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Effects of wildfire on soil nutrients in Mediterranean ecosystems

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A B S T R A C T

High-intensity and fast-spreading wildfires are natural in the Mediterranean basin. However, since 1960, wildfire occurrence has increased because of changes in land use, which resulted in extensive land abandonment, increases in the fuel load and continuity in the landscape. The level of soil degradation related to wildfire occurrence depends on fire recurrence, topography of the site, intensity of the soil erosion processes and plant cover post-fire regeneration rate. Therefore assessing fire impacts on soil properties is critical to quantify land degradation processes and to assess post-fire restoration plans. This article reviews the changes in soil nutrient status of Mediterranean ecosystems affected by wildfires by focusing on the interactions between the different drivers and factors, and the underlying processes of these changes. Articles dealing with wildfires in areas belonging to the Mediterranean basin and characterized by an annual average rainfall of 300–900 mm and a mean annual temperature around 14–19 °C, have been reviewed. The data show that the soil nutrient content in Mediterranean drylands affected by wildfires depends on the vegetation type, fire recurrence and fire intensity. Immediately after a fire, the nutrient content in both the O and A horizons often increases because of ash deposition, nutrient release from the burnt vegetation and formation of stable nutrient forms. Ash deposition persistence on the soil surface is one of the most important factors in determining the soil nutrient content both immediately after a fire and for the long-term. For the restoration of burned habitats it is important to know the content and the spatial distribution of nutrients in the soil because this can act as a limiting factor to vegetation recovery. Carbon and nitrogen pools in the soil have been recognized as fundamental to vegetation recuperation after a fire. To promote the accumulation and retention of nutrients in soil after a fire, it is important to stabilize the burnt site by applying post-fire measures that limit soil erosion, surface runoff and wind loss of the ash. Depending on the plant species and the time elapsing between consecutive wildfires, fire is responsible for the transition from mature ecosystems (i.e. conifer forests) to shrublands, which are poorer in soil nutrient status. Wildfire occurrence can be reduced by planting fire-resilient plants in fire-prone areas. To define the best post-fire and restoration treatments, the impacts of fire on both the O and the A horizon as well as the impacts of different post-fire treatments on the soil nutrient content require further study.

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1. Introduction

Mediterranean basin ecosystems to a large extent belong to the dry sub-humid climate and cover a land area of about 124 Mha (Scarascia-Mugnozza et al., 2000; FAO, 2004). Although high-intensity and fast spreading wildfires are natural factors in the Mediterranean ecosystem, fire prevention and biodiversity conservation policies aim to reduce wildfire occurrence and extent in the burned area (Fernandes, 2012; Carreiras et al., 2014). Since 1960, changes in land use have resulted in extensive land abandonment, increased fuel load and continuity in the landscape (Moreira et al., 2011), factors considered to be the main drivers of increased wildfire occurrence and resulting risk of soil degradation (Giovannini et al., 2001; Pausas et al., 2004; Campo et al., 2006; Pausas et al., 2008; Raison et al., 2009; Shakesby, 2011; Castro et al., 2012). Based on climate change projections, more frequent and longer droughts would further increase the risk of wildfire occurrence in the Mediterranean basin and delay plant recovery (Doerr and Cerdà, 2005; Mayor et al., 2007; Vallejo et al., 2012; Koutsias et al., 2013).

Frequent wildfires show indications of triggering soil degradation by reducing or temporarily eliminating the vegetation cover and expose the soil to erosion agents such as water and wind. In addition to the increased risk of soil erosion, frequent wildfires also induce changes in the water cycle by altering the infiltration capacity of the soil and increasing soil hydrophobicity (Vallejo and Alloza, 1998; Fernández et al., 2012; Carreiras et al., 2014). Therefore assessing the impact(s) of wildfires on soil properties is critical to quantify land degradation processes and to assess post-fire restoration plans (Jurgensen et al. 1997 cited in Marañón-Jiménez et al., 2013). By changing physical, chemical, mineralogical, and biological soil properties (Doerr and Cerdà, 2005), a fire leads to a net change in available and total nutrients in the soil in the short- and long-term (Johnson et al., 1998; Raison et al., 2009; Turrion et al., 2010; Tessler et al., 2012; Marañón-Jiménez et al., 2013). Fire mobilizes nutrients and increase or decrease their availability in the ecosystem (Thomas et al., 1999; Bento-Gonçalves et al., 2012). Fire intensity, duration and recurrence have been identified as main drivers of change in soil nutrient content (Kutiel and Shaviv, 1992; Martín et al., 2012; Guénon et al., 2013). Post-fire soil nutrient content is also influenced by factors such as topography, vegetation type, thickness of the litter layer and soil type (Kutiel and Shaviv, 1992; Wan et al., 2001; Arnan et al., 2007; Martín et al., 2012). The interaction between the different drivers and factors results in processes such as nutrient release from burned vegetation, changes in nutrient form (i.e. formation of black carbon and recalcitrant nutrient forms), nutrient volatilization and nutrient losses by wash off, wind transport of the ash and/or soil erosion. Right after a fire, the soil is vulnerable to wind and water erosion, especially if heavy rains and storms occur (Kutiel and Naveh, 1987a,b; Cerdà, 1998; Pausas et al., 2008). The loss of nutrients by soil erosion and the amount of nutrients lost by burning depends on the vegetation type because of the interrelation between plant canopy, litter and upper soil layer, which is the main layer affected by a fire (Raison, 1979; Cerdà and Lasanta, 2005; Cerdà and Doerr, 2008). Fire effected changes in soil nutrient status have short and long-term implications for soil productivity and the greater environment.

This article reviews the current knowledge regarding changes in soil nutrient status of Mediterranean drylands affected by wildfires through focusing on the interactions between the different drivers and factors, and the underlying processes of these changes. The criteria to study the effect of wildfires on soil nutrients in the Mediterranean basin were: areas characterized by an annual average rainfall of 300–900 mm and a mean annual temperature around 14–19 °C. In order to compare the data in the articles, unburned control sites needed to be similar in soil type, topography, geomorphology and vegetation composition to the burned plots and had to be closely located. Searching the keywords "(wild)fire", "Mediterranean climate", "Mediterranean basin", "soil", "soil nutrients", "nitrogen", "carbon", "phosphorus", "exchangeable cations" and "vegetation (cover)" in the literature bank of Scopus and Google Scholar resulted in more than 160 articles published in the period 1960–2014 that fulfilled our initial criteria. Studies carried out in other Mediterranean-type climates (MTC) such as Sierra Nevada (USA) and Australia have been used to support data coming from the Mediterranean basin and especially the data from Spain, Turkey and Israel. The study areas were mainly characterized by shrublands (*Quercus coccifera* L., *Cistrus clusii*, *Ulex rivasgodayanus*, and *Rosmarinus officinallis*), pine (*Pinus pinea* L., *Pinus brutia*, *Pinus halepensis*, *Pinus canariensis*, and *Pinus pinaster*) and oak (*Quercus suber* and *Quercus ilex*) forests. Data coming from wet Mediterranean areas such as northern Portugal and Spain (i.e. annual average rainfall of 1600–1800 mm) were not considered because the vegetation was not typical of Mediterranean drylands. The data in those papers meeting all criteria were used in this review, the others provided background information.

The effects of wildfires on the nutrient content of the O horizon as well as of the A_1 horizon were analyzed taking into account organic matter, nitrogen, phosphorus and exchangeable cations content such as potassium, sodium, calcium and magnesium. Depending on the source, data showed the change in soil nutrient status immediately after fire as well as in the short- (within one year after fire) and long-term (later than one year after fire). A few studies also allowed tracking of the change in soil nutrient content on the basis of fire recurrence. With respect to drivers such as fire intensity and recurrence, and to factors such as the vegetation type, the main changes in soil nutrient status after fire are presented. By looking at the interactions between drivers and factors, the underlying processes in soil nutrient status after fire are also indicated. On the basis of the review, this article provides considerations for the restoration of Mediterranean drylands affected by wildfires.

2. Wildfire as one of the main soil degradation drivers in drylands

Based on the Koppen–Geiger climate classification, drylands are typical of the Mediterranean climate (classes Csa and Csb) and are characterized by hot dry summers and wet winters (Kottek et al., 2006; Peel et al., 2007). The main vegetation types are: shrublands, savanna woodlands and forests (Keeley et al., 2012). Even though geomorphology and soil type change with the location, soils in drylands are usually not very productive because, in addition to the high rainfall variability and water scarcity, they are low in organic matter content, nutrient content and aggregate stability (FAO, 2004; Millennium Ecosystem Assessment, 2005; Koohafkan and Stewart, 2008; Vallejo et al., 2012). In addition, owing to the long history of extensive land use, uncropped soils are usually shallow and/or stony, and develop on steep slopes (Vallejo and

Alloza, 1998). FAO (2004) and Reynolds et al. (2007) stated that desertification is a major problem in degraded drylands, which take 5–10 times longer to naturally regenerate vegetation cover and soil than wetter environments (United Nations, 2013). Although the vegetation is adapted to wildfires, frequent wildfires can foster land degradation and desertification in Mediterranean drylands by hindering vegetation recovery (Vallejo et al., 2012).

Wildfires may show low or high severity (degree of organic matter consumed above- and belowground) and intensity (heat per area per time unit) (Ice et al., 2004; Certini, 2005; Keeley, 2009; Shakesby, 2011). Fire intensity, duration and recurrence are main factors affecting soil nutrient status (Kutiel and Shaviv, 1992; Martín et al., 2012; Guénon et al., 2013). According to Novara et al. (2011) and Wan et al. (2001), the O horizon and the upper 0–2 cm of the A horizon are the most affected by fire and the ones in which major changes in soil nutrient content are measured. Indeed, depending on the fire intensity and duration, the temperature reached by the soil during a fire decreases more or less sharply with increasing soil depth. Neary et al. (1999) reported that in a chaparral shrubland, light intensity wildfires reached 250 °C at the soil surface, 100 °C at 2.5 cm soil depth and <50 °C at 5 cm soil depth. Moderate intensity wildfires reached 400 °C at the soil surface, 175 °C at 2.5 cm soil depth and 50 °C at 5 cm soil depth. High intensity wildfires reached 675 °C at the soil surface, 190 °C at 2.5 cm soil depth and 75 °C at 5 cm soil depth. Although high intensity wildfire can reach 900 °C at the soil surface depending on the vegetation type, soil temperature at 0–5 cm soil depth rarely exceeds 150 °C (Ice et al., 2004; Certini, 2005). Belowground changes occur as a function of the intensity and duration of the fire and are minor in the case of fast moving fires, which affect only the first few centimeters below the soil surface (Neary et al., 1999; Certini, 2005). Dry soil is a poor conductor of heat and often no heating occurs deeper than 20–30 cm depth (Ice et al., 2004; Certini, 2005). On the basis of fire severity and frequency, and post-fire weather conditions, a fire can cause short-term, long-term or permanent fire-induced changes in physical, chemical, mineralogical and biological soil properties (Lavabre et al., 1993; Certini, 2005; Ekinci and Kavdir, 2005; Pausas et al., 2008; Turrion et al., 2010; Shakesby, 2011).

Ecosystem resilience depends on the size and intensity of the fire, which is related to land use, vegetation cover, topography, exposure and weather conditions. Vegetation cover is one of the most important factors involved in the change of the soil nutrient status after fire. Shrubland, grassland and especially conifer forests have been identified as the most prone vegetation types to burn in drylands because of plant flammability, extension and continuity on the territory of these land covers (Oliveira et al., 2014). Oliveira et al. (2014) stated that regardless of the vegetation type, gentle slopes are more fire prone than areas with steeper slopes (>25%) because of fuel accumulation. Moreover vegetation flammability is related to the direction of the slope and north-facing slopes, receiving less solar radiation, are usually more humid and less fire prone than south-facing slopes (Shakesby, 2011; Oliveira et al., 2014). South-facing slopes are usually characterized by higher soil erosion rates than north-facing slopes because the low plant recovery rate leaves the soil more exposed to erosional agents (Marques and Mora, 1992; Andreu et al., 2001; Pierson et al., 2002; Buhk et al., 2006). Besides its direct impact on vegetation regrowth and fuel accumulation, topography is a key factor in determining fire spread and intensity (Bennett et al., 2010).

Many processes occur during and after a fire. Several studies showed that there is a relation between fire intensity, fire recurrence and available nutrients in the topsoil. In high severity surface fires, the litter layer on the soil surface characterizing forest soils is replaced by an ash layer. This ash layer contains carbon and nutrients from the burnt fuel of the forest, the former litter layer and the burnt topsoil that have not been lost via volatilization and/or smoke convection (Certini, 2005; Shakesby, 2011). The quality and quantity of the ash depend on vegetation type and fire severity, which can be assessed by ash color (higher the severity of the fire, lighter the color of the ash) (Marion et al., 1991; Pereira et al., 2012; León et al., 2013). Studying the effect of fire severity on ash chemical and extractable elements, Pereira et al. (2012) related low severity fires to very dark brown ash, medium severity fires to black and very dark gray ash, and high severity fires to dark gray, light gray and white ash. Depending on fire intensity, phosphorus can be converted to apatite, which is not plant-available in the short-term (Hirschler et al., 1990; Guidry and Mackenzie, 2000; Johnston, 2001; Piccoli and Candela, 2002). Additionally, carbon can be converted to black carbon (BC) and charcoal, recalcitrant forms of carbon highly resistant to microbiological and chemical decomposition (Skjemstad et al., 2002). Carbon and nitrogen are also usually lost by volatilization. The loss of nutrients by volatilization depends on the specific nutrient's temperature of volatilization, i.e. nitrogen and carbon, the most affected nutrients by fire cycles, are volatilized at 200–500 °C (Wan et al., 2001; Ice et al., 2004; Ferran et al., 2005; Pereira et al., 2014; Marañón-Jiménez et al., 2013). Calcium and magnesium are not lost by volatilization because of their high volatilization temperature (Britt, 2007). Kutiel and Naveh (1987b) stated that the flush of available nutrients immediately after a fire can be followed by a rapid growth of herbaceous plants and by a significant increase in plant storage of these same elements. As a consequence, the mineral elements mobilized from the burned woody plants and their litter can return to the system via the herbaceous post-fire flush. Therefore, short-term vegetation recovery is critical in post-fire nutrient conservation. Although in the short-term the quantity of available nutrients in the form of water-soluble ash components tends to increase with increasing fire intensity and occurrence, the ash layer can be easily washed off by surface runoff or soil erosion, or blown away by the wind (Cerdà and Doerr, 2008; Bodí et al., 2014). Ash losses can affect long-term ecosystem nutrient capital and slow down plant growth (Giovannini et al., 1987; Marion et al., 1991; Certini, 2005; Ekinci and Kavdir, 2005; Pausas et al., 2008; Bodí et al., 2011; Shakesby, 2011).

By altering vegetation and soil properties, a fire changes the hydrologic regime and the sediment transport dynamics of the burned site (Swanson, 1981; Cerdà and Lasanta, 2005; Keesstra et al., 2014). Studying the hydrological response of a small Mediterranean basin to wildfire, Lavabre et al. (1993) underlined the significant increase in the annual runoff yield and in the flood regimes after a fire because of the temporal reduction of the vegetation cover and the subsequent reduced evapotranspiration rate (Lavabre et al., 1993; Neary et al., 2011). Although ash deposition can reduce soil losses (Cerdà and Doerr, 2008; Pérez-Cabello et al., 2012; León et al., 2013; Pereira et al., 2013), wildfires often leads to high soil erosion rates especially during the first year after a fire (Shakesby, 2011; Fernández et al., 2012). The average soil erosion rate for the European Mediterranean region has been estimated to be 56 t ha^{-1} (Shakesby, 2011). According to Ekinci and Kavdir (2005) and Shakesby (2011), the soil erosion rate after a fire increases because of the decreased stability of soil aggregates, which is related to the reduction in the amount of soil organic matter (SOM). Due to the loss of SOM, large unstable aggregates (>250 μm) are formed. Although the effects of wildfire on soil aggregates stability are highly variable (Mataix-Solera et al., 2011), Varela et al. (2010) showed that soil aggregate stability is first related to the combustion of the organic matter and then to the direct thermal effect of fire intensity. In contrast to the effects of high intensity wildfires, the occurrence of low-intensity wildfires (temperatures lower than 300 °C) increases aggregate stability by condensing hydrophobic substances onto aggregates, with the consequence that soil water repellency is also likely to increase (DeBano, 2000; Mataix-Solera and Doerr, 2004; Moody et al., 2009; Mataix-Solera et al., 2011; Stoof et al., 2011; Ebel et al., 2012; Rodríguez-Alleres et al., 2012). Related to this, Imeson et al. (1992) and others have stated that the spatial heterogeneity in soil water repellency caused by a fire leads to discontinuous runoff processes.

Frequent fires present a threat to soil conservation in areas characterized by significant climatic fluctuations such as the Mediterranean basin (Johnson et al., 1998; Andreu et al., 2010; Shakesby, 2011; Vallejo et al., 2012). Biomass burning mainly releases water, carbon dioxide, carbon monoxide, methane, nonmethane hydrocarbons(excluding isoprene and terpenes), nitric oxide, ammonia, sulfur gases, methyl chloride, hydrogen, tropospheric ozone, elemental carbon and atmospheric particulate matter (Levine et al., 1995; Garcia-Hurtado et al., 2013; Garcia-Hurtado et al., 2014). According to Certini (2005) and Marañón-Jiménez et al. (2013), charred large woody materials act as reservoirs of nutrients in ecosystems affected by wildfire because they are only superficially burned and gradually release nutrients by decomposition. Additionally, Kutiel and Shaviv (1992) stated that most of the available nutrients in soil are lost at temperature ranging from 250 °C to 600 °C. Prescribed fires characterized by low/moderate severity and occurring under controlled conditions in specific places, are often set up to reduce potential wildfire hazard, limit soil and nutrient losses and restore drylands' vegetation (Elmore and Hitt, 2010). They are able to control undesired vegetation, and increase soil pH(H_2O) and soil available nutrients. Moreover, when compared to wildfires, the low fire intensity (maximum 200–300 °C) characterizing prescribed fires generally decreases the soil erosion rate by lowering the rate of organic matter lost, which allows plants to more quickly recolonize the burned area (Certini, 2005; Ekinci and Kavdir, 2005; Shakesby, 2011; Castro et al., 2012). The adoption of effective fire management practices (i.e. keeping charred woody materials on soil surface, seeding and mulching) is fundamental in preserving soil fertility and promoting vegetation recovery (Certini, 2005; Fernández et al., 2012; Marañón-Jiménez et al., 2013).

The main drivers, factors and processes involved in the change of the soil nutrient status after a fire are shown in Fig. 1.

3. Vegetation restoration after a fire

Wildfire is recognized as an important feature in the vegetation dynamics of Mediterranean drylands. Although many Mediterranean plant communities show high resilience to fire, the increased risk for fire and drought occurrence because of climate change and human activities, may negatively affect the regeneration of the burned vegetation (de Luís et al., 2005; Romanya et al., 2007; Fernandes, 2012; Vallejo et al., 2012; Carreiras et al., 2014). According to de Luís et al. (2005), the time elapsing between two consecutive fire events is getting shorter in some areas of the Mediterranean basin. With respect to *P. halepensis*, one of the most common tree species in the Mediterranean basin, high intensity and repeated fires could lead to the disappearance of the plant community if the interval between two fire events is shorter than 15 years (time required by the pine tree to produce cones) (Pausas et al., 2004). The disappearance of tree communities would foster the diffusion of shrublands in the Mediterranean basin (Fernandes, 2012; Santana et al., 2013), which are more fire prone than forests (Romanya et al., 2007). In addition, by producing less litter and fine roots than woody species, the soil nutrient content would decrease under shrubs (López et al., 1998; Romanya et al., 2007). According to Thomas et al. (2000), litter can significantly reduce particulate nutrient losses and solute losses in pine forests. Among other nutrients, carbon and nitrogen pools in soil have been recognized as fundamental to vegetation recovery after a fire in the long-term (Romanya et al., 2007; Rau et al., 2010). For the restoration of burned habitats it is important to know the content and the spatial distribution of nutrients in the soil because this can act as a limiting factor to vegetation recovery. In addition, knowing the short- and long-term impacts of fire on the soil nutrient content

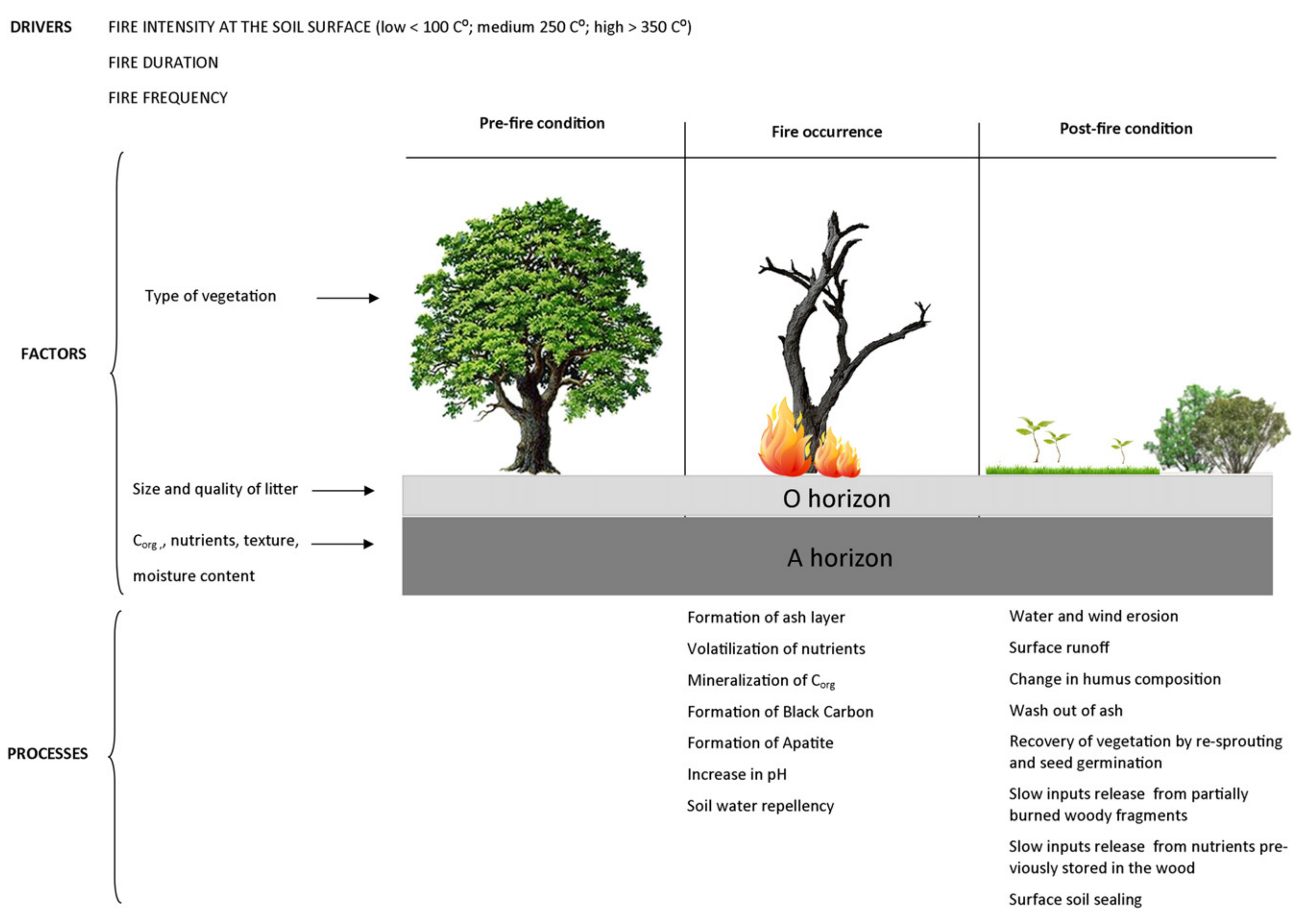

Fig. 1. Main drivers, factors and processes involved in changes in soil nutrient status after a fire.

is of value for assessing or selection of post-fire treatments aimed at promoting the accumulation and retention of nutrients in soil (Rau et al., 2010; Bento-Gonçalves et al., 2012). Raftoyannis and Spanos (2005) stated that post-fire treatments such as log and branch barriers can negatively affect the re-vegetation process by interfering with *Q. coccifera* growth and thus delaying or reducing ecosystem recovery. Although fire suppression is currently promoted in the Mediterranean basin, a complementary option may be the vegetation (fuel) control approach through spread out of autochthonous less-flammable plant species, such as *Juniperus oxycedrus*, and creation of more fire-resilient landscapes (Dimitrakopoulos and Papaioannou, 2001; Fernandes, 2012; Vallejo et al., 2012). The need to move toward more immediate, short-term intervention measures directly after a fire, e.g. to control and reduce soil erosion so that vegetation can more quickly re-establish, rather than to focus only on long-term restoration strategies has been underlined by Bento-Gonçalves et al. (2012) among others.

4. Effects of wildfires on soil organic carbon and nutrients

Soil nutrient changes are studied in both, the O horizon and the A horizon. With respect to the A horizon, data in the papers reviewed refer to a maximum depth of 0–15 cm. The organic matter content has been quantified by the Walkley and Black method, potentiometric titration, dry combustion and (CPMAS)^{13}C Nuclear Magnetic Resonance Spectroscopy. Nitrogen content has been quantified by direct UV absorption at 210 nm, colorimetric analysis, as well as the Kjeldhal and Griess–Ilosvay methods, and expressed as nitrate, nitrite, ammonium, nitrate-N or ammonium-N. Total phosphorus content has been determined by all authors using sulfuric acid–hydrogen peroxide–hydrofluoric acid digestion. Although the majority of the sources used the Olsen-P method to quantify the available phosphorus content in neutral to basic soils, a few authors used the Mehlich-1 method for acidic soils. The ammonium acetate method and the barium chloride–triethanolamine method were used to determine the exchangeable cation content on acidic and calcareous soils, respectively. Besides difficulties coming from differences in the method used, not all the papers reviewed provided quantitative data. Some authors preferred to underline the difference (in %) in nutrient content between burned and unburned plots rather than indicate the quantity of the nutrient in soil before and after burning. Data available were not always comparable because of differences in unit of measurement, accuracy of the method used (margin of error, correction factors) and differences in the experimental design (sampling time, soil depth, choice of the control site). Even when different authors used the same method to quantify the change in nutrient content, differences in sampling time and procedure (i.e. soil depth) often made detailed comparison of the data difficult.

4.1. Soil organic carbon (SOC)

Soil organic matter is an important component in ecosystem dynamics because of its ability to exchange ions, interact with clay minerals, form soil aggregates, absorb and release plant nutrients and hold water. Plant and animal decomposition leads to the formation of soil organic carbon, which is usually scarce in arid regions due to the poor vegetation cover and thus the low litter inputs (Schitzer, 1982; Romanya et al., 2007). The stage of growth of the plant community is also an important factor to consider when determining the soil nutrient content because it is strongly related to the amount of inputs available to the soil. A study conducted in Spain by Romanya et al. (2007) showed that in semiarid conditions, mature hardwood forests have higher soil organic carbon than recently established conifer forests. Soil organic carbon content in grass patches usually ranges from 2.9 g 100 g^{-1} in semiarid conditions to 5 g 100 g^{-1} in dry sub-humid Mediterranean climate zones. In forests and shrublands, SOC content ranges from 2.4 g 100 g^{-1} in semiarid conditions to 5.6 g 100 g^{-1} in dry sub-humid Mediterranean climate zones (Romanya et al., 2007). Romanya et al. (2007) claimed that lower C losses are expected in degraded ecosystems than in mature ecosystems after a fire.

4.1.1. Soil organic C content after a fire in the O horizon

The effects of fire on SOC are highly dependent on, along with other factors, type of fire (crown, surface), fire duration and intensity, soil moisture, soil type and vegetation. In addition, the effects are usually very different in the O horizon than the A horizon. With respect to the O horizon, fire can cause a significant reduction in SOC content in the short-term because of mineralization of C from the organic matter, followed by volatilization, ash removal by wind, soil erosion and surface runoff. Over the long-term, SOC content can increase because of incorporation of the ash into soil, vegetation recovery and decomposition of partially burned woody fragments.

Immediately after a fire, SOC content has been found to be significantly lower in the O horizon of a burnt soil than in the unburnt soil in mixed conifer forests as well as in pine forests and shrublands (DeBano and Conrad, 1978; Caldwell et al., 2002; Murphy et al., 2006). Although one week after a low intensity wildfire SOC content was steady in shrublands, it increased one year later because of the incorporation of the ash into the O horizon and the formation of charcoal (Castelli and Lazzari, 2002). Two as well as three years later, the authors reported similar SOC contents in both the burnt and the control sites. One year after the fire, Mataix-Solera et al. (2002) and Johnson et al. (2007) measured lower SOC contents in the O horizon of a burnt soil than in the unburnt soil in pine forests. In the long-term (10–20 years) Johnson et al. (2005) and Kaye et al. (2010) measured significantly higher SOC contents in the O horizon of burnt soil than in unburnt soil under shrubs. Ferran et al. (2005) noticed that the organic matter content in shrublands affected by wildfires decreases in the L, F and H (O) horizons as the number of fire events increases. The same trend was observed for the litter mass, which was significantly lower in plots affected by 3 wildfire events than in other plots where there were fewer fire events occurred (Carreira et al., 1994; Guénon et al., 2011; Tessler et al., 2012).

4.1.2. SOC content after a fire in the A horizon

SOC changes in the A horizon depend on the depth of burning in the soil (C combustion starts at 200 °C), the incorporation of partially burned litter and wood, the regeneration of litter inputs (depending on the type of vegetation) and on changes in the organic matter decomposition rate caused by a fire, e.g. pH and soil moisture. Although SOC content tends to decrease after a fire in both the short- and the long-term for high severity fires, it may show no significant differences because of the formation of the so called Black Carbon (BC) (Moghaddas and Stephens, 2007; Nave et al., 2011). BC belongs to those fire-generated, highly condensed and oxidation and biological degradation resistant forms of C which are assumed to increase the passive soil OM pool. According to Forbes et al. (2006), the char, ash and charcoal produced during burning of the organic matter are commonly referred to as BC. The charcoal component is particularly important because it represents a form of C that is relatively inert, so that it works as a C sink (MacKenzie et al., 2008). In addition, increases in SOC content are related to decomposition of partially burned woody fragments (Almendros et al., 1990).

While most studies reviewed found lower SOC content in the A horizon following a fire, this was not always the case. Immediately after the occurrence of a high intensity wildfire in shrublands, the SOC content in the A horizon (0–8 cm and 0–15 cm) has been found to be significantly lower in the burnt soil than in the unburnt soil (Badía and Martí, 2003; Rau et al., 2010). At all fire intensities, SOC content of the A horizon (0–5 cm) was significantly lower in the burnt plots than the unburnt plots one month after fire in a mixed oak and pine forest (Kutiel and Naveh, 1987b). The same trend was measured in the 0–5 cm A horizon under pine trees, nine months after a low intensity wildfire (Hernandez et al., 1997). One year after fire, significantly lower SOC contents were measured in the burnt soil in mixed conifer forests (0–4 cm and 4–15 cm) as well as in pine forests (0–10 cm) and mixed oak and pine forests (0–5 cm) (Kutiel and Naveh, 1987b; Mataix-Solera et al., 2002; Johnson et al., 2007). SOC content also changes depending on the season of fire because of differences in fire intensity and post-fire weather conditions. Hamman et al. (2008) stated that in a mixed oak and conifer forest, the impact of early season burns is not as significant as the impact of late season burns on the total SOC. With respect to samples collected one year after a fire in the A horizon (0–15 cm), early season burns did not significantly alter the total SOC content. However, late season burns did significantly lower the total SOC content in the soil. The same decrease in SOC content in the 0–15 cm soil depth was observed by Brockway et al. (2002) two years after a fire in a pine forest. In a pine forest, Mataix-Solera et al. (2002) also measured lower SOC contents in the burnt soil (0–4 cm and 4–15 cm) 18 months after a fire. The same trend in the A horizon (0–5 cm) was measured in both a pine forest and a mixed pine and oak forest by Ekinci and Kavdir (2005) and Kavdir et al. (2005), eight years and 12 years after a fire, respectively. Twenty years after the occurrence of a wildfire in shrublands, Johnson et al. (2005) measured significantly lower SOC contents in the burnt soil than in the unburnt soil (0–7 cm).

In some cases, SOC showed a different response to a fire. Carreira et al. (1994) and Ferran et al. (2005) looked at the change in SOC content under shrubs with respect to a fire recurrence. In general, Ferran et al. (2005) measured higher organic carbon contents in the burned soil than in the unburned soil (0–2.5 cm and 2.5–10 cm). Since in plots being affected by 2 wildfires (0–2.5 cm) the organic carbon content was lower in the burned soil than the unburned soil, Ferran et al. (2005) stated that there was not a clear relation between the SOC content and a fire recurrence. However, Carreira et al. (1994) measured lower SOC contents in the burned soil than in the unburned soil (0–5 cm). According to Carreira et al. (1994), Guénon et al. (2011) and Tessler et al. (2012) the organic carbon content decreases with increasing fire recurrence and its change in soil is highly related to fire frequency. Indeed these authors found that plots being affected by 3 wildfires had significantly lower SOC content than plots being affected by only one wildfire. There are still questions to be answered about the factors affecting SOC content in the A horizon as affected by fire.

4.1.3. Underlying processes

Romanya et al. (2007) showed that the impact of fire on SOC content is greater in mature ecosystems than in degraded ecosystems. Taking into consideration that in semiarid areas the potential for carbon sequestration is lower than in other ecosystems, fire decreases the total organic carbon content in the soil by interrupting the carbon inputs to the soil and by increasing the risk for soil erosion (Romanya et al., 2007; Carreiras et al., 2014). Several authors showed that both, the O and A horizons recover slowly from C losses in the short- as well as in the long-term. In the short-term, organic C losses are mostly related to burning of the organic matter, volatilization and ash removal. By affecting vegetation recovery, soil erosion was responsible for lowering SOC content of the burnt soil in the long-term (Guénon et al., 2013; Hamman et al., 2008; Novara et al., 2011). SOC increases were related to charcoal formation, incorporation of ash into soil, decomposition of partially burnt woody fragments and vegetation recovery (Kutiel and Naveh, 1987b; Krull et al., 2003; González-Pérez et al., 2004; Kavdir et al., 2005; Forbes et al., 2006). Comparing plots characterized by different fire histories, Kavdir et al. (2005) stated that the amount of charcoal in the soil was higher in recently burned plots than in old burnt plots. According to Kaye et al. (2010), while shrubs have a limited capacity to accumulate C in the long-term, tree species are the main C pools even 20–30 years after a fire.

4.2. Nitrogen: total nitrogen, ammonium, nitrate and nitrite

Nitrogen in soil can be present in organic and inorganic forms and be plant-available or not plant-available (EPA, 2013). Biological processes, climatic conditions and soil type influence the amount and form in which nitrogen is present in soil. Although the plant available fraction of nitrogen is only a small percentage of the total nitrogen in the soil, it is the most important for short-term plant growth (Keeney and Nelson, 1982; Ajazi and Maci, 2013). After a fire, the soil N content is usually highly variable because of the spatial heterogeneity in fire severity and N losses during burning. In addition, considering that the effect of fire in the A horizon diminishes with the soil depth, the risk of collecting inaccurate data increases with increasing sampling depth (Hamman et al., 2008). Nitrogen availability in soil influences ecosystem resilience, and nitrogen poor soil compromises dryland restoration after a fire by limiting plant growth (Ajazi and Maci, 2013).

Nitrogen after wildfires can be lost by leaching, soil erosion, runoff and volatilization (Marion et al., 1991; Wan et al., 2001; Ice et al., 2004; Ekinci and Kavdir, 2005; Ferran et al., 2005; Pausas et al., 2008; Marañón-Jiménez et al., 2013). Although fire increases the available N content in soil in the short-term (Serrasolses and Vallejo, 1999), acid hydrolyses occurrence can decrease the amount of hydrolyzable N in soil. Because of the alteration of the N cycle, the ability of the vegetation to recover after a fire may decrease in the long-term (Sanchez and Lazzari, 1999; Castro et al., 2006; Belay-Tedla et al., 2009). While soil erosion and surface runoff spatially move nitrogen in soil, volatilization releases nitrogen into the atmosphere as ammonia gas (NH_3) and NO_x. Belillas and Feller (1998), Ekinci and Kavdir (2005) and Johnson et al. (1998) show that fire can cause a volatilization rate of up to 260 kg inorganic N ha^{-1} at 200 °C. The ability of soil to recover from nitrogen losses is highly dependent on the establishment of N-fixing plant species after a fire. Referring to low intensity wildfires, DeBano et al. (1977) stated that burned soils quickly recover from nitrogen losses. Nevertheless with increasing the fire intensity, the ability of the ecosystem to offset organic and inorganic losses decreases (Prieto-Fernandez et al., 2004; Ekinci and Kavdir, 2005).

4.2.1. Total N

The term total nitrogen refers to the sum of nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_4^+) and organic nitrogen (EPA, 2013).

4.2.1.1. Total N content after a fire in the O horizon. Immediately after a fire, the total nitrogen content in the O horizon of pine forests and mixed conifer forests showed a decrease compared to the control site (Caldwell et al., 2002; Murphy et al., 2006). In shrublands, Badía and Martí (2003), DeBano and Conrad (1978) and Rau et al. (2010) measured a decrease in the total nitrogen content of the O horizon immediately after a fire. According to Rau et al. (2010), almost 80% of the aboveground N as well as almost 90% of herbaceous, litter, and shrub N were removed by fire. In addition, fire led to the loss of more than 30% of the total N at 200–300 °C (DeBano and Conrad, 1978). The same trend was observed by DeBano and Conrad (1978) and by Wienhold and Klemmedson (1992) in the litter layer (0–1 cm and 1–2 cm). Nevertheless, Castelli and Lazzari (2002), measured higher total nitrogen contents in the burnt soil than in the control site immediately after a fire in both, shrublands and grass patches. The authors also reported a decrease in total nitrogen content one year as well as two years later. Data referring to the ash layer (0–2 cm) suggested that eight months after the occurrence of a high intensity wildfire, the total nitrogen content does not significantly decrease in pine forests (Kutiel and Naveh, 1987a). One year after the fire, significantly lower total nitrogen contents were measured by Johnson et al. (2007) in a pine forest. Twenty years after the occurrence of a wildfire in shrublands, the total nitrogen content was higher in the burnt site than in the unburnt site (Johnson et al., 2005).

4.2.1.2. Total N content after a fire in the A horizon. According to Wan et al. (2001), who reviewed the effects of fire on soil nitrogen under different vegetation covers, fire tends to decrease the soil total nitrogen content in the A horizon (0–2.6 cm and 0–5 cm). Experimental studies conducted by Castro et al. (2006) in Galicia, confirmed that the total nitrogen content in the A horizon (0–15 cm) significantly decreases immediately after the occurrence of high intensity wildfires in pine forests. Nevertheless, Kovacic et al. (1986) showed that regardless of the fire intensity, the change in total nitrogen content in the A horizon (0–5 cm) immediately after the fire is not significant. According to Hamman et al. (2008) both early and late season burns significantly increase the total N content in the A horizon (0–5 cm) of mixed conifer and oak forests immediately after the fire. However in shrublands, DeBano and Conrad (1978), Kutiel and Shaviv (1992) and Wienhold and Klemmedson (1992) measured a decrease in the total nitrogen content of the A horizon (0–2 cm and 2–10 cm) immediately after the fire. According to Kutiel and Shaviv (1992), fire decreased the total N content by 34% at 250 °C and by 86% at 600 °C. The same trend was measured by Rau et al. (2010) for a 0–8 cm soil depth and by Badía and Martí (2003) in the 0–15 cm soil depth. According to Kutiel and Shaviv (1992), one year later, the total nitrogen content was still significantly lower in the burnt soil compared to the control site. One month as well as one year after the occurrence of a wildfire in a mixed pine and oak forest, the total nitrogen content of the A horizon (0–5 cm and 2–25 cm) was not significantly lower in the burnt soil than in the control site (Kutiel and Naveh, 1987b). Significant lower total nitrogen contents in the burnt soil were measured by Kutiel and Inbar (1993) one month as well as eight months after the occurrence of a low–medium intensity wildfire in a pine forest. The same trend in pine forests was measured by Johnson et al. (2007) for 0–10 cm and by Ekinci and Kavdir (2005) for 0–5 cm, one year as well as eight years after the fire, respectively. In a mixed oak and pine forest, the total nitrogen content of the A horizon (0–5 cm) was significantly lower in the burned soil than in the unburned soil two weeks as well as two and eight years after the fire. The total nitrogen content was not significantly lower in plots burnt 12 years before the sampling time (Kavdir et al., 2005). Two years after the fire, the total nitrogen content increased in the A horizon (0–15 cm) of a pine forest (Brockway et al., 2002). Carreira et al. (1994) noticed that the total nitrogen content in the A horizon (0–5 cm) decreased with increasing fire recurrence in shrublands. Although the total nitrogen content slightly increased after the occurrence of one wildfire, it decreased in plots being affected by two and three wildfires. In particular, plots being affected by three wildfires showed significantly lower total nitrogen contents than the control site. Ferran et al. (2005) also related the total nitrogen content of the A horizon (0–2.5 cm and 2.5–10 cm) to the fire recurrence in shrublands. However their findings, in contrast to Carreira et al. (1994), were that the total nitrogen content was significantly higher in the burnt soil than in the unburnt soil at all soil depth after the occurrence of one, two and three wildfires.

4.2.1.3. Underlying processes. As total N is mostly organic, the losses are loosely related to the losses of organic C. Nitrogen volatilization is considered the main cause of total nitrogen loss from both the O and the A horizons immediately after a fire (Kutiel and Naveh, 1987a; Castro et al., 2006). Kutiel and Naveh (1987a) reported that N-fixing plant species such as legumes can buffer initial total nitrogen losses in the O horizon. In the short- as well as in the long-term, low N bioavailability in the A horizon is related to the transformation of the nutrient to more recalcitrant and less hydrolyzable forms (Castro et al., 2006). The differences in change in total N with fire recurrence reported by Carreira et al. (1994) and Ferran et al. (2005) may be related to the elapsed time between two consecutive fire events, which was greater in the area studied by Carreira et al. (1994). According to Ferran et al. (2005), total N in the A horizon increased because of the increase in potentially mineralizable N content.

4.2.2. Inorganic nitrogen: nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+)

The presence of inorganic nitrogen in soil is related to nitrification and mineralization processes, which are affected by soil temperature, aeration and pH(H_2O) (USDA, 2011; Ajazi and Maci, 2013).

4.2.2.1. Nitrate, nitrite and ammonium content after a fire in the O horizon. Regardless of the vegetation type, Johnson et al. (1998) and Wan et al. (2001) reported significantly higher NH_4^+ and NO_3^- contents in the O horizon immediately after a fire (T > 100 °C). The same trend was measured by Bauhus et al. (1993) in the ash layer of a sclerophyll forest, immediately after a fire. Higher NH_4^+-N and NO_3^--N contents were measured in the ash layer (0–2 cm) of a pine forest, eight months after the occurrence of a high intensity wildfire (Kutiel and Naveh, 1987a). According to Serrasolses and Vallejo (1999), mineral N concentration is very low in the ashbed right after fire, whereas ammonium concentration shows a significant increase the following months, followed by an increase in nitrate concentration later on. Marion et al. (1991) claimed that as fire severity increases the concentration of NH_4-N in the ash increases while the concentration of NO_3-N decreases. One year, as well as two and three years after fire, Chorover et al. (1994) measured significantly lower NH_4^+ concentrations in the precipitation and stream water of a mixed-conifer forest. Although the author measured significantly higher NO_3^- concentrations one year after the fire, the NO_3^- concentration reached pre-fire levels two years later and significantly decreased during the third years. The literature (Carreira et al., 1994; Castelli and Lazzari, 2002) shows that the NO_3^--N content in the O horizon can increase after a fire in shrublands. Different trends in NO_3^--N content have been observed by Carreira et al. (1994) and Castelli and Lazzari (2002) in relation to fire recurrence. While Carreira et al. (1994) measured a significant increase in soil NO_3^--N content with increasing fire recurrence, and especially after the occurrence of the second fire (time period between fires: nine years), Castelli and Lazzari (2002) measured a significant decrease in soil NO_3^--N content after the occurrence of a second fire (time period between fires: three years). These differences were highly dependent on the time period between fires and the litter accumulation on the soil surface.

4.2.2.2. Nitrate, nitrite and ammonium content after a fire in the A horizon. Badía and Martí (2003) and Wan et al. (2001) reported significant higher NH_4^+ and NO_3^- contents in the A horizon (0–15 cm) immediately after a fire (T > 100 °C) regardless of the vegetation type. In pine forests, the occurrence of low, medium and high intensity wildfires significantly increased the NH_4^+-N content in the A horizon (0–2 cm, 2–10 cm, 0–5 cm, 5–15 cm, 0–10 cm) immediately after a fire as well as in the short-term (Kovacic et al., 1986; Covington and Sackett, 1992; Kutiel and Shaviv, 1992; Kutiel and Inbar, 1993; Hernandez et al., 1997; Rodriguez et al., 2009). At all fire intensities, Kovacic et al. (1986) measured an increase in the $NO_2 + NO_3$-N content of the A horizon (0–5 cm) of a pine forest, immediately after a fire. Bauhus et al. (1993) measured an increase in the mineral N contents of the A horizon (0–5 cm and 5–15 cm) of a sclerophyll forest, immediately after a fire. Nine months after the occurrence of a medium intensity wildfire in a pine forest, the NH_4^+-N and the NO_3^--N contents were similar in the A horizon (0–5 cm and 0–10 cm) of burnt and unburnt plots (Hernandez et al., 1997; Rodriguez et al., 2009). The NH_4^+-N content in the A horizon (0–5 cm) of a mixed oak and pine forest was significantly higher in the burnt soil than in the unburnt soil one month after a fire, and significantly lower fourteen months after a fire. In addition, 14 months after a fire the NH_4^+-N content was not significantly higher under oak trees than under pine trees for the 0–5 cm soil depth and significantly higher under pine trees than under oak trees in the 5–25 cm soil depth (Kutiel and Naveh, 1987b). Several authors (Kutiel and Inbar, 1993; Hernandez et al., 1997; Rodriguez et al., 2009) reported that in the short-term, the occurrence of low, medium and high intensity wildfires significantly increases the NO_3^--N content in the A horizon (0–5 cm and 0–10 cm) of pine forests.

According to Brockway et al. (2002), while the NH_4^+-N content in the A horizon (0–15 cm) was significantly higher in the burnt soil than in the unburnt soil two years after a fire in grass patches, the NO_3^--N content significantly decreased. In shrublands, the NH_4^+ content in the A horizon (0–2 cm) increased one year as well as three years after a fire. At the same time, the NH_4^+ content at 2–10 cm soil depth decreased under oak shrubs and increased under *Cercocarpus* (Wienhold and Klemmedson, 1992). Wienhold and Klemmedson (1992) also measured a significant decrease in NO_3^- content one year as well as three years after a fire at 0–2 cm and at 2–10 cm soil depth. Carreira et al. (1994) stated that the NH_4^+-N content in soil under shrubs decreases with increasing fire recurrence. Plots being affected by three wildfires showed significantly lower NH_4^+-N contents than plots being affected by one, two or no wildfires. The difference in NH_4^+-N content between plots being affected by one or two wildfires and the control turned out to be not significant.

4.2.2.3. Underlying processes. Short- and long-term increases in NH_4^+-N as well as in NO_3^--N and $NO_2 + NO_3$-N contents in both the O and A horizons can be related to ash deposition and to the high nitrification potential of the burned soil in comparison to the unburned soil (Kutiel and Naveh, 1987a,b; Hernandez et al., 1997). According to Rodriguez et al. (2009), nitrogen mineralization and nitrification were promoted by the increase in temperature, pH and soil moisture after a fire. While studying the impacts of fire recurrence on the soil nutrient content, Carreira et al. (1994) and Castelli and Lazzari (2002) underlined the importance of the time elapsing between two consecutive fire events for the accumulation of litter which increases the soil inorganic nitrogen content. In Mediterranean shrublands, at least nine years between two consecutive fire events are required to avoid inorganic nitrogen decreases in soil (Carreira et al., 1994; Castelli and Lazzari, 2002).

4.3. Phosphorous

Schulte and Kelling (1996) reported that soil generally contains 500–1000 mg kg^{-1} total phosphorus in the forms of organic and inorganic P. Despite its high content, phosphorus is a limiting factor to plant growth because it is mainly present in the forms that are not plant available. Droughts can further reduce the amount of plant-available phosphorus in soil and thus affect vegetation recovery (Hinojosa et al., 2012). Wildfire can reduce the total soil phosphorus content by volatilization (770 °C), soil erosion and surface runoff (Marion et al., 1991; Wan et al., 2001; Ice et al., 2004; Ekinci and Kavdir, 2005; Ferran et al., 2005; Pausas et al., 2008; Marañón-Jiménez et al., 2013) and the available content through the formation of apatite. Changes in soil P content depend on fire intensity and duration, which are related to the temperature reached in the soil during fire (Ekinci and Kavdir, 2005). If the ash is not washed away by surface runoff or blown away by wind, higher amounts of phosphorus are found in the burned soil than in the unburned soil.

4.3.1. Total phosphorous

4.3.1.1. Total phosphorous content after a fire in the O horizon. In several studies, the total P content in pine forests significantly decreased immediately after a fire as well as one year later (Murphy et al., 2006; Johnson et al., 2007). In another study, eight months after the occurrence of a high intensity wildfire, the total phosphorus content significantly increased in the ash layer (0–2 cm) of a pine forest (Kutiel and Naveh, 1987a). One year as well as three years after a fire, Wienhold and Klemmedson (1992) measured a reduction in the total P content of the litter layer in shrublands. Under the same vegetation cover, the total P content significantly increased twenty years after a fire (Johnson et al., 2005).

4.3.1.2. Total phosphorous content after a fire in the A horizon. By burning pine forests at low, medium and high fire intensities, the total phosphorus content did not significantly increased in the A horizon (0–5 cm) one month as well as eight and nine months after a fire (Kutiel and Inbar, 1993; Hernandez et al., 1997). The same trend in pine forests was measured by Johnson et al. (2007) for the 0–10 cm soil depth, one year after a fire. In mixed oak and pine forests, the total P content in the A horizon (0–5 cm) did not significantly increase one month or fourteen months after the occurrence of high intensity wildfires. The difference in total P content under pine compared to oak trees was not significant (Kutiel and Naveh, 1987b). One year as well as three years after a fire, the total P content significantly decreased in the A horizon (0–2 cm) under *Cercocarpus* but did not significantly decrease under oak shrubs (Wienhold and Klemmedson, 1992). In the 2–10 cm soil depth, the total P content significantly decreased under *Cercocarpus* and increased under oak shrubs, but not significantly. The total P content in the A horizon (0–15 cm) significantly decreased two years after a fire in grass patches (Brockway et al., 2002). In shrublands, Johnson et al. (2005) reported significantly lower total P contents in the A horizon (0–7 cm), twenty years after a fire.

4.3.1.3. Underlying processes. Although there is limited data available on the total phosphorus content in the O horizon, it appears to be more sensitive to fire than the A horizon. Although burning of the litter layer, surface runoff and wind were responsible for total phosphorous losses in the O horizon, no significant changes in total phosphorous content were measured in the A horizon under pine and oak trees. Decreases in total P content were measured in the A horizon under shrubs. The high volatilization temperature of the nutrient was responsible for its low mobility in the soil and thus for its persistence in the ecosystem.

4.3.2. Available phosphorous

4.3.2.1. Available phosphorous content after a fire in the O horizon. Very little data was found in the literature regarding the available phosphorous content in the O horizon after fire. DeBano and Conrad (1978) reported a decrease in the available phosphorous content of the litter layer (0–1 cm and 1–2 cm) in shrublands, immediately after fire.

4.3.2.2. Available phosphorous content after a fire in the A horizon. In shrublands, DeBano and Conrad (1978) measured a significant decrease in the available phosphorous content of the A horizon (2–10 cm), immediately after a fire. Under the same vegetation cover but for a 0–15 cm soil depth, Badía and Martí (2003) reported a significant increase in available and extractable phosphorous immediately after a fire. Regardless of the fire intensity, Hernandez et al. (1997), Kutiel and Inbar (1993), Kutiel and Shaviv (1992) and Rodriguez et al. (2009) reported a significant increase in the available phosphorus content of the A horizon (0–2 cm, 2–10 cm, 0–5 cm and 0–10 cm) in pine forests, immediately after a fire as well as in the short-term. Under oak trees, the available phosphorus content in the A horizon (0–10 cm) significantly increased six months after the occurrence of a high intensity wildfire (Pardini et al., 2004). Eight years after the occurrence of a medium–high intensity wildfire in a pine forest, the available phosphorous content was not significantly higher in the burnt plots than in the control site at 0–5 cm soil depth (Ekinci and Kavdir, 2005). Castelli and Lazzari (2002) and Ferran et al. (2005) related the available phosphorus content in shrublands to fire recurrence. They found that the available phosphorus content in the A horizon (0–1 cm and 0–2.5 cm) was significantly higher than the control after the occurrence of one as well as of two and three wildfires. However, the available phosphorus content tended to decrease with increasing fire recurrence, so that plots affected by only one wildfire showed higher P contents than plots being affected by two or three wildfires.

4.3.2.3. Underlying processes. The available P content tended to decrease in the O horizon because of burning of the litter layer and the removal of the ash from the soil surface by wind and surface runoff. Eight years after a fire, Ekinci and Kavdir (2005) related the increase in available P content in the A horizon to the deposition and persistence of the ash on the soil surface. In addition, the quick conversion of immediately available phosphorous to mineral P and insoluble P forms such as apatite was responsible for the increase and/or steady persistence of P in the A horizon (Kutiel and Shaviv, 1992; Schulte and Kelling, 1996; Pardini et al., 2004). According to Kutiel and Inbar (1993), although the combustion and mineralization of organic P caused a peak in the amount of available P one month after a fire, the presence of $CaCO_3$ in soil led to the formation of apatite eight months later.

4.4. Exchangeable cations: Ca^{2+}, Mg^{2+}, K^+, Na^+

DeBano and Conrad (1978) stated that the role of fire in nutrient recycling is especially notable when considering the element potassium (K), which increases in soil after a fire because of plant and litter burning and spread apart of potassium-containing minerals such as biotite in high intensity fires reaching the mineral soil (Viro, 1974). According to Marion et al. (1991), the concentrations of Ca^{2+}, Mg^{2+} and K^+ in the ash increase with increasing fire severity. It has been shown that the concentration of $CaCO_3$ in soil and the ash pH increase with increasing fire severity because of burning the wood at temperatures between 350 and 400 °C. Pereira et al. (2012) reported that, depending on the ionic strength of the solution and on the ionic radii of each ion, $CaCO_3$ becomes negatively charged and attracts the cations in solution at a pH(H_2O) between 7 and 10.8. Ions with smaller ionic radii than Ca^{2+} (0.99 Å), i.e. Mg^{2+} (0.72 Å), are incorporated onto the $CaCO_3$ surface through substitution for this element. Although monovalent cations such as Na^+ and K^+ have ion radii bigger than Ca^{2+} (1.02 Å and 1.38 Å, respectively), the previous incorporation of other ions in the structure and the consequent formation of new crystals allow them to be incorporated onto the $CaCO_3$ surface as well. Additionally, if present in high concentration, Na and K compete with Mg and Ca for the available places on the $CaCO_3$ surface. Pereira et al. (2012) showed that water-extractable Na and K in the ash decrease with increasing $CaCO_3$ amount. While researching the effects of thermal shocks on cation leaching dynamics, Cancelo-González et al. (2013) proved that the amount of cations leached increases with increasing severity of the thermal shock. Based on that, high intensity fires lead to the loss of almost 80% of the total exchangeable cations in the soil through surface runoff and subsurface flow. Indeed exchangeable cations (especially Ca^{2+} and Mg^{2+}) are not usually lost by volatilization because of their high volatilization temperature (Britt, 2007).

4.4.1. Exchangeable cation content after a fire in the O horizon

While the exchangeable K^+ and Ca^{2+} content of the O horizon significantly decreased immediately after the occurrence of a medium intensity wildfire in a pine forest, the exchangeable Na^+ and Mg^{2+} contents were not significantly lowered by fire (Murphy et al., 2006). In a mixed conifer forest, Caldwell et al. (2002) measured a non-significant decrease in the exchangeable Ca^{2+} content of the O horizon immediately after a fire. In shrublands, the exchangeable K^+, Mg^{2+}, Ca^{2+} and Na^+ contents significantly increased in the litter layer (0–1 cm and 1–2 cm) immediately after a fire (DeBano and Conrad, 1978). Significantly lower exchangeable K^+, Ca^{2+} and Mg^{2+} contents in the O horizon were also measured by Johnson et al. (2007) one year after a fire. In a mixed-conifer forest, Chorover et al. (1994) measured significantly lower exchangeable Mg^{2+}, Ca^{2+}, Na^+ and K^+ concentrations in the precipitation and stream water one year as well as two and three years after a fire. Shrublands affected by high intensity wildfires showed a significant increase in exchangeable K^+ content and a significant decrease in exchangeable Ca^{2+} content one year after a fire. The exchangeable Mg^{2+} and Na^+ contents did not change significantly after a fire (Castelli and Lazzari, 2002). Twenty years after a fire, Johnson et al. (2005) measured significantly higher exchangeable Mg^{2+}, K^+ and Ca^{2+} contents in burnt shrubland soil compared to the control site.

4.4.2. Exchangeable cation content after a fire in the A horizon

In a pine forest, the exchangeable Ca^{2+}, Mg^{2+}, K^+ and Na^+ contents of the A horizon (0–2 cm) decreased immediately after a fire. While the change in exchangeable Ca^{2+} and Mg^{2+} content was significant, the change in exchangeable Na^+ and K^+ content was not significant (Kutiel and Shaviv, 1992). In shrublands, Badía and Martí (2003) reported a significant increase in the exchangeable K^+, Mg^{2+}, Ca^{2+} and Na^+ contents of the A horizon (0–15 cm) immediately after a fire. While the content of Mg^{2+}, Na^+ and K^+ in soil showed an increase with increasing fire intensity, the Ca^{2+} content tended to decrease with increasing fire intensity. DeBano and Conrad (1978) also found a significant increase in the exchangeable Mg^{2+} content in the A horizon (2–10 cm) immediately after a fire in shrublands, however the exchangeable K^+, Ca^{2+} and Na^{2+} contents significantly decrease. One month after the occurrence of a low–medium intensity wildfire, Kutiel and Inbar (1993) measured a significant increase in exchangeable K^+, Mg^{2+}, Na^+ and Ca^{2+} content in the A horizon (0–5 cm) under pine trees. The authors also reported that while the difference in exchangeable Ca^{2+} and K^+ content between burnt and unburnt plots remained significant over time, the difference in exchangeable Na^+ content was not significant. In addition, eight months after a fire, the exchangeable Mg^{2+} content reached pre-fire levels. Pardini et al. (2004) measured a non-significant increase in the exchangeable K^+ content and a significant increase in the exchangeable Ca^{2+} content of the A horizon (0–10 cm) of an oak forest, six months after a medium intensity wildfire. In addition, the exchangeable Mg^{2+} and Na^+ contents were not significantly lower in the burnt soil than in the control site. According to Brockway et al. (2002), the total K^+ content as well as the exchangeable Ca^{2+}, Mg^{2+} and Na^+ contents decreased in the A horizon (0–15 cm) two years after a fire in grass patches. In shrublands, Johnson et al. (2005) reported a decrease in the exchangeable K^+ and Mg^{2+} contents of the A horizon (0–7 cm) twenty years after a fire. Relating the exchangeable K^+ content to fire recurrence in shrublands, Ferran et al. (2005) stated that the exchangeable K^+ content did not significantly increased in the A horizon (0–2.5 cm and 2.5–10 cm) after fire.

4.4.3. Underlying processes

Exchangeable cation losses in both, the O and the A horizons are mainly related to soil erosion processes leading to the loss of the ash accumulated on the soil surface after fire (Marion et al., 1991; Ekinci and Kavdir, 2005; Pausas et al., 2008). Because of their high volatilization temperature, exchangeable cations are not usually lost by volatilization but by leaching and surface runoff (Lasanta and Cerdà, 2005; Britt, 2007). According to Kutiel and Shaviv (1992) and Lasanta and Cerdà (2005), Ca^{2+} is the most affected cation by fire in soil. The dissolution of the nutrients in the burnt soil increased the Ca^{2+} concentration in the leached soil three months after a fire (Kutiel and Shaviv, 1992). Additionally, the Ca^{2+} concentration in the surface runoff was still significantly high seven months after a fire (Lasanta and Cerdà, 2005).

5. Summarized overview

The soil nutrient content in Mediterranean drylands affected by wildfires varies with: vegetation type, period of time after fire, fire recurrence and fire intensity. Immediately after a fire, the nutrient content in both the O and the A horizon usually increases because of ash deposition, nutrient mineralization and formation of stable forms such as black carbon, charcoal and apatite. Ash deposition and persistence on the soil surface is one of the most important factors in determining the soil nutrient content immediately after a fire as well as in the short- and long-term. In the long-term, the soil nutrient content usually decreases because of nutrient volatilization, transformation of the nutrients to more recalcitrant forms, and ash removal by wind and surface runoff.

Based on the literature reviewed, an overview of the changes in soil nutrient content under pine and oak forests as well as in shrublands and

Table 1
Change in soil nutrient content of burnt plots vs. unburnt plots immediately after a fire. Symbols refer to: (↓) decrease, (↑) increase, (↔) no significant change, (n.d.) no data available.

	Pine and oak forests		Shrublands and grass patches	
	O horizon	A horizon	O horizon	A horizon
SOC	↓	n.d.	↓	↔
Total N	↓	↓	↓	↓
NH_4^+-N	↑	↑	n.d.	↑
NO_3^--N	n.d.	n.d.	n.d.	↑
Total P	↓	n.d.	n.d.	n.d.
Available P	n.d.	↑	↓	↓
Exchangeable Ca^{2+}	↓	↓	n.d.	↑
Exchangeable Mg^{2+}	↓	↓	n.d.	↑
Exchangeable Na^+	n.d.	↓	n.d.	↑
Exchangeable K^+	↓	↓	n.d.	↑

Table 3
Change in soil nutrient content of burnt plots vs. unburnt plots in the long-term (samples collected later than one year after a fire). Symbols refer to: (↓) decrease, (↑) increase, (↔) no significant change, (n.d.) no data available.

	Pine and oak forests		Shrublands and grass patches	
	O horizon	A horizon	O horizon	A horizon
SOC	n.d.	↓	↑	↓
Total N	n.d.	↓	↑	n.d.
NH_4^+-N	n.d.	n.d.	n.d.	n.d.
NO_3^--N	n.d.	n.d.	n.d.	n.d.
Total P	n.d.	↑	↑	↑
Available P	n.d.	↑	n.d.	↓
Exchangeable Ca^{2+}	n.d.	n.d.	↑	n.d.
Exchangeable Mg^{2+}	n.d.	n.d.	↑	↓
Exchangeable Na^+	n.d.	n.d.	n.d.	n.d.
Exchangeable K^+	n.d.	n.d.	↑	↓

grass patches is presented in Tables 1, 2 and 3. The change in nutrient content in both the O and A horizons is related to three time periods: immediately after a fire, within one year after a fire (short-term changes) and later than one year after a fire (long-term changes). The change noted in Table 1, 2 and 3 is based on the findings of the majority of the authors in the papers reviewed.

6. Conclusions

Although the impact of wildfires on soil nutrient content is highly dependent on the nutrient considered, ash deposition and persistence on the soil surface is essential to limiting nutrient losses and fostering vegetation recovery after fire. Soil erosion, surface runoff and wind removal of the ash can significantly reduce the nutrient content of both the O and the A horizons. Depending on the plant type and on the time elapsing between two consecutive wildfires, fire is responsible for the transition from mature ecosystems (i.e. conifer forests) to shrublands. Indeed, high fire frequency may cause the eradication of keystone species, for example pines, which has consequences for the soil nutrient pool recovery. To promote the accumulation and retention of nutrients in soils after a fire, it is important to stabilize the burnt site by applying post-fire measures aiming to limit soil erosion, surface runoff and wind removal of the ash. To assess and select effective post-fire treatments, a site-specific study is recommended taking into consideration: slope gradient and orientation, soil type, climatic conditions, original plant community of the burnt site, fire intensity, duration and recurrence. In addition, the study should consider the plant communities neighboring with the burnt site because those can affect the re-vegetation process by altering the composition in species of the original plant community. The diffusion of less flammable and fire-resilient plants can further help to reduce wildfire occurrence and to foster the development of mature ecosystems in the Mediterranean basin, which are characterized by soils richer in nutrients than the soils of shrublands.

Table 2
Change in soil nutrient content of burnt plots vs. unburnt plots in the short-term (samples collected within one year after a fire). Symbols refer to: (↓) decrease, (↑) increase, (↔) no significant change, (n.d.) no data available.

	Pine and oak forests		Shrublands and grass patches	
	O horizon	A horizon	O horizon	A horizon
SOC	n.d.	↓	n.d.	↓
Total N	↓	↓	n.d.	↓
NH_4^+-N	n.d.	↓	n.d.	↑
NO_3^--N	↑	↑	n.d.	↓
Total P	↓	↑	↓	↓
Available P	n.d.	↑	n.d.	↑
Exchangeable Ca^{2+}	↓	↑	n.d.	↓
Exchangeable Mg^{2+}	↓	↑	n.d.	↔
Exchangeable Na^+	n.d.	↑	n.d.	↔
Exchangeable K^+	↓	↑	n.d.	↑

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