeda X, Mataix-Solera J, Jordan A, Burget M. Spatial models for monitoring the spatio temporal evolution of ashes after fire: a case study in Lithuania. Solid Earth 2013; 4: 153–165.

Pereira P, Jordan A, Martin D, Cerdà A. Editorial: The role of ash in fire-affected ecosystems. Catena 2015b; 135: 337–339.

Pereira P, Oliva M, Misiune I. Spatial interpolation of precipitation of precipitation indexes in Sierra Nevada (Spain): comparing the performance of some interpolation methods. Theo. Appl. Climatol. 2015a, doi: 10.1007/s00704-015-1606-8

Pereira P, Úbeda X, Martin D, Mataix-Solera J, Cerdà A, Burget M. Wildfire effects on extractable elements in ash from a Pinus pinaster forest in Portugal. Hydrol. Process. 2014; 28: 3681–3690.

Pereira P, Úbeda X, Martin D, Mataix-Solera J, Guerrero C. Effects of a low severity prescribed fire in ash water soluble elements from a cork oak (*Quercus suber*) forest located in the northeast part of the Iberian Peninsula. Environ. Res. 2011; 111: 237–247.

Pereira P, Úbeda X, Martin D. Fire severity effects on ash chemical composition and water-extractable elements.Geoderma 2012; 191: 105–114.

Powers EM, Marshall JD, Zhang J, Wei L. Post-fire management regimens affect carbon sequestration and storage in a Sierra Nevada mixed conifer forest. Forest Ecol. Manag. 2013; 291: 268–277.

Prieto-Fernndez A, Carballas M, Carballas T. Inorganic and organic N pools in soils burned or heated: immediate alterations and evolution after forest fires. Geoderma 2004; 121: 291–236.

Raison RJ, McGarity JW. Some effects of plant ash on the chemical properties of soils and aqueous suspensions. Plant Soil 1980; 55: 339–352.

San-Miguel-Ayanz J, Moreno JM, Camia A. Analysis of large fires in European Mediterranean landscapes: Lessons learned and perspectives. Forest Ecol. Manag. 2013; 294: 11–12.

Sarah P. Soil aggregation response to a long- and short-term differences in the rainfall amount under arid and Mediterranean climate conditions. Geomorphology 2005; 70: 1–11.

Sarah P. Soil sodium and potassium adsorption ratio along a Mediterranean-arid transect. J. Arid Environ. 2004; 59: 731–741.

Shakesby R. Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth Sci. Rev. 2011; 105: 71–100.

Smith HG, Sheridan G, Lane PNJ, Nyman P, Haydon S. Wildfire effects on water quality in forest catchments: A review with implications for water supply. J. Hydrol. 2011; 396: 170–192.

Soil Survey Staff (2014) Keys to soil taxonomy, 12th ed. USDA-Natural Resources Conservation Service, Washington, DC

Tessler N, Sapir Y, Wittenberg L, Greenbaum N. Recovery of Mediterranean vegetation after recurrent fires: Insight from the 2010 forest fire on Mount Carmel, Israel. Land Degrad. Develop. 2015: doi: 10.1002/ldr.2419

Thomas AD, Walsh RPD, Shakesby RA. Solutes in overland flow following fire in eucalyptus and pine forests, northern Portugal. Hydrol.Process. 2000; 14: 971–985.

Troeh FR, Thompson LM. Soils and Soil Fertility 6th ed. Blackwell Publishing, 2005.

Úbeda X, Outeiro LR, Sala M. Vegetation regrowth after a differential intensity forest fire in a Mediterranean environment, northeast Spain. Land Degrad. Develop. 2006; 17:429–440.

Úbeda, X., Pereira, P., Outeiro, L., Martin, D. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (*Quercus suber*). Land Degrad. Develop. 2009; 20: 589–608.

Varela ME, Benito E, Keiser JJ. Effects of wildfire and laboratory heating on soil aggregate stability of pine forests in Galicia: The role of Lithology, soil organic matter content and water repellency. Catena 2010; 83: 127–134.

Varennes A. Produtividade dos solos e Ambiente. Escolar Editora, Lisboa, 2003.

Velasco A, Úbeda X. Estabilidad de los agregados de un suelo quemado a diferentes intensidades de fuego dieciocho años después de un incendio forestal. Cuadernos de Investigación Geográfica 2014; 40: 333–352.

Vélez R. Los incendios forestales en la cuenca mediterránea. Introducción. In: Vélez R, editor. La defensa contra incendios forestales. Fundamentos y experiencias. McGraw-Hill. 2000; pp. 3-1.

Versini PA, Velasco M, Cabello A, Sempere-Torres D. Hydrological impact of forest fires and climate change in Mediterranean basin. Nat. Hazards 2013; 66: 609–628.

Wagenbrenner JW, MacDonald LH, Coats RN, Robichaud PR, Brown RE. Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. Forest Ecol. Manag. 2015; 335: 176–193.

White JD, Ryan KC, Key CC, Running SW. Remote sensing of forest fire severity and vegetation recovery. Int. J. Wildland Fire 1996; 6: 125–136,

Woods SW, Balfour V. The effects of ash on runoff and erosion after severe forest wildfire, Montana, USA. Int. J. Wildland Fire 2008; 17: 535–548.

Woods SW, Balfour V. The effects of soil texture and ash thickness on post-fire hydrological response from ash-covered soils. J. Hydrol. 2010; 393: 274–286.

CHAPTER 4

CHAPTER 4. POST-WILDFIRE MANAGEMENT EFFECTS ON SHORT-TERM EVOLUTION OF SOIL PROPERTIES (CATALONIA, SPAIN, SW-EUROPE)

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Post-wildfire management effects on short-term evolution of soil properties (Catalonia, Spain, SW-Europe)

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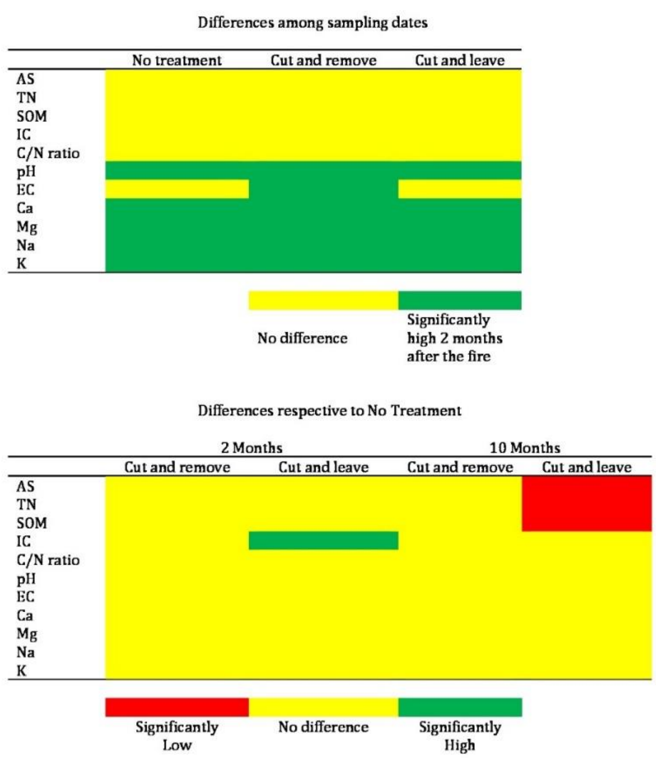
Highlights

Soil C/N ratio, pH, Ca, Mg, Na and K change according to management and sampling date

Manual log and remove operation after a forest fire increase AS, TN and SOM.

Manual log and remove burned wood is a good alternative to mechanical action.

Graphical Abstract



Abstract

Post-fire management practices after wildfires have an important impact on soil properties. Nevertheless, little research has been carried out. The aim of this study is to examine the impact of different post-wildfire forest management practices in a 10-month period immediately after a severe wildfire on soil properties. Two months after a wildfire, three experimental areas were designed, each one with different post-fire management: Cut and Remove (CR) where burned trunks were cutted after fire and removed manually from the area; No Treatment (NT) where no intervention was carried out; and, Cut and Leave (CL) where burned trunks were cut and left randomly on topsoil. In each treatment, we collected nine samples (0-5 cm deep). In total, we sampled 27 samples in each sampling date, two and ten months after the wildfire. The properties analyzed were aggregate stability (AS), total nitrogen (TN), soil organic matter (SOM), inorganic carbon (IC), C/N ratio, pH, electrical conductivity (EC), extractable calcium (Ca), magnesium (Mg), sodium (Na) and potassium (K). Soil C/N ratio was significantly higher in CR and CL treatments 10 months after fire comparing to 2 months after. On the other hand, pH, extractable Ca, Mg and K were significantly higher in all the treatments 2 months after fire than 10 months after. Aggregate stability, TN and SOM were significantly higher in CR comparing to CL, 10 months after the fire. IC was significantly higher in CL than in NT treatment, also, 10 months after the fire. Electrical conductivity was significantly higher in CR and CL treatments 2 months after fire comparing to 10 months after. According to the results, CR and CL post-fire management did not differ importantly from the NT scenario, showing that manual wood management does not have detrimental impacts on soil properties compared to mechanical operations.

Keywords: severe wildfire; post-wildfire treatment; soil aggregate stability; soil nutrients; soil response

1. Introduction

Wildfires are frequent in Mediterranean ecosystems, especially in summer (Bond and Van Wilgen, 1996; Pereira et al., 2005, Tessler et al., 2016). However, in recent decades their number has increased owing to various factors, including the abandonment of traditional activities in rural areas, the lack of management and climate change (Pereira et al., 2015a; Vélez, 2000). Wildfires can have a significant impact on a soil's physical and chemical properties (Mataix-Solera and Guerrero, 2007; Pereira et al., 2012; Úbeda and Outeiro, 2009), the degree of this impact depends on fire severity (Keeley, 2009), rainfall intensity in the immediate period after the fire (Francos et al., 2016a), type of soil and pre- (Francos et al., 2018a) and post-fire management practices (Spanos et al., 2005). Fire severity is defined by the impact that a fire had on ecosystems. This index is especially usefull in wildfires where no information about fire intensity is available. Fire severity assesses the changes induced by the fire in vegetation and soil properties (Keeley, 2009).

Soil aggregate stability (AS) response to wildfire is complex and, in turn, depends on many factors. The literature shows that the impact of wildfire on AS can vary, and depends on temperature, type of burned vegetation, soil type in the post-fire period (Velasco and Úbeda, 2014) and fire severity (Mataix-Solera et al., 2011). Jordán et al. (2011) observed a decrease in stability, while Arcenegui et al. (2008) reported an increase after a wildfire. Total nitrogen (TN) and soil organic matter (SOM) dynamics depend on the type of vegetation affected, land management, topography of the burned area and fire severity (Pereira et al., 2013; Pereira et al., 2015b; Smithwick et al., 2005; Turner et al., 1998). Normally, TN increases in the period immediately after the fire, due to the ash-bed effect in low-to-medium wildfire severities, and decreases in high wildfire severities (Wan et al., 2001). Francos et at. (2018b) observed the same trend in TN in low- and high-severity burned areas.In this work, TN was significantly low to medium term and do not vary to long-term respect to short-term in both areas. A similar response is observed in relation to SOM (Certini et al., 2011; González-Pérez et al., 2004). The incorporation of mineralized material into the soil profile normally decreases the C/N ratio (Baird et al., 1999). Due to the ash-bed effect (Williams et al., 1994)), pH and electrical conductivity (EC) increase in the period immediately after a fire (Pardini et al., 2004). A similar response is observed in extractable calcium (Ca), magnesium (Mg) and potassium (K) (García-Orenes et al., 2017; Kutiel and Shaviv, 1992).

After a wildfire, there are different stabilization treatments to reduce soil degradation. The most common are hillslope treatments (to avoid post-fire erosion and runoff), seeding (to increase the rapid restoration of vegetation and soil cover), mulching (to protect soil from raindrop impact and reduce overland flow), erosion barriers (to trap sediments, reduce overland flow and increase water infiltration), road and railway treatments (to prevent road and railway failures and increase downstream sedimentation), and channel treatments (to reduce the sediment transport from low-order channels to perennial water lines) (Robichaud, 2009). Vegetation treatments are also common in wildfire affected areas such as cutting and extracting either manually or mechanically the burned wood (Francos et al., 2016a; García-Orenes et al., 2017) combined with revegetation under different types of management programs (Calvo et al., 2008; Leverkus et al., 2014; Marañón-Jiménez et al., 2013). The objectives of post-fire management are to reduce soil and water losses, increasing the capacity of ecosystem to recover, restore the ecological functions, and manage the existing residual fuels to mitigate wildfire risk. The intervention in fire-affected areas is a matter of discussion, since some areas can recover naturally and partial cuts may be more effective than salvage logging, soil tilling or tree removal (Muñoz-Rojas et al., 2017).

Despite the importance of post-wildfire management on soil properties, few works have examined their impact on soil properties. The majority of works were focused in overland flow and soil erosion (e.g.

Shakesby et al., 1996; Thomas et al., 2000). The studies available showed that mechanical wood extraction have more negative impacts on soil properties than a non-intervention (García-Orenes et al., 2017; Powers et al., 2013). The majority of works were focused on vegetation (García-Jiménez et al., 2017; Spanos et al., 2005). However, soil is clearly a key element for supporting vegetation and, therefore, for promoting landscape recuperation after a wildfire (Eni et al., 2012; Iwara et al., 2011; Pausas, 2004). For these reasons, to understand vegetation recovery in post-wildfire environments, it is crucial to study the impact of different post-fire management approaches on soil properties (Spanos et al., 2005).

In recent years, the government of Catalonia, a region of North-East of Spain, has opted to invest heavily in post-wildfire forest management, since wildfires are a recurrent problem with severe impacts for the environment, society and economy, as stated in previous works (Francos et al., 2016a; Francos et al., 2016b; García-Jiménez et al., 2017; Velasco and Úbeda, 2014). It is, therefore, essential that we have a better understanding of the effectiveness of these management practices and that we are able to identify which practices have more positive impacts on soil. The aim of this study is to examine the short-term effects of different post-wildfire management programs on the physical and chemical properties of soils. The specific objectives are to observe if the post-wildfire management a) has impacts on soil properties 2 and 10 months after wildfire and b) if they differ importantly among the treatments used. We want to compare the impacts of post-wildfire manual operations with a , scenario of no-intervention.

2. Material and Methods

2.1. Study area

The study area is located in Ódena, Barcelona (41° 38' 28.4'' N - 1° 44' 15.6 E 424 m a.s.l.) in North-East Spain (Fig. 1). This study area was profusely detailed by Francos et al. (2018a). The fire started in El Bruc, on the 26th of July, 2015 and burned a total of 1274 ha during three days. The geological substrate of the burned area is composed of sedimentary rocks from Paleocene shale (Panareda-Clopés and Nuet-Badia, 1993). Soils were classified as *Fluventic Haploxerept* (Soil Survey Staff, 2014). Meteorological data is from the meteorological station of Ódena (41º 36' 19'' N – 1º 38' 18'' E 423 m a.s.l.). The mean annual temperature in the study area is 14.2 °C and the average annual rainfall ranges between 500 and 600 mm. In the first 2 months post-wildfire precipitated a total of 88.1 mm and in the 10 post-wildfire a total of 341.4 mm. In the following 10 months after the wildfire (from August to May), the amount of rainfall were: 42.1 (Aug), 46 (Sept), 26.7 (Oct), 53.2 (Nov), 0.8 (Dec), 0.7 (Jan), 39.2 (Feb), 25.6 (Mar), 70.1 (Apr) and 37 (May) mm. The vegetation of the burned area was mainly *Pinus halepensis* Miller, *Pinus nigra* Arnold and *Quercus ilex* L. Understory vegetation was composed of *Pistacea lentiscus* L. and *Genista scopius* L. This area was affected by wildfires in 1986 and in 2015. In this work we studied the impacts of the wildfire that occurred in 2015.

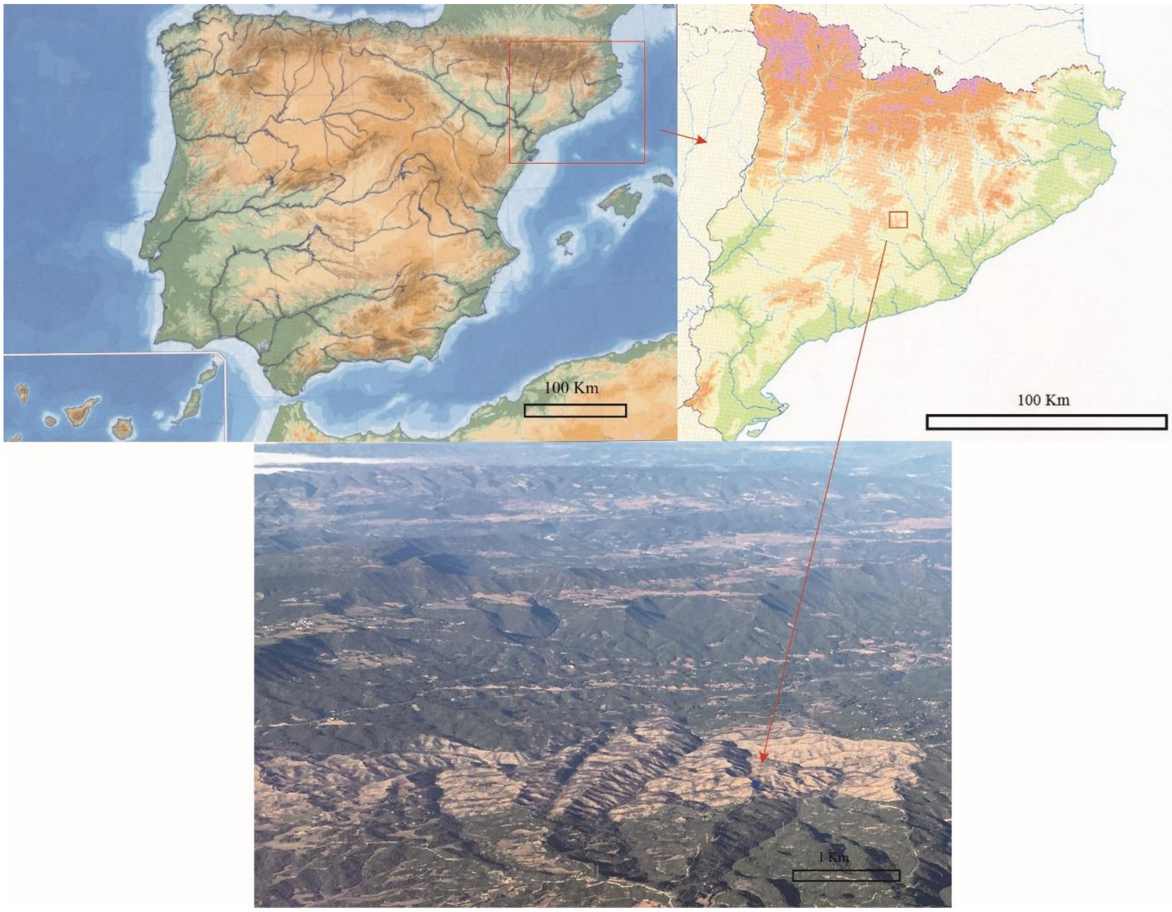


Figure 1. Location of study area

2.2. Experimental design and field sampling

Two months after the wildfire in September 2015, just after the Catalan government initiated its post-wildfire management program and we had obtained permission to conduct our work, an experimental plot was designed and sampled on a homogeneous slope with an incline of approximately 9% with a northern aspect The size of each area is 10x5 m (50m^2) and samples were taken perpendicular to the slope. The selected areas have similar environmental characteristics (i.e. lithology, topography, vegetation and soil type). The same experimental design was carried out in previous studies by García-Orenes et al. (2017) and Francos et al. (2018a). The three post-fire management treatments were as follows: 1) cut and remove (CR), where the trunks were cut and removed manually; 2) no treatment (NT), where no intervention was carried out; and 3), cut and leave (CL), where the trunks were cut and left randomly on the topsoil (Fig. 2). In each area, we sampled nine soil samples (0-5 cm deep) using a steel cylinder. A total of 27 per sampling date.

The second sampling was carried out ten months after the fire (May 2016). Prior to soils sampling in the first sampling date, ash was removed. In the second sampling campaign, no ash remained on the soil surface. The fire was classified as severe because 100% of the tree crowns were consumed (Úbeda et al., 2006); fire effects on soils were considered equal and homogenous within treatment areas. At the second sampling there was no visual evidence of overland flow or sheet or rill erosion that would have moved soil fines on the hillslope.

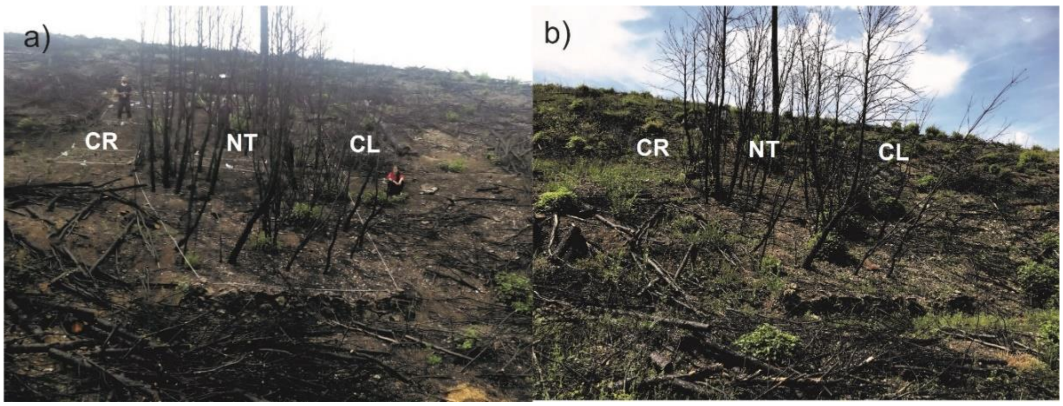


Figure 2. Post-fire forest with three post-fire treatments: CR (Cut and Remove), NT (No Treatment) and CL (Cut and Leave). a) First sampling; b) Second sampling.

2.3. Laboratory methods

Soil samples were dried to constant weight at room temperature (approx. 23 °C) for seven days and sieved using a 2-mm mesh. To analyze the AS of each sample, ten air-dried aggregates (between 4-4.8 mm) were collected. The ten-drop impact (TDI) method (Low, 1954) was used to study AS. This involved placing a 2.8-mm sieve over the aggregates and subjecting them to the impact of ten drops of deionized water, from a droplet placed at a height of 1 m. Each drop had a weight of 0.1 ± 0.001 g and a diameter of 2.8 mm. Aggregates were weighed before the test and the disaggregated material, which passed through the sieve during the test, was also weighed after drying at 105 ºC for 24 h to evaporate the water. The results of the differences are expressed as a percentage of AS, according to the method used by Úbeda et al. (1990).

Total Nitrogen content was analyzed with a Flash EA 112 Series (Thermo-Fisher Scientific, Milan). Data acquisition and calculations were carried out using Eafer 300 software (Thermo-Fisher Scientific, Milan) (Turner et al., 1998). Soil organic matter and inorganic carbon (IC) were measured using the loss-on-ignition (LOI) method described in Heiri et al. (2001). Soil organic matter was estimated after the dried samples had been exposed to a temperature of 550 ºC for four hours. Finally, to calculate soil IC, the samples were subjected to a temperature of 950 ºC for two hours. The results are expressed as percentages.

Soil pH [1:2.5] and Electrical Conductivity (EC) [1:2.5] were analyzed following extraction with deionized water. The amount of soil cations (calcium and magnesium) (Úbeda et al., 2009) and (sodium and potassium) (Knudsen et al., 1986) were extracted from samples using ammonium acetate [1:20]. The extractable cation content was analyzed by inductively coupled plasma mass spectrometry (ICP-MS), using a PerkinElmer Elan-6000 Spectrometer (Norwalk, CT, USA).

2.4. Statistical analysis

Statistical comparisons between sampling dates (time) and treatments were carried out by conducting a two-way ANOVA test. The main effects are time and treatment. Where necessary, data were log or square-root transformed to meet homogeneity assumptions. Normal distribution of the residues was assessed using the shapiro-wilk test and the equality of the variances with the Leven's test. Significant differences were considered at a P value of <0.05. If significant differences were identified, a Tukey HSD post-hoc test was carried out. Relations between the variables were studied with a redundancy analysis (RDA). Statistical analyses were carried out using SPSS 23.0 and CANOCO software.

3. Results

3.1. Aggregate stability, total nitrogen, soil organic matter, inorganic carbon and C/N ratio

The results showed significant differences between treatments in soil AS, TN, SOM and the IC ratio (Table 1). The C/N ratio presented significant differences in relation to time (i.e. date of sampling). No differences were identified between time x treatment interactions. The CR treatment presented a significantly higher AS, TN, SOM than those associated with the CL treatment. In this latter treatment, the IC content was significantly higher than under the NT treatment. With the exception of the IC ratio, the variables where we identified significant differences between treatments (AS, TN and SOM) were observed at the second sampling date. In the CR and CL treatments, the C/N ratio was significantly higher at the second sampling date (Table 2).

Table 1. Two-way ANOVA results. Significant differences at p<0.05*. p<0.01** and p<0.001***. n.s. not significant at a p<0.05.

	Treatment	F value	p value
Aggregate Stability (%)	Time	0.26	n.s
	Treatment	3.21	*

	Time x Treatment	0.66	n.s
Total Nitrogen (%)	Time	1.64	n.s
	Treatment	4.47	*
	Time x Treatment	0.01	n.s
Soil Organic Matter (%)	Time	0.58	n.s
	Treatment	4.77	*
	Time x Treatment	0.04	n.s
Inorganic Carbon (%)	Time	0.02	n.s
	Treatment	4.18	*
	Time x Treatment	0.30	n.s
C/N ratio	Time	20.51	***
	Treatment	0.23	n.s
	Time x Treatment	1.00	n.s
pH	Time	104.43	***
	Treatment	1.96	n.s
	Time x Treatment	0.05	n.s
Electrical conductivity (µS/cm)	Time	7.14	**
	Treatment	4.62	*
	Time x Treatment	1.01	n.s
Extractable Calcium (ppm)	Time	363.66	***
	Treatment	0.69	n.s
	Time x Treatment	1.54	n.s
Extractable Magnesium (ppm)	Time	25.40	***

	Treatment	0.19	n.s
	Time x Treatment	0.09	n.s
Extractable Sodium (ppm)	Time	40.56	***
	Treatment	1.50	n.s
	Time x Treatment	2.67	n.s
Extractable Potassium (ppm)	Time	35.72	***
	Treatment	3.73	n.s
	Time x Treatment	0.22	n.s

Table 2. Descriptive statistics for Aggregate Stability. Total Nitrogen. Soil Organic Matter. Inorganic Matter and C/N Ratio. Different letters represent significant differences at a $p < 0.05$ between sampling dates (capital letters) and treatment (low case letters). First sampling (1) and Second Sampling (2) in three different managements: CR (Cut and Remove). NT (No Treatment) and CL (Cut and Leave) (N=9).

	reatment	Sampling date	Mean	SD	Min	Max
Aggregate Stability (%)	CR	1	96.51	2.16	92.75	98.66
		2	96.96 a	1.90	92.72	98.59
		All	96.74	1.98	92.72	98.66
	NT	1	96.24	1.64	92.86	98.21
		2	94.83 ab	2.39	90.60	98.43
		All	95.54	2.12	90.60	98.43
	CL	1	94.67	4.15	84.95	98.24
		2	94.59 b	1.89	91.04	97.10
		All	95.63	3.12	84.95	98.24
Total Nitrogen (%)	CR	1	0.41	0.13	0.28	0.66

		2	0.38 a	0.14	0.18	0.67
		All	0.39	0.13	0.18	0.67
		1	0.33	0.10	0.22	0.51
	NT	2	0.30 ab	0.08	0.22	0.40
		All	0.31	0.08	0.22	0.51
		1	0.31	0.11	0.19	0.57
	CL	2	0.27 b	0.10	0.16	0.51
		All	0.29	0.10	0.16	0.57
		1	7.11	2.47	4.44	11.27
	CR	2	7.72 a	2.66	4.86	13.37
		All	7.41	2.51	4.44	13.37
		1	5.64	1.54	3.76	8.64
Soil Organic Matter (%)	NT	2	5.86 ab	1.20	4.07	7.84
		All	5.75	1.34	3.76	8.64
		1	5.34	2.02	3.63	10.26
	CL	2	5.75 b	1.63	3.62	8.89
		All	5.54	1.79	3.62	10.26
		1	2.35 ab	0.49	1.57	2.91
	CR	2	2.31	0.51	1.88	3.45
		All	2.33	0.48	1.57	3.45
Inorganic Carbon (%)		1	2.15 b	0.36	1.49	2.77
	NT	2	2.26	0.35	1.68	2.87
		All	2.20	0.35	1.49	2.87
	CL	1	2.67 a	0.46	2.20	3.35

		2	2.55	0.42	2.02	3.14
		All	2.61	0.44	2.02	3.35
C/N ratio	CR	1	20 B	9	11	43
		2	27 A	5	23	38
		All	23	8	11	43
	NT	1	22	9	11	41
		2	27	1	24	28
		All	24	7	11	41
	CL	1	20 B	2	16	22
		2	30 A	4	23	37
		All	25	6	16	37

3.2. pH, electrical conductivity and extractable cations

Soil pH, extractable Ca, Mg, Na, and K showed significant differences between sampling dates, while EC also showed significant differences between treatments at 2 months post-fire. No significant differences were identified in the interaction between time x treatment (Table 1). The results show that soil pH and extractable elements were significantly higher in all treatments at the first sampling date than they were at the second. Significant differences in EC between sampling dates were only identified in the CR and CL treatments. The EC values were significantly higher in the CR treatment than they were in that of the CL at the first sampling date after the fire (Table 3).

Table 3. Descriptive statistics for pH, Electrical Conductivity, Extractable Calcium, Extractable Magnesium, Extractable Sodium and Extractable Potassium. Different letters represent significant differences at a $p < 0.05$ between sampling dates (capital letters) and treatment (low case letters). First sampling (1) and Second sampling (2) in three different managements: CR (Cut and Remove). NT (No Treatment) and CL (Cut and Leave) (N=9).

	Treatment	Sampling date	Mean	SD	Min	Max
pH	CR	1	8.41 A	0.24	7.97	8.67
		2	7.89 B	0.33	7.31	8.36
		All	8.15	0.39	7.31	8.67
	NT	1	8.53 A	0.12	8.36	8.76
		2	8.01 B	0.12	7.81	8.19
		All	8.27	0.30	7.81	8.76
	CL	1	8.51 A	0.16	8.18	8.66
		2	7.95 B	0.06	7.87	8.07
		All	8.23	0.31	7.87	8.66
Electrical Conductivity (µS/cm)	CR	1	395 Aa	142	222	733
		2	273 B	127	142	543
		All	334	145	142	733
	NT	1	277 ab	69	156	368
		2	243	93	124	404
		All	260	81	124	404
	CL	1	270 Ab	59	200	357
		2	216 B	48	136	279
		All	243	59	136	357
Extractable Calcium (ppm)	CR	1	35663 A	4866	27982	43213
		2	17879 B	2462	14578	22552
		All	26771	9885	14578	43213
	NT	1	34239 A	5325	26630	44556
		2	16440 B	2005	13698	20676

		All	25340	9955	13698	44556
	CL	1	36721 A	4156	30495	41971
		2	15212 B	1126	13346	17196
		All	25967	11453	13346	41971
Extractable Magnesium (ppm)	CR	1	1650 A	309	1252	2029
		2	1188 B	330	674	1550
		All	1419	391	674	2029
	NT	1	1699 A	345	1341	2233
		2	1181 B	423	748	1900
		All	1440	460	748	2233
	CL	1	1781 A	474	1255	2511
		2	1209 B	358	854	1776
		All	1495	502	854	2511
Extractable Sodium (ppm)	CR	1	386 A	184	135	635
		2	185 B	87	108	406
		All	286	174	108	635
	NT	1	586 A	287	258	970
		2	222 B	141	80	496
		All	404	288	80	970
	CL	1	628 A	343	208	1051
		2	105 B	40	70	198
		All	367	359	70	1051
xtractable Potassium (ppm	CR	1	642 A	135	486	863
		2	476 B	105	364	684

		All	559	145	364	863
		1	608 A	163	456	1002
	NT	2	369 B	79	320	580
		All	502	166	320	1002
		1	543 A	88	391	656
	CL	2	369 B	81	280	516
		All	456	121	280	656

3.3. Multivariable analysis

The results of the Redundancy Analysis (RDA) for the first and second sampling dates are shown in Figure 3. The first factor of the RDA carried out for the first sampling date explained 33.1% of the variance, while the second factor explained 30.9%. The variables with a high explanatory capacity were EC and IC, while those with a low capacity were the C/N ratio and extractable Mg. Two main groups of variables were identified: 1) AS, TN, SOM and EC; and, 2) pH and extractable Na. The group 1 variables presented high values under CR treatment, while those in group 2 presented low values. CL treatment was associated with high extractable Ca, Mg and IC values. Finally, NT treatment presented high C/N ratio, pH and extractable Na values (Figure 3a). The first factor of the RDA carried out for the second sampling date explained 38.2% of the variance, while the second factor explained 22.5%. Here, again, we identified two groups: 1) AS, TN, SOM, EC, extractable Ca and K; and, 2) IC and C/N ratio. The group 1 variables presented high values under CR treatment and IC and the C/N ratio presented high values under CL treatment. As at the previous sampling date, NT treatment had high values in pH and Na (Figure 3b).

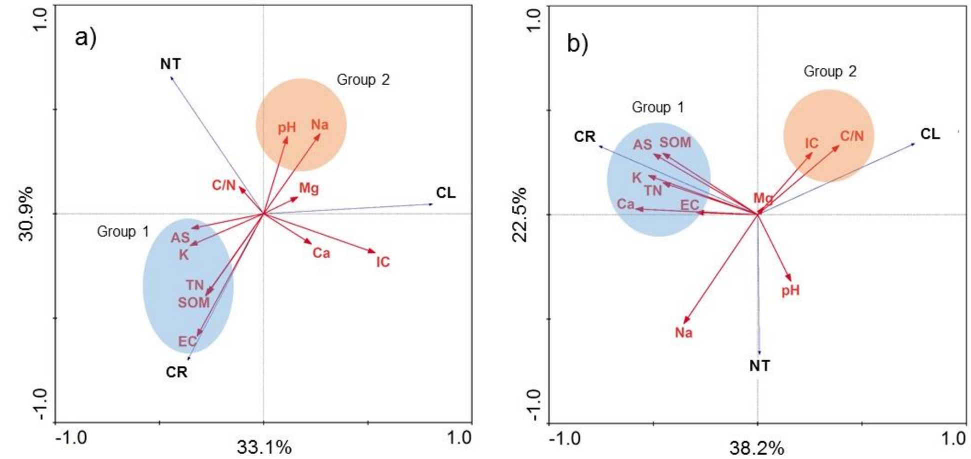


Figure 3. RDA analysis results. a) First sampling; b) Second sampling. Aggregate Stability (AS), Total Nitrogen (TN), Soil Organic Matter (SOM), Inorganic Carbon (IC); C/N ratio, pH, Electrical Conductivity (EC); Extractable Calcium (Ca), Extractable Magnesium (Mg), Extractable Sodium (Na), Extractable Potassium (K), Cut and Remove (CR), No Treatment (NT) and Cut and Leave (CL).

4. Discussion

4.1. Aggregate stability, total nitrogen, soil organic matter, inorganic carbon and C/N ratio

Soil AS, TN and SOM presented significantly lower values under CL treatment than under CR treatment in the second sampling period. These results suggest that the treatment did not have any immediate effects and that they were only observed 10 months after the fire. The removal of burned wood and leaving sawdust residues increased the amount of soil nutrients in CR treatment comparing to CL. Foot traffic during wood removal operations may have contributed also to the incorporation of sawdust residues into soil profile. Soil AS, TN and SOM in CL and CR were not significantly different form NT, showing that the impact of CL and CT management on these soil properties, were not different from a situation of no intervention. . Manual wood removal may not have significant impacts on soil AS, TN and SOM, however, mechanical removal has detrimental on these soil properties, as observed by García-Orenes et al.(2017). Manual wood removal seems to have lower impacts on soil properties comparing to heavy machinery, as previously observed (Wagenbrenner et al., 2016). However, other studies also report that salvage logging either had no effect on (Fernández and Vega, 2016), or it increased (Poirer et al., 2014), the soil organic carbon and TN stocks in the mineral horizon. These outcomes were attributed to the mixing of the forest soil and the organic material with the mineral soil.

Soil IC was significantly higher under CL treatment than under NT treatment at the first sampling date after the fire. Inorganic carbon production is mainly attributable to the total combustion of litter and SOM that occurs in high severity wildfires, such as the one studied here. The ash produced at high temperatures is rich in carbonates and is usually characterized by small, light particles that are easily eroded or rapidly incorporated into the soil profile (Pereira et al., 2012; Pereira et al., 2015b). Since the first sampling was carried out two months after the fire, it is likely that part of the ash had already been incorporated into the first few centimeters of the soil. With the time, there was a non-significant decrease of IC in CL and CR soils, occurring the opposite in NT. This may have been attributed to the disturbance induced by management that contributed to sawdust deposition into soil profile, being leached in the following months after the treatment. In NT, the absence of disturbance may have allow a low rate of deposition and incorporation of ash residues into soil profile and this may explain the increase of IC 10 months after the fire(Pereira et al., 2015a; Wan et al., 2001).

The soil C/N ratio was significantly higher under the CR and CL treatments at the second sampling date, showing that the degree of change was higher in the managed plot comparing to NT. . It is widely known that the C/N ratio decreases after fire as a consequence of the incorporation of ash into the soil profile (Certini et al., 2011; Knicker, 2007), but that it increases in the months following the fire due to ash leaching and erosion (Pereira et al., 2014) and the incorporation of fresh litter from the recovered vegetation into the soil profile (Knicker et al., 2013). This explains why the C/N ratio was significantly higher 10 months after the fire in CL and CR plots. The lower differences of C/N between sampling dates in NT compared to CL and CR may have been a consequence of the reduced rate of mineralized sawdust incorporation into the soil profile. The management practices did not have a positive impact on soil C/N, since the rate of change was higher than in NT. High C/N ratio reduce microbial activity and SOM decomposition, decreasing the availability of nitrogen to plant recuperation (Weintraub et al., 2015; Jiang et al., 2016).

4.2. pH, electrical conductivity and extractable cations

Soil pH, EC and the extractable cations decreased significantly between the first and second sampling dates. A reduction in pH, EC and extractable cations with time has been widely observed in fire-affected areas, which is attributed to ash leaching, erosion and vegetation consumption (Bárcenas-Moreno et al., 2016; Francos et al., 2016a; Granged et al., 2011; Xue et al., 2014). Electrical conductivity values were significantly higher under CR treatment than they were under CL treatment 2 months after the wildfire. This is in line with the results observed for SOM, which was high under CR treatment at the first sampling date after the fire. Although we did not observe significant differences, the high SOM content under CR treatment may have contributed to the high concentration of solutes. Ash incorporated into the soil profile increases both SOM and EC (Bodí et al., 2014; Pereira et al., 2012). Fire decrease SOM but the incorporation of burned vegetation produce an increase of SOM and EC in first centimeters of soil (Badía et al., 2017).

4.3. Overall discussion and implications for landscape management

The RDA identified two groups 2 months after the wildfire: one comprising pH and Na and the other comprising AS, K, TN, SOM and EC (Fig 3a). The elements of the second group had a higher concentration in NT and CL, while the variables grouped in the group 1 had a higher concentration in CR.

At the second sampling date (Fig 3b), once again two groups were identified: one comprising IC and the C/N ratio (with high values in CL) and the other comprising AS, K, Ca, TN, SOM and EC (with high values in CR) Cut and remove treatment was the type of management where were found the highest concentrations of key soil nutrients, 2 (AS, SOM, TN and K) and 10 (AS, SOM, TN, Ca and K) months after the wildfire. It has been widely observed that post-fire management may have negative impacts on soil properties (Baldini et al., 2004; García-Orenes et al., 2017; Gómez-Rey et al., 2013; Powers et al., 2013). In our case, CR or CL treatments did not differ importantly from NT, which is contrary to previous works (Wagenbrenner et al., 2016), and this was attributed to the fact that post-fire management was carried out manually, not mechanically..

Overall, the management that we applied in these areas, cannot be used in a wider scale. At the small scale studied, the managements carried out did not differ from a situation of no intervention. The majority of the post-wildfire management activities are carried out using machinery and affect a larger area, and this may have important negative impacts on soil and vegetation recuperation. Future studies will consider the impact of mechanical vegetation removal after the fire. Other limitations were 1) the fact that we could only sampling two months after the fire due logistic questions, and we could not study the immediate impacts post-fire and 2) only sampled 2 and 10 months after the fire and better resolution studies are important to identify the impacts of post-fire management activities. Further research will consider these two aspects. Despite this, our work is novel and represents an advance to the current knowledge. Our results demonstrated that manual wood removal may be a good management in burned areas in the first 10 months after the fire, especially in specific areas where the fire was more severe and the soil is vulnerable to mechanical treatments.

5. Conclusions

The wildfire had an immediate impact on soil pH and extractable Ca, Mg, Na and K, irrespective to the treatment carried out, decreasing their levels 10 months after the fire. The inverse was observed in C/N ratio, but only in CR and CL treatments. On the other hand, the effects of the post-wildfire management, in soil AS, TN, SOM, and IC were only observed 10 months after the fire. AS, TN and SOM were significantly higher in CR than in CL, while the IC values were significantly higher in CL than in NT. This could be because the saw dust from management activities and the more intense foot traffic that may have been deposited in the topsoil. Soil EC was significantly higher in CR compared to CL 2 months post-fire. The concentration of salts in soil was significantly higher in CR and CL treatments 2 months after the wildfire compared to 10 months after. Overall, the differences between the managements carried out and no intervention scenario were not relevant, showing that manual wood management in soil properties do not have negative impacts on soil properties, compared to salvage logging.

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References

Arcenegui, V., Mataix-Solera, J., Guerrero, C., Zornoza, R., Mataix-Beneyto, J., García-Orenes, F., 2008. Immediate effects of wildfire on water repellency and aggregate stability in Mediterranean calcareous soils. Catena 74, 219-226.

Badía, D., López-García, S., Martí, C., Ortíz-Perpiñá, O., Girona-García, A., Casanova-Gascón, J., 2017. Burn effects on soil properties associated to heat transfer under contrasting moisture content. Sci. Total Environ. 601-602, 1119-1128.

Baird, M., Zabowski, D., Everett, R.L., 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant Soil 209, 233–243.

Baldini, S., Certini, G., Di Fluvio, F., Giovannini, G., Marchi, E., 2007. Environmental impact of postfire logging in a Maritime pine stands in Tuscany (Central Italy). In Proceedings of the "4 International Wildland fire Conference" 13-17 May, Seville, Spain.

Barcenas-Moreno, G., Garcia-Orenes, F., Mataix-Solera, J., Mataix-Beneyto, J., 2016. Plant community influence on soil microbial response after a wildfire in Sierra Nevada National Park (Spain). Sci. Total Environ. 573, 1265–1274.

Bodí, M., Martin, D., Santin, C., Balfour, V., Doerr, S.H., Pereira, P., Cerdà, A., Mataix-Solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 130, 103–127.

Bond, W.J., Van Wilgen, B.W., 1996. Fire and Plants (population and community biology serie 14). Population and Community Biology 263.

Calvo, L., Santalla, S., Valbuena, L., Marcos, E., Tárrega, R., Luis-Calabuig, E., 2008. Post-fire natural regeneration of a Pinus pinaster forest in NW Spain. Plant Ecol. 197, 81-90.

Certini, G., Nocentini, C., Knicker, H., Arfaioli, P., Rumpel, C., 2011. Wildfire effects on soil organic matter quantity and quality in two fire-prone Mediterranean pine forest. Catena 167-168, 148-155.

Eni, D.D., Iwara, A.I., Offiong, R.A., 2012. Analysis of Soil-Vegetation Interrelationships in a South-Southern Secondary Forest of Nigeria. Int. J. Forestry Res. doi:10.1155/2012/469326.

Fernandez, C., Vega, J.A., 2016. Effects of mulching and post-fire salvage logging on soil erosion and vegetation regrowth. Forest Ecol. Manag. 375, 46–54.

Francos, M., Pereira, P., Alcañiz, M., Mataix-Solera, J., Úbeda, X., 2016a. Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain). Sci. Total Environ. 572, 1353-1362.

Francos, M., Pereira, P., Mataix-Solera, J., Arcenegui, V., Alcañiz, M., Úbeda, X., 2018b. How clear-cutting affects fire severity and soil properties in a Mediterranean ecosystem. J. Environ. Manage. 206, 625-632.

Francos, M., Úbeda, X., Pereira, P., Alcañiz, M., 2018a. Long-term impact of wildfire on soils exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula). Sci. Total Environ. 615, 664-671.

Francos, M., Úbeda, X., Tort, J., Panareda, J.M., Cerdà, A., 2016b. The role of forest fire severity on vegetation recovery after 18 years. Implications for forest management of Quercus suber L. in Iberian Peninsula. Global Planet. Change 145, 11-16.

García-Jiménez, R., Palmero-Iniesta, M., Espelta, J.M., 2017. Contrasting Effects of Fire Severity on the Regeneration of Pinus halepensis Mill. And Resprouter Species in Recently Thinned Thickets. Forests 8 (3), 55.

García-Orenes, F., Arcenegui, V., Cherenková, K., Mataix-Solera, J., Moltó, J., Jara-Navarro, A.B., Torres, M.P., 2017. Effects of salvage logging on soil properties and vegetation recovery in a fire-affected Mediterranean forest: A two year monitoring research. Sci. Total Environ. 586, 1057-1065.

Gómez-Rey, M.X., Couto-Vazquez, A., García-Marcos, S., González-Prieto, S.J., 2013. Impact of fire and post-fire management techniques in soil chemical properties. Geoderma 195-196, 155-164.

González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fires on soil organic matter-a review. Environ. Int. 30, 855-870.

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

Heiri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J. Paleolomnol. 5, 101-110.

Iwara, A.I., Ogundele, F.O., Ibor, U.W., Deekor, T.N., 2011. Multivariate Analisys of Soil-Vegetation Interrelationships in a South-Southern Secondary Forest of Nigeria. Int. J. Biol. Sci. 3 (3), 73-82.

Jiang, X., Denef, K., Stewart, C.E., Cotrufo, F., 2016. Controls and dynamics of biochar decomposition and soil microbial abundance, composition, and carbon use efficiency during long-term biochar amended soil incubations. Biology and Fertility of Soils 52, 1-14.

Jordán, A., Zavala, L.M., Mataix-Solera, J., Nava, A.L., Alanís, N., 2011. Effects of fire severity on water repellency and aggregate stability on Mexican volcanic soils. Catena 84, 136-147.

Keeley, J.E., 2009. Fire intensity, fire severity and burnt severity: a brief review and suggests usage. Int. J. Wildland Fire 18 (1), 116-126.

Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 85, 91-118.

Knicker, H., Gonzalez-Vila, F., Gonzalez-Vazquez, R., 2013. Biodegradability of organic matter in fire affected mineral soils of Southern Spain. Soil Biol. Biochem. 56, 31–39.

Knudsen, D., Petersen, G.A., Pratt, P.F., 1986. Lithium, sodium and potassium. In: Soil Science Society of America (ed) Methods of soil analysis. Vol. 2. ASA-SSSA, Madison, WI. pp. 225-246.

Kutiel, P., Shaviv, A., 1992. Effects of soil type, plant composition and leaching on soil nutrients following a simulated forest fire. For. Ecol. Manag. 53, 329-343.

Leverkus, A.B., Lorite, J., Navarro, F.B., Sánchez-Cañete, E.P., Castro, J., 2014. Post-fire salvage logging alters species composition and reduces cover, richness, and diversity in Mediterranean plant communities. J. Environ. Manage. 133, 323-331.

Low, A.J., 1954, The study of soil structure in the field and in the laboratory. J. Soil Sci. 5, 57-74.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordan, A., Zavala, L.M., 2011. Fire effects on soil aggregation. Earth Sci. Rev. 109, 44–60.

Mataix-Solera, J., Gómez, I., Navarro-Pedreño, J., Guerrero, C., Moral, R., 2002. Soil organic matter and aggregates affected by wildfire in a *Pinus halepensis* forest in a Mediterranean environment. Int. J. Wildland Fire 11, 107-114.

Mataix-Solera, J., Guerrero, C., 2007. Efectos de los incendios forestales sobre las propiedades edáficas, in: Mataix-Solera, J. (Ed.), Incendios forestales, suelos y erosión hídrica. Caja Mediterráneo CEMACAM. Alcoi. pp. 5-40.

Marañón-Jiménez, S., Castro, J., 2013. Effect of decomposing post-fire coarse woody debris on soil fertility and nutrient availability in a Mediterranean ecosystem. Biogeochemistry 112, 519–535.

Marañón-Jiménez, S., Castro, J., Querejeta, J.I., Fernández-Ondoño, E., Allen, C.D., 2013. Post-fire wood management alters water stress, growth, and performance of pine regeneration in a Mediterranean ecosystem. Forest Ecol. Manag. 308, 231-239.

Panareda-Clopés, J.M., Nuet-Badia, J., 1993. Tipología y cartografía corológica de las plantas vasculares de Montserrat (Cordillera Prelitoral Catalana). Rev. Geogr. 27, 33-58.

Pardini, G., Gispert, M., Dunjó, G., 2004. Relative influence of wildfire on soil properties and erosion processes in different Mediterranean environments in NE Spain. Sci. Total Environ. 328, 237-246.

Pausas, J.G., 2004. Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean Basin). Clim. Change 63, 337–350.

Pereira, M.G., Trigo, R.M., Camara, C.C., Pereira, J.M.C., Leite, S.M., 2005. Synoptic patterns associated with large summer forest fires in Portugal. Agr. Forest Meteorol. 129, 11–25.

Pereira, P., Cerdà, A., Úbeda, X., Mataix-Solera, J., Arcenegui, V., Zavala, L.M., 2015b. Modelling impacts of wildfire on ash thickness in a short-term period. Land Degrad. Dev. 26, 180-192.

Pereira, P., Cerdà, A., Úbeda, X., Mataix-Solera, J., Martin, D., Jordán, A., Burguet, M., 2013. Spatial model for monitoring the spatio-temporal evolution of ashes after fire: a case study of a burnt grassland in Lithuania. Solid Earth 4, 153-165.

Pereira, P., Jordan, A., Martin, D., Cerdà, A., 2015a. The role of ash in fire-affected ecosystems. Catena 135, 337–339.

Pereira, P., Úbeda, X., Martin, D., 2012. Fire severity effects on ash chemical composition and water-extractable elements. Geoderma 191, 105-114.

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Cerdà, A., Burget, M., 2014. Wildfire effects on extractable elements in ash from a *Pinus pinaster* forest in Portugal. Hydrol. Process. 28, 3681–3690.

Poirer, V., Pare, D., Boiffin, J., Munson, A.D., 2014. Combined influence of fire and salvage logging on carbon and nitrogen storage in boreal soil profiles. Forest Ecol. Manag. 326, 133–141.

Powers, E.M., Marshall, J.D., Zhang, J., Wei, L., 2013. Post-fire management regimes affect carbon sequestration and storage in Sierra Nevada mixed conifer forest. Forest Ecol. Manag. 291, 268–277.

Robichaud, P., 2009. Post-fire stabilization and rehabilitation, in: Cerda, A., Robichaud, P. (Eds.), Fire effects on soils and restoration strategies. Science Publishers, Enfield(NH), pp. 299–320.

Muñoz-Rojas, M., Pereira, P., Brevik, E.C., Cerda, A., Jordan, A., 2017. Soil mapping and process models for sustainable land management applied to modern challenges, in: Pereira, P., Brevik, P., Munoz-Rojas, M., Miller, B.A. (Eds.), Soil mapping and process modeling for sustainable land use management. Elsevier, Amsterdam, The Netherlands, pp. 151–190.

Shakesby, R.A., Boakes DJ, Coelho, COA, Bento Goncalves AJ, Walsh, R.P.D., 1996. Limiting the soil degradational impacts of wildfire in pine and eucalyptus forests in Portugal: A comparison of alternative post-fire management practices. Appl. Geogr. 16, 337–355.

Smithwick, E.A.H., Turner, M.G., Mack, M.C., Stuart Chapin III, F., 2005. Postfire Soil N Cycling in Northern Conifer Forests Affected by Severe, Stand-Replacing Wildfires. Ecosystems 8, 163-181.

Soil Survey Staff, 2014. Keys to soil taxonomy. 12th edn. USDA-Natural Resources Conservation Service, Washington, DC.

Spanos, I., Raftoyannis, Y., Goudelis, G., Xanthopoulou, E., Samara, T., Tsiontsis, A., 2005. Effects of postfire logging on soil and vegetation recovery in a Pinus halepensis Mill. forest of Greece. Plant Soil 278, 171-179.

Tessler, N., Wittenberg, L., Greenbaum, N., 2016. Vegetation cover and species richness after recurrent forest fire in the Eastern Mediterranean ecosystem of Mount Carmel, Israel. Sci. Total Environ. 572, 1395-1402.

Thomas, A.D., Walsh, R.P.D., Shakesby, R.A., 2000. Post-fire forestry management and nutrient losses in eucalyptus and pine plantations, Northern Portugal. Land Degrad. Dev. 11, 257-271.

Turner, M.G., Baker, W.L., Peterson, C., Peet, R.K., 1998. Factors influencing succession: lessons from large, infrequent natural disturbance. Ecosystems 1 (5), 11-23.

Úbeda, X., Outeiro, L.R., 2009. Physical and chemical effects of fire in oil, in: Cerdà, A., Robichaud, P.R. (Eds.), Fire Effects on Soils and Restoration Strategies. Science Publishers, Enfield, NH. pp. 105-133.

Úbeda, X., Pereira, P., Outeiro, L., Martin, D.A., 2009. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (*Quercues suber*). Land Degrad. Dev. 20, 589-608.

Úbeda, X., Outeiro, L.R., Sala, M., 2006. Vegetation regrowth after a differential severity forest fire in a Mediterranean environment, Northeast Spain. Land Degrad. Dev. 17, 429-440.

Úbeda, X., Sala, M., Imerson, A., 1990. Variaciones en la estabilidad y consistencia de un suelo forestal antes y después de ser sometido a un incendio. Actas de la I Reunión Nacional de Geomorfología. Teruel, pp. 677-685.

Velasco, A., Úbeda, X., 2014. Estabilidad de los agregados de un suelo quemado a diferentes intensidades de fuego dieciocho años después de un incendio forestal. Cuadernos de Investigación Geográfica 40, 333–352.

Vélez, R., 2000. Los incendios forestales en la cuenca mediterránea. Introducción, in: Vélez, R. (Ed.), La defensa contra incendios forestales. Fundamentos y experiencias. McGraw-Hill, pp. 3-1.

Wagenbrenner, J.W., Robichaud, P.R., Brown, R.E., 2016. Rill erosion in burned and salvage logged western montane forests: Effects of logging equipment type, traffic level, and slash treatment. J. Hydrol. 541, 889–901.

Wan, S., Hui, D., Luo, Y., 2001. Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: a meta-analysis. Ecol. Appl. 11, 1349-1365.

Weintraub, S.R., Taylor, P.G., Porder, S., Cleveland, C.C., Asner, G.P., Townsend, A.R., 2015. Topographic controls on soil nitrogen availiability in a lowland tropical forest. Ecology 96, 1561-1574.

Williams, J.E, Whelan, R.J., Gill, A.M., 1994. Fire and environmental heterogeneity on southern temperate forest ecosystems: Implications for management. Aust. J. Bot. 42, 125–137.

Xue, L., Li, Q., Chen, H., 2014. Effects of a wildfire on selected physical, chemical and biochemical soil properties in a *Pinus massoniana* Forest in South China. Forests 5, 2947–2966.

CHAPTER 5

CHAPTER 5. LONG-TERM IMPACT OF WILDFIRE ON SOILS EXPOSED TO DIFFERENT FIRE SEVERITIES. A CASE STUDY IN CADIRETES MASSIF (NE IBERIAN PENINSULA)

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Long-term impact of wildfire on soils exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula)

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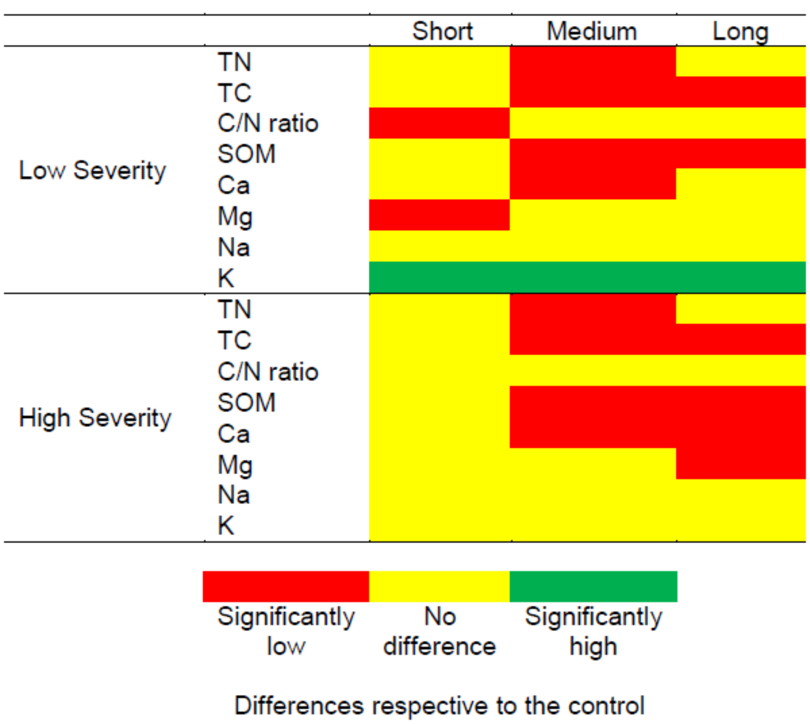
Highlights

Severe wildfires affects soil properties to long-term

Soil nutrients responded differently to time and to fire severity

Soils need more time to recover its properties in high- than in low-severity areas

Graphical Abstract



Abstract

Wildfires affect ecosystems depending on the fire regime. Long-term studies are needed to understand the ecological role played by fire, especially as regards its impact on soils. The aim of this study is to monitor the long-term effects (18 years) of a wildfire on soil properties in two areas affected by low and high fire severity regimes. The properties studied were total nitrogen (TN), total carbon (TC), C/N ratio, soil organic matter (SOM) and extractable calcium (Ca), magnesium (Mg), sodium (Na) and potassium (K). The study was carried out in three phases: short- (immediately after the wildfire), medium- (seven years after the wildfire) and long-term (18 years after the wildfire). The results showed that in both fire regimes TN decreased with time, TC and SOM were significantly lower in the burned plots than they were in the control in the medium- and long-terms. C/N ratio was significantly lower at short-term in low wildfire severity area. Extractable Ca and Mg were significantly higher in control plot than in the burned plots in the medium-term. In the long-term, extractable Ca and Mg were significantly lower in the area exposed to a high severity burning. No differences were identified in the case of extractable Na between plots on any of the sampling dates, while extractable K was significantly higher in the plot exposed to low wildfire than it was in the control. Some restoration measures may be required after the wildfire, especially in areas affected by high severity burning, to avoid the long-term impacts on the essential soil nutrients of TC, SOM, extractable Ca and Mg. This long-term nutrient depletion is attributable to vegetation removal, erosion, leaching and post-fire vegetation consumption. Soils clearly need more time to recover from wildfire disturbance, especially in areas affected by high severity fire regimes.

Keywords: Long-term; fire severity; soil chemical properties; fire recurrence; forest resilience.

1. Introduction

Wildfires are a global phenomenon and a natural element of ecosystems, where their impact is dependent upon the fire regime (Bento-Gonçalves et al., 2012; Gill, 1975; González-Pérez et al., 2004). Soils are an essential component of ecosystems, and can undergo substantial modifications because of the direct (e.g. heating) or indirect (e.g. ash) effects of fire; although, they normally return to their pre-fire conditions with time. These effects are largely confined to the first few centimeters of the soil profile. It has been well documented that the impact of fire on soils depends on soil type, fire history, the ecosystem and species burned, the topographical conditions of the affected area, the meteorological conditions during and after the fire and the severity of the fire. Low fire severities can increase soil organic matter (SOM), total nitrogen (TN), total carbon (TC) and total extractable cations, whereas high fire severities consume all the SOM, reduce TN and TC considerably, and substantially increase the amount of major cations in solution. The ability of the burned area to recover its pre-fire levels depends on the ecosystem affected, fire severity, the topography of the fire-affected area and the post-fire meteorological conditions. Soil status in post-fire environments is a key element for ecosystem recovery (Badia et al., 2014; Bodi et al., 2014; Caon et al., 2014; Pereira et al., 2016).

The large majority of studies examining the impact of wildfires on soil focus on their short- (e.g. Romanya et al., 2001; Snyman, 2003; Tessler et al., 2008; Pereira et al., 2017) or medium-term effects (e.g. Lozano et al., 2016; Martinez-Garcia et al., 2017); and owing to constraints of time, logistics and finances, little research has examined their long-term effects. However, long-term studies are essential to gain a good understanding of the ecological role of fire. Most of the long-term studies that have been reported were undertaken in North America (Ojima et al., 1994; Slaughter et al., 1998; Yermakov and Rothstein, 2006; LeDuc and Rothstein, 2010; Johnson et al., 2012), although some data are available from other parts of the world, including South America (Roscoe et al., 2000; Silvana-Longo et al., 2011), Australia (Muñoz-Rojas et al., 2016) and Europe (Kaye et al., 2010). Moreover, while some authors have looked at how soils change up to hundred years after the last known fire (DeLuca et al., 2006; McNamara et al., 2015), they neither monitor the evolution of post-fire soil properties nor record the severity of the fire regime to which they were exposed, a critical parameter in determining the recovery of the ecosystem. As such, there is clearly a pressing need for greater insights into the long-term effects of wildfires on soil properties, especially in fire-prone ecosystems, like the Mediterranean, in order to develop a better understanding of their resilience and capacity to respond to such disturbances. These studies are especially relevant in the current context of climatic change, with predictions that the number and size of fires will increase and that fire regimes will change (e.g. a larger number of high severity fires during longer fire seasons) (Brotons et al., 2013; Turco et al., 2014, 2017).

Although a long-term study of the effects of fire on soil properties in the Mediterranean has been conducted (Kaye et al. 2010), to the best of our knowledge, no study has examined the effects of different fire severity regimes from a long-term perspective. Yet, note that a number of studies have examined the short- and medium-term effects of low (Inbar et al., 2016), moderate (Faria et al., 2015) and high severity fires (Badia et al., 2015; Lombao et al. 2015; Francos et al., 2016b). The importance of long-term studies of the effects of varying fire intensities is that they should shed light on the capacity of soils to respond to different levels of disturbance. This is critical for determining how long low and high fire severity regimes can modify a soil ecosystem. Thus, the aim of the study reported here is to monitor the long-term impact of a wildland fire of different severities on the chemical properties of soil.

2. Materials and methods

2.1. Study area

The study area is located in Cadiretes, Girona, in the north-east of the Iberian Peninsula, at an altitude of between 190 and 250 m a.s.l. (Úbeda et al., 2005) (Figure 1). The parent material consists of metamorphic rock. The massif is covered by dense Mediterranean vegetation, and includes such species as *Quercus suber* L., *Arbutus unedo* L., *Erica arborea* L. *Pinus pinaster* ssp. Mean annual rainfall ranges between 700 and 800 mm, with an autumn maximum and a summer minimum (Úbeda, 2001). Summer temperatures often exceed 25 ºC; while winters are generally mild, rarely below 0 ºC. Evapotranspiration exceeds precipitation in summer months, from June to August (Úbeda, 1998). According to US Soil Taxonomy (Soil Survey Staff, 2014), the soils of the control and low severity zone can be classified as Typic Haploxerept while those in the high severity fire zone are classified as Lithic Haploxerept. The area was affected by a fire that broke out on 5 July 1994 burning an area of 55 ha. In 1994, the area was a plantation of *Pinus pinaster* ssp. with potential for *Quercus suber* L. (Francos et al. (2016a). After the wildfire, no restoration measures were carried out in the forest affected area.

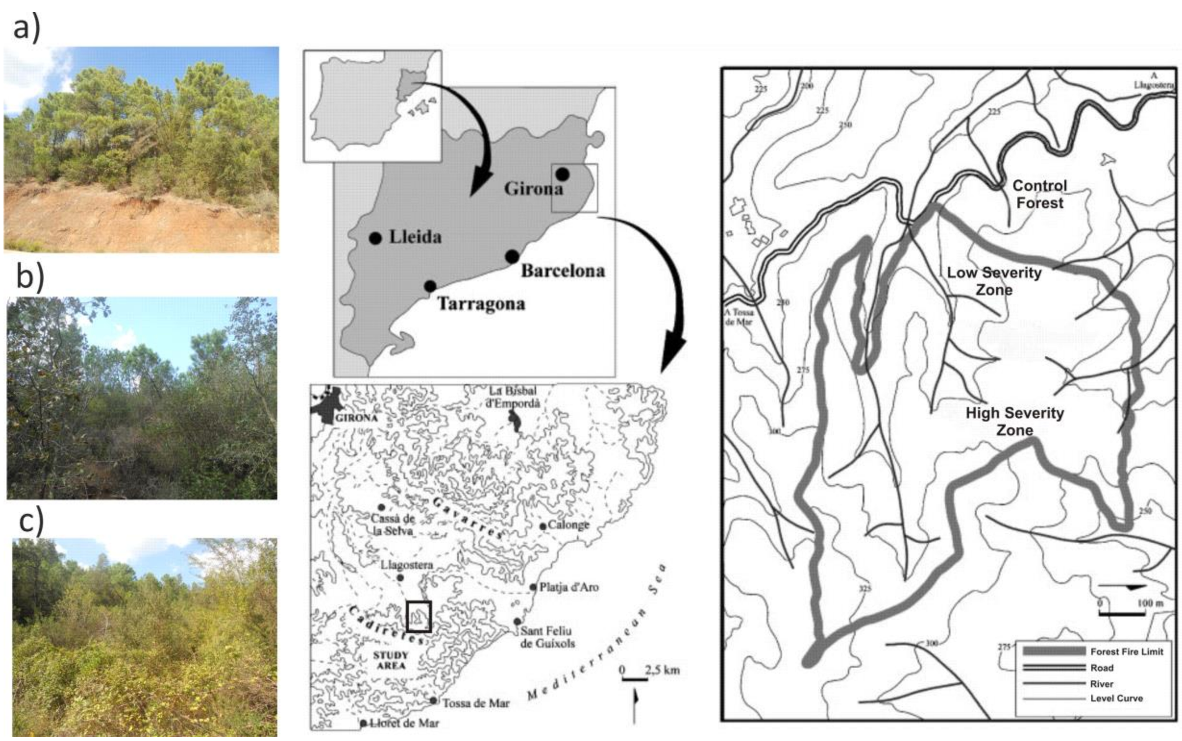


Figure 1. Location of study area. a) Control, b) Low severity and c) High severity. Modified from Úbeda, 1998.

2.2. Experimental design and field sampling

One day after the fire, two plots were designed in an area affected by both low and high severity wildfires (henceforth L-S and H-S plots) respectively,. Both plots were on south-facing slopes with a gradient of approximately 10%. Fire severity was assessed by means of ash color and the number and diameter of surviving branches. Soils covered with black ash were considered to be indicative of low fire severity, while those covered with grey/white ash were deemed indicative of high fire severity (Moreno and Oechel, 1989). An unburned area was selected as control. All three areas present similar environmental characteristics in terms of their parent material, topography and vegetation. Inside each area, we designed a transect, and we sampled the soils each 2 m. In the first sampling campaign, conducted immediately after the fire, we collected five topsoil samples from each plot (0-3 cm). In the second (seven years after the wildfire) and third (18 years after the wildfire) sample campaigns, we collected a further 10 samples per plot per campaign. In total, we collected 75 soil samples. After the fire, no forestry management measures were implemented. Thus, the only impacts on soil properties were those induced by the wildfire and the post-fire natural plant recovery. Two previous studies have been published reporting the results from the data collected in the short- (Ubeda, 2001) and medium-terms (Ubeda et al., 2005).

2.3. Laboratory methods

Samples were dried in the laboratory at room temperature (±23 ºC) for 48 hours. Soils were sieved with a <2 mm mesh to discard the coarser material. TN and TC content were analyzed with a Flash EA 112 Series (Thermo-Fisher Scientific, Milan). Data acquisition and calculations were carried out using Eafer 300 software (Thermo-Fisher Scientific, Milan) (Pereira et al., 2012). SOM was measured using the loss-on-ignition (LOI) method described in Heiri et al. (2001). For each sample, 1 g of soil was pulverized and dried in a muffle furnace at 105 ºC for 24 hours. To estimate SOM, the dried samples were heated at 550 ºC for 4 hours. Soil extractable Ca, Mg, Na and K were analyzed with the method proposed by Knudsen et al. (1986).

2.4. Statistical analysis

Statistical comparisons between sampling times and plots were carried out with a two-way ANOVA test. Significant differences were considered at $p<0.05$. If significant differences were identified, a Tukey HSD post-hoc test was applied. A redundancy analysis (RDA) was carried out to identify the extent to which the variation in one set of variables accounts for the variation in another. Statistical analyzes were carried out using SPSS 23.0 and CANOCO for Windows 4.5 software.

3. Results

3.1. Total Nitrogen, Total Carbon, C/N ratio and Soil Organic Matter.

We found significant differences between time, severity and time *x* severity in the soils TN, TC, and C/N ratio. In the case of SOM, significant differences were only observed between time and severity (Table 1). TN and TC were significantly higher in the plots affected by low and high severity fires in the short-term than they were in the medium- and long-terms. The C/N ratio was significantly lower in both burned plots in the medium-term than it was at the other two sampling times. SOM was significantly higher in the control, L-S and H-S plots in the short-term than it was in the medium- and long-term. In the case of severity, the control plot presented significantly higher TN values in the medium-term. TC content was significantly higher in the control plot compared to corresponding values in the burned plots in the medium- and long-terms. The C/N ratios were significantly lower in the L-S plot than they were in the control plot in the short-term, while SOM was significantly higher in the control compared to the values recorded in the L-S and H-S plots in the medium- and long-terms (Table 2).

Table 1. Two-way ANOVA results. Significant differences at $p<0.05$*. $p<0.01$** and $p<0.001$***. n.s. not significant at a $p<0.05$.

	Sites	p value
Total Nitrogen (%)	Time	***
	Severity	*
	Sampling x Severity	***
Total Carbon (%)	Time	***
	Severity	**
	Sampling x Severity	*
C/N ratio	Time	***
	Severity	***
	Sampling x Severity	**
Soil Organic Matter (%)	Time	***
	Severity	*
	Sampling x Severity	n.s.
Extractable Calcium (ppm)	Time	***
	Severity	**

	Sampling x Severity	*
Extractable Magnesium (ppm)	Time	***
	Severity	*
	Sampling x Severity	n.s.
Extractable Sodium (ppm)	Time	***
	Severity	n.s.
	Sampling x Severity	n.s.
Extractable Potassium (ppm)	Time	***
	Severity	***
	Sampling x Severity	*

Table 2. Descriptive statistics for Total Nitrogen. Total Carbon. C/N Ratio. Soil Organic Matter. Different letters represent significant differences at a $p < 0.05$ between sampling dates (capital letters) and severities (low case letters). Three different Severities: Control, Low Severity and High Severity. Three different Samplings: Short-term (N=5), Medium-term (N=10) and Long-term (N=10).

	Sampling date	Plot	Mean	SD	Min	Max
Total Nitrogen (%)	Short-term	Control	0.39	0.38	0.25	0.54
		Low	0.64 A	0.25	0.49	0.79
		High	0.44 A	0.28	0.29	0.59
		All	0.49	0.28	0.25	0.79
	Medium-term	Control	0.49 a	0.26	0.38	0.59
		Low	0.22 bB	0.09	0.12	0.33
		High	0.16 bB	0.84	0.05	0.26
		All	0.29	0.21	0.05	0.59
	Long-term	Control	0.24	0.07	0.14	0.35
		Low	0.18 B	0.08	0.08	0.28
		High	0.18 B	0.05	0.08	0.28
		All	0.20	0.07	0.08	0.35
Total Carbon (%)	Short-term	Control	10.54	8.61	7.17	13.9
		Low	13.70 A	5.98	10.33	17.06
		High	9.75 A	2.84	6.39	13.12
		All	11.33	6.07	6.39	17.06
	Medium-term	Control	11.23 a	6.07	8.85	13.61
		Low	5.27 bB	1.99	2.9	7.65

		High	3.05 bB	1.84	0.67	5.43
		All	6.52	5.1	0.67	13.61
	Long-term	Control	6.83 a	1.70	4.45	9.21
		Low	4.25 bB	1.75	1.87	6.63
		High	3.9 bB	1.22	1.52	6.28
		All	4.99	2.02	1.52	9.21
C/N ratio	Short-term	Control	50.11 aA	5.56	40.87	55.23
		Low	29.91 bA	14.76	3.66	38.60
		High	38.78 abA	2.70	35.75	42.79
		All	39.60	12.10	3.66	55.23
	Medium-term	Control	16.54 C	9.33	6.62	37.47
		Low	13.98 B	2.17	11.00	18.60
		High	10.84 B	1.94	8.18	13.15
		All	13.79	5.93	6.62	37.47
	Long-term	Control	28.45 B	4.99	31.41	49.57
		Low	23.61 A	5.78	32.02	51.54
		High	20.56 A	4.52	27.30	42.37
		All	38.49	5.22	27.30	51.54
Soil Organic Matter (%)	Short-term	Control	18.16 A	14.85	13.43	22.90
		Low	19.01 A	13.78	14.28	23.75
		High	16.96 A	4.73	12.28	21.70
		All	18.05	11.15	12.28	23.75
	Medium-term	Control	6.39 aB	1.12	3.05	9.74
		Low	3.06 bB	1.16	0.29	6.41
		High	1.77 cC	1.07	1.58	5.11
		All	3.74	2.26	0.29	9.74
	Long-term	Control	9.44 aB	1.94	6.09	12.78
		Low	6.74 bB	2.29	3.39	10.09
		High	6.32 bB	1.06	2.98	9.67
		All	7.50	2.26	2.98	12.78

3.2. Soil Calcium, Magnesium, Sodium and Potassium.

Significant differences were observed in levels of soil extractable Ca and K between time, severity and time *x* severity. In the case of extractable Mg, significant differences were only identified between time and severity, while in that of extractable Na, significant differences were only observed in relation to time (Table 1). Extractable Ca was significantly higher in the areas affected by low fire severity in the medium- and long-terms compared to levels recorded in the short-term. In the H-S plot, extractable Ca was significantly higher in the medium-term sample period than it was in the others. Extractable Mg and Na levels were significantly lower in all the plots studied in the short and long-term periods compared to the levels reported in the medium-term. Extractable K was significantly higher in the medium-term in the control plot when compared to the values recorded in the short- and long-terms (Table 3).

Table 3. Descriptive statistics for Extractable Calcium, Magnesium, Sodium and Potassium. Different letters represent significant differences at a $p < 0.05$ between sampling dates (capital letters) and severities (low case letters). Three different Severities: Control, Low Severity and High Severity. Three different Samplings: Short-term (N=5), Medium-term (N=10) and Long-term (N=10).

	Sampling date	**Severity**	**Mean**	**SD**	**Min**	**Max**
Extractable Calcium (ppm)	Short-term	Control	1289 B	465	447	2132
		Low	1075 B	253	232	1917
		High	1369 C	129	527	2212
		All	1244	244	232	2212
	Medium-term	Control	5170 aA	2032	4574	5766
		Low	3246 bA	669	2650	3842
		High	3248 bA	517	2653	3844
		All	3888	172	2650	5766
	Long-term	Control	3020 aB	678	2425	3616
		Low	2715 aA	1005	2119	3311
		High	1912 bB	289	1316	2508
		All	2549	172	1316	3616
Extractable Magnesium (ppm)	Short-term	Control	392 aB	151	200	584
		Low	213 bC	51	21	405
		High	281 abB	28	89	473
		All	296	56	21	584

	Medium-term	Control	912 A	416	776	1048
		Low	878 A	172	742	1014
		High	785 A	243	649	920
		All	858	39	649	1048
	Long-term	Control	620 aAB	139	484	756
		Low	518 abB	211	383	654
		High	419 bB	48	283	555
		All	519	39	283	756
Extractable Sodium (ppm)	Short-term	Control	347	205	62	631
		Low	380 B	152	96	664
		High	330 AB	166	46	614
		All	352	164	46	664
	Medium-term	Control	554	540	353	755
		Low	854 A	536	653	1055
		High	562 A	288	361	763
		All	657	475	353	1055
	Long-term	Control	194	98	7	394
		Low	102 B	136	99	303
		High	164 B	112	37	364
		All	153	119	7	394
Extractable Potassium (ppm)	Short-term	Control	161 bB	67	95	227
		Low	308 a	110	242	374
		High	278 ab	51	212	344
		All	249	99	95	374
	Medium-term	Control	300 bA	56	254	347
		Low	419 a	119	372	465
		High	252 b	43	205	298
		All	324	105	205	465
	Long-term	Control	216 bB	48	169	262
		Low	307 a	71	261	354
		High	238 ab	71	191	285
		All	254	74	169	354

In the case of severity, levels of soil extractable Ca were significantly higher in the control than they were in the fire affected plots in the medium-term. In the long-term, significantly higher levels were only observed in the case of the H-S plot. Soil extractable Mg content was significantly higher in the control than it was in the L-F plot in the short-term. Extractable Mg was also significantly higher in the control and L-S plots than it was in the H-S plot. Finally, in relation to soil extractable K, we observed significant differences on all the sampling dates. In the short- and long-term, extractable K was significantly higher in the L-S plot than it was in the control. In the medium-term, extractable K content was significantly higher in the L-S plot than it was in the control and the H-S plot.

3.3. Multivariable analysis

Short-term RDA factor 1 explains 41% of the variance, while factor 2 explains 39.6%. In the short- term, the values of TC, TN, SOM, extractable Na and K were highest in the L-S plot. Soil extractable Ca, Mg and C/N presented the highest values in the control plot (Fig. 1a). Medium-term RDA factor 1 explains 45.7% of the variance and factor 2 explains 39.6%. The values of TN, TC, C/N ratio, SOM, extractable Ca and Mg were highest in the control plot, while Na and K were highest in the L-S plot (Fig. 1b). Finally, long-term RDA factor 1 explains 44.4% of the variance and factor 2 explains 33.4%. As for the medium-term sampling period, values of TN, TC, C/N ratio, SOM, extractable Ca and Mg were highest in the control plot. At this sampling date, extractable Na was also highest in the control. Soil extractable K content was still highest in the L-S plot.

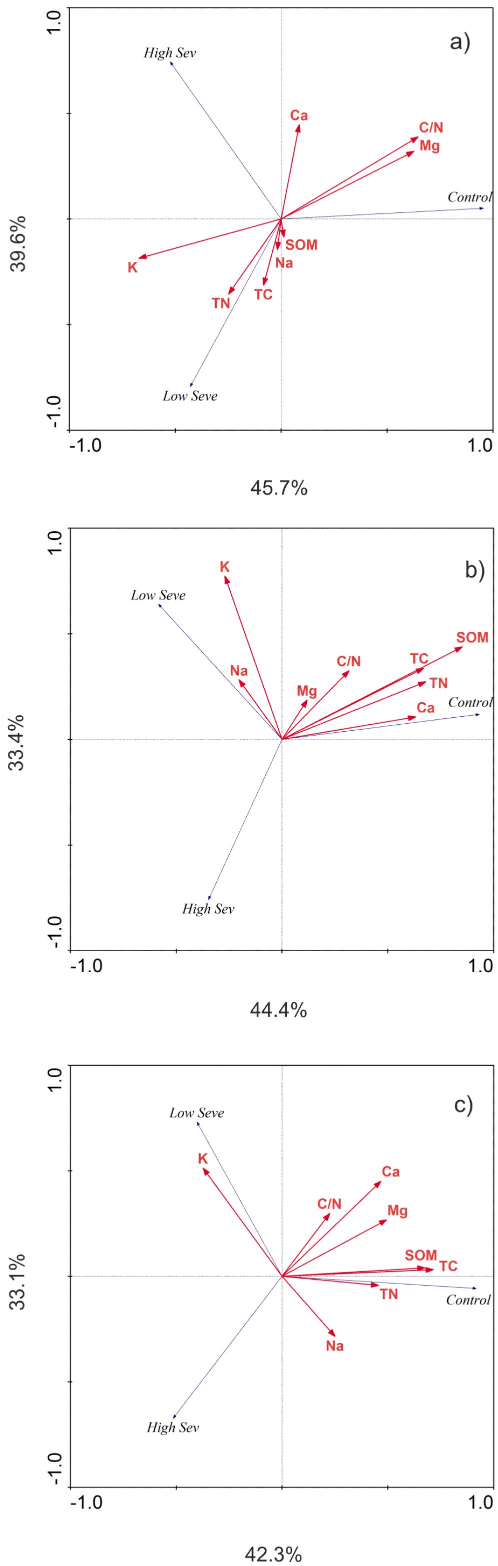


Figure 2. RDA for the relation between Factor 1 and 2. The environmental variables are TN (Total Nitrogen), TC (Total Carbon), C/N (Carbon/Nitrogen ratio), SOM (Soil Organic Matter), Ca (Calcium), Mg (Magnesium), Na (Sodium), K (Potassium). The cases of study are a) Short-term sampling, b) Medium-

term sampling, c) Long-term sampling. The three different severities are: Control (Unburned), Low Sev (Low Severity area) and High Sev (High Severity area).

4. Discussion

4.1. Total Nitrogen, Total Carbon, C/N ratio and Soil Organic Matter.

Soil TN content was significantly higher in the short-term than it was on the other sampling dates in both burned sites. This can be attributed to the incorporation of ash and charcoal into the soil profile in the immediate aftermath of the fire (Goforth et al., 2005; Pereira et al., 2011; Dzwonko et al., 2015). With time, there is a reduction in soil TN content, independent of fire severity, because of post-fire erosion, leaching and plant consumption (see Xue et al., 2014). Differences between plots were identified only in the medium-term, reflecting the marked reduction in TN values in the burned plots for the reasons outlined above. Independent of severity, the fire did not have any short-term implications on soil TN (despite the non-significant increase in both fire-affected plots in relation to the control). Studies conducted elsewhere report different results: for example, Murphy et al. (2006) and Knoepp et al. (2005) observed a significant decrease in soil TN after a wildfire.

As with TN, soil TC content was significantly lower between the first and second sampling dates, which can be attributed to the same causes described above for TN and to the increasing mineralization rate of carbon in the presence of high amounts of nutrients in solution (Alcaniz et al., 2016). In the medium- and long-terms, the fire-affected areas presented significantly lower TC values than those in the control plot. Kvadir et al. (2012) also observed a significantly lower concentration of soil TC in burned soils compared to that found in controls 12 years after a wildfire. Our results showed that after 18 years, the impacts of the wildfire could still be observed in soil TC content, which had failed to recover their pre-fire levels irrespective of the severity of the regime. Reduced levels of litter deposition meant less carbon was returned to the soils, indicating that in the case of soil TC content, the wildfire represented a long-term disturbance. In burned soils the amount of fresh residues – which are rich in labile and easily decomposable materials – is reduced (Kvadir et al., 2005). Silvana-Longo et al. (2011), Kaye et al. (2010) and Johnson et al. (2005) observed that four, 10 and 20 years after a wildfire, respectively, TC levels failed to return to their pre-fire levels.

The C/N ratio was significantly lower in the medium-term than at the other two sampling dates. The fall in the C/N ratio in the medium-term is the consequence of the mineralization effect of fire. The incorporation of burned material into the soil profile reduces the C/N ratio (Pereira et al., 2011) and in fire-affected soils this reduction is due to the fact that N is immobilized (in contrast to C) in recalcitrant heterocyclic structures.

When heated, organic fractions with a low degree of humification increase their relative N content (Badia and Marti, 2003; Certini et al., 2011). This may have contributed to a reduction in the soil C/N ratio reduction in the medium-term (Black and Harden, 1995; Badia and Marti, 2003; Gonzalez-Perez et at., 2004). Pyrogenic organic matter is highly recalcitrant and resistant to microbial decomposition and so may remain in the soil profile for a long time (Santin et al., 2016). Knicker et al. (2013) observed a considerable presence of charcoal in the topsoil five years after a severe wildfire. Eighteen years after the fire, the soil C/N ratio had returned to its pre-fire level, which may be a consequence of the decrease in ash and charcoal, and an increase in fresh litter, which slightly increased SOM content. Our results are in line with Roscoe et al. (2000), who did not identify any significant differences in soil C/N ratio 21 years after a wildfire. In contrast, Johnson et al. (2012) observed that C/N ratio was significantly high in the control plot 12 years after a wildfire. Here, the area affected by the low severity fire had a significantly lower C/N ratio, the consequence of the relative enrichment of soil TN as discussed above. The high C/N ratio observed in the area affected by the high severity fire, similar in that respect to the ratio in the control, can be attributed to the decline in TN. Soto et al. (1991) observed that nitrogen losses are low below temperatures of 460 ºC, but that they increase at greater temperatures. Ubeda et al. (2009) observed that ash produced at high temperatures had a high C/N ratio as result of the marked decrease in TN content.

SOM content in both burned plots was significantly higher in the short-term than in the medium- and long-terms. As expected, it follows the same dynamic as that observed for TC and TN content, and the reasons for the subsequent drop are as above. Our results are in line with previous studies were a decrease in SOM was observed in both short- (Pereira et al., 2014) and long-term studies (Kloss et al., 2012; Rovira et al., 2013; Xue et al., 2014). After the fire, soil needs a certain time to recover. During the post-fire period, there is a negative SOM balance, despite the input of ash and charcoal into the soil profile. The natural input of SOM is interrupted and is only recovered with the reestablishment of vegetation (Rovira et al., 2013). The effects of wildfire on SOM immediately after a fire vary: some studies report an increase (Mataix-Solera et al., 2002; Santin et al., 2008; Muranova and Simanský, 2015), others a decrease (Fernandez et al., 1997; Neff et al., 2005; Murphy et al., 2006), and some, as in our study, report no change (Jordán et al., 2011), independently of the fire severity observed. However, low fire severities typically increase or have no effect on SOM, while high fire severities reduce it (Jordán et al., 2011; Mehdi et al., 2012; Caon et al., 2015). SOM was significantly lower in the burned areas in the medium- and long-terms, as was TC, as a consequence of post-fire erosion, leaching and plant consumption. The fact that SOM was significantly lower in the burned plots in the medium-term is evidence that some erosion may have occurred between the first and second sampling dates. SOM erosion on slopes has been identified previously (Novara et al., 2011). Soil mantle and plant recovery is crucial in the immediate months following a fire to reduce the impact of torrential rainfalls. If this does not happen, erosion may induce a delay in ecosystem reorganization (Pardini et al., 2004).

4.2. Soil Calcium, Magnesium, Sodium and Potassium.

Levels of extractable Ca and Mg were significantly higher in the medium-term compared to the levels recorded at the other two sampling dates (with the exception of extractable Ca between the medium- and long-terms in the L-S plot). Soils were richer in these elements in the medium-term, coinciding with the lowest C/N ratio, and due in all probability to the incorporation of charcoal rich in these nutrients following the first rainfalls. Despite the fall in TC and SOM, the mineralization induced by fire increases the availability of Ca and Mg in ash (Pereira et al., 2012) and this may explain the increase in these nutrients in the medium-term. It also indicates that some of the combusted material was incorporated into the soil profile in the period between the first two sampling dates. The long-term reduction may be the consequence of vegetation consumption. Several studies observed a significant increase in extractable Ca and Mg after wildfires (Kutiel and Inbar, 1993; Pardini et al., 2004). Compared to the figures for the control plot, no significant differences were observed in extractable Ca in the short-term; however, they were seen in extractable Mg. In both cases, nutrient concentrations were lower in the L-S plot. There is no obvious explanation for this, but we hypothesize that the degree of mineralization associated with the fire in the L-S plot was not as great as that in the H-S plot and, hence, nutrient availability was not as high. Previous studies carried out in areas affected by low severity fires observed a decrease in Mg in relation to levels in the control (Thomaz et al., 2014; Fuentes-Ramirez et al., 2015). In the medium-term, the values of extractable Ca and Mg were higher in the control plot, but only significantly so in the case of Ca. The lower content in relation to that in the control may have been due to nutrient depletion induced by erosion, leaching or vegetation consumption between the first and second sampling dates. Finally, in the long-term, the soils of the plot affected by the H-S fire presented significantly lower values of extractable Ca and Mg, compared to those in the control, showing that in this plot nutrients did not recover to their pre-fire levels. However, in stark contrast, Johnson et al. (2005) observed an increase in extractable Ca and Mg in the long-term.

Levels of extractable Na were significantly higher in the fire-affected plots in the medium-term than they were at the other sampling dates. In this respect, extractable Na followed the same patterns as extractable Ca and Mg, a pattern that can be attributed to the incorporation of combusted material into the soil profile. Studies elsewhere also report an increase in extractable Na after fire (Scarenbroch et al., 2012). There were no significant differences in soil extractable K between sampling dates on the fire affected plots. However, in each sampling period, the concentration of this nutrient was always significantly higher in the L-S plot than that in the control plot. Previous studies have reported an increase in extractable K in fire affected areas compared to unburned areas (Marion et al., 1991; Kutiel and Inbar, 1993; Scarenbroch et al., 2012).

4.3. Overall discussion and implications for management of wildfire affected areas

The wildfire changed the soils' nutrient dynamics. In the short-term, some nutrients were identified at high concentrations in the L-S plot (e.g. TN, TC, SOM, extractable Na and K), while in the H-S plot the concentration of the nutrients studied (with the exception of extractable Ca) was always below that of the nutrients in the other two plots. This points to the heterogeneous impact of the wildfire, which, in the short-term, can increase soil nutrients in some areas but reduce them in another. The low concentration of soil nutrients in plots affected by high severity fires is a consequence of the high temperatures reached, the high impact on the soils and the volatilization of nutrients (Caon et al., 2014). The lack of nutrients in the soil limited the recovery of the vegetation and plant diversity six months after the fire, in a comparative study with an area affected by a low severity fire (Francos et al., 2016a). In the medium-term, with the exception of extractable Na and K, the values of the other elements were high in the L-S plot, a consequence of post-fire erosion, leaching and vegetation consumption. Finally, in the long-term, the concentration of nutrients was very similar to that observed in the medium-term (with the exception of extractable K, all the other nutrients were present in high concentrations in the control plot). The wildfire studied had a long-term effect on some of the soil nutrients studied. Extractable Na was unaffected by fire (compared to levels in the control area) in any of the sampling periods, while TN and the C/N ratio returned to pre-fire levels in both the L-S and H-S plots. Extractable Ca and Mg only recovered in the L-S plot, while TC and SOM failed to recover their pre-fire levels 18 years after the wildfire in both burnt plots. This shows that rates of soil nutrient recovery differ after a fire, and in this case, TC and SOM were the most affected, irrespective of the severity of the fire regime. This finding contradicts previous studies that suggest that fires may contribute to long-term carbon stock pools and increase the soil capacity of carbon storage (Bennet et al., 2014; Santin et al., 2015). Extractable K was always significantly higher in the L-S plot, showing the long-term effect of fire on this nutrient.

Several studies conducted in the Mediterranean environment show that vegetation can recover rapidly after wildfires (Diaz-Delgado and Pons, 1995; Wittenberg et al., 2007; Tessler et al., 2016); however, few studies have examined their impact on soils. Indeed, according to our results, certain soil properties (TC, SOM extractable Ca and Mg) can take many years to return to their pre-fire levels. Here, the fact that the wildfire fire changed the plant composition, from a *Pinus* plantation to a *Quercus suber* forest, may have influenced soil recovery to pre-fire levels. To avoid soil degradation in the immediate post-fire period and to contribute to a faster recovery of soil properties, certain restoration measures – including different types of mulching (Santana et al., 2014), namely: organic amendments (Cellier et al., 2014), hydroseeding (Vourlitis et al., 2017), straw (Vega et al., 2014) or forest residues (Prats et al., 2014) – may be important in avoiding soil erosion, nutrient depletion and the long-term effects of fire on soil properties, especially in areas affected by high fire severities. The difficulties soils face in recovering increase with the frequency of wildfires, as observed by Guenon et al. (2013). Changes in land-use and climate change in the Mediterranean region (e.g. temperature increase and precipitation reduction) will increase wildfire frequency and severity (Turco et al., 2015), and therefore, decrease the capacity of soil recuperation to this disturbance. This may cause at long-term soil and land degradation. However, management techniques should not be implemented in the immediate post-fire period since this is when the soil is most vulnerable to human disturbance. Other post-fire engineering measures are also being used in areas affected by fire. They include hillslope treatments, erosion barriers, road and trail treatments (e.g. armoring, flow directions and water passage structures) and channel treatments (Robichaud, 2009). Salvage logging may not be an appropriate measure due to its negative effects on soil properties (Fernandez and Vega, 2016; Garcia-Orenes et al., 2017).

5. Conclusions

The wildfire studied here had different long-term effects on the soils' chemical properties, indicating that soil nutrients respond differently according to the time since wildfire and severity of wildfire. Levels of extractable Na did not differ from those recorded in the control plot on any of the sampling dates, while TN and C/N recovered to pre-fire levels. Major cations only recovered in the area affected by low fire severity and TC and SOM did not recover in either of the burned areas. The low severity fire also led to a long-term increase in extractable K levels. The lack of recovery presented by the essential soil parameters – including, extractable Ca and Mg (in the area affected by high severity fire), TC and SOM – show that restoration measures may be needed, especially after high severity fires. The inability of nutrients to recover their pre-fire levels is clearly a consequence of the disturbances caused by the fire (e.g. vegetation removal, erosion, leaching) and the subsequent loss of vegetation with time, decreasing overall soil nutrient content. Soils need more time to return to their pre-fire levels, especially in areas affected by high severity fire. However, further research is needed to observe when TC and SOM return to pre-fire levels as here this may have been influenced by the change in vegetation cover.

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References

Alcaniz, M., Outeiro, L., Francos, M., Farguell, J., Ubeda, X., 2016. Long-term dynamics of soil chemical properties after a prescribed fire in a Mediterranean forest (Montgri Massif, Catalonia, Spain). Sci. Total Environ. 572, 1329–1335.

Amores, R., Outeiro, L., Asperó, F., Úbeda, X., 2008. Efectos a medio plazo en la vegetación mediterránea de una plantación de *Pinus pinaster* sometida a diferentes intensidades de fuego. AURORA geography journal 2, 13-30.

Badia, D., Marti, C., 2003. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Research and Management 17, 23–41.

Badia, D., Martí, C., Aguirre, A.J., Aznat, J.M., Gonzalez-Pérez, J.A., De la Rosa, J.M., León, J., Ibarra, P., Echevarria, T., 2014. Wildfire effects on nutrients and organic carbon of a Rendzic Phaeozem in NE Spain: Changes at cm-scale topsoil. Catena 113, 267-275.

Bennett, L.T., Aponte, C., Baker, T.G., Tolhurst, K.G., 2014. Evaluating long-term effects of prescribed fires on carbon stocks in a temperate eucalypt forest. For. Ecol. Manage. 328, 219–228.

Bento-Gonçalves, A., Vieira, A., Úbeda, X., Martin, D., 2012. Fire and soils: key concepts and recent advances. Geoderma 191, 3-13.

Black, T.A., Harden, J.W., 1995. Effect of timber harvest on soil carbon storage at Blodgett Experimental Forest, California. Can J For Res 25, 1385-1396.

Bodi, M., Martin, D., Santin, C., Balfour, C., Doerr, S., Pereira, P., Cerda, A., Mataix-Solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 130, 103–127.

Brockway, D.G., Gatewood, R.G., París, R.B., 2002. Restoring fire as an ecological process in shortgrass prairie ecosystems: initial effects of prescribed burning during the dormant and growing seasons. Journal of Environmental Management 65, 135-152.

Brotons, L., Aquilue, N., Caceres, M., Fortin, M.J., Fall, A., 2013. How fire history, fire suppression practices and climate change affect wildfire regimens in Mediterranean landscapes. PlosOne 8, e2392

Caldwell, T.G., Johnson, D.W., Miller, W.W., Qualls, R.G., 2002. Forest floor carbon and nitrogen losses due to prescription fire. Soil Sci. Soc. Am. J. 66, 262-267.

Caon, L., Vallejo, V.R., Coen, R.J., Geissen, V., 2014. Effects of wildfire on soil nutrients in mediterranean ecosystems. Earth-Science Reviews 139, 47-58.

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 angustifolia* L.). Plant Ecology 204, 155-171.

Carreira, J.A., Niell, F.X., 1992. Plant nutrient changes in a semi-arid Mediterranean shrubland after fire. Journal of Vegetation Science 3, 457-466.

Castelli, L., Lazzari, M., 2002. Impact of fire on soil nutrients in central semiarid Argentina. Arid Land Res. Manag. 16, 349-364.

Castro, A., González-Prieto, S.J., Carballas, T., 2006. Burning effects on the distribution of organic N compounds in a N-15 labelled forest soil. Geoderma 130, 97-107.

Cellier, A., Gauquelin, T., Baldy, V., Ballini, C., 2014. Effects of organic amendment on soil fertility and plant nutrients in a post-fire Mediterranean ecosystem. Plant Soil 376, 211–228.

Certini, G., Nocentini, C., Knicker, H., Arfaioli, P., Rumpel, C., 2011. Wildfire effects on soil organic matter quantity and quality in two fire-prone Mediterranean forests. Geoderma 167-168, 148–155.

Certini, G., Nocetini, C., Knicker, H., Arfaioli, P., Rumpel, C., 2011. Wildfire effects on soil organic matter quantity and quality in two fire-prone Mediterranean pine forests. Geoderma 167-168, 148-155.

Corona, P., Ascoli, D., Barbati, A., Bovio, G., Colangelo, G., Elia, M., Garfi, V., Iovino, F., Lafortezza, R., Leone, V., Lovreglio, R., Marchetti, M., Marchi, E., Menguzzato, G., Nocentini, S., Picchio, R., Portoghesi, L., Puletti, N., Sanesi, G., Chianucci, F., 2015. Integrated forest management to prevent wildfires under Mediterranean environments. Annals of Silvicultural Research 39 (1), 1-22.

DeBano, L.F., Conrad, C.E., 1978. The effect of fire on nutrients in a chaparral ecosystem. Ecology 489-497.

DeLuca, T.H., MacKenzie, M.D., Gundale, M.J., Holben, W.E., 2006. Wildfire-Produced charcoal directly influences nitrogen cycling in Ponderosa Pine forests. Soil Sci. Soc. Am. J. 70, 448–453.

DeLuca, T.H., MacKenzie, M.D., Gundale, M.J., Holben, W.E., 2006. Wildfire-Produced Charcoal Directly Influences Nitrogen Cycling in Ponderosa Pine Forests. Soil Science Society of America Journal 70, 448-453.

Diaz-Delgado, R., Pons, X., 2001. Spatial patterns of forest fires in Catalonia (NE of Spain) along the period 1975–1995: Analysis of vegetation recovery after fire. For. Ecol. Managem. 147, 67–74.

Dzwonko, Z., Loster, S., Gawronski, S., 2015. Impact of fire severity on soil properties and development of tree and shrub species in a Scots pine moist forest site in Poland. For. Ecol, Manage. 342, 56–63.

Faria, S., De la Rosa, J.M., Knicker, H., Gonzalez-Perez, J.A., Keizer, J.J., 2015. Molecular characterization of wildfire impacts on organic matter in eroded sediments and topsoil in Mediterranean eucalyptus stands. Catena 135, 29–37.

Fernandes, P.M., Davies, G.M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., Stoof, C.R., Vega, J.A., Molina, D., 2013. Prescribed burning in southern Europe: developing fire management in a dynamic landscape. Frontiers in Ecology and the Environment 11, 4-14.

Fernandez, C., Vega, C., 2016. Effects of mulching and post-fire salvage logging on soil erosion and vegetative regowth in NW Spain. For. Ecol. Manage. 375, 46–64.

Fernandez, I., Cabanairo, A., Carballas, T., 1997. Organic matter changes immediatly after a wildfire an Atlantic forest soil and comparison with laboratory soil heating. Soil Biol. Biochem. 29, 1–11.

Ferran, A., Delitti, W., Vallejo, V.R., 2005. Effects of fire recurrence in *Quercus coccifera* L. shrubland of the Valencia Region (Spain): II. plant and soil nutrients. Plant Ecology 177, 71-83.

Francos, M., Pereira, P., Alcañiz, M., Mataix-Solera, J., Úbeda, X., 2016b. Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain). Science of Total Environment 572, 1353-1362.

Francos, M., Úbeda, X., Tort, J., Panareda, J.M., Cerdà, A., 2016a. The role of forest fire severity on vegetation recovery after 18 years. Implications for forest management of Quercus suber L. in Iberian Peninsula. Global and Planetary Change 145, 11-16.

Fuentes-Ramirez, A., Schafer, J.L., Mudrak, E.L., Schat, M., Parag, H.A., Holzapfel, C., Moloney, K.A., 2015. Spatio-temporal impacts of fire on soil nutrient availability in *Larrea tridentate* shrublands of the Mojave Desert, USA. Geoderma 259-260, 126–133.

Garcia-Orenes, F., Arcenegui, V., Chrenkova, K., Mataix-Solera, J., Molto, J., Jara-Navarro, A.B., Torres, M.P., 2017. Effects of salvage logging on soil properties and vegetation recovery in a fire-affected Mediterranean forest: A two year monitoring research. Sci. Total Environ. 586, 1057–1065.

Gill, A.M., 1975. Fire and the Australian flora: a review. Australian forestry 38 (1), 4-25.

Goforth, B.R. Graham, R.C., Hubbert, K.R, Zanner, C.W., Minnich, R.A., 2005. Spatial distribution and properties of ash thermally altered soils after high-severity forest fires, southern California. International Journal of Wildland Fire 14, 343-354.

González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fire on soil organic matter-a review. Environment International 30 (6), 855-870.

Granged, A.J.P., Jordán, A., Zavala, L., Muñoz-Rojas, M., Mataix-Solera, J. 2011. Short-term effects of experimental fire a soil under eucalyptus forest (SE Australia). Geoderma 167-168, 125-134.

Grove, T.S., O'Conell, A.M., Dimmock, G.M., 1986. Nutrient changes in surface soils after an intense fire in jarrah (Eucalyptus marginata Donn ex Sm.) forest. Australina Journal of Ecology 11, 303-317.

Guenon, R., Vennetier, M., Dupuy, N., Roussos, S., Pailler, A., Gros, R., 2013. Trends in recovery of Mediterranean soil chemical properties and microbial activities after infrequent and frequent wildfires. Land Degrad. Develop. 24, 115–128.

Henri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J. Paleolomnol. 5, 101-110.

Inbar, A., Lado, M., Sternberg, M., Tenau, H., Ben-Hur, M., 2014. Forest fire effects on soil chemical and physicochemical properties, infiltration, and erosion in a semi-arid Mediterranean region. Geoderma 221-222, 131–138.

Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management 140, 227-238.

Johnson, D.W., Murphy, J.F., Susfalk, R.B., Caldwell, T.G., Miller, W.W., Walker, R.F., Powers, R.F., 2005. The effects of wildfire, salvage logging, and post-fire N-fixation on the nutrient budget of a Sierran forest. Forest Ecology and Management 220, 155-165.

Johnson, D.W., Walker, R.F., McNulty, M., Rau, B.M., Miller, W.W., 2012. The Long-Term Effects of Wildfire and Post-Fires Vegetation on Sierra Nevada Forest Soils. Forests 3, 398-416.

Jordan, A., Zavala, L.M., Mataix-Solera, J., Nava, A.L., Alanís, N., 2011. Effect of fire severity on water repellency and aggregate stability on Mexican volcanic soils. Catena 84, 136-147.

Kavdir, Y., Ekinci, H., Yuksel, O., Mermut, A.R., 2005. Soil aggregate stability and ^{13}C CP/MAS-NMR assessment of organic matter in soils influenced by forest wildfires in Canakkale, Turkey. Geoderma 129, 219–229.

Kaye, J.P., Romanyà, J., Vallejo, V.R., 2010. Plant and soil carbon accumulation following fire in Mediterranean woodlands in Spain. Oecologia 164, 533-543.

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

Khanna, P.K., Raison, R.J., Falkiner, R.A., 1994. Chemical properties of ash derived from Eucalyptus litter and its effects on forest soils. Forest Ecology and Management 66, 107-125.

Kloss, S., Sass, O., Geitner, C., Prietzel, J., 2012. Soil properties and charcoal dynamics of burnt soils in the Tyrolean Limestone Alps. Catena 99, 75–82.

Knicker, H., Gonzalez-Vila, F.J., Gonzalez-Vazquez, R., 2013. Biodegradability of organic matter in fire-affected mineral soils of southern Spain. Soil Biology and Biochemistry 56, 31–39.

Knoepp, J.D., DeBano, L.F., Neary, D.G., 2005. Soil Chemistry. In: United States Department of Agriculture (USDA). Wildland Fire in Ecosystems. Effects of Fire on Soils and Water. Vol.4.

Knudsen, D., Petersen, G.A., Pratt, P.F., 1986. Lithium, sodium and potassium. In: Soil Science Society of America (Ed.). Methods of soil analysis. Vol. 2. ASA-SSSA, Madison, WI. 225-246.

Kutiel, P., Inbar, M., 1993. Fire impacts on soil nutrients and soil-erosion in a Mediterranean pine forest plantation. Catena 20, 129-139.

Kutiel, P., Shaviv, A., 1992. Effects of soil type, plant composition and leaching on soil nutrients following a simulated forest-fire. Forest Ecology and Management 53, 329-343.

Lasanta, T., Cerdá, A., 2005. Long-term erosional responses after fire in the Central Spanish Pyrenees: 2. solute release. Catena 60, 80-101.

LeDuc, S.D., Rothstein, D.E., 2010. Plant-available organic and mineral nitrogen shift in dominance with forest stand age. Ecology 91, 708-720.

Lombao, A., Barreiro, A., Cancelo-González, J., Matín, A., Díaz-Raviña, M. 2015. Impact of termal shock on forest soils affected by fires of different severity recurrence. Spanish Journal of Soil Science 5 (2), 165-179.

Lozano, E., Jimenez-Garcia, P., Mataix-Solera, J., Arcenegui, V., Mataix-Beneyeto, J., 2016. Sensivity of gomalin-related soil protein to wildfires: Immediate and medium-term changes. Sci. Tot. Environ. 572, 1238–1243.

Marion, G.M., Moreno, J.M., Oechel, W.C., 1991. Fire severity, ash deposition, and clipping effects on soil nutrients in Chaparral. Soil Sci. Soc. Am. J. 55, 235-240.

Martinez-Garcia, E., Lopez-Serrano, F.R., Dadi, D., Garcia-Morote, F.A., Andres-Abellan, M., Pumpanen, J., Rubio, E., 2017. Medium-term dynamics of soil respiration in a Mediterranean mountain ecosystem: The effects of burn severity, post-fire burnt-wood management, and slope-aspect. Agr. Forest Metereol. 233, 195–208.

Mataix-Solera, J., Gomez, I., Navarro-Pedreno, J., Guerrero, C., Moral, R., 2002. Soil organic matter and aggregates affected by wildfire in Pinus helepensis forest in a Mediterranean environment. International Journal of Wildland Fire 11, 107–114.

McNamara, N.P., Gregg, R., Oakley, S., Stott, A., Tanvir Rahman, Md, Collin Murrell, C., Wardle, D.A., Bardgett, R.D., Ostle, N.J., 2015. Soil methane sink capacity response to long-term wildfire chronosequence in Northern Sweden. PlosOne 10, e0129892

Mehdi, H., Mahdavi, S.A., Mostafa, A. 2012. Effects of different fire severity levels on soil chemical and physical properties in Zagros forests of western Iran. Folia Forestalia Polonica 54 (4), 241-250.

Miesel, J.R., Goebel, P.C., Corace III, R.G., Hix, D.M., Kolka, R., Palik, B., Mladenoff, D., 2012. Fire effects on soils in Lake State Forests: A Compilation of Published Research to Facilitate Long-Term Investigations. Forests 3, 1034-1070.

Monsanto, P.G., Agee, J.K., 2008. Long-term post-wildfire dynamics of a coarse woody debris after salvage logging and implications for soil heating in dry forests of the eastern Cascades, Washington. Forest Ecology and Management 255, 3952-3961.

Moreno, J.M., Oechel, W.C., 1989. A simple method for estimating fire severity after a burn in a California chaparral. Acta Oeolgica 10, 57-68.

Muñoz-Rojas, M., Erickson, T.E., Martini, D., Dixon, K.W., Merritt, D.J. 2016. Soil physicochemical and microbiological indicators of short, medium and long term post-fire recovery in semi-arid ecosystems. Ecological Indicators 63, 14-22.

Muranova, K., Simanský, V. 2015. The effect of different severity of fire on soil organic matter and aggregate stability. Acta fytotechn. zootechn. 18, 1-5.

Murphy, J.D., Johnson, D.W., Miller, W.W., Walker, R.F., Blank, R.R., 2006. Prescribed fire effects on forest floor and soil nutrients in a Sierra Nevada forest. Soil Science 171, 181-199.

Neff, J.C., Harden, J.W., Gleixner, G., 2005. Fire effects on soil organic matter content, composition, and nutrients in boreal interior Alaska. Can. J. For. Res. 35, 2178–2187.

Novara, A., Gristina, L., Bodi, M., Cerda, A., 2011. The impact of fire on redistribution of soil organic matter on a Mediterranean hillslope under maquia vegetation type. Land Degradation & Development 22, 530–536.

Ojima, D.S., Schimel, D.S., Parton, W.J., Owensby, C.E., 1994. Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24, 67-84.

Pardini, G., Gispert, M., Dunjo, G., 2004. Relative influence of wildfire on soil properties and erosion process in different Mediterranean environments in NE Spain. Sci. Tot. Environ. 328, 237–246.

Pereira, P., Cerda, A., Martin, D.A., Ubeda, X., Depellegrin, D., Novara, A., Martinez-Murillo, J.F., Brevik, E.C., Menshov, O., Rodrigo Comino, J., Miesel, J., 2017. Short-term low severity spring grassland fire impacts on soil extractable elements and ratios in Lithuania. Sci. Tot. Environ. 578, 469–475.

Pereira, P., Rein, G., Martin, D., 2016. Editorial: Past and present post-fire environments. Sci. Tot. Environ. 573, 1275–1277.

Pereira, P., Úbeda, X., Martin, D., 2012. Fire severity effects on ash chemical composition and water-extractable elements. Geoderma 191, 105-114.

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Guerrero, C., 2011. Effects of a low prescribed fire in ash water soluble elements in a Cork Oak (Quercus suber) forest located in Northeast of Iberian Peninsula, Environ. Res. 111, 237–247.

Pereira, P., Ubeda, X., Mataix-Solera, J., Oliva, M., Novara, A., 2014. Short-term changes in soilMunsell colour value, organic matter content and soil water repellency after a spring grassland fire in Lithuania. Solid Earth 4, 209–225.

Peterson, D.W., Hessburg, P.F., Salter, B., James, K.M., Dahlgreen, M.C., Barnes, J.A., 2007. Reintroducing fire in regenerated dry forests following a stand-replacing wildfire. In: Powers, R.F. (tech. Ed.), Restoring Fire-Adapted Ecosystems: Proceedings of the 2005 National Silvidulture Workshop. USDA Forest Service General Technical Reports PSW-GTR-203, pp 79-86.

Piqué, M., Castellnou, M., Valor, T., Pagés, J., Larrañaga, A., Miralles, M., Cervera, T., 2011. Integració del risc de grans incendis forestals (GIF) en la gestió forestal: Incendis tipus i vulnerabilitat de les estructures forestals al foc de capçades. Sèrie: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST). Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya.

Prats, S.A., Martins, M.A.S., Malvar, M.C., Ben-Hur, M., Keizer, J.J., 2014. Polyacrylamide application versus forest residue mulching for reducing post-fire runoff and soil erosion. Sci. Total Environ. 468-469, 467–474.

Quintana, J.R., Cala, V., Moreno, A.M., Parra, J.G., 2007. Effects of heating on mineral components of the soil from Spanish juniper (*Juniperus thurifera* L.) woodland. Journal of Arid Environments 71, 45-56.

Rau, B.M., Tausch, R., Reiner, A., Johnson, D.W., Chambers, J.C., Blank, R.R., Luchessi, A., 2010. Influence of prescribed fire on ecosystems biomass, carbon and nitrogen in a pinyon juniper woodland. Rangel. Ecol. Manag. 63, 197-202.

Robichaud, P., 2009. Post-fire stabilization and rehabilitation. In Cerdà, A., Robichaud, P. (Eds.) Fire Effects on Soils and Restoration Strategies. Science Publishers, Enfield, New Hampshire, pp. 299-320.

Romanya, J., Casals, P., Vallejo, V.R., 2001. Short-term effects on soil nitrogen availability in Mediterranean grasslands and shrublands growing in old fields. Forest Ecol. Manage. 147, 39–53.

Romanya, J., Rovira, P., Duguy, B., Vallejo, R., Rubio, A., 2007. C Sequestration issues in the Mediterranean soils. Greenhouse-gas budget of soils under changing climate and land use (BurnOut). COST Action vol. 639, pp. 15-22.

Roscoe, R., Buurman, P., Velthorst, E.J., Pereira, J.A.A., 2000. Effects of fire on soil organic matter in a "cerrado sensu-stricto" from Southeast Brazil as revealed by changes in δ 13C. Geoderma 95, 141-160.

Rovira, P., Romanya, J., Duguy, B., 2012. Long-term effects of wildfires on the biochemical quality of soil organic matter: A study on Mediterranean shrublands. Geoderma 179-180, 9–19.

Santana, V.M., Alday, J.G., Baeza, J., 2014. Mulch application as post-fire rehabilitation does not affect vegetation recovery in ecosystems dominated by obligate seeders. Ecol. Eng. 71, 80–66.

Santin, C., Doerr, S., Merino, A., Bryant, R., Loader, N.J., 2016. Forest floor chemical transformations in a boreal forest fire and their correlations with temperature heating and duration. Geoderma 264, 71–80.

Santin, C., Doerr, S., Preston, C.M., Gonzalez-Rodriguez, G., 2015. Pyrogenic organic matter production from wildfires: a missing link in the global carbon cycle. Global Change Biology 21, 1621–1633.

Scarenbroch, B.C., Nix, B., Jacobs, K.A., Bowles, M.L., 2012. Two decades of low-severity prescribed fire increases soil nutrient availability in a Midwestern, USA oak (*Quercus*) forest. Geoderma 183-184, 80–91.

Silvana-Longo, M., Urcelay, C., Nouhra, E., 2011. Long term effects of fire on ectomycorrhizas and soil properties in Nothofagus pumilio forests in Argentina. Forest Ecology and Management 262, 348-354.

Simard, D.G., Fyles, J.W., Paré, D., Nguyen, T., 2001. Impacts of clearcut harvesting and wildfire on soil nutrient status in the Quebec boreal foret. Can J Soil Sci 81, 229-237.

Slaughter, K.W., Grigal, D.F., Ohmann, L.F., 1998. Carbon storage in southern boreal forests following fire. Scand. J. For. Res. 13, 119-127.

Snyman, H.A., 2003. Short-term response of rangeland following an unplanned fire in terms of soil characteristics in a semi-arid climate of South Africa. J. Arid Environ. 55, 160–180.

Soil Survey Staff., 2014. Keys to soil taxonomy. 12th ed. USDA-Natural Resources Conservation Service, Washington, DC.

Soto, B., Benito, E., Diaz-Fierros, F., 1991. Heat-induced degradation process in forest soils. Int. J. Wildland Fire 1, 147–152.

Tempel, D.J., Gutierrez, R.J., Whitmore, S.A., Reetz, M.J., Stoelting, R.E., Berigan, W.J., Seamans, M.E., Peery, M.Z., 2014. Effects of forest management on California Spotted Owls: implications for reducing wildfire risk in fire-prone areas. Ecol. Appl. 24, 2089–2106.

Tessler, N., Sapir, Y., Wittenberg, L., Greenbaum, N., 2016. Recovery of Mediterranean vegetation after recurrent fires: Insight from the 2010 forest fire on Mount Carmel, Israel. Land Degrad. Develop. 27, 1424–1431.

Tessler, N., Wittenberg, L., Malkinson, D., Greenbaum, N., 2008. Fire effects and short-term changes in soil water repellency – Mt. Carmel, Israel. Catena 74, 185–191.

Thomaz, E.L., Antoneli, V., Doerr, S.H., 2014. Effects of fire on the physicochemical properties of a soil in a slash-and-burn agriculture. Catena 122, 209–215.

Turco, M., Llasat, M.C., Hardenberg, J.V., Provenzalle, A., 2014. Climate change impacts on wildfires in a Mediterranean environment. Climatic change 125, 369–380.

Turco, M., Levin, N., Tessler, N., Saaroni, H., 2017. Recent changes and relations among drought, vegetation and wildfires in the Easthern Mediterranean: The case of Israel. Glob. Planet. Chang. https://doi.org/10.1016/j.gloplacha.2016.09.002

Turrion, M.B., Lafuente, F., Aroca, M.J., López, O., Mulas, R., Ruiperez, C., 2010. Characterization of soil phosphorus in a fire-affected forest *Cambisol* by chemical extractions and ^{31}P-NMR spectroscopy analysis. Sciente of the Total Environment 408, 3342-3348.

Úbeda, X., 1998. Efectes de les diferents intensitats de foc, durant els incendis forestals, en els paràmetres físics i químics del sòl i en l'increment de l'escolament i de l'erosió. Tesi Doctoral. Universitat de Barcelona. 231 pp

Úbeda, X., 2001. Influencia de la intensidad de quemado sobre el suelo. Edafología 8, 41-48.

Úbeda, X., Bernia, S., Simelton, E., 2005. The long-term effects on soil properties from a forest fire of varying intensity in a Mediterranean environment. In: García, C., Batalla, R. (Eds), Catchment dynamic and river processes. Developments in Earth Surface Processes (7), pp. 87-102.

Úbeda, X., Outeiro, L.R., Sala, M., 2006. Vegetation regrowth after a differential severity forest fire in a Mediterranean environment, Northeast Spain. Land Degradation and Development 17, 429-440.

Úbeda, X., Pereira, P., Martin, D.A. (2009) Fire temperature effects on total carbon, total nitrogen, C/N and release of water soluble phosphorous of litter from two Quercus suber trees located in different plots on the Iberian Peninsula, International Meeting of Fire Effects on Soil Properties 2nd Edition**,** 11-15 February, 2009, Marmaris, Turkey.

Vega, J.A., Fernandez, C., Fonturbel, T., Gonzalez-Prieto, S., Jimenez, E., 2014. Testing the effects of straw mulching and herb seeding on soil erosion after fire in a gorse shrubland. Geoderma 223-225, 79–87.

Verma, S., Jayakumar, S., 2012. Impact of forest fire on physical, chemical and biological of soil: a review. Proceedings of the International Academy of Ecology and Environmental Science 2 (3): 168-176.

Vose, J.M., Swank, W.T., Clinton, B.D., Knoepp, J.D., Swift, L.W., 1999. Using stand replacement fires to restore southern Appalachian pine-hardwood ecosystems: effects on mass, carbon, and nutrient pools. Forest Ecology and Management 114, 215-226.

Vourlitis, G.L., Giganavicius, J., Gordon, N., Bloomer, K., Grant, T., Hentz, C., 2017. Hydroseeding increases ecosystem nitrogen retention but inhibits natural vegetation regeneration after two years of chaparral post-fire recovery. Ecol. Eng., 102, 46–57.

Wittenberg, L., Malkinson, D., Beeri, O., Halutzy, A., Tessler, N., 2008. Spatial and temporal patterns of vegetation recovery following sequences of forest fires in a Mediterranean landscape, Mt. Carmel Israel. Catena 71, 76–83.

Xue, L., Li, Q., Chen, H., 2014. Effects of a wildfire on selected physical, chemical and biochemical soil properties in a *Pinus massoniana* Forest in South China. Forests, 5, 2947–2966.

Yermakov, Z., Rothstein, D.E., 2006. Changes in soil carbon and nitrogen cycling along 72-year wildfire chronosequence in Michigan jack pine forests. Oecologia 149, 690-700.

CHAPTER 6

CHAPTER 6. LONG-TERM FOREST MANAGEMENT AFTER WILDFIRE (CATALONIA, NE IBERIAN PENINSULA)

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ORIGINAL PAPER

Long-term forest management after wildfire (Catalonia, NE Iberian Peninsula)

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Abstract: Studies of post-fire soil status in Mediterranean ecosystems are common; however, few have examined the effects of long-term forest management after a wildfire on physicochemical soil properties. Here, we analyzed differences in soil properties attributable to long-term post-fire management and assessed the sustainability of these management practices in relation to the soil properties. The study area is located in Ódena in the northeast region of the Iberian Peninsula consisted of the control forest (burned more than 30 years ago), low density forest (LD; burned in a wildfire in 1986 and managed in 2005) and high density forest (HD; burned in a wildfire in 1986 and no managed). For soils from each plot, we measured soil water repellency, aggregate stability, total nitrogen (TN), soil organic matter (SOM), inorganic carbon (IC), pH, electrical conductivity, extractable calcium, magnesium, sodium, potassium (K), phosphorus, aluminum (Al), manganese (Mn), iron (Fe), zinc, copper, boron, chrome, silicon and sulfur and calculated the ratios of C/N, $Ca+Mg/(Na+K)^{1/2}$, Ca/Al and Ca/Mg. Significant differences were found in TN, IC, SOM, pH, K, Al, Mn, Fe and C/N ratio ($p < 0.05$). All soil properties were found to have largely recovered their pre-fire values. Soils were affected by the post-fire management practices implemented 20 years after the fire, as reflected in their respective physicochemical properties, so that soil properties at the control and LD sites are more similar today than those at the control and HD sites. Thus, sustainable forest

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management can overcome soil degradation in areas affected by wildfire in the medium- and long-term by improving soil properties.

Keywords: soil chemical properties; aggregate stability; post-fire management; wildfire risk; vegetal density.

1. Introduction

Wildfires are natural phenomena in Mediterranean ecosystems (Gill 1975; Bento-Gonçalves et al. 2012). In the Mediterranean region of Catalonia, socioeconomic changes that characterized the end of the 20th century produced forest structures of great vegetation density, which clearly led to an increase wildfire severity (Vélez 2000). Thus, while wildfires usually have short- to medium-term impacts on soil properties, when these burn episodes are especially severe the effects can last much longer. Soil chemical properties, determined by the consumption and depletion of plant nutrients, are closely related to forest characteristics such as vegetation density (Francos et al. 2018a) and constitute a critical relationship for determining post-fire management practices in affected areas (Rezaei and Gilkes, 2005).

The effects of short-term post-fire management have been widely studied (e.g., García-Orenes et al. 2017; Francos et al. 2018b; Taboada et al. 2018), but few have focused on the long-term effects of fire on soil properties. Studies such as that by Pereira et al. (2018) are essential for summarizing and understanding the effect of forest management in fire affected areas. In addition, Francos et al. (2018a) analyzed soil chemical properties 18 years after a wildfire, Ojima et al. (1994) studied the effect of fire on N cycling over a 50-year period, and Johnson et al. (2012) assessed the effects of two fires (1960 and 1981) on soil chemistry and vegetation. In a limited number of cases, most notably LeDuc and Rothstein (2010), Muñoz-Rojas et al. (2016) and Yermakov and Rothstein (2006), the authors also carried out a chronosequence of soil status, although in the majority of studies, only long-term analyses of soil samples were done. Here, in common with DeLuca et al. (2006) and McNamara et al. (2015), we compare the properties of a soil long after a fire with those of an unburned area, which we use as a reference for pre-fire values (control), and focus specifically on the role of vegetation density. In this respect, few studies have analyzed the effects of plant densities on soil properties after fire over the long term, the most notable exceptions being Holz et al. (2000), who studied the long-term effects of sewage sludge amendments on barley yield, and Kaye et al. (2010), who studied long-term soil status following differential tree and shrub regeneration in an area affected by wildfire.

Long-term post-fire management can lead to changes in plant density that in turn affect soil properties. For example, soil water repellency (SWR) can be affected for periods that range from a few months to years, depending on fire severity (Dyrness 1976). Indeed, the persistence of SWR depends more on fire severity than on the length of time since burning (Huffman et al. 2001). However, others have shown that SWR reaches pre-fire values roughly 1 year after the fire, regardless of the severity and no longer differs between burned and unburned areas (MacDonald and Huffman 2004). Likewise, there is no clear pattern for the long-term effects of fire on a soil's aggregate stability (AS) (Mataix-Solera et al. 2011). Soil C storage and, consequently, soil organic matter (SOM) and inorganic C (IC) levels varied with plant density (Kaye et al. 2010). The time since the last wildfire (Johnson et al. 2012) and vegetation density (Zhang et al. 2010) also influence soil N, pH, electrical conductivity (EC) and levels of major nutrients. A high C/N ratio can impede plant growth due to the close relationship with SOM and N availability (Jiang et al. 2016). However, we lack information regarding the effects of variations in plant density on the minor elements in the soil and their ratios.

Here we determined the soil status in areas subject to different long-term post-fire management practices after a wildfire event. The objectives were (1) to analyze differences in soil properties associated with post-fire management practices; (2) study how plant density affects soil properties; and (3) identify long-term sustainable post-fire management practices as revealed by soil properties.

2. Materials and methods

2.1. Study site

The study site is located in Ódena in the northeastern region of the Iberian Peninsula (Figure 1) was burned by a forest fire in 1986. The predominant vegetation in the study area is *Pinus halepensis* Miller, *Pinus nigra* Arnold and *Quercus ilex* L. with an understory vegetation of *Pistacea lentiscus* L., *Genista scopius* L. and *Hedera helix* L. The substrate consists of sediments from Paleocene shale (Panareda-Clopés and Nuet-Badia, 1993). The soil is classified as Fluventic Haploxerept (Soil Survey Staff 2014). The site has a typical Mediterranean climate, with an annual temperature of 14.2°C and a mean annual rainfall between 500 and 600 mm.

2.2. Experimental design and sampling

Three 1-ha plots were set up and sampled in October 2015, almost 30 years after the fire: a reference plot (control) that was not affected by the 1986 wildfire or subjected to any management; a low density (LD) plot burned in the 1986 wildfire and subjected to management in 2005; and a high density (HD) burned in the 1986 wildfire but not subjected to any management. The management in 2005 (almost 20 years after the wildfire) at the LD site involved a clear-cutting operation, which left a density of 1,000 trees/ha and the cut vegetation on the soil. At each site, we collected nine topsoil samples (0–5 cm depth), for a total of 27 samples for the study area. The three sites occupied areas with a similar soil type, vegetation composition and topographical characteristics (slope < 3% with a northeast aspect). Wildfire severity in 1986 was classified as high according to Tarrant (1956) and Úbeda et al. (2006), given that 100% of tree crowns were burned.

2.3. Laboratory methods

The analysis of the physicochemical properties of soil are described in great detail in Francos et al. (2016a) and Francos et al. (2018b). SWR was determined using the water drop penetration time (WDPT) test (Wessel 1988). AS was analyzed using the 10-drop impact (TDI) method (Low 1954). TN of pulverized soil was determined using a Flash EA 112 Series (Thermo-Fisher Scientific, Milan) and Eafer 300 software (Thermo-Fisher Scientific, Milan) (Pereira et al. 2012). SOM and IC were determined using the loss-on-ignition (LOI) method described by Heiri et al. (2001). Soil pH and EC were determined by deionized water extraction [1:2.5]. Extractable elements (Ca, Mg, Na and K) were determined by ammonium acetate extraction [1:20], in line with the method described by Knudsen et al. (1986) Available P was analyzed following the Olsen Gray method (Olsen et al. 1954). Soil Al, Mn, Fe, Zn, Cu, B, Cr, Si, and S were determined using ammonium acetate extraction [1:20]. Extractable cations were expressed as mg/kg of soil. The soil C/N ratio was calculated with organic C and TN. The ratio (extractable Na+extractable K)/(extractable Ca+extractable Mg)$^{1/2}$ (SPAR) was obtained using the method of Sarah (2004). We also calculated the ratio Ca/Al and Ca/Mg.

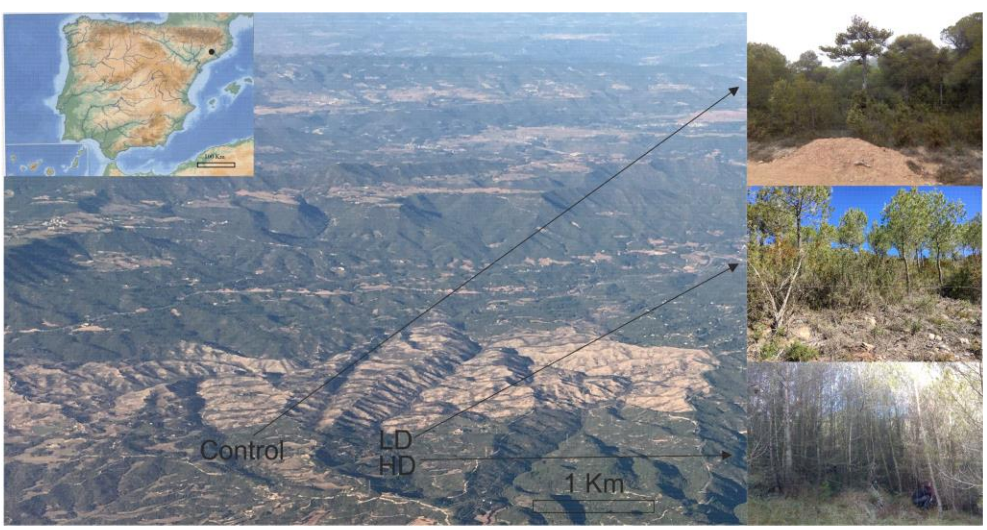

Fig. 1 Location of study area.

2.4. Statistical analysis

Data normality and homogeneity were assessed using the Shapiro-Wilk test and Levene's test. We applied a one-way ANOVA test in those cases with a Gaussian distribution and homogeneity of variances. For those that did not satisfy normality and homogeneity requirements, we applied the nonparametric Kruskal-Wallis ANOVA test. If significant differences were identified at $p < 0.05$, a Tukey post hoc test was applied to identify differences within treatments. A redundancy analysis (RDA) was carried out to identify the extent to which the variation in one set of variables accounts for the variation in another. The soil properties used in the RDA were SWR, AS, TN, SOM, IC, pH, EC, Ca, Mg, Na, K, P, Al, Mn, Fe, Zn, Cu, B, Cr, Si, S, C/N, SPAR, Ca/Al and Ca/Mg. Statistical analyses were carried out using SPSS 23.0 (IBM, Armonk, NY, USA) and CANOCO for Windows 4.3 (Microcomputer Power, Ithaca, NY, USA).

3. Results

3.1. Soil physicochemical properties

No significant differences in SWR and AS were recorded between the three sites (control, HD and LD). Soil TN was significantly higher at the control site than at the HD site. SOM was significantly higher at the control than at the LD and HD sites. IC levels were significantly higher at the LD site than at the Control and HD sites. Soil pH was significantly higher at the LD than at the HD site and significantly higher at the HD than at the control site. There were no significant differences in EC between the three sites (Table 1).

3.2. Major and minor soil elements

The three plots did not differ in Ca, Mg, Na and available P levels. Soil K was significantly higher at the LD than at the HD plot. Soil Al and Fe were significantly higher at the control plot than at the HD. Soil Mn was significantly higher at the HD plot than at the control and LD plots. Soil Zn, Si and S did not differ among the three sites (Table 2).

Table 1. Descriptive statistics for physicochemical characteristics in soils from three plots in northeastern Iberian Peninsula.

Soil property	Study site	Mean	SD	Significance level
Soil water repellency (sec.)	Control	3.80	4.48	n.s.[a]

	LD	3.08	4.28	
	HD	2.20	4.46	
Aggregate stability (%)	Control	95.64	2.24	n.s.[a]
	LD	93.94	2.78	
	HD	95.85	2.01	
Total nitrogen (%)	Control	0.34 a	0.07	*[a]
	LD	0.30ab	0.06	
	HD	0.25b	0.06	
Inorganic carbon (%)	Control	4.89b	0.31	***[b]
	LD	5.66a	0.75	
	HD	4.61b	0.28	
Soil organic matter (%)	Control	6.56a	1.10	***[a]
	LD	5.23b	0.90	
	HD	4.15b	0.77	
pH	Control	7.73c	0.13	***[a]
	LD	8.18a	0.13	
	HD	7.89b	0.11	
EC (µS/cm)	Control	215.33	100.69	n.s.[a]
	LD	198.78	35.50	
	HD	148.44	38.51	

Notes: LD = low density, HD = high density. Different letters indicate significant differences at $p < 0.05$, $N = 9$; *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. [a]One-way ANOVA, [b]Kruskal-Wallis test.

Table 2. Descriptive statistics of major and minor elements characteristics in soils from three plots in northeastern Iberian Peninsula.

Soil property	Study site	Mean	SD	Significance level
Extractable Ca (mg/kg)	Control	19450	1609.10	n.s.[a]
	LD	19593	808.96	
	HD	18613	1336.75	
Extractable Mg (mg/kg)	Control	1136	279.49	n.s.[a]
	LD	1137	232.20	
	HD	1093	317.25	
Extractable Na (mg/kg)	Control	237	96.06	n.s.[a]
	LD	237	51.24	
	HD	288	84.68	
Extractable K (mg/kg)	Control	289 ab	46.11	*[a]

	LD	337a	59.82	
	HD	277b	29.07	
	Control	11.62	10.33	
Available P (mg/kg)	LD	5.40	7.02	n.s.[a]
	HD	15.64	13.89	
	Control	5.56a	5.58	
Extractable Al (mg/kg)	LD	2.43ab	1.64	*[a]
	HD	1.42b	0.93	
	Control	46.12b	12.12	
Extractable Mn (mg/kg)	LD	37.23b	10.37	***[b]
	HD	98.67a	29.03	
	Control	10.54a	7.66	
Extractable Fe (mg/kg)	LD	7.98ab	3.35	**[a]
	HD	2.72b	1.27	
	Control	3.70	1.05	
Extractable Zn (mg/kg)	LD	3.61	1.34	n.s.[a]
	HD	2.84	0.89	
	Control	104.81	103.30	
Extractable Si (mg/kg)	LD	176.88	38.57	n.s.[1]
	HD	116.09	87.58	
	Control	58.28	14.17	
Extractable S (mg/kg)	LD	62.64	11.66	n.s.[1]
	HD	57.18	14.94	

Notes: LD = low density, HD = high density. Different letters indicate significant differences at $p < 0.05$, $N = 9$; *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. [a]One-way ANOVA, [b]Kruskal-Wallis test.

3.3. Soil ratios

C/N ratio presented significantly higher values at the control plot than at the HD. There were no significant differences in SPAR, Ca:Al and Ca:Mg among the three sites (Table 3).

Table 3. Descriptive statistics of soil ratios.

Soil property	Study site	Mean	SD	Significance level
	Control	19.15 a	2.15	
C/N	LD	17.82ab	1.65	*
	HD	16.75b	1.22	

	Control	0.509	0.085	
SPAR	LD	0.556	0.082	n.s.
	HD	0.572	0.085	
	Control	5319.30	2865.97	
Ca:Al	LD	113220.76	304130.25	n.s.
	HD	23437.40	24615.93	
	Control	18.08	4.78	
Ca:Mg	LD	17.88	3.93	n.s.
	HD	18.27	4.97	

Notes: LD = low density, HD = high density. Different letters indicate significant differences in a one-way ANOVA.at $p < 0.05$, $N = 9$; *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$.

3.4. Multivariable analysis

The RDAs for the three sites are shown in Fig. 2. The RDA explained a total of 98.7% of the variance: 49.7% was explained by factor 1 and 49% by factor 2. The variables with the highest explanatory capacity were pH, Mn and SOM; the variables with the lowest explanatory capacity were Ca/Mg, Mg and SWR. In this principal component analysis (PCA), we found a cluster composed of Ca, Zn, EC, Fe, TN, SOM, C/N and Al. Another cluster was formed by IC, K, Si, Ca/Al and S (Fig. 2).

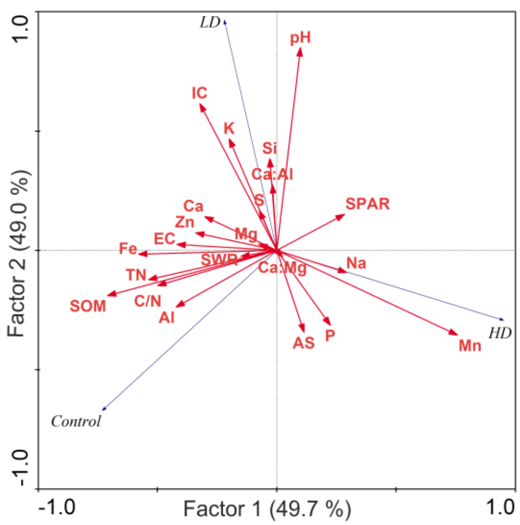

Figure 2. RDA showing the relation between factors 1 and 2. Variables: soil water repellency (SWR), aggregate stability (AS), total nitrogen (TN), inorganic carbon (IC); soil organic matter (SOM); pH; electrical conductivity (EC); extractable Ca, Mg, Na, K, P, Al, Mn, Fe, Zn, Si, S, C/N; (extractable Na+extractable K)/(extractable C+extractable Mg)$^{1/2}$ (SPAR), Ca/Al and Ca/Mg. The three plots are control (unburned in 1986), low density and high density.

4. Discussion

4.1. Soil physicochemical properties

Although the study sites were affected by a wildfire, almost 30 years later no differences are detected in SWR or AS between the three plots. Other studies have recorded lower long-term values in burned soils after a fire (Mataix-Solera et al. 2008), but in our study area we did not observe any changes in these soil properties, with values returning to pre-fire levels just months after the fire (DeBano 2000). Post-fire management has been reported not to affect short-term AS after a wildfire (Francos et al. 2018b). Here, the fact that SWR and AS recovered their pre-fire values and were not affected by differences in forest management or time elapsed since the wildfire can be attributed to their high spatiotemporal variability.
That soil TN was significantly higher at the control site than at the HD site might be attributed to the fact that losses due to volatilization were more than offset by N fixation by vegetation. Given the absence of any differences with the LD site, the lower TN values can be explained by the higher vegetation density; in other words, the post-fire management almost 20 years after the fire at the LD site did not affect soil TN.

Soil organic matter was significantly higher at the control site than at the LD and HD sites, a finding in line with that of Johnson et al. (2012) who determined that, 46 years after a fire, soil carbon had not yet reached pre-fire values. Rezaei and Gilkes (2005) observed similarly high values of SOM at sites with a predominant canopy cover. Their result is in line with our findings here, given that the HD site is abundant in herbaceous plants, shrubs and thin pines, which prevent an increase in SOM and TN. TN and SOM was found by Francos et al. (2018a) to need more time than other variables to return to their pre-fire values; thus, the effects of fire were enduring. The same conclusion was reached by Silvana-Longo et al. (2011), who reported that the effects of wildfire on SOM persist more than 10 years. This result may explain why in the present study values at the control site were significantly higher than at sites burned in 1986. A comparison of the low (LD) and high (HD) density sites reveals the same dynamics as described by Zhang et al. (2010) who compared secondary forest sites, with LD having higher TN and SOM than at HD sites, although the differences did not reach statistical significance. Rapid plant regrowth after fire allows C and N to be fixed and ensures that the decrease in these nutrients is not too great (Johnson and Curtis, 2001). In the present study, due to rapid plant regrowth, long-term soil properties after the fire had values similar to those recorded for the pre-fire forest. However, this similarity was greater at the LD site than at the HD site. This latter difference can be attributed to the absence of forest management and the consequent high plant density at the HD site, which delays the restoration of soil properties to their pre-fire values.

The fact that IC levels were significantly higher at the LD site than at the other two is in line with studies that show the relation of this soil property with fire severity (Pereira et al. 2012). Here, the C (organic and inorganic) balance was higher at the control and LD sites than at the HD. As for pre-fire conditions (Francos et al. 2018c), plant regeneration and Ca and Mg dynamics are important for IC levels (Sainju et al. 2007). Barbera et al. (2010) observed a similar dynamic to the one we describe here, where high plant density and root biomass led to the precipitation of IC compounds at lower layers.

Our findings in the case of soil pH, which was significantly higher at the LD than at HD plot and at the HD plot compared with the control, are in line with the positive correlation found by Zhang et al. (2010) between the number of years since an area had been abandoned and an increase in pH. In our study, the higher pH values at the LD are probably related to the site's lower SOM and TN values in agreement with the study of Francos et al. (2018c) in which high pH values were correlated with high SOM and TN values and low pH was correlated with low SOM and TN. Although Rezaei and Gilkes (2005) obtained significantly high EC values in areas with high vegetation density, EC value did not differ significantly among plots in our study probably because EC values are closely related to levels of certain soil nutrients, as explained below.

4.2. Major and minor soil elements

There were no significant changes in the extractable elements (Ca, Mg, Na and available P) with the exception of K, which was significantly higher at the LD than at the HD site. In our study, the absence of changes in EC is closely related to the similar values presented by Ca, Mg, Na and P among the three sites. In contrast, forest management after a disturbance (such as abandonment or a fire) can produce differences in soil K, with levels being higher in managed areas than in non-managed areas (Zhang et al. 2010), as was the case here with the significantly higher K at the LD than at the HD site. Caon et al. (2010) concluded that, following a wildfire, there is likely to be a long-term decrease in nutrients as a result of volatilization and their transformation to recalcitrant forms, all as a result of post-fire erosion.

Soil Al and Fe were significantly higher at the control than at the HD site. In this case, we cannot attribute the higher values at the control plot to the time since the wildfire because of the absence of any differences between the LD and control sites. As such, these changes can be attributed to the post-fire management and the consequent differences in vegetation density. Soil Fe is especially important for plant fertility, and low Fe levels can hinder recovery (García-Marco and González-Prieto, 2008). Here, the high density resulting from the absence of any management led to a combined soil stress for Al and Fe. Unlike Fe, soil Mn was significantly higher at the HD site. This finding is in line with the negative correlation that Jones (2003) reported between Fe and Mn. García-Marco and González-Prieto (2008) found that higher values of Mn may be related to better seed germination and plant growth. In the present study, the higher values at the HD site seem to reflect the fact that higher plant density produces either extremely high or low values for elements, which in turn are indicative of problematic soils. Soil Zn, Si and S levels were similar at all three sites, and according to Johnson et al. (2012), pre-fire values can be restored in less than two decades, while long-term post-fire management does not significantly change soil properties, as appears to have occurred in the present study. One of the causes of the general increase in K, Al and Fe at the LD site may be the presence of wood residues on the topsoil, which may have triggered a long-term increase in these nutrients (Van Lear 1998). This long-term increase also seems to have an impact on C gains, with the result that the IC, SOM and C/N ratio at the LD site are more similar to those of the control than the HD site.

4.3. Soil ratios

Of the soil ratios, only the C/N ratio differed significantly, being significantly higher at the control than at the HD site, similar to the results obtained by Francos et al. (2018a) where the control area had a higher long-term (though not significant) ratio than in the burned areas after fire. In the present study, the statistically significant differences can be attributed to the significantly higher SOM values obtained at the control than at the HD site. The same conclusion was drawn by Johnson et al. (2012) who determined that almost 46 years after a wildfire, the burned areas had not reached their pre-fire values. One explanation could be the high resistance of organic material to microbial decomposition, so that the changes persisted over the long term after a wildfire (Santin et al. 2016). Francos et al. (2018b) did not find any short-term C/N ratio differences after fire that were attributable to post-fire management. In this study, the differences in the ratio between the LD and HD sites are not significant, even though the values obtained at the LD are more similar to those recorded at the control. No significant differences were recorded in the SPAR, Ca/Al and Ca/Mg between the three sites. Changes in these ratios are confined to just a few months after a wildfire event and, in many cases, appear to be related to erosion, changes in soil structure or strong soil perturbations (Sarah 2005). Accordingly, the absence of differences in these ratios here seems to suggest that the ratios were restored to their pre-fire values and that long-term post-fire management does not alter soil properties. Soil Ca, Mg and Al were lower in the HD plot than in the control, and the difference for Al were statistically significant. The dynamics of these nutrients are similar to that of Ca/Al and Ca/Mg; the impact of fire on the two ratios is short term (Pereira et al. 2017). More studies of the relationship between these ratios and other nutrients are needed to gain a better understanding of their dynamics. To the best of our knowledge, only a few studies (e.g., Pereira et al. 2017) have analyzed the effect of wildfire or post-fire management on soil Ca/Al and Ca/Mg.

4.4. Implications for forest management

The RDA explained a total of 98.7% of the variance: 49.7% was explained by factor 1 and 49% by factor 2. In the PCA, we found one cluster composed of Ca, Zn, EC, Fe, TN, SOM, C/N and Al, grouped near the control. Another cluster was formed by IC, K, Si, Ca/Al and S (Fig. 2) and was grouped close to the LD site. Only one variable, Mn, was close to the HD site. The control and LD sites cluster close to each other because their soil properties are more similar than they are with those at the HD site. The control and the LD sites explained a greater number of properties and a higher variance than the HD site.

Post-fire management may have negative effects on soil properties (Baldini et al. 2007; Gómez-Rey et al. 2013). Here, as the post-fire management was carried out 20 years after the wildfire, the soil properties have had 10 years to improve. Indeed, the absence of management in these intervening years is one of the causes of higher vegetation density and consequent soil degradation. Post-fire management actions can have a marked impact on soil properties depending on the particular action, and postponing management for more than a year (Ginzburg and Steinberger, 2012) or for longer after wildfire (Francos et al. 2018a) has been recommended. Despite this, Mediterranean ecosystems are well adapted to fire, and in the long-term, their soil properties recover pre-fire values (López-Poma et al. 2014). In the long-term, forests usually regain a high vegetation density (as here, even with a management action), and the best time to carry out post-fire management may be when the direct and indirect effects of wildfire have disappeared (Silvana-Longo et al. 2011).

Post-fire management treatments should not be carried out immediately after the fire to protect the seed bank (Madrigal et al. 2011), but are best implemented in the medium to long-term (Francos et al. 2016b). This strategy also lessen fire risk by reducing the accumulation of fuel and disrupting the fuel continuity, while at the same time reducing the overall costs of forest management (Francos et al. 2016b). Almost 30 years after the fire, despite differences between the treated and untreated sites, there were no signs of post-fire erosion or soil degradation. This absence might be attributable to the rapid regrowth of vegetation after wildfire that characterizes Mediterranean ecosystems and protects soils (Cerdà and Doerr, 2005). Forest management actions are necessary (and flagged as a priority at the HD site) to reduce the density of small trees and consequent fire risk (van Mantgem 2011). The fact that burned areas over the long-term after a fire have been shown to recover their pre-fire nutrient values is one of the best examples for promoting post-fire management, as for the LD site in our study area (Bento-Gonçalves et al. 2012). Post-fire forest management of this kind is not detrimental to the soil properties; on the contrary, the managed areas had values more similar to pre-fire values than in the nonmanaged control site. In addition, the managed areas present a lower fire risk due to the non-accumulation of forest fuel and their lower vegetation density. The reduced plant density at the LD site in 2005 benefitted some soil properties and reduced root competition for water and nutrients (Castro et al. 2011). In the same vein, Francos et al. (2018c) proposed forest management to prevent the outbreak of new wildfires, being able to maintain the effect of these actions for 10 years, the period of time that salvage logging remains effective (Donato et al. 2006, 2015). According to Francos et al. (2016b), 18 years after a fire, the vertical and horizontal accumulation of fuel represents a high fire risk. Sustainable forest management strategies are necessary if we consider future climate change scenarios and associated changes in fire activity (Girardin et al. 2013) using forest management tools to reduce fuel loads and restore historical fire conditions (Hunter et al. 2011).

5. Conclusions

Our study on soil properties at two sites subjected to different post-fire management actions revealed that two qualities, SWR and AS, were notably unaffected. The absence of differences in these and other soil properties between the control and the two burned sites (LD and HD) allows us to conclude that the soils have completely recovered. More specifically, the LD and control sites have very similar soil properties, and are more similar than are the soils of the control and HD sites, on the one hand, and those of the HD and LD sites, on the other. The differences that emerged, however, between the low (LD) and high density (HD) sites can be directly attributed to the forest treatment in 2005 and the consequent differences in plant density. Significant differences were recorded between the LD and HD sites in the following soil properties: IC, pH, K and Mn, while non significant differences were recorded for TN, P, Al, Fe and C/N. All in all, our findings point to the positive effects of a management action conducted almost 20 years after a wildfire and the more harmful (though not dramatically so) effects of high plant density at the HD site, which reveal problems in the recovery of certain soil properties. Indeed, various authors advocate forest management to keep fire risk low and promote better soil properties. Further studies, however, are needed to establish when the differences between the soil properties of the LD and HD sites become significant.

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References

Baldini S, Certini G, Di Fluvio F, Giovannini G, Marchi E (2007) Environmental impact of postfire logging in a Maritime pine stands in Tuscany (Central Italy). In Proceedings of the "4 International Wildland fire Conference" 13-17 May, Seville, Spain.

Barbera V, Poma I, Gristina L, Novara A, Egli M (2010) Long-term cropping systems and tillage management effects on soil organic carbon stock and steady state level of C sequestration rates in a semiarid environment. Land Degrad Dev 23: 82–91.

Bento-Gonçalves A, Vieira A, Úbeda X, Martin D (2012) Fire and soils: key concepts and recent advances. Geoderma 191: 3–13.

Castro J, Allen CD, Molina-Morales M, Marañon-Jiménez S, Sánchez-Miranda A, Zamora R (2011) Salvage logging versus the use of burnt Wood as nurse object to promote post-fire tree seedling establishment. Restor Ecol 19 (4): 537–544.

Caon L, Vallejo VR, Coen RJ, Geissen V (2014) Effects of wildfire on soil nutrients in Mediterranean ecosystem. Earth-Sci Rev 139: 47–58.

Cerdà A, Doerr SH (2005) Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. Int J Wildland Fire 14: 423–427.

DeBano LF (2000) The role of fire and soil heating on water repellency in wildland environments: a review. J Hydrol 231-232: 195-206.

DeLuca TH, MacKenzie MD, Gundale MJ, Holben WE (2006) Wildfire-Produced charcoal directly influences nitrogen cycling in Ponderosa Pine forests. Soil Sci Soc Am J 70: 448–453.

Donato DC, Fontaine JB, Campbell JL, Robinson WD, Kauffman JB, Law BE (2006) Post-wildfire logging hinders regeneration and increased fire risk. Science 311 (4759): 352.

Donato DC, Fontaine JB, Campbell JL (2015) Ten years after the Biscuit Fire: Evaluating vegetation succession and post-fire management effects. Joint Fire Science Program 11-1-1-4.

Dyrness CT (1976) Effects of wildfire on soil wettability in the high Cascades of Oregon. USDA Forest Service Research Paper PNW-202, 18p.

Francos M, Pereira P, Alcañiz M, Mataix-Solera J, Úbeda X (2016a) Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain). Sci Total Environ 572: 1353-1362.

Francos M, Pereira P, Alcañiz M, Úbeda X (2018b) Post-wildfire management effects on short-term evolution of soil properties (Catalonia, Spain, SW-Europe). Sci Total Environ 633: 285-292.

Francos M, Úbeda X, Pereira P, Alcañiz M (2018a) Long-term impact of wildfire on soils exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula). Sci Total Environ 615: 664-671.

Francos M, Pereira P, Mataix-Solera J, Arcenegui V, Alcañiz M, Úbeda X (2018c) How clear-cutting affects fire severity and soil properties in a Mediterranean ecosystem. J Environ Manage 206: 625-632.

Francos M, Úbeda X, Tort J, Panareda JM, Cerdà A (2016b) The role of forest fire severity on vegetation recovery after 18 years. Implications for forest management of Quercus suber L. in Iberian Peninsula. Global Planet Change 145: 11-16.

García-Marcos S, González-Prieto S (2008) Short- and medium-term effects of fire and the fire-fighting chemicals on soil micronutrient availability. Sci Total Environ 407: 297-303.

García-Orenes F, Arcenegui V, Cherenková K, Mataix-Solera J, Moltó J, Jara-Navarro AB, Torres MP (2017) Effects of salvage logging on soil properties and vegetation recovery in a fire-affected Mediterranean forest: A two year monitoring research. Sci Total Environ 586: 1057-1065.

Gill AM (1975) Fire and the Australian flora: a review. Aust forestry 38 (1): 4-25.

Ginzburg O, Steinberger Y (2012) Salvage logging versus natural regeneration post-fire practices in a forest: Soil chemical and microbial aspects. Open Journal of Ecology 2 (1): 29-37.

Girardin MP, Ali AA, Carcaillet C, Gautgier S, Hély C, Le Goff H, Terrier A, Bergeron Y (2013) Fire in managed forest of eastern Canada: Risk and options. For Ecol Manage 294: 238-249.

Gómez-Rey MX, Couto-Vazquez A, García-Marcos S, González-Prieto SJ (2013) Impact of fire and post-fire management techniques in soil chemical properties. Geoderma 195-196: 155-164.

Heiri O, Lotter AF, Lemcke G (2001) Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J Paleolimnol 5: 101-110.

Holz SC, Ingelmo F, Canet R (2000) Long-term effects of the application of sewage sludge and vegetal cover on some physical and physicochemical properties of a degraded arid soil. Agrochimica 44: 132-139.

Huffman EL, MacDonald LH, Stendnik JD (2001) Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrol Proc 15: 2877-2892.

Hunter ME, Iniguer JM, Lentile LB (2011) Short- and long-term effect on fuels, forest structure, and wildfire potential from prescribed fire and resource benefit fire in Southern Forests USA. Fire Ecol 7 (3): 108-120.

Jiang X, Denef K, Stewart CE, Cotrufo F (2016) Controls and dynamics of biochar decomposition and soil microbial abundance, composition, and carbon use efficiency during long-term biochar amended soil incubations. Biol Fertil Soils 52: 1-14.

Johnson DW, Curtis PS (2001) Effects of forest management on soil C and N storage: meta analysis. For Ecol Manag 140: 227-238.

Johnson DW, Walker RF, McNulty M, Rau BM, Miller WW (2012) The Long-Term Effects of Wildfire and Post-Fires Vegetation on Sierra Nevada Forest Soils. Forests 3: 398-416.

Jones JB (2003) Agronomic Handbook: management of crops, soils and their fertility. Boca Raton. CRC Press, 450 p.

Kaye JP, Romanyà J, Vallejo VR (2010) Plant and soil carbon accumulation following fire in Mediterranean woodlands in Spain. Oecologia 164: 533-543.

Knudsen D, Petersen GA, Pratt PF (1986) Lithium, sodium and potassium. In: Soil Science Society of America (ed) Methods of soil analysis. Vol. 2. ASA-SSSA, Madison, WI, pp. 225-246.

LeDuc SD, Rothstein DE (2010) Plant-available organic and mineral nitrogen shift in dominance with forest stand age. Ecology 91: 708-720.

López-Poma R, Orr BJ, Bautista S (2014) Successional stage after land abandonment modulates fire severity and post-fire recovery in a Mediterranean mountain landscape. Int J Wildland Fire 23: 1005-1015.

Low AJ (1954) The study of soil structure in the field and in the laboratory. J Soil Sci 5: 57-74.

MacDonald LH, Huffman EL (2004) Post-fire soil Water Repellency: Persistence and Soil Moisture Thresholds. Soil Sci Soc Am J 68: 1729-1734.

Madrigal J, Hernando C, Guijarro M (2011) El papel de la regeneración natural en la restauración tras grandes incendios forestales: el caso del pino negral. Boletín del CIDEU10: 5-22.

Mataix-Solera J, Cerdà A, Arcenegui V, Jordán A, Zavala LM (2011) Fire effects on soil aggregation: A review. Earth-Sci Rev 109: 44-60.

Mataix-Solera J, Navarro MA, Zornoza R, Guerrero C, Gómez I, García-Orenes F, Bárcenas GM (2008) Fire effects on soil organic carbon in turn affecting physical, chemical and biochemical soil properties. Geophysical Research Abstracts 10: 01280.

McNamara NP, Gregg R, Oakley S, Stott A, Tanvir Rahman Md, Collin Murrell C, Wardle DA, Bardgett RD, Ostle NJ (2015) Soil methane sink capacity response to long-term wildfire chronosequence in Northern Sweden. PlosOne 10: e0129892.

Muñoz-Rojas M, Erickson TE, Martini D, Dixon KW, Merritt DJ (2016) Soil physicochemical and microbiological indicators of short, medium and long term post-fire recovery in semi-arid ecosystems. Ecol Indic 63: 14-22.

Olsen SR, Cole CV, Frank SW, Dean LA (1954) Estimation of Available Phosphorus in soils by Extraction with Sodium Bicarbonate. USDA Circular No. 939. US Government Printing Office: Washington, DC.

Ojima DS, Schimel DS, Parton WJ, Owensby CE (1994) Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24: 67-84.

Panareda-Clopés JM, Nuet-Badia J (1993) Tipología y cartografía corológica de las plantas vasculares de Montserrat (Cordillera Prelitoral Catalana). Rev Geogr 27: 33-58.

Pereira P, Cerdà A, Martin D, Úbeda X, Depellegrin D, Novara A, Martínez-Murillo JF, Brevik EC, Menshov O, Rodrigo-Comino J, Miesel J (2017) Short-term low-severity spring grassland fire impacts on soil extractable elements and soil ratios in Lithuania. Sci Total Environ 578: 469-475.

Pereira P, Francos M, Brevik EC, Úbeda X, Bogunovic I (2018) Post-fire management. Current Opinion in Environmental Science & Health 5: 26-32.

Pereira P, Úbeda X, Martin D (2012). Fire severity effects on ash chemical composition and water-extractable elements. Geoderma 191, 105–114.

Rezaei SA, Gilkes RJ (2005) The effects of landscape attributes and plant community on soil chemical properties in rangelands. Geoderma 125: 167-176.

Sainju UM, Caesar-Tonthat T, Lenssen AW, Evans RG, Kolberg R (2007) Long-Term tillage and Cropping Sequence Effects on Dryland Residue and Soil Carbon Fractions. Soil Sci Soc Am J 71: 1730-1739.

Santin C, Doerr S, Merino A, Bryant R, Loader NJ (2016) Forest floor chemical transformations in a boreal forest fire and their correlations with temperature heating and duration. Geoderma 264: 71–80.

Sarah P (2004) Soil sodium and potassium adsorption ratio along a Mediterranean-arid transect. J Arid Environ 59: 731-741.

Sarah P (2005) Soil aggregation response to a long- and short-term differences in the rainfall amount under arid and Mediterranean climate conditions. Geomorphology 70: 1-11.

Silvana-Longo M, Urcelay C, Nouhra E (2011) Long term effects of fire on ectomycorrhizas and soil properties in Nothofagus pumilio forests in Argentina. For Ecol Manag 262: 348-354.

Soil Survey Staff (2014) Keys to soil taxonomy. 12th ed. USDA-Natural Resources Conservation Service, Washington, DC.

Taboada A, Fernández-García V, Marcos E, Calvo L (2018) Interactions between large high-severity fires and salvage logging on a short-return interval reduce the growth of fire-prone serotinous forests. For Ecol Manag 414: 54-63.

Tarrant RF (1956) Effects of slash burning on some soils of the Douglas-fir Region 1. Soil Sci Soc Am J 20: 408-411.

Úbeda X, Outeiro LR, Sala M (2006) Vegetation regrowth after a differential severity forest fire in a Mediterranean environment, Northeast Spain. Land Degrad Dev 17: 429-440.

Van Lear D (1998) Effects of harvesting intensity on forest productivity and soil carbon storage: Clemson site. Progress Report to NCASI.

van Mantgem PJ, Stephenson NL, Knapp E, Keeley JE (2011) Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada, California. For Ecol Manag 261: 989-994.

Vélez R (2000) Los incendios forestales en la cuenca mediterranea. Introducción. In: Vélez R (ed) La defensa contra los incendios forestales. Fundamentos y experiencias. McGraw-Hill 3-1.

Wessel AT (1988) On using the effective contact angle and the water drop penetration time for classification of water repellency in dune soils. Earth Surf Proc Land 13 (6): 555-561.

Yermakov Z, Rothstein DE (2006) Changes in soil carbon and nitrogen cycling along 72-year wildfire chronosequence in Michigan jack pine forests. Oecologia 149: 690-700.

Zhang K, Dang H, Tan S, Wang Z, Zhang Q (2010) Vegetation community and soil characteristics of abandoned agricultural land and pine plantation in the Qinling Mountains, China. For Ecol Manag 259, 2036–2047.

RESUMEN DE LOS ARTÍCULOS PRESENTADOS

RESUMEN CAPÍTULO 1

Manejo del suelo después del incendio

El fuego es un elemento natural de los ecosistemas y forma parte de los biomas del mundo tal y como los conocemos hoy en día. El uso del fuego por los seres humanos produce cambios sin precedentes en los ecosistemas. Hoy en día, no se puede entender algunos ecosistemas de la Tierra sin la presencia del fuego, como por ejemplo el Mediterráneo. Las medidas excesivas de supresión del fuego aumentan la flamabilidad de los ecosistemas. El fuego cambia las propiedades del suelo directamente por el calentamiento producido durante la ignición, e indirectamente, por las cenizas que se depositan después del paso del fuego. El grado del impacto dependede diversos factores como las propiedades del suelo, el tipo de vegetación, la topografía, las condiciones meteorológicas ocurridas durante el incendio, la recurrencia del fuego, la intensidad y la severidad del mismo. Normalmente los fuegos de baja severidad (ej. quemas de pastos y quemas prescritas), pueden tener impactos beneficiosos en los suelos como consecuencia de la deposición de las cenizas y el material carbonizado en la superficie del suelo, incrementando temporalmente los nutrientes del suelo. El calentamiento producido por ese tipo de fuegos es bajo y los impactos directos en los suelos son reducidos. Por otro lado, los fuegos de alta severidad tienen impactos perjudiciales en el suelo como consecuencia de las altas temperaturas alcanzadas, consumiendo una importante cantidad de materia orgánica y pueden inducir a la degradación del suelo. Esos efectos adversos son el tipo de impacto que debe ser evitado, y son cada vez más frecuentes e intensos como consecuencia del incremento en la frecuencia de los fuegos de alta severidad. Hay muchos aspectos importantes para entender el grado de impacto de los fuegos en los suelos, como el uso del suelo previo al incendio y el manejo forestal realizado en la zona estudiada. El grado de impacto del fuego en los suelos depende del historial de fuegos, las propiedades de las cenizas, la topografía, la meteorología posterior al incendio, el grado de recuperación de la vegetación y la gestión posterior al incendio. La manera en la que intervenimos en las zonas quemadas determina el grado de degradación del suelo. Prácticas como la tala y extracción de la madera utilizando maquinaria pesada, incrementa la degradación del suelo, mientras la aplicación de mantillo la reduce. A pesar del impacto de los fuegos de alta severidad en las propiedades del suelo, los ecosistemas desarrollan mecanismos de respuesta a las perturbaciones del fuego. Por esta razón debemos preguntarnos si debemos intervenir en esas áreas que son extremadamente vulnerables, especialmente en el periodo inmediato después del fuego. Muchas de las medidas de gestión posterior al incendio son desastrosas para el suelo (ej. tala y extracción), por lo tanto, en muchos de los casos la mejor opción es no intervenir. La “no intervención” es una opción que debe ser considerada seriamente. Esto no significa que algunas áreas no deben necesitar algún tipo de restauración, pero la intervención debe ser llevada a cabo mediante prácticas sostenibles como la aplicación de mantillo. Lo importante a retener es que la gestión posterior al incendio puede reducir o desencadenar la degradación del suelo y esto depende en gran medida de como actuemos después del incendio.

RESUMEN CAPÍTULO 2

Cómo afectan los clareos a la seriedad del fuego y las propiedades del suelo en un ecosistema Mediterráneo

Los ecosistemas Mediterráneos se caracterizan por la frecuente ocurrencia y recurrencia del fuego. Las prácticas de gestión forestal en los ecosistemas Mediterráneos son empleadas frecuentemente para reducir el riesgo y la severidad de los incendios forestales. Sin embargo, estos tratamientos pre-incendio pueden influir en los efectos que tienen los incendios forestales en las propiedades del suelo. El objetivo de este estudio es examinar los efectos de un incendio forestal a corto plazo en las propiedades del suelo en tres sitios con diferente gestión forestal: dos expuestos a prácticas de gestión en diferentes años – 2005 (sitio M05B) y 2015 (sitio M15B) – y uno donde no se realizó ninguna gestión (NMB) y comparar sus propiedades con las encontradas en una parcela (Control) que no fue afectada por el incendio de 2015. Se analizaron diferentes propiedades edáficas entre las que se encuentran: estabilidad de agregados (AS), el contenido de materia orgánica del suelo (SOM), nitrógeno total (TN), ratio carbono/nitrógeno (C/N), carbono inorgánico (IC), pH, conductividad eléctrica (EC), Calcio (Ca), Magnesio (Mg), Sodio (Na), y Potasio (K) extraíbles, carbono de la biomasa microbiana (C_{mic}) y respiración basal edáfica (BSR). En las parcelas gestionadas, la operación de clareo fue realizada cortándose parte de la vegetación que se dejó cubriendo la superficie del suelo. Los valores de AS registrados en el sitio Control fueron significativamente más altos que los registrados en M05B, mientras que los valores de TN y SOM en NMB fueron significativamente más altos que los registrados en M05B. IC fue significativamente mayor en M05B que en las otras parcelas. No hubo diferencias significativas en la relación C/N entre los sitios analizados. El pH del suelo en M05B fue significativamente mayor que el valor registrado en la gráfica de Control. El Ca extraíble fue significativamente mayor en NMB que en M05B y Control, mientras que el Mg extraíble fue significativamente más bajo en M05B que en NMB. El potasio (K) extraíble fue significativamente menor en el Control que en las tres parcelas afectadas por el fuego. C_{mic} fue significativamente mayor en NMB que en el Control. Los valores de BSR, BSR/C y BSR/C_{mic} en los sitios afectados por el fuego fueron significativamente más bajos que los registrados en el Control. No se identificaron diferencias significativas en C_{mic}/C. En general, la comparación de los tratamientos previos al incendio mostró que NMB fue la práctica que tuvo los menores efectos negativos en las propiedades del suelo estudiadas, seguido de M15B y que la severidad del fuego fue más alta en M05B debido a la acumulación de combustible de plantas muertas siendo éste, según las propiedades analizadas, la gestión menos conveniente. Es necesario continuar con este estudio y ver cómo evolucionan las propiedades edáficas a lo largo del tiempo y si esas diferencias se amplian o se disminuyen.

RESUMEN CAPÍTULO 3

Impacto de un evento de lluvia torrencial en las propiedades del suelo después de un incendio forestal en un ecosistema Mediterráneo (Noreste de España)

Los eventos intensos de lluvia después de incendios severos pueden tener impacto en las propiedades del suelo, sobre todo en el entorno mediterráneo donde la confluencia de ambos eventos cada vez está siendo más común. Este estudio busca examinar el impacto inmediato y el efecto después de un año de un evento de lluvia intensa en un bosque mediterráneo afectado por un incendio forestal de alta severidad. El trabajo analiza las siguientes propiedades del suelo: estabilidad de los agregados del suelo, nitrógeno total, carbono total, carbono orgánico e inorgánico, relación C/N, carbonatos, pH, conductividad eléctrica, calcio extraíble, magnesio, sodio, potasio, fósforo disponible y la relación de adsorción de sodio y potasio (SPAR). Muestreamos los suelos en el área quemada antes, inmediatamente después y un año después del evento de lluvia. Los resultados mostraron que el evento de lluvia intensa no tuvo un impacto inmediato en la estabilidad de los agregados del suelo, pero se registró una diferencia significativa un año después. La precipitación intensa no dio lugar a ningún cambio significativo en el nitrógeno total del suelo, el carbono total, el carbono inorgánico, la relación C/N y los carbonatos durante el período de estudio. Las diferencias sólo se registraron en el carbono orgánico del suelo. El contenido de carbono orgánico del suelo fue significativamente mayor después de la lluvia que en los otros muestreos. El evento de lluvia aumentó el pH del suelo, la conductividad eléctrica, los cationes mayoritarios, el fósforo disponible y el SPAR. Un año después del incendio, se observó una disminución significativa en la estabilidad de agregados del suelo que puede atribuirse a los altos niveles de SPAR y la intervención humana, mientras que la reducción de elementos extraíbles puede atribuirse a la lixiviación del suelo y el consumo de la vegetación. En general, el evento de lluvia intensa, otros eventos de lluvia después del incendio y la intervención humana no tuvieron un impacto perjudicial en las propiedades del suelo con toda probabilidad debido a la topografía plana del terreno.

RESUMEN CAPÍTULO 4

Efectos a corto plazo de la gestión post-incendio en las propiedades del suelo (Cataluña, España, SO Europa)

Las prácticas de gestión forestal posteriores al incendio tienen un impacto importante en las propiedades del suelo. Sin embargo, poca investigación se ha llevado a cabo sobre esto. El objetivo de este estudio es examinar el impacto de diferentes prácticas de manejo forestal después de incendios forestales en las propiedades del suelo en un período de 10 meses inmediatamente después de un incendio forestal de alta severidad. Dos meses después del incendio forestal, se diseñaron tres áreas experimentales, cada una con diferente gestión posterior al incendio: Cortar y quitar (CR) donde los troncos quemados se cortaron después del fuego y se sacaron manualmente del área; Sin tratamiento (NT) donde no se llevó a cabo ninguna intervención; y, Cortar y dejar (CL) donde los troncos quemados se cortaron y se dejaron al azar sobre el suelo. En cada tratamiento, recogimos nueve muestras (0-5 cm de profundidad). En total, muestreamos 27 muestras en cada fecha de muestreo, dos y diez meses después del incendio forestal. Las propiedades analizadas fueron estabilidad de agregados (AS), nitrógeno total (TN), materia orgánica del suelo (SOM), carbono inorgánico (IC), relación C/N, pH, conductividad eléctrica (EC), calcio (Ca), magnesio (Mg), sodio (Na) y potasio (K) extraíbles. La relación C/N del suelo fue significativamente mayor en los tratamientos CR y CL 10 meses después del fuego en comparación con 2 meses después. Por otro lado, el pH, el Ca, Mg y K extraíbles fueron significativamente más altos en todos los tratamientos 2 meses después del fuego que 10 meses después. La estabilidad de agregados, TN y SOM fueron significativamente más altos en CR que en CL, 10 meses después del incendio. IC fue significativamente mayor en CL que en NT, también, 10 meses después del incendio. La conductividad eléctrica fue significativamente mayor en los tratamientos de CR y CL 2 meses después del fuego en comparación con 10 meses después. De acuerdo con los resultados, el manejo posterior al fuego de CR y CL no difirió significativamente del escenario NT, lo que demuestra que el manejo basado en la extracción manual de la madera no tiene impactos perjudiciales sobre las propiedades del suelo en comparación con los otros tipos de gestión post-incendio. Este tipo de gestión forestal manual es muy interesante para ser levada a cabo en pequeñas zonas y lugares concretos donde no se quiera favorecer la degradación y erosion del suelo y se pretenda conseguir un determinado paisaje con posibles fines como el turístico.

RESUMEN CAPÍTULO 5

Efectos a largo plazo de un incendio forestal en suelos expuestos a diferentes severidades de fuego. Caso de estudio del Macizo de Cadiretes (NE Península Ibérica)

Los incendios forestales afectan a los ecosistemas dependiendo del régimen de incendios. Se necesitan estudios a largo plazo para comprender el papel ecológico que desempeña el fuego, especialmente en lo que respecta a su impacto en los suelos. El objetivo de este estudio es monitorizar los efectos a largo plazo (18 años) de un incendio forestal en las propiedades del suelo en dos áreas afectadas por los regímenes de baja y alta severidad de fuego. Las propiedades estudiadas fueron nitrógeno total (TN), carbono total (TC), relación C/N, materia orgánica del suelo (SOM) y calcio (Ca), magnesio (Mg), sodio (Na) y potasio (K) extraíbles. El estudio se llevó a cabo en tres fases: corto (inmediatamente después del incendio forestal), medio (siete años después del incendio forestal) y a largo plazo (18 años después del incendio forestal). Los resultados mostraron que en ambos regímenes de incendios TN disminuyó con el tiempo, TC y SOM fueron significativamente más bajos en las parcelas quemadas que en el control a medio y largo plazo. La relación C/N fue significativamente más baja a corto plazo en el área de baja severidad de incendios forestales. El Ca y el Mg extraíbles fueron significativamente mayores en la parcela de control que en las parcelas quemadas a medio plazo. A largo plazo, el Ca y Mg extraíbles fueron significativamente más bajos en el área expuesta a un fuego de alta severidad. No se identificaron diferencias en el caso del Na extraíble entre parcelas en ninguno de los muestreos, mientras que el K extraíble fue significativamente más alto en la parcela expuesta a incendios forestales de baja severidad que en el Control. Es posible que se requieran algunas medidas de restauración después del incendio forestal, especialmente en áreas afectadas por fuegos de alta severidad, para evitar los impactos a largo plazo en los nutrientes esenciales del suelo como TC, SOM, Ca y Mg extraíbles. Este agotamiento de nutrientes a largo plazo se puede atribuir a la eliminación de vegetación, la erosión, la lixiviación y el consumo de vegetación después del incendio. Los suelos claramente necesitan más tiempo para recuperarse de la perturbación de los incendios forestales, especialmente en las áreas afectadas por regímenes de fuego de alta severidad.

RESUMEN CAPÍTULO 6

Gestión forestal a largo plazo después de un incendio (Cataluña, NE Península Ibérica)

Los estudios del estado del suelo después del fuego en los ecosistemas mediterráneos son comunes; sin embargo, pocos han examinado los efectos del manejo forestal a largo plazo después de un incendio forestal en las propiedades físico-químicas del suelo. El objetivo de este estudio es analizar las diferencias en las propiedades del suelo atribuibles a la gestión forestal a largo plazo después del incendio y evaluar la sostenibilidad de estas prácticas de gestión post-incendio en relación con las propiedades del suelo. El área de estudio se encuentra en Ódena, en el NE de la Península Ibérica. Las parcelas analizadas fueron i) Control (bosque quemado hace más de 30 años), LD (bosque de baja densidad quemado en un incendio forestal en 1986 y gestionado en 2005) y HD (bosque de alta densidad quemado en un incendio forestal en 1986 y no gestionado). La gestión llevada a cabo en 2005 implicó una tala selectiva hasta conseguir una densidad de 1000 árboles/ha dejando la vegetación cortada sobre la superficie del suelo. Las siguientes propiedades del suelo fueron examinadas: repelencia al agua del suelo (SWR), estabilidad de agregados (AS), nitrógeno total (TN), materia orgánica del suelo (SOM), carbono inorgánico (IC), pH, conductividad eléctrica (EC), calcio extraíble (Ca), magnesio (Mg), sodio (Na), potasio (K), fósforo (P), aluminio (Al), manganeso (Mn), hierro (Fe), zinc (Zn), cobre (Cu), boro (B), cromo (Cr), silicio (Si) y azufre (S). Los ratios calculados para el suelo fueron la relación carbono/nitrógeno (C/N), la relación SPAR $(Ca + Mg) / (Na + K)^{1/2}$, la relación calcio: aluminio y la relación calcio: magnesio. Se observaron diferencias significativas en TN, IC, SOM, pH, K, Al, Mn, Fe y en la relación C/N ($p < 0{,}05$). Se encontró que todas las propiedades del suelo han recuperado en gran medida sus valores previos al fuego. Los suelos se habían visto afectados por las prácticas de manejo posteriores al incendio implementadas veinte años después del incendio, como se refleja en sus respectivas propiedades físico-químicas. Las prácticas de gestión han producido que las propiedades del suelo registradas en los sitios Control y LD sean más similares hoy en día que las de los sitios Control y HD. El manejo forestal sostenible puede superar la degradación del suelo mediante el manejo de áreas afectadas por incendios forestales a medio y largo plazo, mejorando las propiedades del suelo con estas prácticas de manejo forestal y disminuyendo el riesgo de que ocurran nuevos incendios forestales y éstos sean de alta severidad.

GENERAL CONCLUSIONS

GENERAL CONCLUSIONS

The research carried out in this doctoral thesis seeks to contribute to our knowledge of the effects of fires of different severities, and of forest management practices implemented both pre- and post-wildfire, on soil properties. The studies have focused on the physical, chemical and microbiological properties of soils and via their analysis has sought to answer the specific questions raised at the outset of this work and to address the gaps identified in the literature. The results obtained from each of the separate studies making up this thesis, both in the laboratory and in the field, have allowed the following conclusions to be drawn:

1. Soils can be significantly affected by fire, the extent of this impact and the extent of changes in soil properties depending on fire severity. Low-severity fires can be beneficial to soils, while high-severity fires negatively impact soil properties leading to soil degradation. The degree of this impact depends on many factors, including pre-fire land use (the most visible factor), the site's fire history, the type of ash produced, the topography, the post-fire weather conditions and the degree of vegetation recovery. These factors are interdependent and tend to augment or mitigate the impact of fire. Recent studies have, in addition, begun to examine the impact of human intervention and it is clear that more work is need to determine the effects of this factor. Human interventions and the way that areas affected by high-severity fires are managed are crucial for controlling soil degradation. Here, sustainable measures, most notably mulching, can play an important role in contrast to more invasive techniques, such as salvage logging, that result in considerable soil degradation. In short, post-fire management interventions either trigger or reduce soil degradation in areas affected by high-severity fire.
2. Forest management practices, primarily clear-cutting, have been implemented in many forest areas to reduce the forest fuel load and to curb the risks of wildfire. Clear-cutting and the time lapse between this management intervention and a wildfire have a direct impact on soil properties. Failure to clear cut vegetation from the soil surface impinges on the severity of the fire and can provoke more alterations to soil properties. There is a dearth of studies evaluating pre-fire management practices and the effectiveness of such interventions to avoid detrimental effects on soil properties and determining the lapse time during which such practices are effective. Clear-cutting is recommended as a means of preventing the outbreak of new forest fires, and cut vegetation should not be allowed to accumulate on the soil surface so as to reduce fire severity in potential medium- to long-term episodes of wildfire. More studies like the one reported here are needed to ensure the implementation of appropriate forest management practices and to determine whether differences between treated and untreated sites increase or disappear over time.
3. The post-wildfire environment is extremely fragile and torrential rainfall can further damage soil properties. In this study, one year after the fire, as a consequence of ash and soil erosion, tree removal, vegetation recovery and changes in soil chemical properties, it was observed that:

a) The intense rainfall event period did not have a significant impact on soil aggregate stability in the area affected by high-severity fire. Significant impacts were only observed one year after the fire as a consequence of ash and soil erosion, tree removal, vegetation recovery and changes in soil chemical properties, such as the sodium and potassium adsorption ratio (SPAR).

b) The fire did not have an impact on total nitrogen, total carbon, inorganic carbon, the C/N ratio or carbonate levels. However, particles of charred material, either transported from other areas or which were more resistant to erosion, may have contributed to a significant increase in organic carbon.

c) Extractable cations and available phosphorous increased after the rainfall event as a consequence of a favorable soil pH. SPAR values were also significantly higher after the rainfall event, indicating that they may have contributed to the decrease in soil aggregate stability.

d) One year after the fire, a significant decrease was observed in aggregate stability and extractable elements. This was probably due to the high SPAR levels (in the case of the decrease in soil aggregate stability), human intervention and soil leaching provoked by other rainfall events and vegetation consumption.

Thus, based on the results of the analyses conducted in this study, the intense rainfall event and other post-fire processes (attributable to natural or human causes) did not greatly modify the soil properties as recorded before the intense rainfall episode due, it would appear, to the flat study area.

4. Post-wildfire dynamics can be affected by forest management practices. The results from this study showed that the wildfire had an immediate impact on soil pH and on the major extractable elements, irrespective of the treatment implemented, reducing their levels 10 months after the fire. The inverse was observed in the case of the C/N ratio, but only in cut-and-remove (CR) and cut-and-leave (CL) interventions. In contrast, the effects of post-wildfire management on soil AS, TN, SOM, and IC were only observed 10 months after the fire. AS, TN and SOM were significantly higher after CR than after CL, while the IC values were significantly higher after CL than after no treatment (NT). This could be because of the saw dust generated by the management practices and the more intense foot traffic to which the topsoil may have been exposed. Soil EC was significantly higher after CR than after CL 2 months post-fire. The concentration of salts in the soil was significantly higher following CR and CL treatments 2 months after the wildfire compared to 10 months after. Overall, the differences between the management interventions taken and the no intervention scenario were not relevant, showing that manual forest management practices do not have any negative impacts on soil properties, compared to salvage logging.

5. Depending on soil properties, fire severity, and post-fire weather conditions, the effects of a wildfire may still be noticeable even in the long term. The particular wildfire studied here presented different long-term effects on the soils' chemical properties, indicating that soil nutrients respond differently depending on the amount of time elapsed since the wildfire and on the severity of the wildfire. Levels of extractable

Na did not differ from those recorded in the control plot on any of the sampling dates, while TN and C/N recovered their pre-fire levels. Major cations only recovered in the area affected by low-fire severity, while TC and SOM did not recover in either of the burned areas. The low-severity fire also resulted in a long-term increase in extractable K levels. The lack of recovery presented by the essential soil parameters – including, extractable Ca and Mg (in the area affected by high-severity fire), TC and SOM – show that restoration measures may be needed, especially after high-severity fires. The inability of nutrients to recover their pre-fire levels is clearly a consequence of the disturbances caused by the fire (e.g. vegetation removal, erosion, leaching) and the subsequent loss of vegetation with time, decreasing overall soil nutrient content. Soils need more time to return to their pre-fire levels, especially in areas affected by high severity fire. However, further research is needed to observe when TC and SOM return to pre-fire levels as here this may have been influenced by the change in vegetation cover.

6. Wildfires form a part of the Mediterranean ecosystem and medium- to long-term forest management practices implemented after a fire, aimed at avoiding severe fire episodes and reducing the risk of recurrence, are needed. This last study detected differences in the soil properties at two sites exposed to different post-fire management actions but found that two qualities, SWR and AS, were largely unaffected. The absence of differences in these and other soil properties at the control and burned sites – low density (LD) and high density (HD) allows us to conclude that the soils have been completely recovered. More specifically, the LD and control sites presented very similar soil properties, more similar than those of the control and HD sites, on the one hand, and those of the HD and LD sites, on the other. The differences that emerged, however, between the LD and HD sites can be directly attributed to the forest treatment conducted in 2005 and the consequent differences in plant density. Significant differences were recorded between the LD and HD sites in relation to the following soil properties: IC, pH, K and Mn, while non-significant differences were recorded in the cases of TN, P, Al, Fe and the C/N ratio. All in all, our findings point to the positive effects of a management action conducted almost 20 years after a wildfire and the more harmful (though not dramatically so) effects of high plant density at the HD site, which reveal some problems in the recovery of certain soil properties. Indeed, various authors advocate forest management interventions to keep the fire risk low and to achieve better soil properties. Further studies, however, are needed to establish the point in time at which the differences between the soil properties of the LD and HD sites became significant.

CONCLUSIONES GENERALES

CONCLUSIONES GENERALES

La investigación llevada a cabo en esta Tesis Doctoral representa una contribución importante en el conocimiento de los efectos de los incendios forestales de diferente severidad y el manejo forestal antes y después de un incendio en las propiedades del suelo. Estas propiedades analizadas han sido físicas, químicas y microbiológicas y, a través de su análisis, se ha intentado responder las preguntas planteadas al comienzo de la Tesis Doctoral y las lagunas científicas en las que el autor ha tratado de aportar conocimiento a través de los resultados de su investigación. Los resultados incluidos tanto en el trabajo de laboratorio como en el campo, han permitido alcanzar las siguientes respuestas que se han enmarcado dentro de cada uno de los artículos compilados en la presente Tesis Doctoral y que se resumen a continuación:

1. Los suelos pueden ser afectados de manera importante por el fuego, este impacto depende de la severidad del fuego. Los incendios de baja severidad pueden beneficiar las propiedades del suelo, mientras que los incendios de alta severidad producen impactos negativos en las propiedades del suelo y, en consecuencia, la degradación del suelo. El grado del impacto depende de muchos factores como el uso del suelo antes del incendio (el factor más visible), el historial de incendios del lugar, el tipo de ceniza producida, la topografía, el clima posterior al incendio y el grado de recuperación de la vegetación. Estos factores son interdependientes y pueden aumentar o mitigar el impacto del fuego. La intervención humana y su impacto en las propiedades edáficas es otro factor que en los últimos años se ha estudiado y del que se necesita mayor investigación. La intervención humana y la forma en que manejamos las áreas afectadas por incendios de alta severidad es crucial para el aumento o la disminución de la degradación del suelo. Las medidas sostenibles, como el “mulch”, son importantes, al contrario que las técnicas de extracción mecánica de madera que causan una importante degradación del suelo. En general, las opciones de manejo posterior al incendio pueden desencadenar o reducir la degradación del suelo en áreas afectadas por incendios de alta severidad. Por ello este tipo de estudios que analizan las prácticas forestales sostenibles son imprescindibles para una adecuada gestión forestal en la que se consideren los incendios forestales como lo que son, un elemento natural de los ecosistemas.
2. Las prácticas de gestión forestal, como la tala, se realizan en muchas áreas forestales para disminuir la carga de combustible vegetal y en consecuencia disminuir el riesgo de incendios forestales. El clareo y el lapso entre este manejo y un incendio forestal, tienen efecto en las propiedades del suelo. No eliminar la vegetación cortada de la superficie del suelo influye en la severidad del fuego y produce cambios sustanciales en las propiedades del suelo. Hay una falta importante de estudios que evaluen estas gestiones previas al incendio y la eficacia de la gestión para evitar impactos dramáticos en los suelos y evaluar el tiempo de espera durante el cual esta gestión es efectiva. Recomendamos este tratamiento como una forma de prevenir la generación de nuevos incendios forestales, pero se deben evitar grandes acumulaciones de

vegetación cortada que cubra la superficie del suelo para reducir la severidad de los incendios en episodios potenciales de incendios forestales a medio y largo plazo. Se necesitan más estudios como este para asegurar la implementación de un manejo forestal adecuado y para determinar si las diferencias entre los sitios tratados y no tratados aumentan o desaparecen con el tiempo.

3. El estado del suelo posterior al incendio es muy frágil y una lluvia torrencial puede dañar sus propiedades. En este estudio, un año después del incendio, como consecuencia de la erosión de las cenizas y del suelo, la remoción de árboles y la recuperación de la vegetación produjeron cambios en las propiedades químicas del suelo. En general, nuestro estudio observó y concluyó que:

a) Este período de lluvia intensa no tuvo un impacto significativo en la estabilidad del agregado del suelo en esta área afectada por un incendio de alta severidad. Se observaron impactos significativos solo un año después del incendio como consecuencia de la erosión de las cenizas y del suelo, la remoción de árboles, la recuperación de la vegetación y los cambios en las propiedades químicas del suelo, como el SPAR.

b) El fuego no tuvo un impacto en el nitrógeno total, el carbono total, el carbono inorgánico, la relación C/N o los niveles de carbonatos. Sin embargo, las partículas de material carbonizado, transportadas desde otras áreas o que eran más resistentes a la erosión, pueden haber contribuido a un aumento significativo del carbono orgánico.

c) Los cationes extraíbles y el fósforo disponible aumentaron después del evento de lluvia como consecuencia de un pH favorable. Los valores de SPAR también fueron significativamente más altos después del evento de lluvia, lo que indica que puede haber contribuido a la disminución de la estabilidad de agregados del suelo.

d) Un año después del incendio, se observó una disminución significativa en la estabilidad de agregados y los elementos extraíbles. Esto probablemente se debió a los altos niveles de SPAR (en el caso de la disminución en la estabilidad de agregados del suelo), la intervención humana y la lixiviación del suelo provocada por otros eventos de lluvia además del consumo de vegetación.

Por lo tanto, según los resultados de los análisis realizados en este estudio, el evento de lluvia intensa y otros procesos posteriores al incendio (atribuibles a causas naturales o humanas) no modificaron en gran medida las propiedades del suelo a corto plazo como se registró antes del episodio de lluvia intensa debido a la superficie plana en la que se encuentra el área de estudio.

4. La dinámica posterior a los incendios forestales puede verse afectada por las prácticas de manejo forestal. En nuestro estudio, los resultados mostraron que el incendio forestal tuvo un impacto inmediato en el pH del suelo y en los elementos mayoritarios extraíbles, independientemente del tratamiento realizado, disminuyendo sus niveles 10 meses después del incendio. Lo inverso se observó en la relación C/N, pero solo en los tratamientos de CR y CL. Por otro lado, los efectos del manejo posterior a los incendios forestales en las propiedades del suelo AS, TN, SOM y IC se observaron 10 meses después del incendio.

AS, TN y SOM fueron significativamente más altos en CR que en CL, mientras que los valores de IC fueron significativamente más altos en CL que en NT. Esto podría deberse a que la viruta de la tala que provino de las actividades de manejo y el tránsito peatonal más intenso pudo haberse depositado en el la superficie del suelo. La EC del suelo fue significativamente mayor en CR en comparación con CL 2 meses después del incendio. La concentración de sales en el suelo fue significativamente mayor en los tratamientos de CR y CL 2 meses después del incendio forestal en comparación con 10 meses después. En general, las diferencias entre los manejos realizados y el escenario de no intervención no fueron relevantes, lo que demuestra que el este tipo de gestión manual es una buena alternativa para pequeñas áreas concretas donde se pretende evitar el uso de maquinaria pesada y con ello la erosión y pérdida de suelo provocando con todo ello la degradación del suelo.

5. En algunos casos, los efectos del fuego pueden permanecer a largo plazo y esto depende de factores como la propiedad estudiada del suelo, la severidad del fuego y las condiciones climáticas posteriores al incendio. Los incendios forestales tienen diferentes efectos a largo plazo en las propiedades químicas del suelo, lo que indica que los nutrientes del suelo responden de manera diferente según el tiempo transcurrido desde el incendio y varían en gran medida dependiendo de la severidad de dichos incendios. Los niveles de sodio extraíble no difirieron de los registrados en el control en ninguna de las fechas de muestreo, mientras que TN y C/N se recuperaron a niveles anteriores al fuego. Los cationes mayoritarios solo se recuperaron en el área afectada por la baja intensidad del fuego y TC y SOM no se recuperaron en ninguna de las áreas quemadas. El fuego de baja severidad también condujo a un aumento a largo plazo en los niveles de potasio extraíble. La falta de recuperación presentada por los parámetros esenciales del suelo, incluidos Ca y Mg extraíbles (en el área afectada por incendios de alta severidad), TC y SOM, muestra que las medidas de restauración pueden ser necesarias, especialmente después de incendios de alta severidad. La incapacidad de los nutrientes para recuperar sus niveles previos al fuego es claramente una consecuencia de las perturbaciones causadas por el fuego (por ejemplo, remoción de vegetación, erosión, lixiviación) y la subsiguiente regeneración vegetal con el tiempo, disminuyendo el contenido general de nutrientes del suelo. Los suelos necesitan más tiempo para regresar a sus niveles anteriores al fuego, especialmente en las áreas afectadas por el fuego de alta severidad. Sin embargo, se necesita más investigación para observar cuándo TC y SOM vuelven a los niveles previos al fuego, ya que esto puede verse afectado por el cambio en la cubierta vegetal.

6. Los incendios forestales forman parte del ecosistema mediterráneo y la gestión forestal de medio a largo plazo después del incendio es necesario para evitar los efectos dañinos causados por incendios forestales de alta severidad y reducir el riesgo de incendio. Nuestro último estudio detectó diferencias en las propiedades del suelo de dos sitios expuestos a diferentes acciones de manejo posteriores al incendio y reveló que dos propiedades, SWR y AS, no fueron afectadas. La ausencia de diferencias en estas y otras

propiedades del suelo en los sitios de control y quemados (LD y HD) nos permite concluir que los suelos se han recuperado completamente. Más específicamente, los sitios LD y Control presentan propiedades del suelo muy similares, siendo más similares que los suelos de los sitios Control y HD, por un lado, y los de los sitios HD y LD, por el otro. Sin embargo, las diferencias que surgieron entre los sitios de baja (LD) y alta densidad (HD) pueden atribuirse directamente al tratamiento forestal realizado en 2005 y las consiguientes diferencias en la densidad vegetal. Se registraron diferencias significativas entre los lugares LD y HD en las siguientes propiedades del suelo: IC, pH, K y Mn, mientras que no se registraron diferencias significativas en TN, P, Al, Fe y relación C/N. En resumen, nuestros hallazgos apuntan a los efectos positivos de la gestión realizada casi 20 años después de un incendio forestal y los efectos más dañinos (aunque no dramáticos) de la alta densidad vegetal en el sitio de HD, que revelan algunos problemas en la recuperación de ciertas propiedades del suelo. De hecho, varios autores abogan por el manejo forestal para mantener bajo el riesgo de incendio y para lograr mejores propiedades del suelo. Sin embargo, se necesitan estudios adicionales para establecer el punto en el tiempo en que las diferencias entre las propiedades del suelo de los sitios LD y HD se vuelvan significativas.

REFERENCES

REFERENCES

Abram, N.J., Gagan, M.K., McCulloch, M.T., Chappell, J., Hantoro, W.S., 2003. Coral reef death during the 1997 Indian Ocean Dipole linked to Indonesia wildfires. Science 301 (5635), 952-955.

Achard, F., Eva, H.D., Mollicone, D., Beuchle, R., 2008. The effect of climate anomalies and human ignition factor on wildfires in Russian boreal forests. Philosophical Transactions of the Royal Society of London B: Biological Science 363 (1501), 2329-2337.

Adams, P.W., Boyle, J.R. 1980. Effects of Fire on Soil Nutrients in Clearcut and Whole-tree Harvest Sites in Central Michigan 1. Soil Science Society of America Journal 44 (4), 847-850.

ADCIF., 2002. Los incendios forestales en España, decenio 1991-2000. Ministerio de Medio Ambiente. 80 pp.

Alcañiz, J.M., serrasolsas, I., Vallejo, R., 1996. Efectes del's incendis forestals sobre el sòl. En: Terradas, J., (Ed.). Ecologia del foc. Editorial Proa. 111-130 pp.

Alcañiz, M., Outeiro, L., Francos, M., Úbeda, X., 2018. Effects of prescribed fires on soil properties: A review. Science of the Total Environment 613, 944-957.

Almendros, G., González-Vila, F.J., González-Pérez, J.A., Knicker, H., De la Rosa, J.M., Dettweiler, C., Hernández, Z. 2012. On the structural factors of soil humic matter related to soil water repellence in fire-affected soils. In EGU General Assembly Conference Abstracts (Vol. 14, p. 8712).

Almendros-Martín, G., González-Vila, F. J., & Martín Martínez, F. (1990). Fire induced transformation of soil organic matter from an oak forest. An experimental approach to the effects of fire on humic substances.

Anderson, J.P.E., 1982. Soil respiration. methods of soil analysis. Part 2. Chemical and microbiological properties, 9, 831-871.

Anderson, T. H. (2003). Microbial eco-physiological indicators to asses soil quality. *Agriculture, Ecosystems & Environment*, *98*(1-3), 285-293.

Arcenegui, V., Mataix-Solera, J., Guerrero, C., Zornoza, R., Mayoral, A. M., Morales, J. 2007. Factors controlling the water repellency induced by fire in calcareous Mediterranean forest soils. European Journal of Soil Science, 58 (6), 1254-1259.

Archibald, S., Lehmann, C., Gómez-Dans, J.L., Bradstock, R.A., 2013. Defining pyromes and global sindromes of FIRE regimes. In: Proocedings of the national Academy of Science of the United States of America-PNAS 110 (16), 6442-6447.

Arianoutsou, M., Beard, J.S., Ferrés, L., Folch, R., Trabaud, L.V., 1993. La vida a les formacions escleròfil·les. En: Folch, R., (Ed.). Mediterrànies. Barcelona, Edit. MAB, UNESCO. 64-109 pp.

Badía, A., Saurí, D., Cerdan, R., Llurdés, J.C., 2002. Causality and management of forest fires in mediterranean environaments: an example from Catalonia. Environmental Hazards 4, 23-32.

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

Badía, D., Martí, C. 2003. Effect of simulated fire on organic matter and selected microbiological properties of two contrasting soils. Arid Land Research and Management 17 (1), 55-69.

Badía-Villas, D., González-Pérez, J.A., Aznar, J.M., Arjona-Gracia, B., Martí-Dalmau, C. 2014. Changes in water repellency, aggregation and organic matter of a mollic horizon burned in laboratory: soil depth affected by fire. Geoderma 213, 400-407.

Bai, S. H., Blumfield, T. J., Reverchon, F. 2014. The impact of mulch type on soil organic carbon and nitrogen pools in a sloping site. Biology and fertility of soils 50 (1), 37-44.

Belcher, C.M., 2013. Fire phenomena and the earth System: An Interdisciplinary guide to fire science. First edition. John Wiley and Sons, Ltd.

Benayas, J.R., Martins, A., Nicolau, J.M., Schulz, J.J., 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB reviews: Perspectives in agriculture, veterinary science, nutrition and natural resources 2 (57), 1-14.

Bento-Gonçalves, A., Vieira, A., Úbeda, X., Martin, D., 2012. Fire and soils: key concepts and recent advances. Geoderma 191, 3-13.

Binkley, D., Son, Y., & Valentine, D. W. (2000). Do forests receive occult inputs of nitrogen?. Ecosystems, 3(4), 321-331.

Bissett, J., Parkinson, D. 1980. Long-term effects of fire on the composition and activity of the soil microflora of a subalpine, coniferous forest. Canadian Journal of Botany 58 (15), 1704-1721.

Bodí, M.B., Cerdà, A., Mataix-Solera, J., Doerr, S.H., 2012. Efectos de los incendios forestales en la vegetación mediterránea: revisión bibliográfica. Boletín de la AGE 58, 33-55.

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

Bond, W.J., Keeley, J.E., 2005. Fire as a global "herbivore": the ecology and evolution of flammable ecosystems. Trends in ecology and evolution 20 (7), 387-394.

Bravo, S., Kunst, C., Grau, R., Aráoz, E., 2010. Fire-rainfall relationships in Argentine Chaco savannas. Journal of Arid Environments 74, 1319-1323.

Bremner, J. M., & Mulvaney, C.S. 1982. Nitrogen—Total 1. Methods of soil analysis. Part 2. Chemical and microbiological properties, (methodsofsoilan2), 595-624.

Cade-Menun, B.J., Berch, S.M., Preston, C.M., Lavkulich, L.M. 2000. Phosphorus forms and related soil chemistry of Podzolic soils on northern Vancouver Island. II. The effects of clear-cutting and burning. Canadian Journal of Forest Research 30 (11), 1726-1741.

Caldwell, T.G., Johnson, D.W., Miller, W.W., Qualls, R.G. 2002. Forest floor carbon and nitrogen losses due to prescription fire. Soil Science Society of America Journal, 66, 262-267.

Caon, L., Vallejo, V. R., Ritsema, C. J., Geissen, V. 2014. Effects of wildfire on soil nutrients in Mediterranean ecosystems. Earth-Science Reviews, 139, 47-58.

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

Castellnou, M., Nebot, E., 2007. El papel del fuego en la gestión del paisaje. 4ª Conferencia Internacional sobre incendios forestales. Sevilla.

Castillo-Soto, M., 2015. Diagnosis of Forest Fires in Chile. In: Bento-Gonçalves, A., Vieira, A. (Eds.). Wildland fires – a Worlwide reality. Nova Science Publishers. pp. 211-224.

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

Certini, G., Nocentini, C., Knicker, H., Arfaioli, P., Rumpel, C. 2011. Wildfire effects on soil organic matter quantity and quality in two fire-prone Mediterranean pine forests. Geoderma 167, 148-155.

Cervera, T., Garrabou, R., Tello, E., 2015. Política forestal y evolución de los bosques en Cataluña desde el siglo XIX hasta la actualidad. Investigaciones de Historia Económica 11 (2), 116-127.

Collins, B.M., Stephens, S.L., 2007. Managing natural wildfires in Sierra Nevada wilderness areas. Frontiers in Ecology and the Environment 5 (10), 523-527.

Corona, P., Ascoli, D., Barbati, A., Bovio, G., Colangelo, G., Elia, M., Garfi, V., Iovino, F., Lafortezza, R., Leone, V., Lovreglio, R., Marchetti, M., Marchi, E., Menguazzato, G., Nocentini, S., Picchio, R., Portoghesi, l., Puletti, N., Sanesi, G., Chianucci, F., 2015. Integrated forest management to prevent wildfires under Mediterranean environments. Annals of Silvicultural Research 39, 1-22.

Cram, D.S., Baker, T.T., Fernald, A.G., Cibils, A.F., VanLeeuwen, D.M. 2015. Fuel and vegetation trends after wildfire in treated versus untreated forests. Forest Science 61 (4), 753-762.

Cromer, R.N., 1967. The significance of the "ashbed effect" in Pinus radiate plantations. Appita 20, 104-112.

de la Rosa, J. M., Knicker, H., González-Pérez, J. A., Vila, F. J. G., Morillo, N. T. J., Martín, G. A. 2014. Revisión de las investigaciones a nivel molecular sobre el impacto del fuego en la materia orgánica del suelo: Progresos en el siglo XXI. Cuadernos de investigación geográfica (40), 387-402.

Deak, B., Valkó, O., Torok, P., Vegvari, Zs., Hartel, T., Schmotzer, A., Kapocsi, I., Tothmeresz, B., 2014. Grassland fires in Hungary – Experiences of nature conservationists on the effect of fire on biodiversity. Applied Ecology and Environmental Research 267-283.

DeBano, L. F., Dunn, P. H., Conrad, C.E. 1977. Fire's effect on physical and chemical properties of chaparral soils. I. Soil nitrogen. Soil Science Society of America Journal 43, 504-509.

DeBano, L. F., Neary, D.G., Ffolliott, P.F. 1998. Fire effects on ecosystems. John Wiley & Sons.

DeByle, N.V. 1976. Fire, logging, and debris disposal effects on soil and water in northern coniferous forests. International Union of Forest Research Organizations.

DeLuca, T.H., Zouhar, K.L. 2000. Effects of selection harvest and prescribed fire on the soil nitrogen status of ponderosa pine forests. Forest Ecology and Management, 138 (1-3), 263-271.

Demeyer, A., Nkana, J.V., Verloo, M.G. 2001. Characteristics of wood ash and influence on soil properties and nutrient uptake: an overview. Bioresource technology 77 (3), 287-295.

Díaz-Raviña, M., Prieto, A., Acea, M.J., Carballas, T. 1992. Fumigation-extraction method to estimate microbial biomass in heated soils. Soil Biology and Biochemistry 24 (3), 259-264.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 1998. Spatial variability of soil hydrophobicity in fire-prone eucalyptus and pine forests, Portugal. Soil Sci. 163, 313-324.

Doerr, S. H., Cerdà, A. 2005. Fire effects on soil system functioning: new insights and future challenges. International Journal of Wildland Fire, 14 (4), 339-342.

Doerr, S.H., Dekker, L.W., Shakesby, R.A., Ritsema, C.J., Bryant, R., 2002. Water repellency of soils: the influence of ambient relative humidity. Soil Sci. Soc. Am. J. 66, 401-405.

Doran, J. C. (1994). CSIRO Division of Forestry. Forest Genetic Resources, 12.

Durán Humia, J., Rodríguez Pereiras, A., Gallardo Correa, A., Fernández Palacios, J.M. 2008. Nutrient availability of a fires chronosequence in Pinus canariensis forests. Cuadernos de la Sociedad Española de Ciencias Forestales (España).

Eck, B., Barrett, M., McFarland, A., Hauck, L. 2010. Hydrologic and water quality aspects of using a compost/mulch blend for erosion control. Journal of Irrigation and Drainage Engineering 136 (9), 646-655.

Ela, S.D., Gupta, S.C., Rawls, W.J., 1992. Macropore and surface seal interactions affecting water infiltration into soil. Soil Science Society of America Journal 56 (3), 714-721.

Ershad, M., Hemmati, V., Hashemi, S.A., Foroozan, A.H. 2013. The Effects of Forest Fires on the Chemical Properties of Soils in Northern Iran: A Case Study on Pinus Taeda Stands. Bull. Env. Pharmacol. Life Sci, 2 (9), 51-54.

Etiegni, L., Campbell, A.G. 1991. Physical and chemical characteristics of wood ash. Bioresource technology 37 (2), 173-178.

Fernandes, P. M., Davies, G. M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., Stoof, C.R., Vega, J.A., Molina, D. 2013. Prescribed burning in southern Europe: developing fire management in a dynamic landscape. Frontiers in Ecology and the Environment 11(s1), e4-e14.

Fernandes, P.M., Rigolot, E., 2007. The fire ecology and management of maritime pine (Pinus pinaster Ait.) Forest Ecology and Management 241 (1-3), 1-13.

Fernandes, P.M., Botelho, H.S., 2003. A review of prescribed burning effectiveness in ire hazard reduction. International Journal of Wildland Fire 12, 117-128.

Fernández, I., Cabaneiro, A., Carballas, T. 1997. Organic matter changes immediately after a wildfire in an Atlantic forest soil and comparison with laboratory soil heating. Soil Biology and Biochemistry 29 (1), 1-11.

Ferreira-Leite, L., Bento-Gonçalves, A., Lourenço, L., Úbeda, X., Vieira, A., 2013. Grandes incêndios em Portugal Continental como resultado das perturbaçoes nos redimes de fogo no mundo mediterraneo. Silva Lusitana 21, 127-142.

Ferreira-Leite, F., Bento-Gonçalves, A., Vieira, A., da Vinha, A., 2015b. Forest fires in Portugal- Brief Characterization. In: Bento-Gonçalves, A., Vieira, A. (Eds.). Wildland fires – a Worlwide reality. Nova Science Publishers. pp. 115-126.

Ferreira-Leite, F., Bento-Gonçalves, A., Vieira, A., da Vinha, A., 2015a. Mega-fires around the world: a literatura review. In: Bento-Gonçalves, A., Vieira, A. (Eds.). Wildland fires – a Worlwide reality. Nova Science Publishers. pp. 15-34.

Francos, M., Pereira, P., Alcañiz, M., Mataix-Solera, J., Úbeda, X., 2016a. Impact of an intense rainfall event on soil properties following a wildfire in a Mediterranean environment (North-East Spain). Sci. Total Environ. 572, 1353-1362.

Francos, M., Pereira, P., Alcañiz, M., Úbeda, X., 2018a. Post-wildfire management effects on short-term evolution of soil properties (Catalonia, Spain, SW-Europe). Sci. Total Environ. 633, 285-292.

Francos, M., Pereira, P., Mataix-Solera, J., Arcenegui, V., Alcañiz, M., Úbeda, X., 2018c. How clear-cutting affects fire severity and soil properties in a Mediterranean ecosystem. Journal of Environmental Management 206, 625-632.

Francos, M., Úbeda, X., Pereira, P., 2018d. Long-term forest management after wildfire (Catalonia, NE Iberian Peninsula). Journal of Forestry Research. https://doi.org/10.1007/s11676-018-0867-3.

Francos, M., Úbeda, X., Pereira, P., Alcañiz, P., 2018b. Long-term impact of wildfire on soil exposed to different fire severities. A case study in Cadiretes Massif (NE Iberian Peninsula). Sci. Total Environ. 615, 664-671.

Francos, M., Úbeda, X., Tort, J., Panareda, J.M., Cerdà, A., 2016b. The role of forest fire severity on vegetation recovery after 18 years. Implications for forest management of Quercus suber L. in Iberian Peninsula. Global Planet. Change 145, 11-16.

Fritze, H., Pennanen, T., & Pietikäinen, J. 1993. Recovery of soil microbial biomass and activity from prescribed burning. Canadian Journal of Forest Research, 23 (7), 1286-1290.

García-Orenes, F., Arcenegui, V., Chrenkova, K., Mataix-Solera, J., Moltó, J., Jara-Navarro, A. B., Torres, M. P. 2017. Effects of salvage logging on soil properties and vegetation recovery in a fire-affected Mediterranean forest: A two year monitoring research. Science of the Total Environment 586, 1057-1065.

Ginzburg, O., Steinberger, Y. 2012. Effects of forest wildfire on soil microbial-community activity and chemical components on a temporal-seasonal scale. Plant and soil 360 (1-2), 243-257.

Giovannini, G. (2012). Fire in agricultural and forestal ecosystems: the effects on soil. ETS.

Giovannini, G., Lucchesi, S., Giachetti, M. 1988. Effect of heating on some physical and chemical parameters related to soil aggregation and erodibility. Soil Science, 146 (4), 255-261.

Girardin, M.P., Ali, A.A., Carcaillet, C., Gautgier, S., Hély, C., Le Goff, H., Terrier, A., Bergeron, Y., 2013. Fire in managed forest of eastern Canada: Risk and options. For. Ecol. Manage. 294, 238-249.

Goforth, B.R., Graham, R.C., Hubbert, K.R., Zanner, C.W., Minnich, R.A. 2005. Spatial distribution and properties of ash and thermally altered soils after high-severity forest fire, southern California. International Journal of Wildland Fire, 14 (4), 343-354.

Gómez-Rey, M. X., Couto-Vázquez, A., García-Marco, S., González-Prieto, S. J. 2013. Impact of fire and post-fire management techniques on soil chemical properties. Geoderma 195, 155-164.

González, J.R., Palahí, M., Trasobares, A., Pukkala, T., 2006. A fire probability model for forest stands in Catalonia (north-east Spain). Annals of forest Science, Springer Verlag/EDP Science 63 (2), 169-176.

González, J.R., Pukkala, T., 2007. Characterization of forest fires in Catalonia (north-east Spain). Eur. J. Forest. Res. 126 (3), 421-429.

González-Pérez, J. A., González-Vila, F. J., Almendros, G., Knicker, H. 2004. The effect of fire on soil organic matter—a review. Environment international 30 (6), 855-870.

Grove, T.S., O'connell, A.M., Dimmock, G.M. 1986. Nutrient changes in surface soils after an intense fire in jarrah (Eucalyptus marginata Donn ex Sm.) forest. Australian Journal of Ecology 11 (3), 303-317.

Grove, T.S., O'connell, A.M., Malajczuk, N. 1980. Effects of fire on the growth, nutrient content and rate of nitrogen fixation of the cycad Macrozamia riedlei. Australian Journal of Botany 28 (3), 271-281.

Guerrero, C., Gómez, I., Solera, J. M., Moral, R., Beneyto, J. M., Hernández, M. T. 2000. Effect of solid waste compost on microbiological and physical properties of a burnt forest soil in field experiments. Biology and Fertility of Soils 32 (5), 410-414.

Guerrero, C., Mataix-Solera, J., Gómez, I., García-Orenes, F., Jordán, M.M. 2005. Microbial recolonization and chemical changes in a soil heated at different temperatures. International Journal of Wildland Fire 14 (4), 385-400.

Guerrero, C., Mataix-Solera, J., Navarro-Pedreño, J., García-Orenes, F., Gómez, I. 2001. Different patterns of aggregate stability in burned and restored soils. Arid Land Research and Management 15 (2), 163-171.

Guerrero, C., Mataix-Solera, J., Rodríguez, F., García-Orenes, F., Gómez, I., Moral, R. 2003. Carbono microbiano edáfico en suelos afectados por fuego y enmendados con diferentes residuos orgánicos. Edafología 10 (2), 185-190.

Heiri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J. Paleolimnol. 5, 101-110.

Hernández, T., Garcia, C., Reinhardt, I. 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biology and fertility of soils 25 (2), 109-116.

Hubbert, K. R., Oriol, V. 2005. Temporal fluctuations in soil water repellency following wildfire in chaparral steeplands, southern California. International Journal of Wildland Fire, 14 (4), 439-447.

Hudspith, V.A., Belcher, C.M., Barnes, J., Dash, C.B., Kelly, R., Hu, F.S., 2017. Charcoal reflectance suggests heating duration and fuel moisture affected burn severity in four Alaskan tundra wildfires. International Journal of Wildland Fire 26 (4), 306-316.

Ice, G.G., Neary, D.G., Adams, P.W. 2004. Effects of wildfire on soils and watershed processes. Journal of Forestry, 102 (6), 16-20.

Imeson, A. C., Verstraten, J. M., Van Mulligen, E. J., Sevink, J. 1992. The effects of fire and water repellency on infiltration and runoff under Mediterranean type forest. Catena, 19(3-4), 345-361.

Inbar, M., Tamir, M. I., Wittenberg, L. 1998. Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 24 (1), 17-33.

Insam, H., Domsch, K. H. 1988. Relationship between soil organic carbon and microbial biomass on chronosequences of reclamation sites. *Microbial ecology 15* (2), 177-188.

Jennings, T.N., Smith, J. E., Cromack, K., Sulzman, E. W., McKay, D., Caldwell, B. A., Beldin, S. I. 2012. Impact of postfire logging on soil bacterial and fungal communities and soil biogeochemistry in a mixed-conifer forest in central Oregon. Plant and soil 350 (1-2), 393-411.

Jiménez-Ruano, A., Mimbrero, M.R., de la Riva Fernández, J., 2017. Understanding wildfires in mainland Spain. A comprehensive analysis of fire regime features in a climate-human context. Applied geography 89, 100-111.

Johnson, D. W., Murphy, J. F., Susfalk, R. B., Caldwell, T. G., Miller, W. W., Walker, R. F., Powers, R. F. 2005. The effects of wildfire, salvage logging, and post-fire N-fixation on the nutrient budgets of a Sierran forest. Forest Ecology and Management 220 (1-3), 155-165.

Johnson, D., Murphy, J.D., Walker, R.F., Glass, D.W., & Miller, W.W. 2007. Wildfire effects on forest carbon and nutrient budgets. Ecological engineering 31 (3), 183-192.

Jones, B.A., Thacher, J.A., Chermak, J.M., Berrens, R.P., 2016. Wildfire smoke health costs: a methods case study for a Shouthwestern US “mega-fire”. Journal of Environmental Economics and Policy 5 (2), 181-199.

Jones, O.D., Nyman, P., Sheridan, G.J., 2014. Modelling the effects of fire and rainfall regimes on extreme erosion events in forested landscapes. Stochastic environmental research and risk assessment 28 (8), 2015-2025.

Karlen, D.L., Ditzler, C.A., Andrews, S.S. 2003. Soil quality: why and how?. Geoderma, 114 (3-4), 145-156.

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

Khanna, P.K., Raison, R., Falkiner, R. 1994. Chemical properties of ash derived from Eucalyptus litter and its effects on forest soils. Forest Ecology and Management 66 (1-3), 107-125.

Kharuk, V.I., Dvinskaya, M.L., Sergey, T., 2010. There’s no forest without fire. Wildfires in the taiga as a natural factor. Science First Hand 27, 120-129.

Knoepp, J. D., DeBano, L. F., Neary, D. G. 2005. Soil chemistry. In: Neary, Daniel G.; Ryan, Kevin C.; DeBano, Leonard F., eds. 2005. Wildland fire in ecosystems: effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-vol. 4. Ogden, UT: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 53-71, 42, 53-71.

Knudsen, D., Petersen, G.A., Pratt, P.F., 1986. Lithium, sodium and potassium. In: Soil Science Society of America (Ed.) Methods of soil analysis. Vol. 2. ASA-SSSA, Madison, WI. pp. 225-246.

Lima, H.N., Costa, O.V., de Mello, J.W.V., Corrêa, M.M. 2002. A comparison of methods used for determination of cation exchange capacity in brazilian soils. Revista Ceres (Brazil).

Low, A.J., 1954. The study of soil structure in the field and in the laboratory. J. Soil Sci. 5, 57-74.

Mandre, M., Korsjukov, R., Ots, K. 2004. Effect of wood ash application on the biomass distribution and physiological state of Norway spruce seedlings on sandy soils. Plant and soil 265 (1-2), 301-314.

Marcos, E., Luis, E., Tarrega, R. 1998. Chemical soil changes in shrubland after experimental fire. Fire Management and Lanscape Ecology 3-11.

Marcos, E., Tárrega, R., Luis, E. 2007. Changes in a Humic Cambisol heated (100–500 C) under laboratory conditions: the significance of heating time. Geoderma, 138 (3-4), 237-243.

Maksimova, E., Abakumov, E., 2017. Micro morphological characteristics of sandy forest soils recently impacted by wildfires in Russia. Solid Earth 8 (2), 553-560.

Marlon, J.R., Bartlein, P.J., Gavin, D.G., Long, C.J., Scott-Anderson, R., Briles, C.E., Brown, K.J., Colombaroli, D., Hallett, D.J., Power, M.J., Scharf, E.A., Walsh, M.K., 2003. Long-term perspectives on wildfires in the western USA. Proceedings of the National Academy of Science 109 (9), 535-543.

Marion, G.M., Moreno, J.M., Oechel, W.C. 1991. Fire severity, ash deposition, and clipping effects on soil nutrients in chaparral. Soil Science Society of America Journal, 55 (1), 235-240.

Marino, E., Hernando, C., Planelles, R., Madrigal, J., Guijarro, M., Sebastián, A., 2014. Forest fuel management for wildfire prevention in Spain: a quantitative SWOT analysis. International Journal of Wildland Fire 23 (3), 373-384.

Martínez-Alier, J., Roca-Jusment, J., 2000. Economía ecológica y política ambiental. Fondo de Cultura Económica de España. ISBN: 9789681459981. 493 p.

Marques, M. A., Mora, E. 1998. Effects on erosion of two post-fire management practices: clear-cutting versus non-intervention. Soil and Tillage Research 45 (3-4), 433-439.

Mataix-Solera, J., 1999. Alteraciones físicas, químicas y biológicas en suelos afectados por incendios forestales. Contribución a su conservación y regeneración. Tesis Doctoral. Facultad de Ciencias. Universidad de Alicante. 330 pp.

Mataix-Solera, J., Arcenegui, V., Guerrero, C., Mayoral, A. M., Morales, J., González, J., García-Orenes, F., Gómez, I. 2007. Water repellency under different plant species in a calcareous forest soil in a semiarid Mediterranean environment. Hydrological Processes: An International Journal, 21 (17), 2300-2309.

Mataix-Solera, J., Arcenegui, V., Tessler, N., Zornoza, R., Wittenberg, L., Martínez, C., Caselles, P., Pérez-Bejarano, A., Malkinson, D., Jordán, M.M. 2013. Soil properties as key factors controlling water repellency in fire-affected areas: Evidences from burned sites in Spain and Israel. Catena 108, 6-13.

Mataix-Solera, J., Cerdà, A., 2009. Incendios forestales en España. Ecosistemas terrestres y suelos. In: Efectos de los incendios forestales sobre los suelos en España: el estado de la cuestión por los científicos españoles. Cátedra Divulgación de la Ciencia, 25-54.

Mataix-Solera, J., Doerr, S.H. 2004. Hydrophobicity and aggregate stability in calcareous topsoils from fire-affected pine forests in southeastern Spain. Geoderma, 118 (1-2), 77-88.

Mataix-Solera, J., Guerrero, C. 2007. Efectos de los incendios forestales en las propiedades edáficas. Incendios Forestales, Suelos y Erosión Hídrica. Caja Mediterráneo CEMACAM Font Roja-Alcoi, Alicante, 5-40.

Mataix-Solera, J., Guerrero, C., García-Orenes, F., Bárcenas, G.M., Torres, M.P., Barcenas, M. 2009. Forest fire effects on soil microbiology. Fire effects on soils and restoration strategies 5, 133-175.

Mataix-Solera, J., Lloret, E., Zornoza, R., Guerrero, C., García-Orenes, F., Mataix-Beneyto, J., Navarro, J., Gómez, I. 2006. Soil organic carbon, microbial biomass and basal respiration 11 years after a forest fire in a Mediterranean environment. In Geophysical Research Abstracts (Vol. 8, p. 01249).

Mataix-Solera, J., Moltó, A., Guerrero, C., Pedreño, J. N., Moral, R., Gómez, J. 2001. Aplicación de un compost de lodo a un suelo forestal quemado. Efecto en los contenidos de N, P, K, Na, Ca, Mg, Fe, Mn, Cu, Zn, y B en Brachypodium phoenicoides. In Congreso Forestal Español.

Mataix-Solera, J., Navarro-Pedreño, J., Guerrero, C., Gómez, I., Marco, B., Mataix, J. (2002). Effects of an experimental fire on soil microbial populations in a Mediterranean environment. In Rubio, J.L., Morgan. R.P.C., Asins, S., Andreu, V. (Eds.). Man and soil at the third millennium, 2, 1607-1614. Geoforma Ediciones.

Maza, A., 1995. Prevenció d'incendis forestals i territorio. Dossiers Agraris ICEA, Incendis forestals 37-41.

McIver, J. D., Starr, L. 2000. Environmental effects of postfire logging: literature review and annotated bibliography. Gen. Tech. Rep. PNW-GTR-486. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. pp. 72.

Moreno, J. M., Zuazua, E., Pérez, B., Luna, B., Velasco, A., Resco de Dios, V., 2011. Rainfall patterns after fire differentially affect the recruitment of three Mediterranean shrubs. Biogeosciences 8 (12), 3721-3732.

Morugán-Coronado, A., García-Orenes, F., Cerdà, A., 2015. Changes in soil microbial activity and physicochemical properties in agricultural soils in Eastern Spain. Spanish Journal of Soil Science 5-3. DOI: 10.3232/SJSS.2015.V5.N3.02

Naidu, C.V., Srivasuki, K.P. 1994. Effect of forest fire on soil characteristics in different areas of Seshachalam hills. Annals of Forestry, 2 (2), 166-173.

Naveh, Z., 1975. The evolutionary significance of fire in the Mediterranean region. Vegetatio 29 (3), 199-208.

Neary, D. G. (2009). Post-wildland fire desertification: can rehabilitation treatments make a difference?. Fire Ecology, 5(1), 129-144.

Neary, D.G., Klopatek, C.C., DeBano, L.F., Ffolliott, P., 1999. Fire effects on belowground sustainability: a review and synteshis. Forest Ecology and Management 122, 51-71.

Oades, J. M. 1993. The role of biology in the formation, stabilization and degradation of soil structure. In Soil Structure/Soil Biota Interrelationships (pp. 377-400).

Olsen, W. 2016. Effects of wildfire and post-fire salvage logging on rill networks and sediment delivery in California forests. PhD Theses. Michigan Technological University. pp. 150.

Outeiro, L., Asperó, F., Mataix-Solera, J., Úbeda, X., 2007. El Foc com a causant de canvis en les propietat del sòl: incendis forestals i cremes prescrites. Treballs de la Societat Catalana de Geografia 229-250.

Outeiro, L., Asperó, F., Úbeda, X., 2008. Geostatistical methods to study spatial variability of soil cations after a prescribed fire and rainfall. Catena 74 (3), 310-320.

Pausas, J.G., Keeley, J.E., 2009. A burning history: the role of fire in the history of life. BioScience 59 (7), 593-601.

Pausas, J.G., Llovet, J., Rodrigo, A., Vallejo, R., 2008. Are wildfires a disaster in the Mediterranean basin? – A review. International journal of Wildland Fire 17, 713-723.

Pearce, G., Dyck, B., Frampton, R., Wingfield, M., Moore, J., 2000. Biophysical risk to forests- New Zealand compared to the rest of the world. Proceedings, New Zealand Institute of Forestry Conference. 17-19 pp.

Peix, J., 1999. Foc Verd II. Programa de gestió del risc d'incendi forestal. PrimerSegon Edicions. Barcelona. 231 p.

Pereira, P., Cerda, A., Martin, D., Úbeda, X., Depellegrin, D., Novara, A., Martínez-Murillo, J., Brevik, E.C., Menshov, O., Rodrigo-Comino, J., Miesel, J. 2017. Short-term low-severity spring grassland fire impacts on soil extractable elements and soil ratios in Lithuania. Science of the Total Environment, 578, 469-475.

Pereira, P., Cerdà, A., Misiune, I., 2015a. Wildfires in Lithuania. In: Bento-Gonçalves, A., Vieira, A. (Eds.). Wildland fires – a Worlwide reality. Nova Science Publishers. pp 185-198.

Pereira, P., Francos, M., Brevik, E.C., Úbeda, X., Bogunovic, I., 2018b. Post-fire soil management. Current Opinion in Environmental Science & Health 5, 26-32.

Pereira, P., Francos, M., Úbeda, X., Brevik, E.C. 2018a. Fire impacts in European Grassland Ecosystems. In: Bento-Gonçalves, A.J., Vieira, A.A.B., Melo-Costa, M.R., Marques-Aranha, J.T. (Eds.) Wildfires: Perspectives, Issues and Challenges of the 21st Century. Nova Science Publisher. ISBN: 978-1-53612-890-1. Chapter 1. pp. 1-27.

Pereira, P., Oliva, M., Baltrenaite, E., 2010. Modelling extreme precipitation in hazardous mountainous areas. Contribution to landscape planning and environmental management. Journal of Environmental Engineering and Landscape Management 18 (4), 329-342.

Pereira, P., Oliva, M., Cerdà, A., 2016. Fire impacts on European Boreal Soils: A review. In EGU General Assembly Conference Abstracts 18, 15644.

Pereira, P., Pranskevicius, M., Bolutiene, V., Jordán, A., Zavala, L., Úbeda, X., Cerdà, A. 2015b. Short term spatio-temporal variability of soil water extractable Al and Zn after a low severity grassland fire in Lithuania. Flamma 6 (2), 50-57.

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Guerrero, C. 2011. Effects of a low severity prescribed fire on water-soluble elements in ash from a cork oak (Quercus suber) forest located in the northeast of the Iberian Peninsula. Environmental Research 111 (2), 237-247.

Pereira, P., Úbeda, X., Mataix-Solera, X., Martin, D., Oliva, M., Novara, A., 2013. Short-term spatio-temporal spring grassland fire effects on soil color, organic matter and water repellency in Lithuania. Solid Earth Discussions 5 (2).

Pereira, P., Úbeda, X., Mataix-Solera, J., Oliva, M., Novara, A. 2014. Short-term changes in soil Munsell colour value, organic matter content and soil water repellency after a spring grassland fire in Lithuania. Solid Earth, 5 (1), 209-225.

Pew, K.L., Larsen, C.P.S., 2001. GIS analysis of spatial and temporal patterns of human caused wildfires in the temperate rain forest of Vancouver Island, Canada. Forest Ecology and Management 140, 1-18.

Pietikäinen, J., Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Canadian Journal of Forest Research 23 (7), 1275-1285.

Piqué, M., Castellnou, M., Valor, T., Pagés, J., Larrañaga, A., Miralles, M., Cervera, T. (2011). Integració del risc de grans incendis forestals (GIF) en la gestió forestal: Incendis tipus i vulnerabilitat de les estructures forestals al foc de capçades. Sèrie: Orientacions de gestió forestal sostenible per a Catalunya (ORGEST).Centre de la Propietat Forestal. Departament d'Agricultura, Ramaderia, Pesca, Alimentació i Medi Natural. Generalitat de Catalunya

Plana, E., 2004. Anàlisi d'escenaris de prevenció i extinció d'incendis des de la perspectiva socioambiental. En: Plana, E. (Ed.). Incendis Forestals, dimensió socioambiental, gestió del risc i ecología del foc. Solsona: Xarxa Alinfo XCT200-00061, DL: L-501.

Plana, E., 2011. Cultura del risc i comunicación sobre el foc i els incendis forestals. Treballs de la Societat Catalana de Geografía 71-72, 265-282.

Plana, E., Domínguez, G., Sangrà, G., Ga, F., 2001. Análisis de la problemática de los incendios forestales de gran dimensión: un enfoque integral. Montes para la sociedad del nuevo milenio. III Congreso Forestal Nacional 5, 846-851.

Prieto-Fernández, A., Acea, M.J., Carballas, T. 1998. Soil microbial and extractable C and N after wildfire. Biology and Fertility of Soils 27 (2), 132-142.

Pyne, S.J., 2016. Fire in the mind: changing understandings of fire in Western civilization. Phil. Trans. R. Soc. B. 371, 1696.

Rab, M. A. 1996. Soil physical and hydrological properties following logging and slash burning in the Eucalyptus regnans forest of southeastern Australia. Forest Ecology and Management 84 (1-3), 159-176.

Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant and soil 51 (1), 73-108.

Raison, R.J., Keith, H., Khanna, P.K. 1990. Effects of fire on the nutrient-supplying capacity of forest soils. FRI Bulletin (159), 39-54.

Rapp, V. 2007. Tested by Fire: What Happens When Wildfires Meet Fuel Treatments?. JFSP Briefs 53. U.S. Joint Fire Science Program.

Robichaud, P. R., Elliot, W. J., MacDonald, L., Saveland, J., Busskohl, C., Howes, S., Everett, A., Coats, R., Wagenbrenner, J.W., Lewis, S.A., Ashmun, L. E., Brown, R.E. 2011. Evaluating post-fire salvage logging effects on erosion.JFSP Research Project Reports. 66.

Rodrigues, M., Jiménez-Ruano, A., Peña-Angulo, D., de la Riva, J., 2018. A comprehensive spatial-temporal analysis of driving factors of human-caused wildfires in Spain using Geographically Weighted Logistic Regression. Journal of Environmental Management 225, 177-192.

Rodriguez-Trejo, D.A., 2012. Génesis de los incendios forestales. Revista Chapingo serie ciencias forestales y del ambiente 18 (3), 357-373.

Romero,, F., Senra, F., 2006. Grandes incendios forestales. Causas y efectos de una ineficaz gestión del territorio. WWF/Adena 34 pp.

Saá, A., Trasar-Cepeda, M.C., Carallas, T., 1998. Soil P status and phosphomonoesterase activity of recently burnt and unburnt soil following laboratory incubation. Soil Biology and Biochemistry 30, 419-428.

Santín, C., Doerr, S.H. 2016. Fire effects on soils: the human dimension. Phil. Trans. R. Soc. B, 371(1696), 20150171.

Serrasolses, I., Vallejo, V. R. 1999. Soil fertility after fire and clear-cutting. In Ecology of Mediterranean Evergreen Oak Forests (pp. 315-328). Springer, Berlin, Heidelberg.

Shive, K. L., Fulé, P. Z., Sieg, C. H., Strom, B. A., Hunter, M. E. 2014. Managing burned landscapes: evaluating future management strategies for resilient forests under a warming climate. International Journal of Wildland Fire 23 (7), 915-928.

Simard, S.J. 1991. Fire severity, changing scales, and how things hang together. International Journal of Wildland Fire 1 (1), 23-34.

Soto, B., Benito, E., Díaz-Fierros, F. 1991. Heat-induced degradation processes in forest soils. International Journal of Wildland Fire 1 (3), 147-152.

Soto, B., Diaz-Fierros, F. 1993. Interactions between plant ash leachates and soil. International Journal of Wildland Fire 3 (4), 207-216.

Spanos, I., Raftoyannis, Y., Goudelis, G., Xanthopoulou, E., Samara, T., Tsiontsis, A. 2005. Effects of postfire logging on soil and vegetation recovery in a Pinus halepensis Mill. forest of Greece. Plant and Soil 278 (1-2), 171-179.

Stanley, D.M., Negri, J.A., Kean, J.W., Laber, J.L., Tillery, A.C., Youberg, A.M., 2016. Prediction of spatially explicit rainfall intensity-duration thresholds for post-fire debris-flow generation in the western United States. Geomorphology 278, 149-162.

Stevens-Rumann, C., Shive, K., Fulé, P., Sieg, C. H. 2013. Pre-wildfire fuel reduction treatments result in more resilient forest structure a decade after wildfire. International Journal of Wildland Fire 22 (8), 1108-1117.

Strom, B. A., Fulé, P. Z. 2007. Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. International Journal of Wildland Fire 16 (1), 128-138.

Strauss. D., Bednar, L., Mees, R., 1989. Do one percent of the forest fires cause ninety –nine percent of the damage? Forest Science 35 (2), 319-328.

Summers, W., Coloff, S., Conard, S.G., 2011. Fire history and global change. Joint fire science program. Project 09-2-01-09. 190 pp.

Tsibart, A.S., Gamova, N.S., Koshovskii, T.S., Gennadiev, A.N., 2015. Wildfires in Russia: features, regimes and consequences. In: Bento-Gonçalves, A., Vieira, A. (Eds.). Wildland fires – a Worlwide reality. Nova Science Publishers. pp. 199-210.

Úbeda, X., 1998. Efectes de les diferents intensitats de foc, Durant els incendis forestals, en els paràmetres físics i químics del sòl i en l’increment de l’escolament i de l’erosió. Tesi Doctoral. Universitat de Barcelona. 231 pp.

Úbeda, X., Francos, M. 2018. Incendios forestales, un fenómeno global. *Biblio3w, Revista Bibliográfica de Geografía y Ciencias Sociales*. Barcelona: Universidad de Barcelona, vol. XXIII, 1.253. [ISSN: 1138-9796].

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

Úbeda, X., Pereira, P., Outeiro, L., Martin, D.A. 2009. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (Quercus suber). Land degradation & development, 20 (6), 589-608.

Úbeda, X., Sala, M., 2001. Chemical concentrations in overland flow from different forested areas in a Mediterranean Environment: burned forest at different fire intensity and unpaved road. Zeitschrift für Geomorphologie NF, 225-238.

Úbeda, X., Sala, M., Imerson, A. 1990. Variaciones en la estabilidad y consistencia de un suelo forestal antes y después de ser sometido a un incendio. I Reunión Nacional de Geomorfología. Teruel. pp. 667-685.

Úbeda, X., Sarricolea, P., 2016. Wildfires in Chile: A review. Global and Planetary Change 146, 152-161.

Ulery, A.L., Graham, R.C., Amrhein, C. 1993. Wood-ash composition and soil pH following intense burning. Soil science, 156 (5), 358-364.

Valendik, E.N., Verkhovets, S.V., Ponomarev, E.I., Ryzhkova, V.A., Kisilyakhov, Y.K., 2014. Large wildfires in Taiga subzones of central Siberia. Journal of Siberian Federal University. Biology 7. p. 43.

Vallejo, V.R., Alloza, J.A. 1998. The restoration of burned lands: the case of eastern Spain. Large forest fires, 91-108.

Vance, E.D., Brookes, P.C., Jenkinson, D.S., 1987. An extraction method for measuring soil microbial biomass C. Soil Biol Biochem. 19, 703-707.

Vanha-Majamaa, I., Lilja, S., Ryoma, R., Kotiaho, J.S., Laaka-Lindberg, S., Lindberg, H., Puttonen, P., Tamminen, P., Toivanen, T., Kuuluvainen, T., 2007. Rehabilitation boreal forest structure and species composition in Finland through logging, dead Wood creation and fire: The EVO experiment. Forest Ecology and Management 250, 77-88.

Varela, M. E., Benito, E., Keizer, J. J. 2015. Influence of wildfire severity on soil physical degradation in two pine forest stands of NW Spain. Catena 133, 342-348.

Vélez, R., 2000. Los incendios forestales en la cuenca mediterránea. Introducción. En: Vélez, R. (Ed.). La defensa contra los incendios forestales. Fundamentos y experiencias. McGraw-Hill. 3-1.

Versini, P.A., Velasco, M., Cabello, A., Sempere-Torres, D., 2013. Hydrological impact of forest fires and climate change in Mediterranean basin. Nat. Hazards 66, 609-628.

Vierek, L.A., Wildfire in the taiga of Alaska. Quaternary Research 3 (3), 465-495.

Volkova, L., Meyer, C. M., Murphy, S., Fairman, T., Reisen, F., Weston, C. 2014. Fuel reduction burning mitigates wildfire effects on forest carbon and greenhouse gas emission. International Journal of Wildland Fire 23 (6), 771-780.

Wagenbrenner, J. W., MacDonald, L. H., Coats, R. N., Robichaud, P. R., Brown, R. E. 2015. Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. Forest Ecology and Management 335, 176-193.

Wagenbrenner, J. W., Robichaud, P. R., Brown, R. E. 2016. Rill erosion in burned and salvage logged western montane forests: Effects of logging equipment type, traffic level, and slash treatment. Journal of Hydrology 541, 889-901.

Wagenbrenner, J. W., Robichaud, P. R., MacDonald, L. H., Brown, R. E. 2014. Effects of Post-Fire Salvage Logging on Erosion Rates at Multiple Scales. In AGU Fall Meeting Abstracts.

Wessel, A.T., 1988. On using the effective contact angle and the water drop penetration time for classification of water repellency in dune soils. Earth Surface Processes and Landforms 13 (6), 555-561.

Wittenberg, L., Malkinson, D., Beeri, O., Halutzy, A., Tesler, N. 2007. Spatial and temporal patterns of vegetation recovery following sequences of forest fires in a Mediterranean landscape, Mt. Carmel Israel. Catena 71 (1), 76-83.

Zhang, R., 1997. Determination of soil sorptivity and hydraulic conductivity from the disk infiltrometer. Soil Sci. Soc. Am. J. 61 (4), 1024-1030.