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SOIL HYDROLOGY AND ECOLOGY AFTER PRESCRIBED FIRE AND MULCHING WITH FERN IN MEDITERRANEAN FORESTS

Ph.D. THESIS

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Abstract

In order to reduce the wildfire impacts on the forest ecosystem, prescribed fire is considered as a primary option to remove the fuel that can generate a high-intensity fire. Despite the beneficial effects, uncertainties remain about the impacts of prescribed fires on forest ecology and hydrology. As a matter of fact, the runoff and erosion rates increase and the recruitment of forest species may perform differently after fire. In order to control the post-fire hydrological response of burned soils after prescribed fires, mulching is one of the most common management options, but this technique can also have negative effects on the forest ecosystem.

In spite of ample literature, the studies about the changes in soil properties and hydrological response after the prescribed fire and mulching are contrasting. For some forest species, such as oak, the research about the effectiveness of prescribed burning on plant regeneration is not unanimous. Moreover, the effectiveness of mulching with fern residues on soil hydrology and ecology has not been previously evaluated as post-fire management strategy.

Considering these literature gaps, this PhD thesis evaluates the short-term effects of the prescribed fire and post-fire mulching with fern residues on hydrology and other properties of soil as well as on plant regeneration in forests (pine, chestnut and oak) of Southern Italy. These evaluations have been carried out adopting both a two-year monitoring activity and a modelling approach regarding the hydrological variables.

A first investigation, carried out using a portable rainfall simulator at the point scale in the three forest sites, has demonstrated that the prescribed fire may decrease water infiltration and increase soil water repellency, but mulching is effective in reducing the runoff generation capacity immediately after fire mainly in broadleaves species and less in conifers. The hydrological effects of both burning and mulching decrease over time until being negligible some months after fire.

The evaluation of surface runoff and erosion in the same forest sites after natural precipitation at the plot scale has shown that, immediately after the prescribed fire, runoff and soil losses significantly increase in all forest stands compared to the unburned soils. This window of disturbance after fire is limited to three-four months, and the pre-fire runoff and erosion rates of the soils are practically restored after five months. Soil mulching with fern is effective to limit the increase in the hydrological response observed in the burned soils.

Another study focusing the soil properties and covers has evidenced that prescribed fire, although being a low-intensity fire, is able to induce significant changes in soil chemistry and surface runoff, and that the magnitude of these changes depends on the soil property and forest species. The effects of the prescribed fire and mulching are often transient. Mulching with fern has been shown to be unable to limit the changes in chemical properties of soils.

It has been also pointed out that low-intensity burning enhances the initial recruitment of oak plants and that soil mulching may be partly synergistic with the prescribed fire. This post-fire technique does not increase acorn emergence and plant survival compared to the burned and untreated sites, but enhances plant height and root mass.

A modelling approach has confirmed the applicability of the SCS-CN and USLE-M models to predict surface runoff and erosion, respectively, in forests burned by prescribed fire and treated with fern under Mediterranean conditions. In contrast, poor predictions of the modelled hydrological variables were provided by the models in unburned plots, and by the Horton and MUSLE models for all the soil conditions. Moreover, optimal values of the input parameters of the tested models, have been proposed for future model applications.

Overall, this PhD thesis has shown by experimental investigations that, in Mediterranean forests, the use of prescribed fire is promising to reduce the wildfire risk, although this practice is surprisingly of uncommon use in Italy. However, we must pay caution to the worsening of the hydrological response of burned soils immediately after fire. A cheap mulch material, such as the fern residues, is effective to control the soil’s response to fire and avoid the hydrogeological risks in the delicate forest environments.

Riassunto

Al fine di ridurre gli impatti degli incendi sull'ecosistema forestale, il fuoco prescritto è considerato un'opzione primaria, perchè rimuove il materiale combustibile che può generare un incendio ad alta intensità. Nonostante gli effetti benefici, permangono incertezze sugli impatti del fuoco prescritto sull'ecologia e l'idrologia forestale. In effetti, l’intensità di deflusso ed erosione aumentano e l’entità della rigenerazione delle specie forestali può essere modificata dopo il fuoco. Per controllare la risposta idrologica post-incendio dei suoli soggetti al fuoco prescritto, il mulching è una delle opzioni di gestione più comuni, ma questa tecnica può anche avere effetti negativi sull'ecosistema forestale.

Nonostante l'ampia letteratura, gli studi sulle modifiche delle proprietà del suolo e della risposta idrologica dovuti al fuoco prescritto ed il mulching riportano risultati contrastanti.

Ad esempio, per alcune specie forestali, come la quercia, la ricerca sull'efficacia del fuoco prescritto sulla rigenerazione delle piante non è unanime. Inoltre, l'efficacia del mulching con residui di felce sull'idrologia e sull'ecologia del suolo non è stata precedentemente valutata come strategia di gestione post-incendio.

In relazione a queste lacune della letteratura, questa tesi di dottorato valuta gli effetti di breve periodo del fuoco prescritto e del mulching post-incendio con residui di felce sull'idrologia e sulle proprietà del suolo, nonché sulla rigenerazione delle piante in boschi (pino, castagno e quercia) del Sud Italia. Tali valutazioni sono state effettuate adottando sia un'attività di monitoraggio biennale, sia un approccio modellistico relativo alle variabili idrologiche.

Una prima indagine, condotta a scala puntuale utilizzando un simulatore di pioggia portatile nei tre siti forestali, ha dimostrato che il fuoco prescritto può ridurre l'infiltrazione e aumentare l'idrorepellenza del suolo e che il mulching è efficace nel ridurre la capacità di generazione del deflusso subito dopo il fuoco (principalmente nelle latifoglie e meno nelle conifere). Gli effetti idrologici sia della combustione, sia del mulching diminuiscono nel tempo fino a divenire trascurabili alcuni mesi dopo il fuoco.

La valutazione del deflusso superficiale e dell'erosione in parcelle forestali a seguito di precipitazioni naturali ha dimostrato che, immediatamente dopo il fuoco prescritto, il deflusso e le perdite di suolo aumentano significativamente in tutti i soprassuoli forestali rispetto ai suoli non trattati. Questa cosiddetta "finestra di disturbo" post-incendio è limitata a tre-quattro mesi e i tassi di deflusso ed erosione dei suoli pre-incendio vengono praticamente ripristinati dopo cinque mesi. Il mulching con felce è efficace per limitare l'aumento della risposta idrologica osservata nei suoli percorsi dal fuoco.

Un altro studio incentrato sulle proprietà chimiche e le coperture del suolo ha evidenziato che il fuoco prescritto, pur essendo un incendio a bassa intensità, è in grado di indurre modifiche significative nei parametri chimici del suolo superficiale e che l'entità di queste modifiche dipende dalla proprietà del suolo e dalla specie forestale considerata. Gli effetti del fuoco prescritto e del mulching sono spesso transitori. È stato dimostrato che il mulching con felce non è in grado di limitare le modifiche delle proprietà chimiche dei suoli.

È stato inoltre sottolineato che la combustione a bassa intensità migliora la rigenerazione iniziale delle piante di quercia e che il mulching del suolo può essere in parte sinergica con il fuoco prescritto. Questa tecnica post-incendio non aumenta l'emergenza delle ghiande e la sopravvivenza delle piante rispetto ai siti bruciati e non trattati, ma incrementa l'altezza della pianta e la massa delle radici.

Un approccio di modellistica idrologica ha confermato l'applicabilità dei modelli SCS-CN e USLE-M per la previsione del deflusso superficiale e della perdita di suolo rispettivamente, nelle aree forestali soggette al fuoco prescritto e trattate con felci in condizioni mediterranee. Al contrario, le previsioni delle variabili idrologiche fornite dai modelli Horton e MUSLE nelle parcelle non bruciate sono risultate molto insoddisfacenti per tutte le condizioni del suolo. Inoltre, sono stati proposti valori ottimali dei parametri di input dei modelli testati per future applicazioni del modello.
Nel complesso, questa tesi di dottorato ha dimostrato con i risultati di indagini sperimentali che, nelle foreste mediterranee, l'uso del fuoco prescritto è una tecnica sostenibile e promettente per ridurre il rischio di incendi boschivi, sebbene questa pratica sia sorprendentemente di uso non comune in Italia. Tuttavia, bisogna tener conto del peggioramento della risposta idrologica dei suoli bruciati immediatamente dopo l'incendio. Un materiale economico per il mulching, come i residui di felce, è efficace per controllare la risposta del suolo al fuoco ed evitare i rischi idrogeologici nei delicati ambienti forestali.

Keywords: infiltration rate; surface runoff; erosion; soil properties; forest regeneration; hydrological model.

General introduction

Fire is a natural and anthropogenic agent with a long history of influence on terrestrial and aquatic ecosystems (Francos and Úbeda, 2021a; Niemeyer et al., 2020; Zema, 2021). This ecological factor impacts on many components of forest, such as soil, air, water, plants and fauna, e.g. (DeBano et al., 1998; Lucas-Borja et al., 2019b; Kozlowski, 2012) as well as on ecosystem services, society and economy (Nadal-Romero et al., 2018; Pereira et al., 2018a). When at high severity, such as wildfires, fire may pose pervasive effects on human goods and assets (Wittenberg and Pereira, 2021a).
The fire effects on the forest ecosystem depend on several factors, such as fire history, intensity and severity, fuel quantity, properties and topography of soils, vegetation species, density and cover, weather patterns, etc. (Zavala et al., 2014; Pereira et al., 2018b; Francos et al., 2018; Zema, 2021). The vegetation burning coupled to changes in the physico-chemical and biological properties of forest soils determines spoilage of vegetation cover with loss of biodiversity. Moreover, wildfire modifies soil hydrology with possible increases in surface runoff as well as soil erosion and degradation (Zema, 2021). The alteration of

soil's hydrological response after fire increases the magnitude and frequency of floods, sediment transport, landslides and debris flow (Shakesby, 2011; Moody et al., 2013; Zema, 2021).

The fire impacts on the forest ecosystem can be heavy in the Mediterranean Basin, where the semi-arid climate conditions are favourable for wildfires (Cerdá and Mataix-Solera, 2009; Hueso-González et al., 2018; Inbar et al., 2014). Under these climatic conditions, the frequency, extent and intensity of wildfires are associated with an increased climate warming in the last decades and in the future (Pausas and Fernández-Muñoz, 2012). This is due to the specific weather conditions (e.g., low humidity, high temperature, and strong winds) (Morán-Ordóñez et al., 2020), and hydrological regime (extreme and flash storm events with very high rainfall amounts and erosivity) (Diodato and Bellocchi, 2010; Giorgi and Lionello, 2008). Soil losses after extreme rainfall events are very high in the Mediterranean basin, where up to 80% of annual precipitation is recorded after the dry summers, when wildfires usually occur (Shakesby, 2011). The increase in erosion rates is presumably the most evident abiotic indicator of environmental degradation due to wildfire (Certini, 2005; Shakesby, 2011).

In order to limit the negative impacts of high-severity fires, preventing strategies have been adopted since long time (Ferreira et al., 2015). Among these strategies, prescribed fire – that is defined as "the planned use of low-intensity fire to achieve very different goals given certain weather, fuel and topographic conditions" (Alcañiz et al., 2018; Fernandes et al., 2013) - is considered as a primary and integrated option to remove or reduce the fuel that can generate a high-intensity fire (Vega et al., 2005; Alcañiz et al., 2018). A low-intensity fire, such as the prescribed fire, which has also a low severity and burn patchiness (Cawson et al., 2012; Pereira et al., 2021), is effective in reducing the wildfire risk in forests (Vega et al., 2005; Alcañiz et al., 2018). Prescribed fire avoids high temperature in soil and tree crown burning, which are the most adverse effects of wildfire on soil and plants. In general, the changes in soil properties after a prescribed fire are limited (Alcañiz et al., 2018; Lucas-Borja et al., 2019b). Moreover, the literature has shown that this practice supports regeneration of some plant species (Francos and Úbeda, 2021b; Scharenbroch et al., 2012; Williams et al., 2012) and is also beneficial for many ecosystem services, such as natural hazard regulation and pest and disease control (Pereira et al., 2021). It results that the use of the prescribed fire has been increasingly gaining popularity among forest managers (Neary and Leonard, 2021; Úbeda et al., 2005) mainly in USA, Australia and the Iberian Peninsula (Klimas et al., 2020). Many regions in these countries are prone to wildfire, and authorities have prioritized prescribed fire due to the numerous experiments that have demonstrated its feasibility to limit the wildfire risk and its environmental sustainability (Alcañiz et al., 2018; Lucas-Borja, 2021). In Italy, although being allowed by the national and regional legislation, the use of prescribed fire is not common, and this increases the vulnerability of the forest heritage to wildfire.

Despite these beneficial effects of prescribed fire, uncertainties remain about its impacts on forest ecology and hydrology (Fuentes et al., 2018). On an ecological approach, the post-fire regeneration of forest species may perform differently after prescribed fires (Lucas-Borja et al., 2016), and even the results of relevant research on the fire effect on tree regeneration are often contrasting. About the hydrological impacts, the increase of soil's response after the prescribed fires is lower compared to wildfires, but the hydrological risks are still present also in the case of prescribed fire (Morris et al., 2013; Shakesby et al., 2015), since low-intensity fires can also exert negative effects on soil properties e.g. infiltration, soil water repellency, bulk density and soil aggregate stability (Alcañiz et al., 2018; Francos and Úbeda, 2021). In the Mediterranean forests, the modifications in the soil's hydrological response may be even more intense compared to other environmental contexts (Fortugno et al., 2017), since the soils are generally shallow and show low aggregate stability, and organic matter and nutrient contents (Cantón et al., 2011). Due to the combination of these climate and soil characteristics, the Mediterranean forests may be more exposed to excessive runoff and soil erosion rates compared to other ecosystems (Zema et al., 2020a, 2020b). The hydrological response of burned soils may become critical and hazardous especially during the so-called "window of disturbance" (Prosser and Williams, 1998), that is that period occurring immediately after fire throughout about one year, when the soil is bare and the pre-fire levels of its properties have not yet recovered. In the Mediterranean forests, the hydrological processes generating surface runoff are dominated by the infiltration-excess mechanism (Lucas-Borja et al., 2018). In this environment, the reduction in water infiltration rate of soil can enhance flooding and hydrogeological instability after the frequent and intense rainstorms. Furthermore, infiltration can be further decreased by soil water repellency, which is common in Mediterranean forest soils (Cerdà and Doerr, 2007; Plaza-Álvarez et al., 2019), and determines soil hydrophobicity.

Moreover, the hydrological processes in burned soils are very complex, since several factors (weather, fire severity, vegetation cover, soil properties, morphology and land management) influence the hydrological response of soil after fire (Moody et al., 2013; Nunes et al., 2018;

Zema, 2021b). It should be also kepp in mind that both wildfires and rainstorms are thought to become more frequent and intense and post-fire recruitment of forest plants may be more difficult and slower according to the forecasted scenarios of climate change (Badia and Marti, 2008). According to these authors, both wildfires and rainstorms are thought to become more frequent and intense, especially in the Mediterranean environment. Therefore, the effects of prescribed fires on soil hydrology and the related variables (infiltration rates, soil water repellency, surface runoff and soil erosion) in delicate environments, such as the Mediterranean areas, needs a particular attention (Hubbert et al., 2006; Hueso-González et al., 2018).

Generally, the prediction of the hydrological effects of forest fires is very challenging (Soulis et al., 2021). Computer-based hydrological models are essential tools to better understand and predict the post-fire hydrological processes in a cost-effective and time-efficient way (Filianoti et al., 2020). Empirical methods, such as the Soil Conservation Service (SCS)-Curve Number (CN) (henceforth “SCS-CN model”), Horton and USLE-family models, are of easier and quicker applicability compared to the most complex physically-based models, but their prediction capacity may be considered as acceptable for many needs and uses (Aksoy and Kavvas, 2005; Lucas-Borja et al., 2020). Regarding the uses of the empirical models to predict runoff and erosion prediction after fire, the variability of their performance for post-fire conditions highlights the difficulties in applications using observed data (Soulis, 2018). Moreover, the input factors of the hydrological models proposed for burned forests by literature are not valid for environments with contrasting characteristics. From this, there is the need of more modelling experiences that should test the accuracy of the empirical models in Mediterranean conditions and under different management scenarios.

Concerning the ecological impacts of prescribed fires, for some forest species, studies of fire effects should ensure the feasibility of this practice on plant regeneration, especially on two critical stages of early recruiting, such as seed emergence and seedling recruitment. This issue is essential in Mediterranean mountainous areas, where the fire hazard is high and the post-fire recruitment of vegetation is limited by drought and some adverse soil characteristics (Moody et al., 2013; Shakesby, 2011).

In order to reduce the soil’s susceptibility to runoff and erosion after burning (including the prescribed fire), several treatments have been proposed and their effectiveness has been verified in many environmental contexts (Lucas-Borja, 2021; Zema, 2021a). Among the ecological engineering techniques, which use vegetative residues for soil conservation, mulching is one of the most common post-fire management options (Lucas-Borja et al., 2019a; Prosdocimi et al., 2016). The objective of mulching is protecting soil with ground cover and improving soil quality by supplying vegetal matter to soil, and this is achieved if mulch is properly and timely applied (Prosdocimi et al., 2016; Zituni et al., 2019). However, post-fire mulching can also have negative effects. In some cases, mulching reduces the soil hydraulic conductivity under unsaturated conditions compared to untreated soils, particularly in the drier season (Lucas-Borja et al., 2018). Mulching material is selected based on its availability, resistance to degradation, weed spreading risk and other factors (Parhizkar et al., 2021; Prats et al., 2015). Straw is often used as mulch cover in fire-affected areas (Bontrager et al., 2019; Keizer et al., 2018), but its residues can be displaced by wind in some areas, leaving slopes bare, or accumulated in thick layer in other areas, with possible reductions in post-fire vegetation emergence (Robichaud et al., 2020). Moreover, agricultural straw may contain seeds, chemicals and parasites, which can be the sources of non-native vegetation and plant diseases. Forest residues (e.g. wood strands, chips or shreds) or dead plants may replace straw, because these substrates do not carry non-native seeds or chemical residues, and are more resistant to wind displacement (Robichaud et al., 2020). In Mediterranean forest floor, fern - *Pteridium aquilinum* (L.) Kuhn - is widely available (which avoids the transport need from other locations). Fern may be directly cut on the floor (without being transported from distant sites) of forests, where the fire and erosion risks are lower, and can be applied as it is (that is, without desiccation, as usually carried out for straw). This native species does not bring non-native seeds or chemical residues into the forest ecosystem, and fresh fern may be more rapidly and easily degraded and incorporated in soil compared to other dried residues.

Therefore, the vegetal residues of fern may replace straw as mulching material in fire-affected areas, which is essential when the soil is left bare and the soil changes (e.g., reduced infiltration, soil water repellency and ash cover) can be significant compared to the unburned areas (Cawson et al., 2012; Francos and Úbeda, 2021b; Klimas et al., 2020; Wittenberg and Pereira, 2021b). However, there is the need to evaluate the hydrological and ecological effectiveness of fern residues to protect the burned soil from runoff and erosion, restore soil properties after fire and support plant regeneration. Moreover, the validation of soil erosion models under post-fire management treatments - such as mulching - is particularly welcome, considering the large variability of climatic and geomorphological conditions (Fernández et al., 2010; Robichaud et al., 2007), and the applicability of SCS-CN method and USLE-family models may questionable without targeted modelling experiences.

State-of-the-art and research needs

Regarding the hydrological effects of prescribed fire and soil mulching, in spite of ample literature, the studies about the changes in soil water repellency and infiltration rates after fires of different severity are not consistent and sometime contrasting (Cawson et al., 2012). For instance, increases in soil water repellency were observed by (Plaza-Álvarez et al., 2018) in pine forests burned by prescribed fires in SE Spain; these effects were found to be even higher compared to high-severity fires (Huffman et al., 2001). Other authors report slight or no changes in soil water repellency in soils burned at low temperatures (Doerr et al., 2006; Robichaud, 2000). Therefore, no direct associations between fire severity and soil water repellency can be found, as resulting from several factors, such as the natural variability in the study sites (soil water content and texture, and vegetation type) (Cawson et al., 2012). These inconsistencies also affect soil infiltration rates (Wittenberg and Malkinson, 2009), with noticeable reductions (Robichaud, 2000) or not significant changes (Plaza-Álvarez et al., 2019) in infiltration rates after prescribed fires. Ash released by fire plays an important role in driving the hydrological properties of burned soils, but also in this case the effects can be contrasting e.g., increase in water retention and reduction in sediment transport (Cerdà and Doerr, 2008), or clogging of soil pores and sealing of the soil surface (Keesstra et al., 2014; Onda et al., 2008). Furthermore, infiltration rates and soil water repellency can be different in soils having the same properties, but covered by trees of different species or age (Zema et al., 2021a, 2021b).

Runoff and erosion increases have been observed after prescribed fires in different ecosystems, such as heathlands, shrublands and gorse (Vega et al., 2005). However, despite an ample literature about the impacts of fire on soil hydrology, the studies about the runoff and erosion rates after prescribed fires are not exhaustive and often contrasting (Cawson et al., 2012; Shakesby et al., 2015). According to González-Pelayo et al., (2010) and Vega et al., (2005), increases in runoff and erosion by one and two orders of magnitude, respectively, may be observed after a prescribed fire compared to unburned areas (Cawson et al., 2013). In contrast, (Coelho et al., 2004) and (de Dios Benavides-Solorio and MacDonald, 2005) reported minimal erosion after prescribed fire (Morris et al., 2013). (Keesstra et al., 2014) reported even lower erosion in areas burned with prescribed fire compared to unburned forests, despite comparable runoff.

On a modeling approach, several authors have evaluated the prediction capacity of hydrological models in burned forest areas with or without post-fire treatments (Zema, 2021b). About runoff prediction, the SCS-CN method is commonly used in fire-affected forests, but the dataset of the CN values, an essential parameter to achieve accurate runoff predictions (Shrestha et al., 2006), is not complete for burned conditions (Soulis, 2018; Springer and Hawkins, 2005).

The USLE-family models (USLE, RUSLE, MUSLE, USLE-M, HUSLE, etc.) were developed to estimate soil erosion in agricultural lands, and their applicability in burned forest is not predictable, since the fire impacts on vegetation and soil properties are different, and complicated by several factors (e.g., soil water repellency, ash effects), especially in the Mediterranean environment (Lopes et al., 2021). Moreover, these equations have been applied mainly for multi-year predictions, while the modelling experiences in the window of disturbance have been carried out using RUSLE and mainly after wildfires (e.g., (Fernández et al., 2010; Karamesouti et al., 2016; Larsen and MacDonald, 2007)). To the authors' best knowledge, no applications are available in the literature about the use of MUSLE and USLE-M models after prescribed fires. From this, the optimal values of C-factors, which are key factors for accurate estimations of erosion (Panagos et al., 2015), are not yet identified in burned conditions and post-fire management (such post-fire management actions). Moreover, the validation of soil erosion models applied in soils with post-fire treatments - such as mulching - is particularly scarce all over the world (Fernández et al., 2010; Robichaud et al., 2007), and the applicability of SCS-CN method and USLE-family models may be questionable without targeted modelling experiences (Zema, 2021b).

Moreover, some studies which have analysed soil changes after prescribed fires detected increases in pH, electrical conductivity, total carbon and nitrogen, magnesium, calcium and potassium in burned soils (e.g., Úbeda et al. 2005; Scharenbroch et al. 2012; Alcañiz et al. 2020). However, how these compounds or elements vary in soils after low-intensity fires differs. For instance, total carbon content may decrease or its changes may be extremely small or not significant (Alcañiz et al., 2018; Úbeda et al., 2005). Likewise, the changes in nitrogen may increase (Alcañiz et al., 2016; Kennard and Gholz, 2001; Shakesby et al., 2015), be stable (Lavoie et al., 2010; Moghaddas and Stephens, 2007; Neill et al., 2007) or even decrease (Alcañiz et al., 2020; Badía et al., 2017; San Emeterio et al., 2016).

The duration of the effects of the prescribed fire on soil properties is also debated in the literature. Some authors report unaltered pH values and ephemeral variations in electrical conductivity (Alcañiz et al., 2020; Badía et al., 2017), while, one year after the prescribed fire, other research shows marked decreases in nitrogen (Blankenship and Arthur, 1999; Muqaddas et al., 2015) as well as the recovery to pre-fire contents of magnesium and calcium (Alcañiz et al., 2020). It has been found that the recovery of pre-fire soil properties may take place over short (Zhao et al., 2015) or long (Alcañiz et al., 2016) time spans, depending on fire temperature and residence time, topography of the burned area, rainfall, and degree of vegetation regeneration (Girona-García et al., 2019; Úbeda et al., 2018).

Concerning forest regeneration after prescribed fires, for some forest species, such as oak, which is predominant in Mediterranean fire-prone areas and has adapted to fire (Curt et al., 2009), studies of fire effects on regeneration have been shown varying results (Petersson et al., 2020). Oak woodlands rely on the ability of this species to survive fire, resprout efficiently after fire and regenerate from seeds (Arthur et al., 2012; Pausas, 2006). However, failure of oak regeneration from acorns is frequent in Mediterranean countries (Curt et al., 2009; Maltez-Mouro et al., 2007). Oak is fire-adapted and moderately shade-intolerant, and therefore prescribed fire can be used as a management tool to decrease competition and increase light levels, thus promoting oak regeneration in some ecosystems (Brose et al., 2013; Izbicki et al., 2020). Prescribed fire also eliminates excess fire-sensitive vegetation and reduces litter, favouring the emergence of acorns (Blankenship and Arthur, 2006). Many studies have shown that fire alone or in combination with other treatments, such as partial canopy removal, can improve the establishment and growth of regenerating oak trees (Hutchinson et al., 2005). However, several oak species have exhibited unsuccessful regeneration in recent years, and it is difficult to identify the reasons, due to a multitude of biotic and abiotic factors influencing this ecological processes (Curt et al., 2009; Pausas et al., 2004; Royse et al., 2010).

Research questions

From this short analysis of the state-of-the-art, it is evident that, in the literature about the effects of prescribed fire on forest ecosystem, there is a large gap of knowledge about several aspects. It results that several research questions remain open after decades of relevant studies (Figure 1).

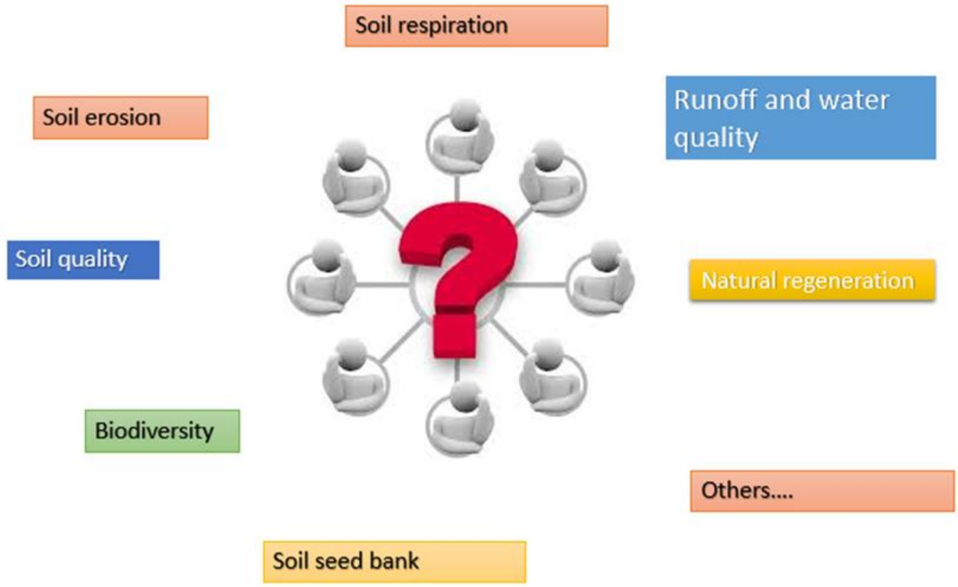

Figure 1 - Open questions regarding effects of prescribed fire on different ecosystem components.

First, since the literature about the changes in infiltration rates and water repellency of soils after the prescribed fire is contrasting and therefore not exhaustive (Cawson et al., 2012), it is not still clear whether infiltration rates increase or not and how much soil water repellency raises up. Therefore, we aim to test whether: (i) water infiltration varies by fire in the short term, and mulch cover with fern can limit this decrease; and (ii) soil water repellency can affect forest soil immediately after fire, and this effect is temporarily (that is, it disappears some months after fire).

As mentioned above, also the knowledge about surface runoff and erosion rates in Mediterranean fire-prone forests is not complete, because the magnitude and duration of the increases in these rates has not sufficiently quantified. About this issue, the research questions are the following: (i) how much does the prescribed fire affects runoff and erosion rates on the short term after its application? (ii) how long is the window of disturbance of soil hydrology due to fire? (iii) are the fern residues effective as mulching cover to reduce runoff and erosion after fire?

Further research is also needed to better understand the effects of prescribed fires on soil changes in environments with characteristics that contrast the previous studies (Hubbert et al., 2006; Hueso-González et al., 2018), especially where the wildfire and erosion risks are high, due to climatic and geomorphological characteristics. To better understand these effects, we want to explore whether and how much: (i) prescribed fire modifies the soil properties in the short term after burning; and (ii) mulching with fern is able to reduce these changes.

On the modelling approach, more studies should assess the prediction capacity of surface runoff and soil loss offered by SCS-CN, Horton and USLE models in burned forest plots of Mediterranean areas under commonly adopted pre-fire and post-fire management, such as prescribed fire and soil mulching. To this aim, there is the need to reply to the following research questions: (i) are the tested models reliable and accurate to predict surface runoff and soil erosion in Mediterranean burned forests? (ii) which are the optimal values of the input parameters of the tested models (CNs and C factors)?

Finally, following the ecological approach, very few evaluations about the effect of fern on early recruitment of oak species in burned soil are available in literature. It results that the question of how prescribed fire and post-fire mulching affects the early seedling recruitment of this important plant species of the Mediterranean forest ecosystems is still not understood. Since oak is a forest species that is adapted to fire, we hypothesise that prescribed burning enhances the initial recruitment of plants, and this effect may be synergistic with mulching application.

Novel aspects

To the authors' best knowledge, the present study is the first investigation that has been carried out on the effects of prescribed fire and post-fire management on the forest ecosystems of Southern Italy, where the post-fire recruitment of vegetation may be difficult due to climate and soil characteristics, and the fire hazard is high and the hydrological response is dangerous for assets and human lives. The latter issues are important concerns in ecosystems when risk of wildfire and heavy rainstorms endanger forests on steep slopes with highly erodible soils, such as in many environments of Southern Europe.

Moreover, mulching with fern residues as post-fire management strategy has not been previously carries out and evaluated in burned forests worldwide. Therefore, no evidence exists in the literature about the effectiveness of fern mulching at controlling hydrological response and restore soil properties after fire, and at enhancing plant regeneration in the short-term after the prescribed fire.

General objective and organisation

According to the research needs and to fill the literature gaps identified as above, this PhD thesis evaluates the short-term effects of the prescribed fire and post-fire mulching on hydrology and other properties of soil as well as on plant regeneration in Mediterranean forests. A targeted investigation has been properly designed and carried out after the prescribed fire and mulching in three forest stands (pine, chestnut and oak) of Southern Italy. To this aim, several plots have been installed in as many experimental sites close to the municipality of Samo (Reggio Calabria) between March and May 2019. A prescribed fire has been deliberately applied in early June 2019 and the burned soil has been mulched with fern residues immediately after burning. These evaluations have been carried out adopting both a two-year monitoring activity and a modelling approach regarding the hydrological variables.

The PhD thesis is presented as a collection of five articles published or submitted in peer-reviewed SCOPUS and ISI indexed, international journals and consists of the following chapters.

The first article investigates water infiltration, runoff volume and peak flow (using a portable rainfall simulator), and soil water repellency (using Water Drop Penetration Test) at the point scale in the three forest sites. The second article evaluates surface runoff volumes and soil losses in the plots of the same forest sites after natural precipitation. The third article analyses the short-term changes in selected soil properties and covers. The fourth article assesses acorn emergence and seedling survival as well as some important biometric characteristics (height, diameter, and dry weight) of oak plants and roots in the same plots as in the second article. The fifth article applies a modelling approach to evaluate the prediction capacity of runoff and soil loss using the SCS-CN, Horton and USLE-family models (MUSLE and USLE-M) in the experimental plots.

A final chapter of the Ph.D. thesis reports future perspectives and identifies further research needs with the aims to give forest managers insight about the feasibility of the experimented strategies (prescribed fire and post-fire soil mulching with fern) to reduce the wildfire hazard and control the soil hydrological response of forests of Southern Italy.

CHAPTER 1

Water Infiltration after Prescribed Fire and Soil Mulching with Fern in Mediterranean Forests

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Abstract: Prescribed fire is commonly used to reduce the wildfire risk in Mediterranean forests, but the soil's hydrological response after fire is contrasting in literature experiences. The mulch treatment can limit the increases in runoff and erosion in the short term after a fire. The use of fern is preferable to straw, due its large availability in forests. However, no experiences of post-fire treatment with fern mulch have been found in the literature and therefore the mulching effectiveness has not been evaluated. This study has measured water infiltration rate (IR) and water repellency (SWR) using a rainfall simulator in three Mediterranean forest stands (pine, oak and chestnut) of Calabria (Southern Italy) after a prescribed fire and mulching treatment with fern in comparison to unburned soil. Prescribed fire reduced water infiltration in all forests in the short term compared to the unburned conditions, and increased SWR in pine and oak forests. These reductions in IR in the time window of disturbance after fire increased the runoff generation capacity in all soils, but had a lower effect on peak flows. However, soil mulching with fern limited the runoff rates and peak flows compared to the burned soils, but this treatment was less effective in pine forest. One year after fire, IR increased in burned soils (treated or not) over time, and SWR disappeared. The effects of mulching have disappeared after some months from fire. The study confirms the usefulness of mulching in broadleaves forest in the short term, in order to

control the hydrological effects of prescribed fire in Mediterranean forests. Both post-fire management techniques should be instead adopted with caution in conifer forests.

Keywords: water infiltration rate; rainfall simulator; time to peak; peak discharge; surface runoff; soil water repellency

1. Introduction

Prescribed fire is the planned use of low-intensity fire to achieve very different goals given certain weather, fuel and topographic conditions (Alcañiz et al., 2018; Fernandes et al., 2013). This management tool has been adopted since long time in many countries under different climatic conditions (e.g., Mediterranean ecosystems) to mitigate the impact of large-scale wildfire in forested environments (Neary et al., 1999). The main objective of prescribed fire is the creation of forest areas with low fuel, in order to reduce intensity, severity and damage of large wildfires (Plaza-Álvarez et al., 2018). However, prescribed fire leads to other beneficial effects, since it facilitates the germination and growth of understory vegetation, increases landscape heterogeneity, and improves pastures for livestock (Lucas-Borja et al., 2019; Neary et al., 1999; Úbeda et al., 2018).

Prescribed fires are usually of low intensity and their effects depend on type and amount of fuel load and soil moisture (Alcañiz et al., 2018; Plaza-Álvarez et al., 2019). The forest areas, where fire is applied in each operation, are variable in size, and depend on the topographic and climatic conditions (Neary and Leonard, 2021). The effects of the prescribed fire are thought to have small impacts on the different components of the forest ecosystems, due to the low fire severity and patch nature (Cawson et al., 2012; Francos and Úbeda, 2021). Literature has shown that prescribed fires do not have generally detrimental impacts on soil properties, since the burning temperature and fire duration are low (Cawson et al., 2016; González-Pelayo et al., 2015; Pereira et al., 2018). However, research is not unanimous about the effects of prescribed fire on the hydrological response of forest soils. For instance, González-Pelayo et al. (González-Pelayo et al., 2010) and Vega et al. (Vega et al., 2005) report increases in runoff and erosion by one and two orders of magnitude, respectively, compared to unburned areas (Cawson et al., 2013), while, by contrast, Coelho et al. (Coelho et al., 2004), and de Dios Benavides-Solorio and MacDonald (de Dios Benavides-Solorio and MacDonald, 2005) reported minimal erosion after prescribed fire (Morris et al., 2013). This depends on both the operation parameters of fire application (intensity, cover, duration, temperature) and the physical properties of the treated soils (texture, aggregate stability, water content) (Alcañiz et al., 2018; Cawson et al., 2012). In general, fire removes vegetation and modifies the hydrologic properties of soil (Cawson et al., 2012; Certini, 2005; Shakesby, 2011). A soil left bare due to vegetation removal becomes more susceptible to raindrop impact and particle detachment (Zema, 2021). Moreover, water infiltration rates, which are generally high in undisturbed forest soils (Robichaud, 2000), can decrease. Both these fire effects may increase surface runoff and erosion rates (Shakesby et al., 2015).

Particular attention has been paid to Mediterranean ecosystems, where the intrinsic climate and soil characteristics may enhance the hydrological hazards not only in burned forests, but also in unaffected valley areas (Fortugno et al., 2017; Shakesby, 2011). Although the prescribed fires have lower impacts on soil hydrology compared to wildfire, changes in the hydrological response of burned areas may be important (Certini, 2005; Shakesby, 2011; Cawson et al., 2012), with noticeable increases in runoff and erosion rates in the time "window of disturbance" immediately after fire. In the Mediterranean forests, the hydrological processes generating surface runoff are dominated by the infiltration-excess mechanism (Lucas-Borja et al., 2018).In this environment, the reduction in water infiltration rate (IR) of soil can enhance flooding and hydrogeological instability after the frequent and intense rainstorms that are typical of the Mediterranean climate (Fortugno et al., 2017). Moreover, IR can be further decreased by soil water repellency (SWR), which is common in Mediterranean forest soils (Cerdà and Doerr, 2007; Plaza-Álvarez et al., 2019), which is primarily determined by the accumulation of long-chain organic compounds in/around soil particles.

In spite of ample literature, the studies about the changes in SWR and IR after fires of different severity are not consistent and sometime contrasting (Cawson et al., 2012). For instance, increases in SWR were observed by (Plaza-Álvarez et al., 2018) in pine forests burned by prescribed fires in SE Spain; these effects were found to be even higher compared to high-severity fires (Huffman et al., 2001). Other authors report slight or no changes in SWR in soils burned at low temperatures (Doerr et al., 2006; Robichaud, 2000). Therefore, no direct associations between fire severity and SWR can be found, as resulting from several factors, such as the natural variability in the study sites (soil water content and texture, and vegetation type) (Cawson et al., 2012). These inconsistencies also affect soil infiltration rates (Wittenberg and Malkinson, 2009), with reductions (Robichaud, 2000) or not significant changes (Plaza-Álvarez et al., 2019) in IR after prescribed fires. Ash released by fire plays an important role in driving the hydrological properties of burned soils, but also in this case the effects can be contrasting (e.g., increase in water retention and reduction in sediment transport (Cerdà and Doerr, 2008), or clogging of soil pores and sealing of the soil surface (Keesstra et al., 2014; Onda et al., 2008). Furthermore, IR and SWR can be different in soils having the same properties, but covered by trees of different species or age (Zema et al., 2021a, 2021b).

It is evident that the characteristics of the soil surface after fire are critical for its hydrological response at both hillslope and catchment scales (Keesstra et al., 2014). In order to reduce soil's susceptibility to runoff and erosion after fire, several treatments have been proposed and their effectiveness has been verified in many environmental contexts (Lucas-Borja, 2021; Zema, 2021). Mulching is one of the most common post-fire management techniques, particularly when vegetation residues are used (Lucas-Borja et al., 2019; Prosdocimi et al., 2016). The objective of mulching is primarily the prevention of raindrop impact and decrease in hydraulic connectivity, but a mulch cover increases the infiltration rates and soil quality, and soil protection with ground cover, if used properly and at the correct time (Prosdocimi et al., 2016). However, mulching can also have negative impacts on burned soils, since infiltration rates can be reduced when soil is in unsaturated and dry conditions (Lucas-Borja et al., 2018).

Straw is commonly applied as mulching material on burned soils, and, in this case, the mulch cover can be removed by wind in some areas and become too thick in others, which hamper vegetation regeneration (Robichaud et al., 2020). In addition, seeds, agro-chemicals and parasites may be transported by straw, determining the development of non-native vegetation and diseases to plants (Bento-Gonçalves et al., 2012). Forest residues and herbaceous plants may properly replace straw as mulch cover, since do not contain non-native seeds or chemical residues, and are more resistant to wind (Zema, 2021). In Mediterranean forest floor, residues of fern (*Pteridium aquilinum* (L.) Kuhn) - a vascular plant that abundantly grows also in semi-arid climates (where the water competition among plants is high, (Bombino et al., 2019b), - contains low lignin (and thus can be easily incorporated into the soil over time) and can be easily transported inside the forestland. Therefore, its use as mulch material in fire-affected areas could be preferable to straw. However, to the best authors' knowledge, no experiences about using fern to protect burned soil from runoff and erosion impacts after fire are available in literature. The effectiveness of soil mulching with forest residues, such as fern, are of utmost importance, to control runoff and erosion generation. Therefore, the beneficial effects of this species on the hydrological properties of burned soils should be explored.

This study evaluates surface hydrology of three Mediterranean forest stands (pine, oak and chestnut) in Calabria (Southern Italy) after a prescribed fire and mulching treatment with fern in comparison to unburned soil. More specifically, IR, surface runoff and peak flow were measured immediately and one year after fire using a portable rainfall simulator, and SWR was determined using Water Drop Penetration Test. We aimed to test if: (i) water infiltration is significantly reduced by fire in the short term, but the mulch cover with fern can limit this decrease, and (ii) SWR can affect forest soil immediately after fire, but its effect is temporarily, since it disappears after some months.

2. Materials and Methods

2.1. Study area

The experiments were carried out in three forest sites close to the municipality of Samo (Calabria, Southern Italy) (Figure 1). The first site ("Calamacia", UTM coordinates 590293 E; 4215327 N) was a reforested stand of pine (*Pinus pinaster* Aiton, age of 20 years old) at an elevation between 650 and 700 m above sea level. The second forest ("Rungia", 588635 E; 4216172 N) consisted of a natural stand of oak (*Quercus frainetto* Ten.) between 900 and 950 m. A third stand was a site ("Orgaro", 590389 E; 4215530 N) reforested in 1990 with chestnut (*Castanea sativa* Mill.) at 700-750 m. In all sites, shrub formations mainly consist of *Quercus ilex L., Rubus ulmifolius S., Leucanthemum vulgare Lam.* (pine forest), *Cyclamen hederifolium, Leucanthemum vulgare Lam*. (oak) *and Rubus ulmifolius S., Pteridium aquilinum L., Leucanthemum vulgare Lam.* (chestnut). None of these forest stands have been managed from their plantation.

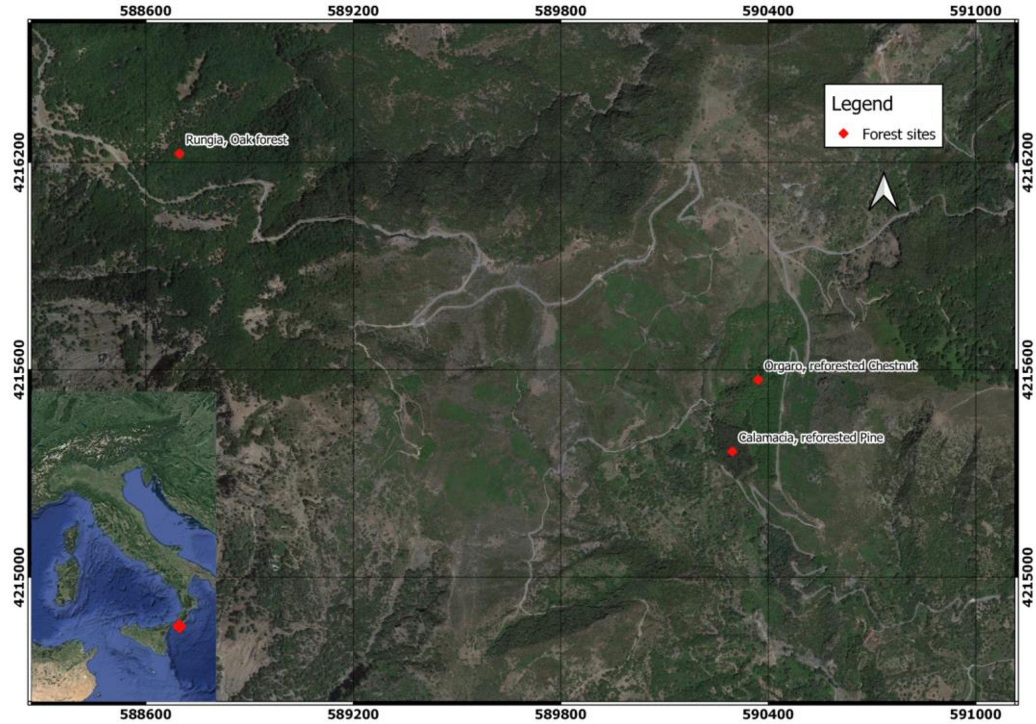

Figure 1. Location of the experimental site (Samo, Calabria, Southern Italy

The climate of the area is typically semi-arid ("Csa" class, "Hot-summer Mediterranean" climate, according to Koppen (Kottek et al., 2006)). Winters are mild and rainy, while summers are warm and dry. The mean precipitation is 1102 mm per year, while the mean annual temperature is 17.4 °C with minimum and maximum values of - 4.3 and 43.1 °C, respectively (weather station of Sant'Agata del Bianco, UTM coordinates 4217548 N, 595159 E, period 2000-2020).

2.2. Experimental design

In each experimental site, two areas were delimited in forest hillslopes with the same gradient (19.8 ± 0.62%) (Figure A1).

A first area (about 200 m^2) was burned in early June 2019, simulating a prescribed fire. The fire operations were carried out with the support of the Forest Regional Agency ("Calabria Verde") and the surveillance of the National Corp of Firefighters. During fire, air temperature was on the average 26 (Calamacia, pine forest) to 43 (Rungia, oak) °C (Figure A2) The mean and maximum soil temperatures, measured by a thermocouple (at a distance of 0.50 from fire flame) connected to a datalogger at a soil depth of 2 cm, were 22.0 and 22.7 °C (pine), 21.0 and 26.9 °C (oak), and 24.7 and 28.8 (chestnut) °C. Fire operations lasted from 15 to 30 minutes, and flame had a maximum temperature between 645 (chestnut) and 720 (oak) °C. Wind was practically absent and air humidity was between 50 and 60%.

In this burned area, a small portion (about 3 m^2) was covered with small branches of fern, which was cut from an adjacent zone and spread on the ground at a dose of 500 g/m^2 of fresh weight (Figure A2) in the same day after the prescribed fire application. The mulch dose is equivalent to a dry matter of 200 g/m^2 of straw, usually applied in burned and mulched areas (Lucas-Borja et al., 2018; Vega et al., 2014).

A second area was not burned and left undisturbed and was considered as "control" for each experimental site.

The soils of the experimental sites were of loamy sand texture (10.6 ± 2.57% of silt, 8.76 ± 0.61% of clay, and 80.7 ± 2.68% of sand), except for the unburned area of the pine forest in Calamacia, which was sandy loam (10.1 ± 1.01% of silt, 9.0 ± 0.01% of clay, and 81.0 ± 0.99% of sand).

Overall, the experimental design consisted of three forest stands (pine, oak and chestnut) × three soil conditions (unburned, burned and not treated, and burned and mulched).

2.3. Hydrological measurements and analysis

For each forest stand and soil conditions, rainfall simulations were carried out in three points randomly chosen. An Eijelkamp® rain simulator was used (Hlavčová et al., 2019; Iserloh et al., 2013), according to the method setup by Bombino et al. (Bombino et al., 2019a) (Figure A3). This simulator allows reproducing rainfall with maximum height and intensity of 18 mm and 360 mm h^{-1} (drop diameter of 5.9 falling 40 cm from the ground) and collecting runoff and sediments in a small bucket. Previously, the simulator was calibrated by generating a rainfall of 3.0 mm at an intensity of 37.8 mm h^{-1} over a surface area of 0.305 m x 0.305 m. The water volume in the sprinkler tank (about 0.3 litres) was dosed by varying the pressure head, as suggested in the operating manual. In more detail, the pressure head on the sprinklers has been properly tuned by moving the aeration tube upward or downward. Throughout the simulated rainfall (300 s), the surface runoff volume was collected in a small graduated bucket on a time scale of 30 s. The time to peak, peak flow and cumulated runoff volume were measured. More specifically, the time to peak was the time measured from the rainfall start to peak flow occurrence.

Moreover, the infiltration curves were determined by subtracting the runoff from the rainfall at each 30-s time interval. The infiltration test stopped when three equal time measurements of instantaneous infiltration had been recorded. The final infiltration rate was assumed as the IR.

Finally, SWR was measured immediately beside (about 0.25 m) to the measurement point of IR, using the Water Drop Penetration Test (WDPT) method (Letey, 1969; Woudt, 1959) on a soil with a natural water content. According to this method, 15 drops of distilled water were released, using a pipette, on the soil surface. The time necessary for drops to completely penetrate into the soil was measured. SWR was classified according to the values of WDPT proposed by (Bisdom et al., 1993): (i) non water-repellent or wettable soil (class 0, WDPT < 5 s); (ii) slightly water-repellent soil (class 1, 5 < WDPT < 60 s); (iii) strongly water-repellent soil (class 2, 60 < WDPT < 600 s); (iv) severely water-repellent soil (class 3, 600 < WDPT < 3600 s); and (v) extremely water-repellent soil (WDPT > 3600 s) (Figure A4) .

SWR is strongly influenced by soil humidity (Alagna et al., 2017; Dekker and Ritsema, 1994). For this reason, soil water content (SWC) was also measured simultaneously to SWR by a device (Vegetronix VG400, accuracy of 2%, measurement range 0-50%), placed on the soil surface and connected to a data logger (UX120 4-channel Analog Logger, Onset HOBO, Massachusetts, USA).

Although there was no difference between burned and not treated as well as burned and mulched soils immediately after fire (because the mulch material was not incorporated into the soil), the WDPT tests were carried on both soil conditions since fire and mulch application.

2.4. Statistical treatment

The statistical significance of the differences among soil conditions for each forest stands was determined applying 2-way ANOVA test to the hydrological variables measured (IR, SWR, time to peak, cumulated runoff and peak flow). The latter was considered as dependent variables, while the simulation date (1 or 365 days from fire) and the soil condition (unburned, burned, and burned and mulched) were the independent factors. The pairwise comparison by Tukey's test (at $p < 0.001$) was also used to evaluate the statistical significance of the differences in the hydrological variables SWR among factors. In order to satisfy the assumptions of the statistical tests (equality of variance and normal distribution), the data were subjected to normality test or were square root-transformed whenever necessary and separately for each forest species. The statistical analysis was carried out using the XLSTAT release 2019 software.

3. Results

Soil infiltration rates started from values around 37-38 mm/h (chestnut and pine) and 18-20 mm/h (oak for unburned and burned and mulched soils immediately after the prescribed fire) and decreased over time, when soil progressively was saturating. The time to steady conditions was short, from 180 (unburned soil in oak forest) to 390 s. This time was generally lower for measurements made one year after fire for pine and oak forests, without differences among the soil conditions. IR decreased faster in unburned soils immediately after fire, and throughout similar times one year after (Figure 2).

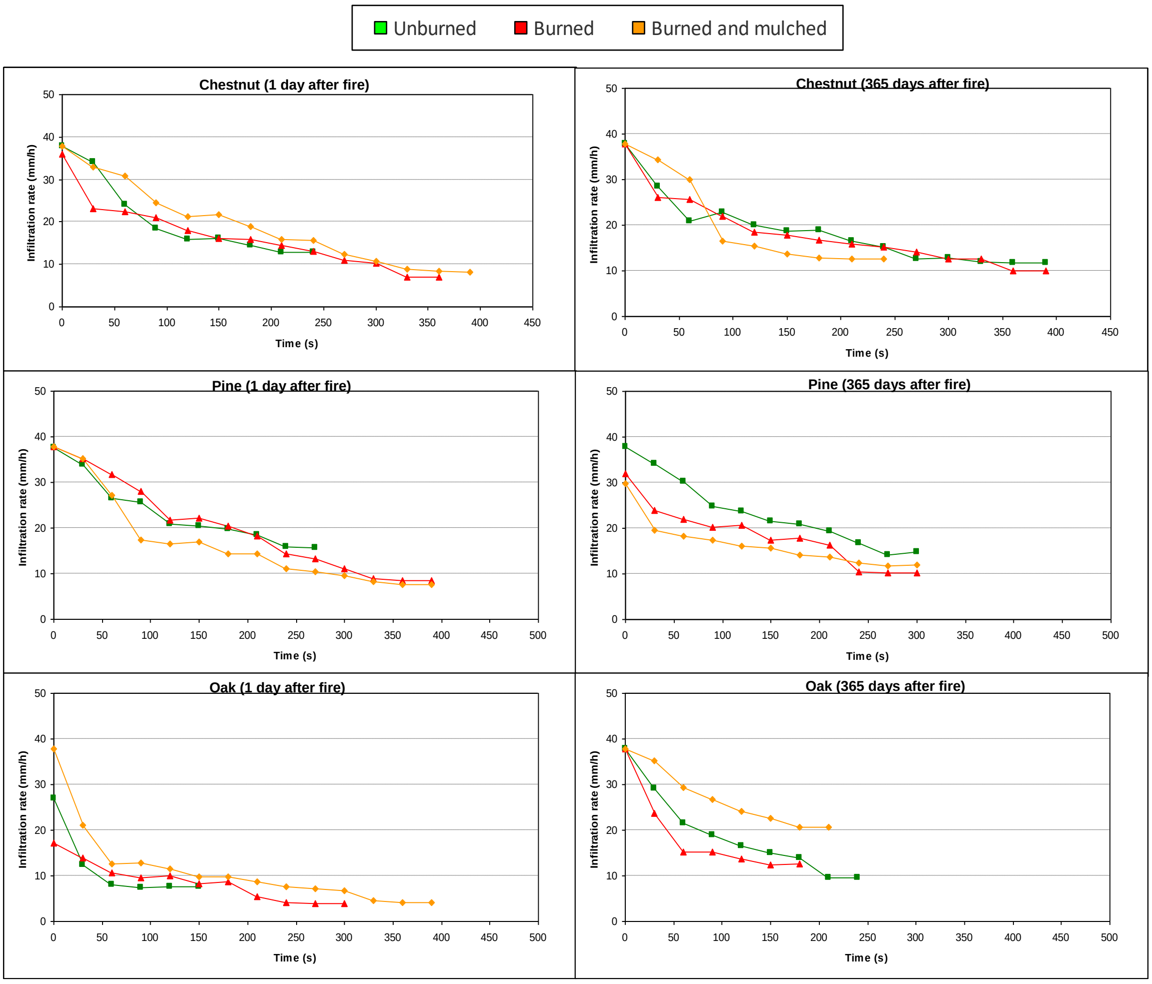

Figure 2. Infiltration curves measured using a portable rainfall simulator after prescribed fire and soil mulching with fern in the experimental site (Samo, Calabria, Southern Italy).

The analysis of water infiltration in unburned soils under steady conditions shows the highest IR in pine forest and the lowest in oak soil, while the soil covered by chestnut had intermediate values. This IR was on average stable over time, with very slight reductions in chestnut and pine forests, and a slight increase in oak soil one year after fire (Figure 3).

Although being of low severity, prescribed fire reduced the mean IR in soils of all forest species compared to the unburned conditions (Figure 3), and these differences were significant for chestnut and pine soils. The application of fern mulch helped in slightly increasing the mean IR in chestnut and oak soils, but not in pine (Figure 3). These variations were significant only for chestnut forest compared to burned soils, while the differences in IR between unburned as well as burned and mulched soils were significant only for pine soil.

One year after fire, the differences between the two survey dates were significant for oak and chestnut. The mean IR of burned soils increased compared to the measurements made immediately after fire. This effect was more noticeable in oak soils compared to chestnut and pine forests. The soils mulched with fern showed the highest IR among all soil conditions, except for pine soils. These values were always higher compared to the burned soils with a noticeable increase detected in oak soils (Figure 3).

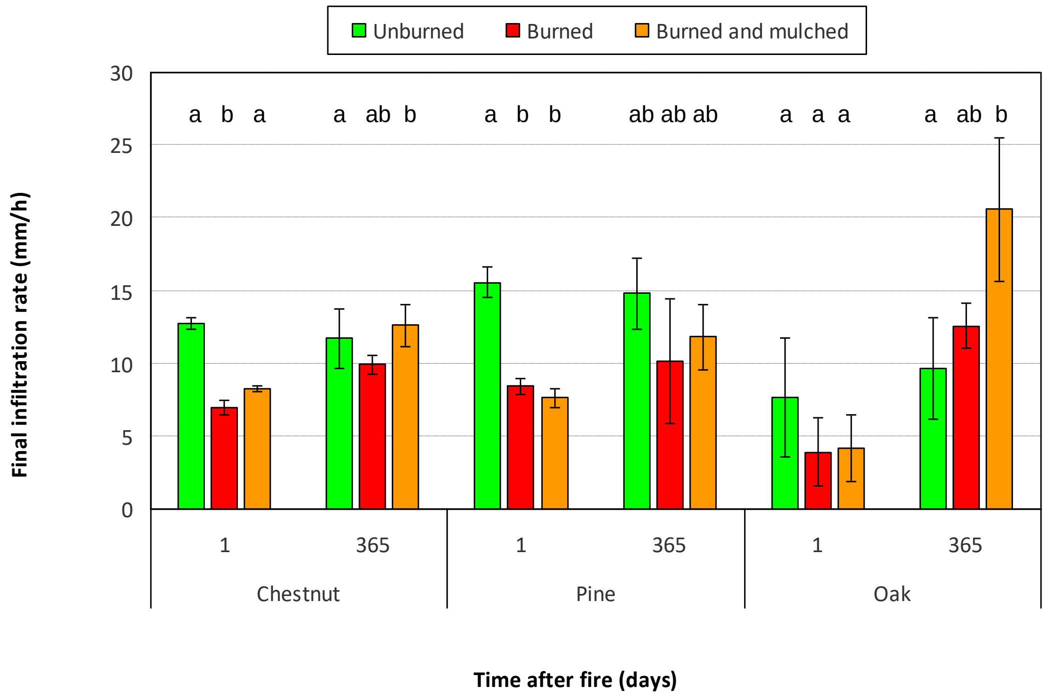

Figure 3. Water infiltration rate measured using a portable rainfall simulator after prescribed fire and soil mulching with fern in the experimental site (Samo, Calabria, Southern Italy). Note: different lowercase letters indicate significant differences in the interaction between soil conditions and time after fire, respectively, after Tukey's test ($p < 0.05$).

SWC did not show significant variations among all soil conditions in the two measurement dates when the SWR measurements were carried out. In more detail, mean SWC was always in the range 15-20% for all forests and soil conditions at both measurement dates, except for oak soils immediately after fire, when SWC was on average between 16 and 22%, although this difference was not significant (Figure 4).

The similarity in SWC among the three soil conditions for each survey makes possible the comparisons of SWR values in the forest soils (De Jonge et al., 1999). Immediately after fire, the unburned soils showed only a slight repellency (chestnut and pine) or no SWR (oak). Prescribed fire determined a slight repellency in chestnut soils, while a strong repellency was found in both pine and oak, with or without mulch cover (Figure 5). The difference between the mulched and not treated soils was due to the natural variability of the soil properties. Only the burned soil of oak forest had a significantly different SWR compared to unburned forest, while SWR of burned and mulched soils were always significantly different compared to unburned sites. Fire increased WDPTs of all soils, and the difference over time was significant for all forest species. These variations did not alter SWR of chestnut forest (still in class 1), but made the pine and oak soils strongly repellent (SWR of class 3). For the soils treated with fern mulch, WDPT values increased in chestnut and pine, respectively, and decreased in oak. The SWR class was the same as the burned soils for all forests.

One year after fire all soils became non-repellent (SWR of class 1), as shown by the WDPT values that were equal to one (Figure 5).

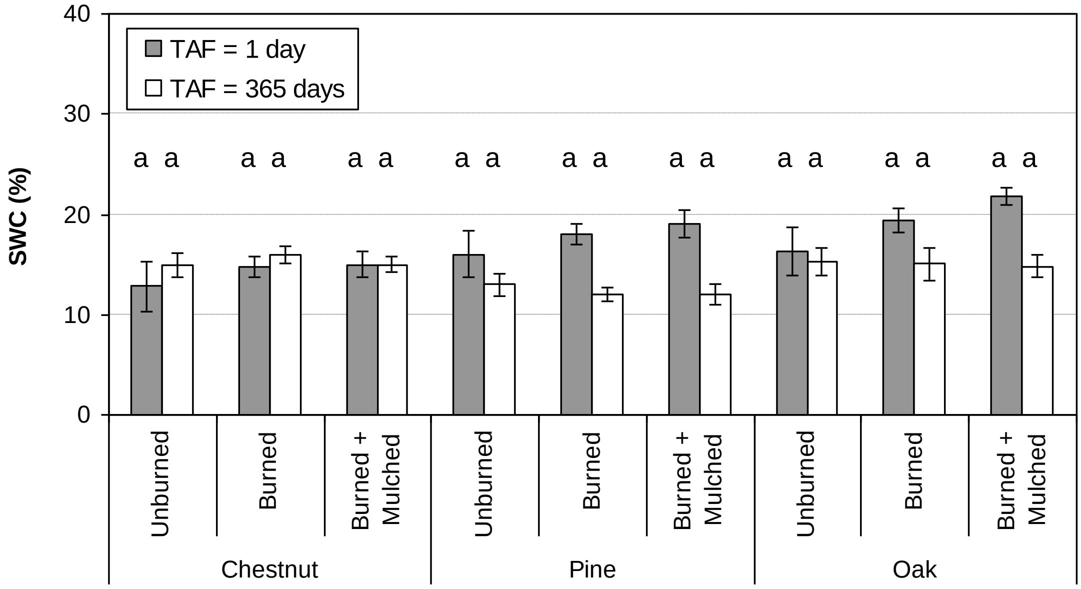

Figure 4. Soil water content (SWC) measured after prescribed fire and soil mulching with fern in the experimental site (Samo, Calabria, Southern Italy). Notes: bars report mean ± std. dev. (n = 3); TAF = time after fire. Different lowercase letters indicate significant differences in the interaction between soil conditions and time after fire, respectively, after Tukey's test ($p < 0.05$).

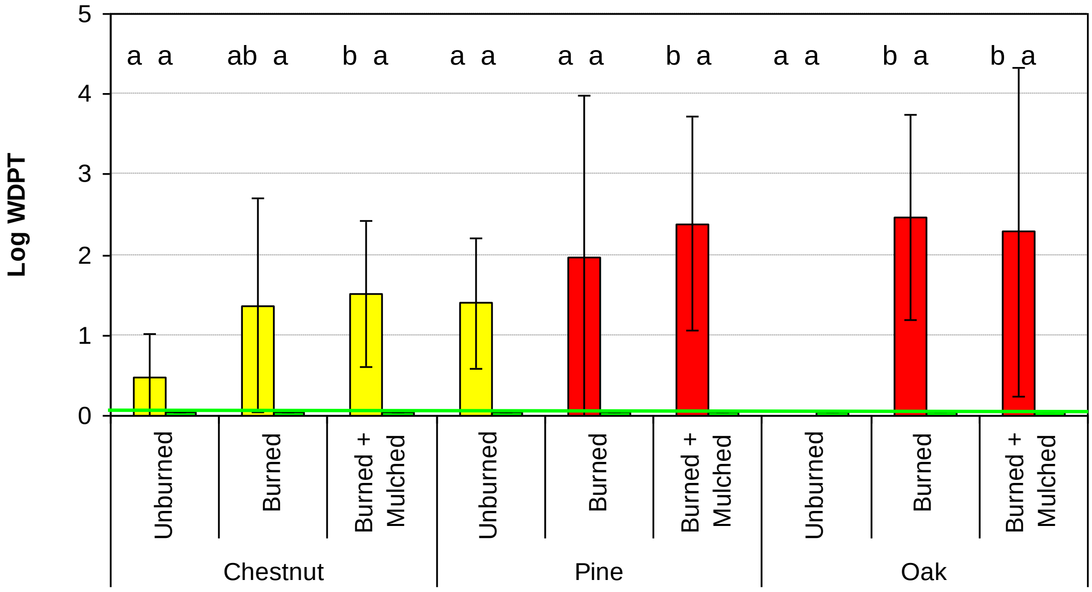

Figure 5. Soil water repellency (as WDPT) measured after prescribed fire and soil mulching with fern in the experimental site (Samo, Calabria, Southern Italy). Notes: Bars report mean ± std. dev. (n = 3); left and right bars of chart are the values measured one day and one year after the prescribed fire, respectively. Green, yellow and red colours refer to SWR class 0, 1

and 2, respectively. The green line identifies the non-repellency condition of soil. Different lowercase letters indicate significant differences in the interaction between soil conditions and time after fire, respectively, after Tukey's test ($p < 0.05$).

The variations of the monitored hydraulic properties determined changes in the hydrological response of the three forest soils among the different conditions. More specifically, the runoff generated in the unburned soils by the simulated rainfall did not show significant variations at the two dates of the experiment. Fire negatively affected the runoff generation capacity in chestnut and mainly in oak soils (but only in this case significantly), while this increase was negligible for pine. Soil mulching with fern was effective, although not significantly, in reducing the runoff response in chestnut soils, and less in oak forests, while the runoff measured in pine soils was basically the same as the other soil conditions. One year after fire, the hydrological response of oak soils was statistically similar among the three conditions, while the burned soils of pine and chestnut showed a higher runoff generation capacity (Table 1).

Small (±11.5%, pine and oak soils) or negligible (+5.2%, chestnut soil) variations in peak flow were measured in all forest soils immediately after the prescribed fire. The mulch application reduced the peak flow in chestnut (by 50%) and oak (by 26.5%) soils down to values that were even lower (by 18%, oak, and 47.4%, chestnut) than in unburned soils. In contrast, in the pine forest treated with mulching, the peak of the hydrograph increased by 13.6% compared to the unburned soil. One year after fire, the increase in peak flow between burned and unburned soils became high only in the pine soil (+85.8%), while these variations were lower in the other forest soils (+10.3%, chestnut, and +27.9%, oak). The mulch application did not noticeably alter the peak flow measured in the unburned soils of chestnut (+6.9%) and oak (-3.3%) forests, which instead increased very much in pine soils (+92.9%) (Table 1).

The time to peak (between 90 and 120 s in unburned soils) generally decreased (to 60 s) in burned soils at both the dates of rainfall simulations, with one exception (150 s, pine forest immediately after fire). Compared to unburned soils, mulch application increased (chestnut and oak soils, 150 s) or did not vary (pine forest) the time to peak immediately after fire. One year after, this time was the same as in the unburned soils (chestnut and oak forest), while decreased to 90 s in pine soils (Table 1).

Table 1. Runoff volume, peak flow and time to peak measured after prescribed fire and soil mulching with fern in the experimental site (Samo, Calabria, Southern Italy). Mean ± std. dev. (n = 3).

Species	Time after fire (days)	Soil condition		
		Unburned	Burned	Burned and mulched
		Runoff (mm)		
Chestnut	1	0.44 ± 0.07 a	0.60 ± 0.11 a	0.29 ± 0.17 a
	365	0.40 ± 0.24 a	0.56 ± 0.05 a	0.49 ± 0.12 a
Pine	1	0.33 ± 1.00 a	0.35 ± 0.03 a	0.36 ± 0.01 a
	365	0.41 ± 0.26 a	0.64 ± 0.57 a	0.65 ± 0.19 a
Oak	1	0.54 ± 0.40 ab	1.21 ± 0.00 c	1.06 ± 0.30 a
	365	0.63 ± 0.06 ab	0.65 ± 0.02 bc	0.61 ± 0.05 bc
		Peak flow (mm h^{-1})		
Chestnut	1	11.46 ± 0.85 a	12.06 ± 0.01 a	6.03 ± 5.12 b
	365	11.66 ± 5.70 a	12.86 ± 2.41 a	12.46 ± 3.04 a
Pine	1	9.55 ± 0.28 a	8.44 ± 1.84 a	10.85 ± 10.23 a
	365	8.44 ± 1.65 a	15.68 ± 19.61 b	16.28 ± 1.21 b
Oak	1	24.52 ± 5.57 a	27.34 ± 5.12 a	20.10 ± 3.48 a
	365	13.67 ± 4.18 b	17.49 ± 0.70 b	13.22 ± 3.04 b
		Time to peak (s)		
Chestnut	1	120 ± 60 a	60 ± 30 a	150 ± 30 b
	365	90 ± 7.67 a	60 ± 30 a	90 ± 0 a
Pine	1	120 ± 30 a	150 ± 30 b	120 ± 60 a
	365	120 ± 30 a	60 ± 60 c	90 ± 0 a
Oak	1	90 ± 0 a	90 ± 0 a	115 ± 30 a
	365	120 ± 15 b	60 ± 15 c	120 ± 30 c

Note: different lowercase letters indicate significant differences in the interaction between soil conditions and time after fire, respectively, after Tukey's test ($p < 0.05$).

4. Discussion

The Mediterranean climate is characterized by flash rainfall event with very low duration and, therefore, high intensity (Fortugno et al., 2017). Moreover, the dominant runoff generation mechanism is "infiltration-excess" (Shakesby, 2011). When a high-intensity rainfall exceeds the water infiltration rate of soil, runoff can be generated also in the early stage of a rainstorm, and this makes significant our experimental rainfall simulation. Moreover, the measurement of the IR carried out in this study relates to the unsaturated fire-affected soil, and this parameter may be initially more important than saturated hydraulic conductivity at the time scale of convective rainfall, which is typically short and only lasts 20-60 min, but common in many post-wildfire response domains (Lucas-Borja et al., 2018; Moody et al., 2013). Therefore, the rainfall simulations carried out in the experimental site helped in identifying the changes in IR and SWR after the prescribed fire, which are somewhat contrasting in literature (Cawson et al., 2012). The IR of burned soils decreased more rapidly compared to the other soil conditions (mainly one year after fire), and this may be a detrimental effect for soil. The reduction in IR in the short term may be due to two synergistic effects. First, the presence of the ash layer left by fire may have clogged soil pores and induced sealing of the soil surface, (Keesstra et al., 2014; Onda et al., 2008), while was not able to increase water adsorption before infiltration (Balfour and Woods, 2007; Cerdà and Doerr, 2008), due to its small thickness. Other studies about prescribed fire have shown that ash contributed to reduce infiltration, creating a thin layer (few mm) of low porosity and permeability (Balfour and Woods, 2007). Second, fire has induced water repellency in the soils of all forest species, with particular intensity in pine and oak forests (where repellency was strong). This result is basically in line with some previous studies, which showed that increases in SWR are common after low-intensity fires (Plaza-Álvarez et al., 2018). However, literature results are not consistent about occurrence and extent of SWR after prescribed fires (Alcañiz et al., 2018), since water repellences may increase (Granged et al., 2011), does not change (Robichaud, 2000) or decrease (Pierson et al., 2008).

Reductions in water infiltration rates have been shown by some authors after prescribed fires (Robichaud, 2000; Wittenberg and Malkinson, 2009), but these effects are limited to a few months after burning and generally are not significant (Fernández et al., 2008; Plaza-Álvarez et al., 2019). In contrast, significant decreases in steady-state infiltration rates were found by other studies in areas burned with prescribed fires, and these reductions were attributed to the synergistic effects of increased water repellency and sealing, and reduced vegetation cover (Cawson et al., 2016).

In the oak wood of our experimental site, the noticeable SWR did not lead to as much significant decrease in IR, and this may be surprising, since an increase in repellency is generally associated to lower water infiltration (Wittenberg and Malkinson, 2009). IR and

SWR (measured by WDPT) are physical parameters that quantify two different processes (infiltration and hydrophobicity). The variability of these processes may be opposite (increases in hydrophobicity can be associated to decreases in water infiltration), but, in some cases, may be the same (Zema et al., 2021a), and this depends on a number of physical and chemical properties of soil (mainly organic matter, texture, plant species). As a matter of fact, Cawson et al. (Cawson et al., 2012) stated that the increase in SWR is not the only reason of infiltration reduction after low-intensity fire. The low content of clay in the experimental soils could have been a reason of the limited effect of SWR affecting the burned soils, since soils with prevailing fine fractions generally exhibit the highest effects of repellency (De Jonge et al., 1999). Moreover, it can not be excluded that prescribed fire only had a punctual effect on soil repellency, without affecting the entire soil surface exposed to rainfall. This is in close accordance with Neary and Leonard (Neary and Leonard, 2021), who stated that water repellency occurs in very localized zones after prescribed fires, and these spots usually do not play significant effects over larger areas. It is also possible that the initially low infiltration rate of oak soils may have gradually increased as water repellency was broken down, as reported by (Robichaud, 2000; Cawson et al., 2012), also when water repellency is moderately strong. Following some of these studies, these increases are generally of low extent, and this makes the effects of SWR on soil hydraulic properties negligible (Doerr et al., 2006; Robichaud, 2000). A possible explanation of IR decrease measured in the pine forest may be the partial sealing of soil due to wax, aromatic oil and resin of trees (released after fire). Instead, water repellency did not affect soils with species (oak and chestnut) that do not release these compounds (Doerr et al., 2000; Cawson et al., 2016).

The irregular distribution of litter and herbaceous covers in chestnut forest let the prescribed fire had a patchy nature, which smoothed its effects on the short-term changes in soil hydraulic properties. This is in accordance with Cawson et al. (Cawson et al., 2012), who state that low-fire severity and burn patchiness are the main reasons of the small impacts that follow prescribed burning. The decoupling of SWR and IR in soils covered by different tree species was noticed also by Zema et al. (Zema et al., 2021a) in similar forest ecosystems covered by pines, junipers and oaks. However, all these effects require more investigations, since it is quite difficult to disentangle the actions of soil coating due to resin release and water repellency due to the OM content (Cawson et al., 2016; Doerr et al., 2003).

The reductions in IR measured immediately after fire in soils of all forest species were of 45-50% compared to the unburned conditions, and worsened the soil’s hydrological response. As a matter of fact, while the runoff generation capacity of the unburned soil was stable, fire increased the runoff generation capacity up to 120% in oak forest (presumably due to the complete litter burning in this broadleaves species, which retain rainfall and shadows soil from erosion). At plot scale, also Vega et al. (Vega et al., 2005) Morales et al. (Morales et al., 2000) and Vieira et al. (Vieira et al., 2015) measured significantly higher runoff volumes compared to unburned soils throughout one year after a prescribed fire.

Conversely, the application of the prescribed fire had a lower effect on peak flows (maximum increase of 11%). In other words, the prescribed fire did not affect the peak values of the runoff hydrograph measured throughout the rainfall simulations, and this means that fire is not able to enhance the flood risk due to increased peak flows. This complies with the statements by Robichaud et al. (Robichaud, 2000), who report constant runoff hydrographs in rainfall simulations after low intensity burns. In contrast, Keesstra et al. (Keesstra et al., 2014) noticed in runoff hydrographs after rainfall simulations a sudden increase in the runoff rate and then a quick decrease until a constant value of runoff. However, since in our study the time to peak of burned soils sometime decreased compared to unburned soils, an increase of the time of concentration in flood events may be possible, and this requires attention.

Soil treatment with fern mulch, whose immediate effects on soil were only mechanical, limited the reduction in water infiltration due to fire in oak and chestnut forests (IR lower by 46% and 35% compared to unburned soils). Consequently, the runoff measured in these soils decreased compared to fire-affected areas (by 10 to 50%), and was higher by 96% (oak) or even lower by 35% (chestnut) compared to the unburned areas. Also peak flows were noticeably lower than in both unburned and burned conditions (-20 to -50%, respectively). The lower hydrological response in the mulched areas may be also due to the soil roughness, which was increased by the vegetal residues distributed over ground. Soil mulching also reduced the time to peak in the short term. In contrast, no effects of mulching on IR, runoff, peak flow and time to peak were detected in pine soils. This may be due to the ineffectiveness of mulching treatment in reducing SWR, which is the main reason of IR decrease in pine soils. Conversely, Bento-Goncalves et al. (Bento-Gonçalves et al., 2012) think that pine needles fell after prescribed fire are able to adequately protect soil as a mulch cover, thus reducing overland flow.

The increase in IR of burned soils (treated or not) over time is expected (Alcañiz et al., 2018; Cawson et al., 2012). This increase may be primarily due to the disappearance of ash, sealing and repellency - with a vanished effect on water infiltrability - and secondarily to the incorporation of litter residues into soils in burned areas, which sums up to the vegetal residues of fern in mulched soils. SWR after prescribed fires is usually of short duration, since the hydrophobic compounds are slightly water soluble and dissolve generally one or two years after fire (Robichaud, 2000). In some Mediterranean pine forests, the effects of SWR even disappeared just after one month from fire (Plaza-Álvarez et al., 2018). Soil enrichment in organic matter is beneficial for the hydraulic properties of soils. Although no measurements in OM were made in this study (that focuses the hydraulic response of soils rather than the physical and chemical properties), the high amounts of litter material in oak forest visually noticed on the survey area compared to the other species may support this explanation. As a matter of fact, the IR increase was more noticeable in oak soils compared to chestnut and pine forests, and the same effect was observed in mulched soils, which showed the highest IR. It is worth to highlight that oak soils treated with fern mulch not only showed the highest IR among all the investigated soil conditions, but their infiltrability was even higher by 110% compared to the unburned soils. In any case, IR of mulched soils was by 15% (for pine soils) to 60% (oak forest) higher than in burned areas, and this proves the effectiveness of mulching in improving the hydrological properties of soils after burning. Despite the increases measured in IR in mulched areas, one year after the prescribed fire the runoff generation capacity of the treated soils was not significantly different compared to the other soil conditions in chestnut or oak forests. As a matter of fact, variations in runoff between -3 and +20%, and in peak flow between -3 and -7% were detected, and the same was noticed for the times to peak, which means that the soil's hydrological response was not influenced by mulching. In pine forest, mulching determined increases in runoff and peak flow by 2 and 4%, respectively, which means that the hydrological response of burned and treated soils (runoff generation, peak of hydrograph and time to concentration) even worsened compared to burned but not treated forests. Although fern has a limited leaf cover (at least in comparison to broadleaf species), the residues can retain water, which afterwards can evaporate, thus limiting surface runoff. However, water drops may flow over the stems and leaves of the residues, with quicker path compared to the overland flow, and therefore a runoff velocity that is higher. To avoid this undesired process, a patchy distribution of fern residues over ground can be suggested, which creates a hydraulic disconnectivity in these quick water flow paths. Unfortunately, the small size of the simulator area does not allow evaluating the impacts of mulching in decreasing hydraulic connectivity, which is a beneficial effects in reducing the travel times of water and sediment flows at the catchment scale (Hooke, 2003). An increased hydraulic connectivity after wildfire may have serious off-site effects, such as increased risk of flooding and pollution of water bodies. The vegetal residues of the mulch cover over ground increases soil roughness and represent obstacles against the water and sediment flows, disrupting the continuity of runoff and sediment pathways.

5. Conclusions

This study has evaluated the changes in soil infiltration of pine, chestnut and oak forests due to prescribed fire and post-fire treatment with fern mulch in field experiments using a rainfall simulator. The fire application reduced water infiltration immediately after fire compared to the unburned conditions, and this reduction has been ascribed to the released ash and SWR. This latter effect was more noticeable in pine forest and less intense in oak soil. These reductions in IR in the window of disturbance after fire increased the runoff generation capacity (conversely stable in unburned soils) in all soils, but had a lower effect on peak flows. Due to the lower reduction in infiltration in oak and chestnut forests due to fern application after fire, soil mulching limited runoff rates and peak flows compared to the burned soils. In contrast, the treatment was less effective in pine forest.

One year after fire, IR increased in burned soils (treated or not) over time, since ash cover, sealing and SWR disappeared and the residues of litter and mulch were incorporated in the soil. Mulched soils in oak forest showed the higher IR increase compared to chestnut and pine forests. However, this increase was not able to determine significant reductions in runoff and peak flow in treated soils compared to burned and unburned conditions. Therefore, the effects of mulching have disappeared after some months from fire.

Overall, this investigation has confirmed both the working hypothesis that prescribed fire may worsen the soil's hydrological response to burning, but mulching is particularly effective in reducing runoff volumes and peak flow immediately after fire in broadleaves species (but not in conifers). Its hydrological effects decrease over time until being not significant or even negligible some months after fire, and, in conifers, even detrimental. This study has been carried out at local scale and under simulated rainfall, and has not evaluated the erosion response of burned soils (treated or not). Further research is needed with field scale extension to plots or hillslopes (perhaps incorporating the time variability of natural rainfall and comparing soil loss in the different conditions). This extension is welcome in order to consider the patchy nature of prescribed fire and the effects of mulching on hydraulic connectivity, and to give indications about the most suitable distribution of mulch material over soil (e.g., homogenous cover or patchy distribution).

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Appendix A

Figure A1. Prescribed fire operations (a) and mulching application (b) in the experimental site (Samo, Calabria, Southern Italy).


a


b

Figure A2. Experimental areas unburned or subjected to prescribed fire and mulching with fern (Samo, Calabria, Southern Italy). (a) oak; (b) chestnut; (c) pine.

Figure A3. Measurements of soil hydraulic conductivity (using a portable rainfall simulator) and fern mulch application into the rainfall simulator at the experimental site (Samo, Calabria, Southern Italy).

Figure A4. Measurements of soil water repellency (by the WDPT method) in the experimental site (Samo, Calabria, Southern Italy).

CHAPTER 2

Prescribed fire and soil mulching with fern in Mediterranean forests: Effects on surface runoff and erosion

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Abstract

Prescribed burning is increasingly used to reduce the wildfire risk, and the need to limit runoff and erosion suggest treating burned soils with mulching. To this aim, fern residues may be more advisable compared to the commonly used straw, since this material is directly available in forests and has lower drawbacks. However, the post-fire hydrological effects of both prescribed fire and soil mulching are contrasting in literature, and fern has not previously experimented as mulching material in Mediterranean forests. To fill these gaps, this study has evaluated the soil hydrological response in small plots installed in three Mediterranean forests (pine, chestnut and oak) after a prescribed fire and mulching treatment with fern. Compared to the unburned soils, runoff and erosion significantly increased immediately after fire (by 150% to 375% for the runoff coefficients, and by 100% to 800% for the soil losses). However, these increases are much lower compared to the highest values reported by some studies. The negative impacts on the hydrological response in burned soils were limited to three-four months. Subsequently, the pre-fire runoff and erosion rates of the burned soils were practically restored, and the hydrological changes were not significant compared to the unburned soils. In the short term after prescribed fire application, soil mulching with fern residues was effective to limit the increase in the

hydrological response of the burned and not treated soils, since runoff coefficients and erosion were reduced by 25-30% in oak soils and 70-80% in chestnut and pine forests. The changes surveyed in soil hydrology were associated with variations in the infiltration rates and water repellency immediately after fire, previously detected in the same experimental site. The restoration of water infiltration rates and disappearance of soil repellency gained importance over time, and the incorporation of mulch residues become beneficial in driving the short-term runoff and erosion response of the burned soils.

Keywords: ecological engineering techniques; post-fire management; hydrological response; pine; chestnut; oak.

1. Introduction

Fire, a key ecological factor in the earth system (Francos and Úbeda, 2021), impacts on many components of ecosystems (soil, air, water, plants and fauna, e.g. (DeBano et al., 1998; Lucas-Borja et al., 2019b; Kozlowski, 2012) as well as on ecosystem services, society and economy (Nadal-Romero et al., 2018; Pereira et al., 2018a). These effects depend on several factors, such as fire history, intensity and severity, fuel quantity, properties and topography of soils, vegetation species, density and cover, weather patterns, etc. (Zavala et al., 2014; Pereira et al., 2018b; Francos et al., 2018; Zema, 2021).

With specific regard to the environmental impacts, wildfire removes vegetation and reduces its capacity to recover, and determines long-lasting changes in soil properties (Neary et al., 1999; Certini, 2005; Shakesby, 2011). Vegetation removal (which leaves soil bare) and soil changes (with increase in water repellency, destruction of aggregates and reduction in water infiltration) due to wildfire increase the surface runoff and erosion rates. Moreover, transport of nutrients and contaminants downstream of burned forests is enhanced (Neary et al., 1999; Certini, 2005; Shakesby and Doerr, 2006; Cawson et al., 2012; Vieira et al., 2015; Zema, 2021). Moreover, the runoff and erosion rates come back to the pre-fire values after five to ten years (Inbar et al., 1998). The increase in flooding and erosion risks after fire is an essential problem for land owners and catchment managers (Prats et al., 2015).

In order to limit the negative impacts of high-severity fires, preventing strategies have been adopted since long time (Ferreira et al., 2015). Among these strategies, prescribed fire - the planned use of low-intensity fire to achieve very different goals given certain weather, fuel and topographic conditions (Fernandes et al., 2013) - is considered as a primary and integrated option to remove or reduce the fuel that can generate a high-intensity fire (Vega et al., 2005; Alcañiz et al., 2018). Prescribed fire, which has low-severity and burn patchiness (Cawson et al., 2012; Pereira et al., 2021), avoids high temperature in soil and tree crown burning, which are the most adverse effects of wildfire on soil and plants. In addition, prescribed fire supports regeneration of some plant species (Francos and Úbeda, 2021; Scharenbroch et al., 2012; Williams et al., 2012). Increases in runoff and erosion after prescribed fires are lower compared to wildfires, but these risks are still present (Morris et al., 2013; Shakesby et al., 2015). Runoff and erosion increases have been observed after prescribed fires in different ecosystems, such as heathlands, shrublands and gorse (Vega et al., 2005). In the Mediterranean forests, these increases may be even more intense compared to other rainstorms (Fortugno et al., 2017) and the soils are generally shallow with low aggregate stability, and organic matter and nutrient contents (Cantón et al., 2011). Due to the combination of these climate and soil characteristics, the Mediterranean forests may be more exposes to excessive runoff and soil erosion rates compared to other ecosystems (Zema et al., 2020a, 2020b). Therefore, there is a need for an improved knowledge about soil hydrology in Mediterranean fire-prone forests, also considering that both wildfires and rainstorms are thought to become more frequent and intense according to the forecasted climate scenarios (Badia and Marti, 2008). However, despite an ample literature about the impacts of fire on soil hydrology, the studies about the hydrological effects of prescribed fire are not exhaustive and often contrasting (Cawson et al., 2012; Shakesby et al., 2015). According to (González-Pelayo et al., 2010) and (Vega et al., 2005), increases in runoff and erosion by one and two orders of magnitude, respectively, may be observed compared to unburned areas (Cawson et al., 2013). In contrast, (Coelho et al., 2004) and (de Dios Benavides-Solorio and MacDonald, 2005) reported minimal erosion after prescribed fire (Morris et al., 2013). (Keesstra et al., 2014) reported even lower erosion in areas burned with prescribed fire compared to unburned forests, despite comparable runoff.

In order to reduce soil's susceptibility to runoff and erosion to fire, several treatments have been proposed and their effectiveness has been verified in many environmental contexts (Lucas-Borja, 2021; Zema, 2021). Among the ecological engineering techniques, which use vegetative residues for soil conservation, mulching is one of the most common post-fire management options (Lucas-Borja et al., 2019a; Prosdocimi et al., 2016). The objective of mulching is protecting soil with ground cover and improving soil quality, if used properly and at the correct time (Prosdocimi et al., 2016; Zituni et al., 2019). However, post-fire mulching can also have negative effects. In some cases, mulching reduces the soil hydraulic conductivity under unsaturated conditions compared to untreated soils, particularly in the drier season (Lucas-Borja et al., 2018). Mulching material is selected based on its availability, resistance to degradation, weed spreading risk and other factors (Parhizkar et al., 2021; Prats et al., 2015). Straw is often used as mulch cover in fire-affected areas (Bontrager et al., 2019; Keizer et al., 2018), but its residues can be displaced by wind in some areas, leaving slopes bare, or accumulated in thick layer in other areas, with possible reductions in post-fire vegetation emergence (Robichaud et al., 2020). Moreover, agricultural straw may contain seeds, chemicals and parasites, which can be the sources of non-native vegetation and plant diseases. Forest residues (e.g. wood strands, chips or shreds) or dead plants may replace straw, because these substrates do not carry non-native seeds or chemical residues, and are more resistant to wind displacement (Robichaud et al., 2020). In Mediterranean forest floor, fern - *Pteridium aquilinum* (L.) Kuhn - is widely available (which avoids the transport need from other locations) and its lignin content is lower compared other agro-forest residues (which allows a fast degradation into soil). Therefore, its use as mulching material in fire-affected areas is preferable to straw. However, at the best authors' knowledge, no evaluations about the use of fern to protect burned soil from runoff and erosion impacts are available in literature. Therefore, the effectiveness of fern mulching to restore the hydrological properties of soils should be assessed, and particularly in the short-term after fire, when the soil is left bare and the soil changes (e.g., reduced infiltration, soil water repellency and ash cover) can be significant compared to the unaffected areas (Cawson et al., 2012; Francos and Úbeda, 2021; Klimas et al., 2020; Wittenberg and Pereira, 2021). A previous study, carried out in the same environment using a rainfall simulator, showed that soil mulching with fern did not increase water infiltration and did not alter soil water repellency of burned soils in the measurement point immediately after a prescribed fire. One year after the soil treatment, the soil hydraulic conductivity noticeably increased and repellency completely disappeared (Carrà et al., 2021).

To fill the research gaps and extend the previous investigation to the plot scale, this study has evaluated the hydrological response of soils in three forest stands of Calabria (Southern Italy) after a prescribed fire and mulching treatment with fern in comparison to undisturbed soils. More specifically, surface runoff volumes and soil losses were measured after natural precipitation throughout one year after fire together with soil covers in pine, oak and chestnut forests. The specific research questions are the following: (i) how much does the prescribed fire affects runoff and erosion rates on the short term after its application? (ii) how long is the "window of disturbance" (Prosser and Williams, 1998) of soil hydrology due to fire? (iii) are the fern residues effective as mulching cover to reduce runoff and erosion after fire?

The experimental replies to these questions study may be of help to promote the use of both prescribed fire against the wildfire risks and soil mulching with fern as ecological engineering technique for forest soil conservation.

2. Material and methods

2.1. Study area

The study was carried out in three forest sites (municipality of Samo, Calabria, Southern Italy) between 600 and 900 m above sea level (Figure 1 and Table 1), of which: (i) the first area ("Calamacia") was a pine (*Pinus pinaster* Aiton) stand reforested in 1984; (ii) the second site ("Rungia") is a natural oak stand (*Quercus frainetto* Ten.); and (iii) the third zone ("Orgaro") was a chestnut stand (Castanea sativa Mill., about 29-year old). No management actions were carried out in the three forest stands. Table 1 reports the main characteristics (coordinates, altitude and soil slope) of the experimental site, while the main characteristics of tree and shrub species are depicted in Table 2.

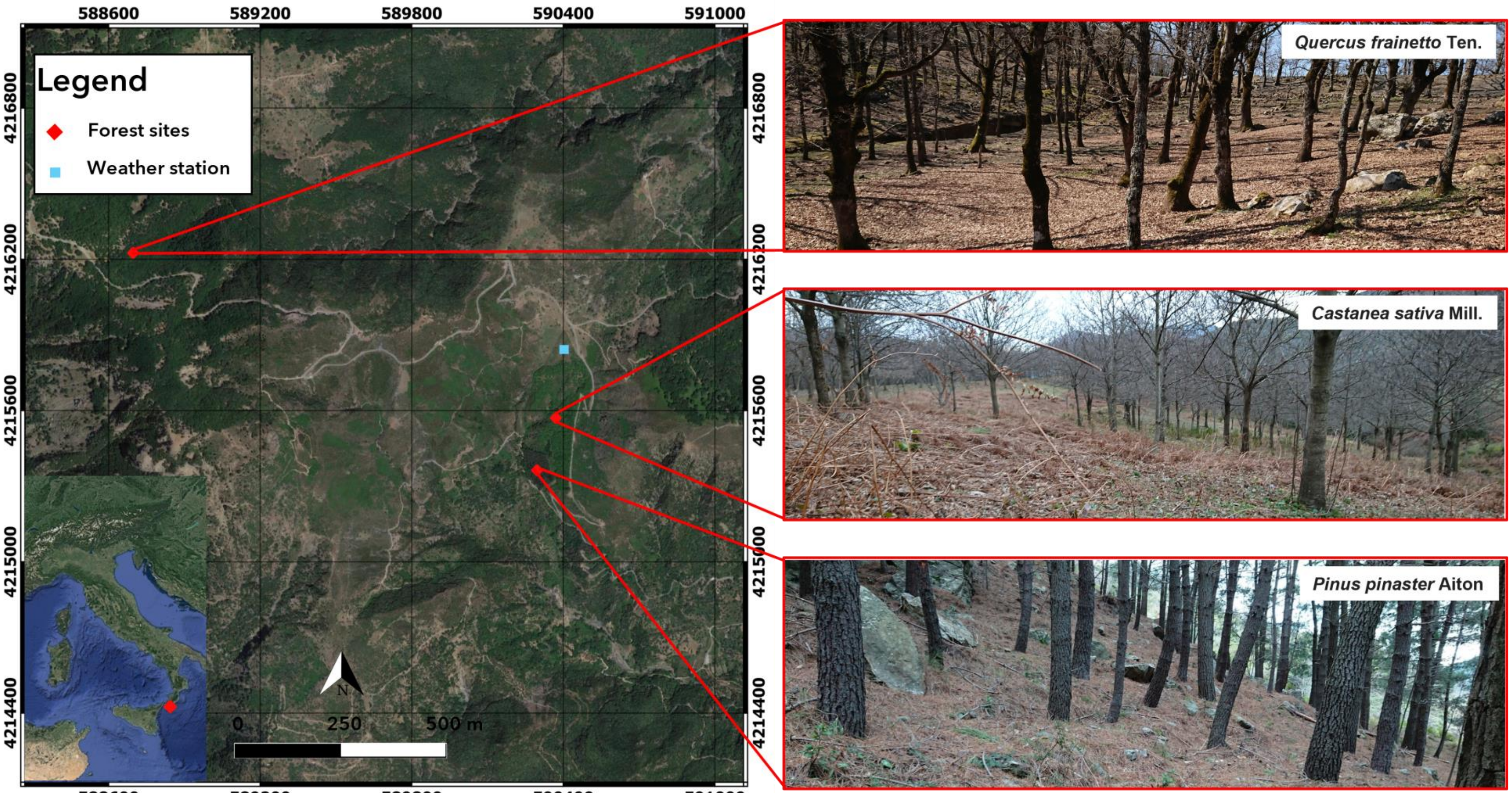


Figure 1 - Location of the experimental site (Samo, Calabria, Southern Italy).

Table 1 - Main characteristics of the experimental site (Samo, Calabria, Southern Italy).

Site	**Main forest species**	**U.T.M. coordinates**	**Aspect**	**Altitude (m a.s.l.)**	**Slope (%)**
Calamacia	Pine	590293 E 4215327 N	South-West	650-700	20.0 ± 0.82
Rungia	Oak	588635 E 4216172 N	North-East	900-950	19.1 ± 1.65
Orgaro	Chestnut	590389 E 4215530 N	West	700-750	20.3 ± 0.96

Table 2 - Main characteristics of the three forest stands in the experimental site (Samo, Calabria, Southern Italy).

Site	**Tree**					**Litterfall layer depth (cm)**	**Shrub species**
	species	**density (n/ha)**	**diameter at breast height (cm)**	**height (m)**	**basal area (m^2/ha)**		
Calamacia	pine (*Pinus pinaster* Aiton)	950 ± 86.4	28.3 ± 9.4	20.5 ± 1.4	67.9 ± 6.5	11.7 ± 4.6	*Quercus ilex* L., *Rubus ulmifolius* S., *Bellis perennis* L.
Rungia	oak (*Quercus frainetto* Ten.)	225 ± 44.7	40.7 ± 8.9	18.2 ± 1.9	31.1 ± 3.6	12.2 ± 3.9	*Cyclamen hederifolium, Bellis perennis* L.
Orgaro	chestnut (*Castanea sativa* Mill.)	725 ± 89.1	20.2 ± 5.6	9.6 ± 1.2	24.3 ± 4.4	6.1 ± 4	*Rubus ulmifolius* S., *Pteridium aquilinum L.*, *Bellis perennis* L.

The climate of the area is typical of the semi-arid environment ("Csa" class, "Hot-summer Mediterranean" climate, according to Koppen (Kottek et al., 2006). Winters are mild and rainy, while summers are warm and dry. The mean annual precipitation and temperature are 1102.3 mm and 17.4 °C, respectively. The minimum temperature is - 4.3 °C, while the maximum is 43.1 °C (weather station of Sant'Agata del Bianco (RC), UTM coordinates 4217548 N, 595159 E, period 2000-2020).

Table 3 shows the main characteristics of these soils for each experimental condition (unburned, burned and not treated, and burned and mulched soils). To summarize, all soils were loamy, except the unburned area of the pine forest, which was sandy loam.

Table 3 - Main characteristics of the soils in the experimental site measured immediately after the prescribed fire and before the mulching treatment (Samo, Calabria, Southern Italy).

Site	**Main forest species**	**Soil condition**	**Texture**			**Type**
			silt (%)	clay (%)	sand (%)	
Calamacia	pine	unburned	10.0 ± 1.01	9.0 ± 0.00	81.0 ± 0.99	sandy loam
		burned	6.3 ± 3.06	8.7 ± 0.58	85.0 ± 3.61	loamy sand
Rungia	oak	unburned	12.7 ± 1.53	9.7 ± 0.58	77.7 ± 1.15	
		burned	10.3 ± 2.25	8.7 ± 0.58	81.0 ± 2.02	
Orgaro	chestnut	unburned	12.3 ± 2.31	8.0 ± 1.73	79.7 ± 0.58	
		burned	11.3 ± 1.53	8.7 ± 0.58	80.2 ± 1.04	

2.2. Prescribed fire operations and mulching application

The prescribed fire was carried out in early June 2019 with the support of the Environmental Regional Agency (Calabria Verde) and the surveillance of the National Corp of Firefighters (Figure 2a).

The main conditions during fire application to the experimental site (temperatures of fire flame, air and soil) are reported in Table 4. These variables were measured by a thermocouple connected to a datalogger at a soil depth of 2 cm. Wind was practically absent and air humidity between 50 and 60%.

Table 4 – Main conditions during prescribed fire application to the experimental site (Samo, Calabria, Southern Italy).

Site	**Main forest species**	**Temperature**					
		fire flame		**air**		**soil**	
		mean	max	mean	max	mean	max
Calamacia	pine	88.3	712	25.7	102	21.9	22.7
Rungia	oak	98	720	43.0	180	21.0	26.9
Orgaro	chestnut	75	645	29.1	139	24.7	28.8

In the burned area, one day after fire, some plots (see section 2.3) were covered with small pieces (maximum length of 5 cm) of fern. The plants were cut from an adjacent zone in the same forest and the fresh residues were spread on the ground. The applied dose was 500 g/m^2 of fresh weight, which is equivalent to 200 g/m^2 of dry matter (commonly used as straw mulching after fire (Lucas-Borja et al., 2018; Vega et al., 2014) (Figure 2b).

(a)

(b)

Figure 2 – Prescribed fire operations (a) and fern mulch applied to three plots of oak (b) in the experimental site (Samo, Calabria, Southern Italy).

2.3. Experimental design

One of the most useful tools to study the fire effects is applying experimental fires and measuring their effects on soil hydrology in plots; this allows the control and evaluation of the fire and soil conditions before, during and after the experiment (González-Pelayo et al., 2010). The current study has adopted the suggested approach and, in each experimental site, nine small plots (three series, each one with three replicated plots) were delimited on forest hillslopes with the same gradient (Table 1). The plots were at a reciprocal distance between 1.5 and 20 m. Three plots were setup in the unburned soils (considered as "control"), while six plots were in the burned area. In the latter soils, three plots were subjected to mulching with fern. Overall, the experimental design consisted of three forest stands (pine, oak and chestnut) × three soil conditions (unburned, burned and not treated, and burned and mulched) × three replicated plots, for a total of 27 plots (Figure 3).

2.4. Plot construction

Immediately after fire, the plots (each one being 3-m long and 1-m wide and covering an area of 3 m^2) were hydraulically isolated in each forest area (unburned, burned and not treated, and burned and mulched soil), using 0.3-m high metallic sheets inserted up to 0.2 m below the ground surface, in order to prevent the flow of surface water (Figure 2b).

Downstream of each plot, a transverse channel was installed, to intercept the water and solid material flows. These flows were collected through a pipe into a 100-litre tank.

Figure 3 – Scheme and plot layout of the experimental design used for the hydrological monitoring after prescribed fire and soil mulching using fern (Samo, Calabria, Southern Italy)

2.5. Monitoring of the hydrological variables

Hydrological measures started immediately after site installation (mid-June 2019) and were carried out throughout 15 months (until mid-September 2020).

A weather station with a tipping bucket rain gauge (measuring sub-hourly data) was installed at a maximum distance of 1 km from the experimental sites, to measure precipitation height, storm duration, and rainfall intensity. Mean rainfall intensity was the total rainfall divided by the storm duration. Moreover, an additional rain gauge (measuring only the rainfall height) was installed in each forest site, in order to estimate the rainfall intercepted by the tree canopy and check the spatial variability of the rainfall measured by the main weather station.

The surface runoff and sediment concentration produced by the monitored rainfalls were measured following the procedures suggested by Lucas-Borja et al. (2019b) and Bombino et al. (2021). To summarize, runoff samples were collected by mixing the water in the tank and collecting three separate samples, totalling about 0.5 litres. The samples were brought to the laboratory, where they were oven-dried at 105 °C for 24 hours. After drying, the sediments were weighted and referred to the sample volume, in order to calculate the sediment concentration. The soil loss produced by the rainfall-runoff event was estimated by the product of the runoff volume by the sediment concentration. The runoff coefficients were also calculated as the ratio of runoff to rainfall.

2.6. Statistical analyses

One-way ANOVA with repeated measures (at each rainfall-runoff event) was applied to the runoff volume and soil loss (response variables) separately for the three forest species, assuming as factor the soil condition (unburned, burned and not treated, and burned and mulched). The pairwise comparison by Tukey's test (at $p < 0.05$) was also used to evaluate the statistical significance of the differences in the response variables. In order to satisfy the assumptions of the statistical tests (equality of variance and normal distribution), the data were subjected to normality test or were square root-transformed whenever necessary. All the statistical tests were carried out with the XLSTAT software.

3. Results

3.1. Rainfall characterization

Throughout the monitoring period 516 rainfall events with a total depth of 1120 mm were recorded at the rain gauging station. Of these events, only seven were classified as erosive events (that is, with depth over 13 mm), according to Wischmeier and Smith (1978). The height of these events was in the range 22.4 (14 July 2020) - 156 (11 March 2020) mm, while their duration varied between 7 (14 July 2020) and 41 (11 November 2019) hours. The latter was the most intense event (mean intensity of 26.2 mm/h), while the maximum intensity was recorded for the event of 23 November 2019 (Table 5). One event (dated 24 July 2020) produced runoff and erosion only in the chestnut plots. The spatial variability of the precipitation among the three forest sites was very low (< 5%) for all the monitored events. The net rainfall (due to the interception) was between 4-10% (pine and chestnut forests) and 6-12% (oak site) of the total precipitation (Table 5).

Table 5 - Main hydrological variables of rainfall events monitored in the experimental site (Samo, Calabria, Southern Italy).

Date	**Height (mm)**	**Net height (mm)***			**Duration (h)**	**Intensity (mm/h)**	
		pine	**oak**	**chestnut**		**max**	**mean**
15 Jul 2019	65	61.8	59.8	60.5	36	22.2	1.99
9 Oct 2019	49.9	45.4	43.9	44.9	26	14.6	1.85
11 Nov 2019	142.8	135.7	132.8	132.8	41	26.2	3.49
23 Nov 2019	87.1	82.7	81.0	81.9	19	24.7	4.58
5 Dec 2019	147.2	141.3	138.4	139.8	30	19	4.90
24 Mar 2020	155.9	149.7	146.5	149.7	32	13.8	2.86
14 Jul 2020	22.4	20.6	19.7	20.4	7	12.8	2.58

Note: recorded at the rain gauge station under tree canopy in each forest.

3.2. Runoff

In the unburned plots, the maximum runoff (from 13.1 ± 11.2 mm in the pine forest to 18.1 ± 12.9 mm in chestnut forest) was measured always after the rainfall with the highest height (156 mm, 24 March 2020). In oak forest, a high runoff (16.4 ± 3.11 mm) was also collected after the event with the highest mean intensity (26.2 mm/h on 11 November 2019, 143 mm in 41 hours). In one event (9 October 2019, 50 mm), having the lowest height among the erosive rainfalls, no runoff was collected in the unburned chestnut and oak forests (Table 6).

Conversely, the highest runoff volume in the burned plots was always collected after the first of the monitored events. More specifically, on 15 July 2019 (one month after the prescribed fire), the runoff was 22.3 ± 1.35 mm in the chestnut forest, 22.3 ± 4.21 mm in the pine stand, and 31.3 ± 2.29 in oak plots. The first rainfall event produced the highest runoff also in the burned and mulched plots of chestnut and oak forests (6.61 ± 1.16 mm and 23 ± 3.69 mm, respectively), while in the pine plots the maximum runoff (10.4 ± 0.80 mm) was measured after the second event (11 November 2019, 143 mm of rainfall) (Table 6).

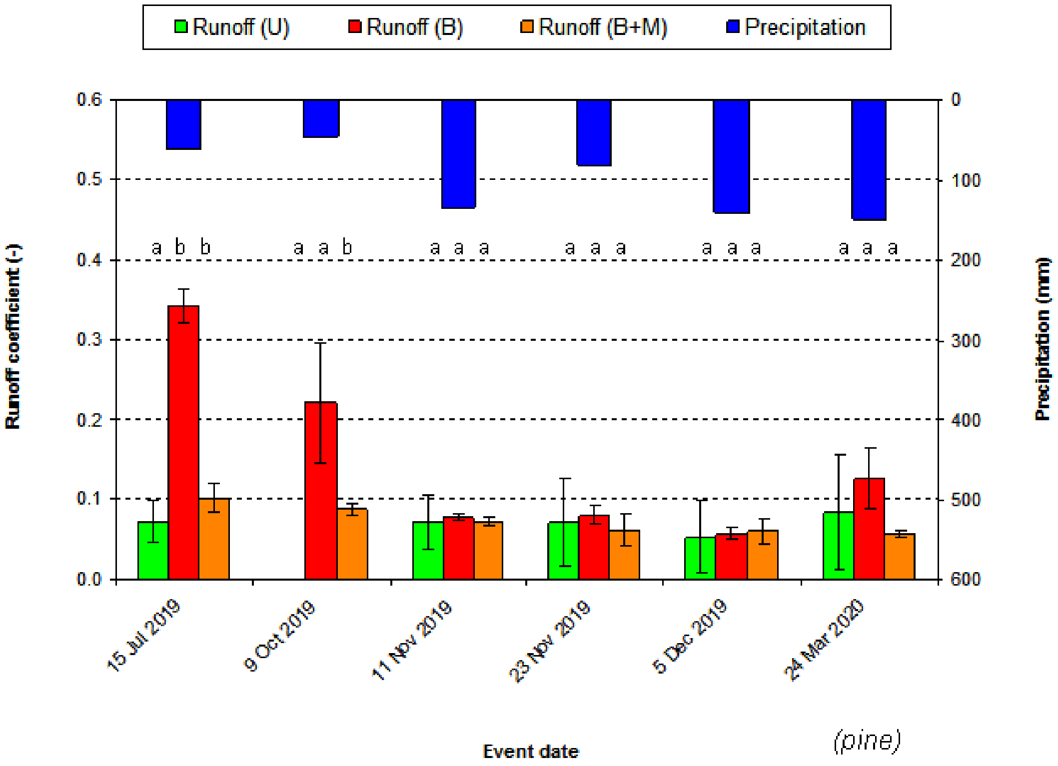
As mentioned above, the hydrological response of the three soil conditions was interpreted in terms of runoff coefficient, which standardizes runoff to the unit rainfall. In unburned plots, this coefficient showed a low variability (0-0.08, pine, 0.07-0.17, chestnut, and 0.00-0.19, oak forest) (Figure 4).

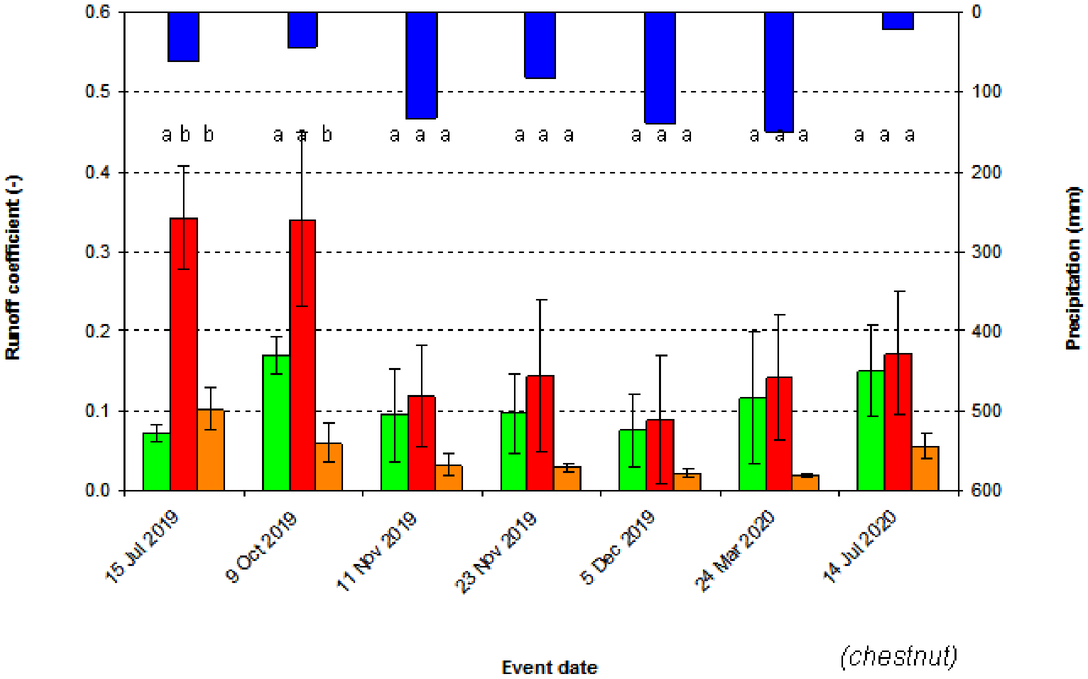
In contrast, immediately after the prescribed fire, the runoff coefficient suddenly increased in all forest plots (up to 0.34 ± 0.02, pine, 0.34 ± 0.06, chestnut, and even 0.48 ± 0.04, oak forest). In pine and chestnut plots, a high runoff coefficient was also noticed also after the second storm (0.22 ± 0.08 and 0.34 ± 0.11, respectively), while, in the oak forest, this coefficient decreased to values (0.20 ± 0.06) that were very similar to the unburned soil and remained in the range 0.14 ± 0.02 to 0.18 ± 0.08. In the pine and chestnut plots, the runoff coefficients decreased over time, and, after the third precipitation event, returned to very low values (between 0.06 ± 0.01 and 0.13 ± 0.04 for pine, as well as 0.09 ± 0.08 and 0.17 ± 0.08 for chestnut), close to the undisturbed soils (Figure 4).

Mulching with fern was effective in decreasing the runoff generation capacity immediately after the prescribed fire particularly in pine and chestnut plots. In these forests, the runoff coefficients after the first rainfall event were 0.10 ± 0.02 and 0.10 ± 0.03, respectively, which means that the runoff volume collected in the plot tanks was less than one third compared to the burned soils. In contrast, in oak plots, the runoff coefficient was 0.35 ± 0.06, about 27% less than in the burned plots. Over time, in burned and mulched plots of pine and oak forests the runoff coefficient restored to the values of the unburned soils (0.06 ± 0.01 to 0.09 ± 0.01, and 0.07 ± 0.06 to 0.12 ± 0.01, respectively, while, in the chestnut plots, these coefficients decreased to values significantly lower (0.02 ± 0.01 to 0.06 ± 0.02) compared to the control soils (Figure 4).

Table 6 - Runoff volume and its sediment concentration measured in plots after prescribed fire and soil mulching using fern (Samo, Calabria, Southern Italy).

Event date	**Runoff volume (mm)**						**Sediment concentration (g/l)**					
	Unburned soil		*Burned soil*		*Burned and mulched soil*		*Unburned soil*		*Burned soil*		*Burned and mulched soil*	
	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.
Pine												
15 Jul 2019	4.69	1.74	22.31	1.35	6.63	1.16	1.20	0.34	2.35	0.36	1.64	0.37
9 Oct 2019	0.00	0.00	11.03	3.74	4.37	0.33	0.00	0.00	1.47	0.14	0.26	0.02
11 Nov 2019	10.22	4.80	11.12	0.53	10.35	0.80	0.47	0.45	0.21	0.13	0.18	0.02
23 Nov 2019	6.18	4.78	7.01	1.02	5.41	1.73	0.11	0.12	0.19	0.10	0.03	0.02
5 Dec 2019	7.85	6.59	8.44	1.02	8.91	2.40	0.00	0.00	0.07	0.05	0.00	0.00
24 Mar 2020	13.06	11.16	19.77	5.98	8.81	0.60	0.02	0.02	0.03	0.02	0.03	0.01
Chestnut												
15 Jul 2019	4.69	0.68	22.30	4.21	6.61	1.69	1.65	0.54	2.32	0.18	2.17	0.24
9 Oct 2019	8.44	1.16	16.98	5.44	3.00	1.23	1.86	0.59	2.08	1.14	0.58	0.26
11 Nov 2019	13.45	8.25	16.93	9.04	4.64	1.93	0.27	0.05	0.43	0.13	0.19	0.04
23 Nov 2019	8.39	4.32	12.49	8.29	2.51	0.41	0.30	0.03	0.55	0.11	0.10	0.02
5 Dec 2019	11.10	6.60	13.03	11.86	3.24	0.71	0.10	0.00	0.07	0.06	0.00	0.00
24 Mar 2020	18.13	12.92	22.11	12.07	2.98	0.29	0.16	0.11	0.25	0.12	0.13	0.05
14 Jul 2020	3.37	1.28	3.85	1.72	1.25	0.37	0.00	0.00	0.00	0.00	0.00	0.00
Oak												
15 Jul 2019	12.55	2.90	31.34	2.29	22.98	3.69	1.58	0.15	1.48	0.37	1.51	0.09
9 Oct 2019	0.00	0.00	10.00	3.13	3.27	2.83	0.00	0.00	0.90	0.26	0.51	0.89
11 Nov 2019	16.35	3.11	20.64	3.05	17.81	1.68	0.31	0.06	0.34	0.13	0.32	0.06
23 Nov 2019	7.70	2.80	15.86	6.84	9.66	0.17	0.59	0.22	1.12	0.96	0.89	0.29
5 Dec 2019	11.01	1.30	21.11	10.64	16.52	0.86	0.40	0.17	0.48	0.20	0.40	0.09
24 Mar 2020	16.36	6.01	22.11	5.32	18.78	1.20	0.32	0.06	0.37	0.05	0.29	0.03


Runoff (U)
Runoff (B)
Runoff (B+M)
Precipitation
Runoff coefficient (-)
Precipitation (mm)
15 Jul 2019
9 Oct 2019
11 Nov 2019
23 Nov 2019
5 Dec 2019
24 Mar 2020
Event date
(pine)


Runoff coefficient (-)
Precipitation (mm)
15 Jul 2019
9 Oct 2019
11 Nov 2019
23 Nov 2019
5 Dec 2019
24 Mar 2020
14 Jul 2020
Event date
(chestnut)

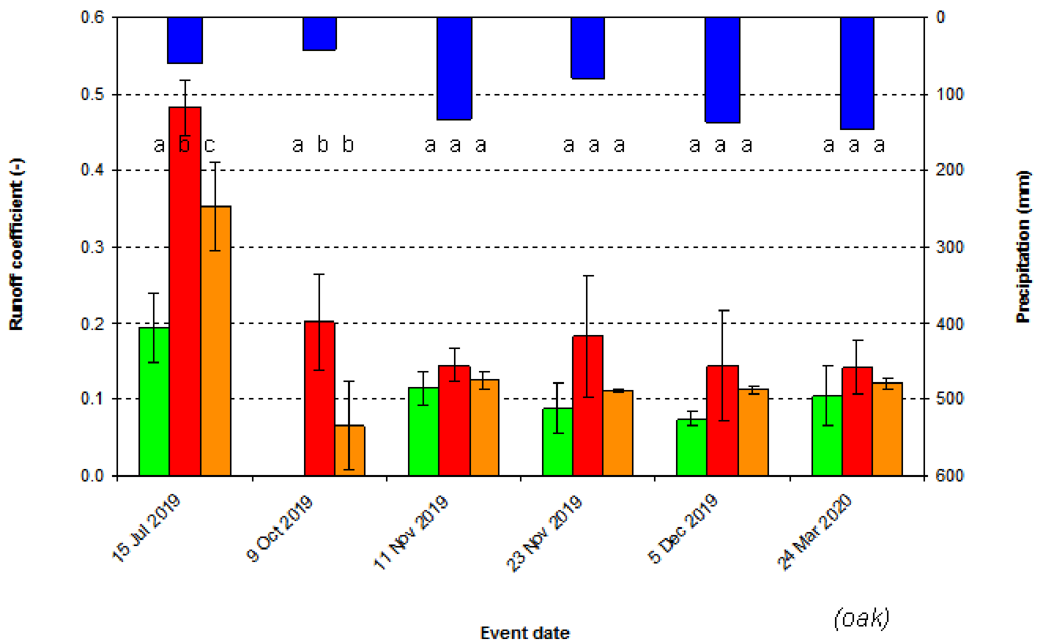


Figure 4 - Precipitation and runoff coefficients measured in plots after prescribed fire and soil mulching using fern (Samo, Calabria, Southern Italy).

Notes: U = unburned soils; B = burned and not treated soils; B + M = burned and mulched soils. Different letters indicate statistically significant differences after Tukey's test ($p < 0.05$).

3.3. Erosion

Sediment concentration in the collected runoff was always the highest for the first event after the prescribed fire (15 Jul 2019) with one exception. More specifically, for this event, the burned and not treated soils showed the maximum sediment concentration (from 1.48 ± 0.37 g/L., oak, to 2.35 ± 0.36 g/L, pine). In pine and chestnut plots, - the latter giving 2.32 ± 0.18 g/L of sediment concentration, - this variable was about 2-fold the sediment concentration measured in the unburned soils (1.20 ± 0.34 g/L and 1.65 ± 0.54 g/L, respectively). However, in the chestnut forest, this event did not give the maximum sediment concentration, which was instead measured after the second event after the fire (1.86 ± 0.59 g/L, 9 Oct 2019) (Table 6).

Compared to the burned soils, mulch application allowed a noticeable decrease in sediment concentration for this first event only in pine forest (1.64 ± 0.37 g/L), while this variable was only slightly lower in chestnut (2.17 ± 0.24 g/L) and even higher in oak (1.51 ± 0.09 g/L) plots. In general, the relatively high erosion surveyed in the unburned soil was due to the lack of precipitation in the 2-3 months before and immediately after fire, which made the soil drying and therefore exposed to higher rainfall erosivity (Table 6).

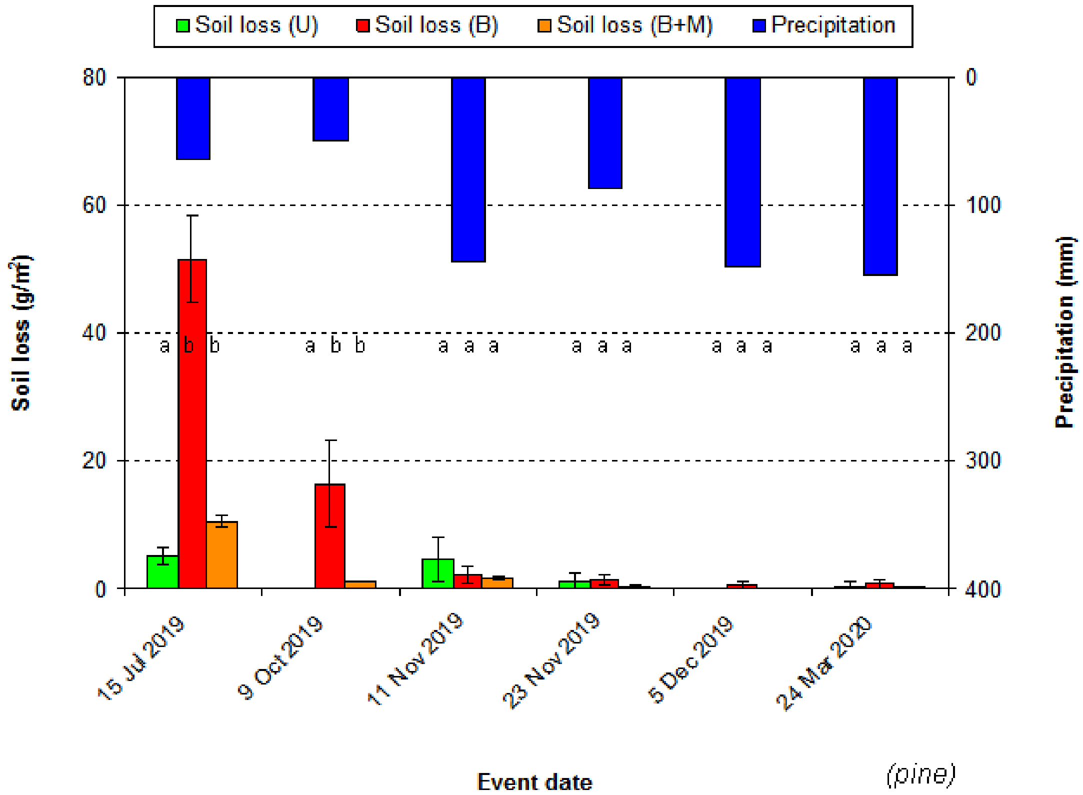
Throughout the monitoring period after the first event, the sediment concentration was still noticeable for the second event in burned plots (treated or not) of all forest species and also in unburned plots of chestnut, with values over 0.90 ± 0.26 g/L (the latter measured in burned plots of oak without treatment). After 4-5 months, this concentration decreased to very low values in all soil conditions. Mulch cover was able to decrease the sediment concentration to values that were noticeably lower compared to the burned soils and in many cases also to unburned soils (Table 6).

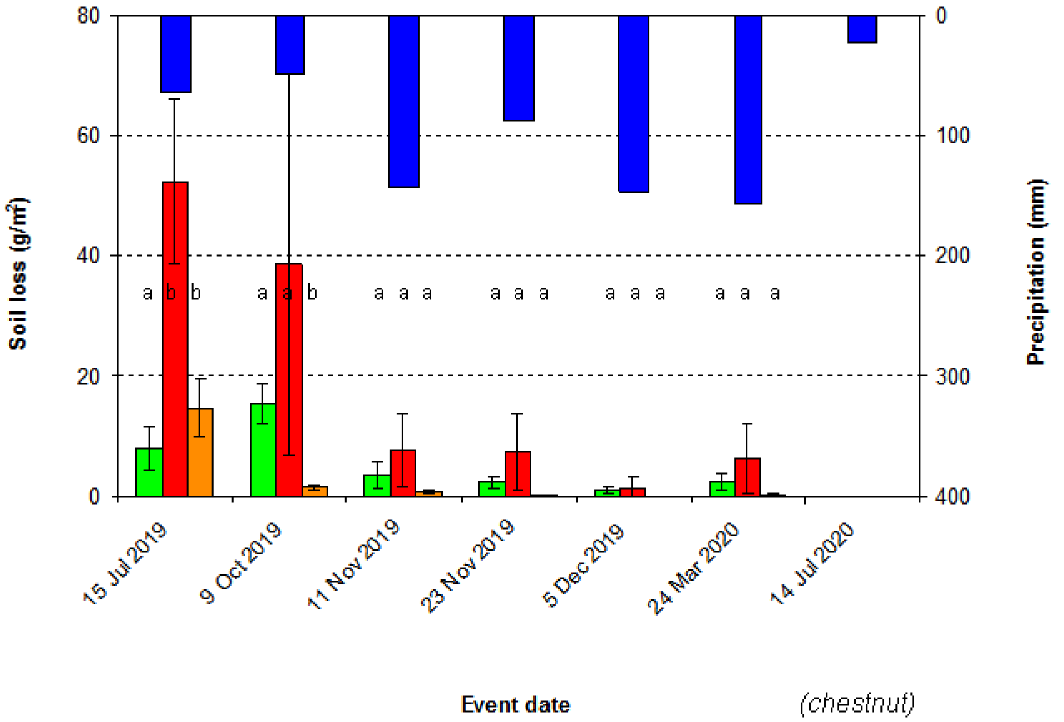
While no temporal trend in this decrease was noticed in unburned soils, a monotonic lowering was detected in burned plots of pine (with or without mulching). Moreover, for some events with relatively low precipitation sediment was collected in burned soils (e.g., 9 Oct 2019, 50 mm and 14.6 mm/h, for pine and oak forests), but not in the paired unburned plots. Another event with high precipitation height (5 Dec 2019, 147 mm and 19.0 mm/h) gave runoff, but no erosion, due to the fact that this precipitation was snow (which smelt immediately producing surface water), which has a negligible detachment capacity (Table 6).

As expected, the soil loss, estimated as the product of the runoff volume collected in the tanks by the corresponding sediment concentration, was of low amount in the unburned plots (up to 5.31 ± 1.40 g/m^2 in pine forest, and 7.37 ± 2.72 g/m^2 in oak, both estimated in the first event of 15 Jul 2019, 65 mm and 22.2 mm/h, and 15.34 ± 3.21 g/m^2 in chestnut, the latter after the second event of 9 Oct 2019, 50 mm and 14.6 mm/h) (Figure 5). For these two events, erosion increased very much in burned soils of all forests, and mainly in pine and chestnut soils. In these plots, maximum values of soil loss equal to 51.61 ± 6.92 and 52.26 ± 13.67 g/m^2 (first event of 15 Jul 2019, 65 mm and 22.2 mm/h) were detected, while in oak soils erosion was noticeably lower, 15.12 ± 2.87 g/m^2 (although higher compared to the unburned plots). However, mulching was effective to reduce these soil losses, whose maximum values were equal to 10.62 ± 0.99 g/m^2 (pine forest), 14.58 ± 4.80 g/m^2 (chestnut), and 11.53 ± 2.23 g/m^2 (oak), always after the first events (15 Jul 2019) (Figure 5).

After the first two events, soil loss showed a low variability in unburned soils (in the range 0 to 4.63 ± 3.57 g/m^2 for pine, 3.55 ± 2.13 g/m^2 for chestnut, both on 11 Nov 2019, 143 mm and 26.2 mm/h, and 5.44 ± 2.79 g/m^2, for oak, estimated after the last event of 14 Jul 2020, 22.4 mm and 12.8 mm/h). In burned and not treated soils erosion decreased over time, but this decrease let soil loss be similar as the values estimated in the unburned plots only in the pine forests (from 0.59 ± 0.47 g/m^2 to 2.35 ± 1.43 g/m^2). In contrast, in the plots of the other forest species, the soil losses were higher compared to the unburned soils (up to 7.78 ± 6.01 g/m^2 in chestnut, and to 14.16 ± 6.13 g/m^2 in oak, both occurring in the third and fourth event, respectively) (Figure 5).

Covering soil with fern mulch was able to reduce erosion mainly in pine and chestnut forests, and less in the oak soils compared to fire-affected plots. In more detail, maximum soil losses were equal to 1.87 ± 0.33 and 0.81 ± 0.16 g/m^2 (both surveyed in the third event of 11 Nov 2019, 143 mm and 26.2 mm/h), respectively, while erosion was always over 5.40 ± 0.81 g/m^2 in oak plots (event of 24 Mar 2020, 156 mm and 13.8 mm/h). In pine and mainly chestnut plots, the estimated soil losses for all monitored events were even lower in comparison to unburned soils, while the pre-fire erosion rates were restored in oak forests only for two events (11 Nov 2019 and 24 Mar 2020) (Figure 5).





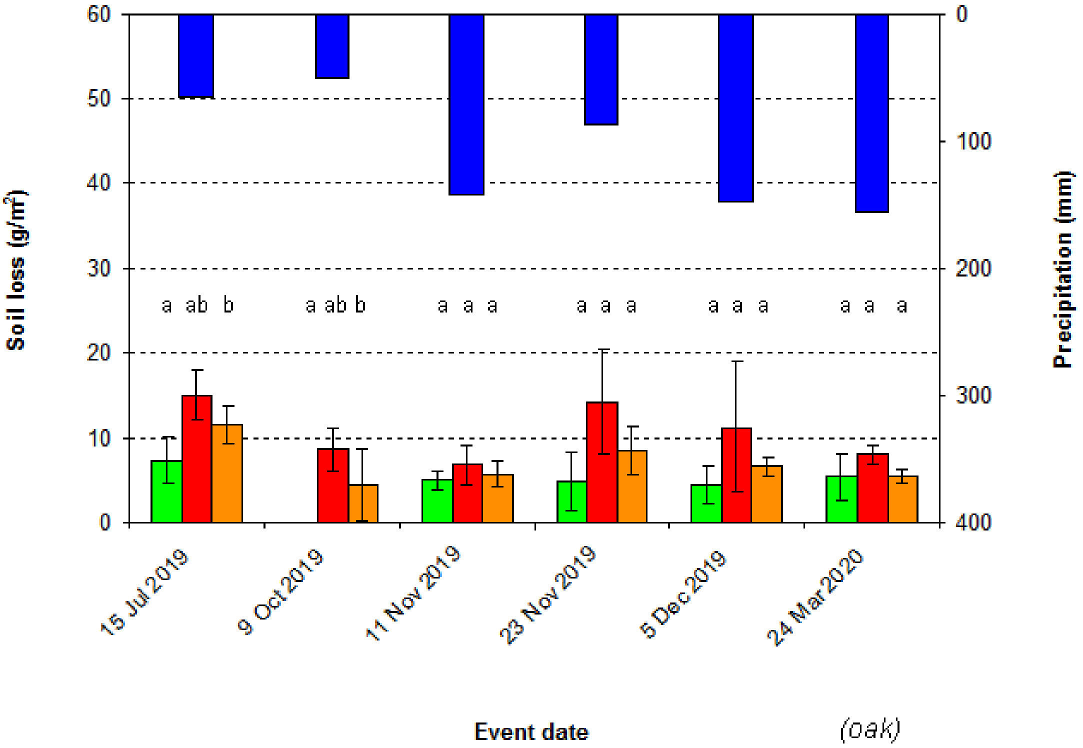


Figure 5 - Precipitation and soil loss measured in plots after prescribed fire and soil mulching using fern (Samo, Calabria, Southern Italy).

Notes: U = unburned soils; B = burned and not treated soils; B + M = burned and mulched soils. Different letters indicate statistically significant differences after Tukey's test (p < 0.05).

4. Discussion

4.1. Effects of prescribed fire on runoff and erosion

Soil hydrology is altered by after fire (also in the case of low intensity, as the prescribed fire) (Cawson et al., 2012; Pereira et al., 2018b; Zema, 2021), and the fire-induced changes influence the hydrological response, which in this study has been quantified by the runoff coefficient and soil loss. It is important to limit as much as possible the increases in runoff and erosion rates adopting suitable post-fire management techniques implemented at the hillslope scale (Lucas-Borja, 2021), and this study has evaluated the effectiveness of soil mulching using fern.

All forest soils showed low runoff coefficients (not higher than 0.20), which means that, also after very intense storms (100-150 mm, having return interval estimated in 3-5 years in this area), the runoff generation capacity of these soils is basically limited. This is mainly due to the high water losses occurring in forest environments, on which high soil infiltration (mainly due to the noticeable organic matter content), tree canopy interception (especially in the broadleaf tree species), water retention by litter and understory, and evapo-transpiration rates are beneficial, e.g., (Imeson et al., 1992; Llorens et al., 2011; Nadal-Romero et al., 2016). The low runoff generation measured in the undisturbed soils also limited the erosion rates, whose maximum value was 0.15 tons per hectare (chestnut forest) for the most intense rainstorm. Cumulating all the monitored erosive events, erosion never exceeded 0.33 tons/ha throughout the monitored year, and this value is well below the tolerance limit between 3 and 11.2 tons/ha per year (Bazzoffi, 2009; Wischmeier and Smith, 1978).

Immediately after the prescribed fire application, the runoff generation capacity of the soil significantly increased in all forest plots. For the first rainfall event, this increase was quantified between 150% (for oak) and 375% (for pine and chestnut forests) compared to the pre-fire values (represented by the unburned soils). The higher overland flow immediately after burning was presumably due to the decrease in roughness of surface soil (Stoof, 2011) and reduction in soil’s water storage due to vegetation and litter removal (Govers et al., 2000; Shakesby et al., 2015).

The surveyed increase in runoff is in accordance with (Andreu et al., 2001), who reported that the maximum runoff is reached during the early storms after the prescribed fire, the first months being the most critical period for runoff production (González-Pelayo et al., 2010; Rubio et al., 2003). The significant runoff generation observed in this period in this study (about 2 to 4-fold the values measured in the unburned plots) complies with the results of

(Vega et al., 2005), who found increases in runoff between 2 and 5 times the control values in gorse shrublands of Galicia (NW Spain), although having a wetter climate compared to Southern Italy. In disagreement with the latter studies, (González-Pelayo et al., 2010) reported 10-fold runoff after prescribed burning in a Mediterranean shrub ecosystem close to Valencia (Spain).

In our study, immediately after the fire erosion was in the range 0.09 (oak) to 0.59 (chestnut) tons/ha. Throughout one to five years after prescribed burning, other authors reported erosion in the range 0.2-4.1 tons/ha under natural rainfall in Mediterranean shrubland and grassland (Vega et al., 2005). In contrast, according to (Shakesby et al., 2015), soil losses at hillslope scale were never higher than 2.41 tons/ha in the first year after the prescribed fire. A large range of soil loss is shown by (Neary and Leonard, 2021), from 0.1 to 15 tons/ha per year after low-intensity fires.

The soil loss in our burned plots was much higher compared to the unburned soils throughout four to five months after burning. Immediately after application, fire made the soil exposed to erosion particularly in pine and chestnut forests, and less in oak plots. The increase in the erosion rates due to fire is variable from 500% in chestnut to 800% in pine for the first event, while increase was only 100% in oak forest. The erosion rates surveyed in pine and chestnut forests are higher than the values reported by (Soto et al., 1994) and close to those of (Soler et al., 1994). The 2-fold soil loss surveyed in oak forest compared to the unburned soils is similar to the increases in burned areas reported by (Vega et al., 2005). Therefore, erosion is not minimal following prescribed, in contrast with (Morris et al., 2013), (Coelho et al., 2004), and (de Dios Benavides-Solorio and MacDonald, 2005), but never remarkable, as found by other research. For instance, according to (González-Pelayo et al., 2010), Inbar et al., (1998), Campo et al., (2006), and Cawson et al., (2013), erosion can increase even by 100 times the unburned soils after prescribed fire.

The worsened hydrological response of burned soils in the experimental plots was mainly ascribed to two effects: (i) the reductions in the water infiltration rates (in soils of all forest species); and (ii) the occurrence of soil water repellency (particularly in pine and oak soils). These statements are supported by the results of the previous study carried out by (Carrà et al., 2021), who have evaluated the water infiltration rate (IR) and soil water repellency (SWR) in the same forest stands, using a portable rainfall simulator to measure IR, and the Water Drop Penetration Test (Bisdom et al., 1993; Letey, 1969; Woudt, 1959), to estimate SWR.

In more detail, regarding the water infiltration measurements, the prescribed fire reduced the mean IR in the soils of all forest species compared to the unburned conditions. The increase in SWR may also have played an important role in increasing runoff and erosion immediately after fire in pine and oak soils. While in chestnut soils the prescribed fire did not alter the slight repellency found in unburned plots, burning determined a strong repellency in both pine and oak soils, with or without mulch cover (Carrà et al., 2021).

Presumably, also the litter and vegetation removal by fire may have played an influence. Since litter and shrub covers were almost completely removed by fire in pine and oak forests, the soil was left bare and thus exposed to the effects of overland flow in runoff and soil detachment as well as to rainsplash erosion. This fire effect was lower in chestnut forest, where the litter amount over ground was much lower compared to the other soils.

The changes in the hydrological response of the burned soils were not permanent, but noticeable throughout 3-4 months. Five months after burning, the low pre-fire runoff generation capacity of the unburned soils was practically restored, and the same decreasing trend was noticed for erosion in pine soils (where, one year after fire, the soil loss became very similar as the control plots). Although declined over time in all forest plots, the increased erosion rates, noticed in the chestnut and oak forests, are still evident after more than one year elapsed from fire application, but, in any case, these changes in soil hydrology were not significant. This means that the recovery of the pre-fire hydrological conditions in burned soils is not complete, although this does not play significant effects on runoff and erosion rates. According to the previous study by (Carrà et al., 2021), this recovery may be ascribed to the increase in the mean IR SWR disappearance, both detected one year after fire. SWR disappeared in few months, losing importance on hydrology of burned soils, which became non-repellent. Moreover, in our experimental plots, we visually noticed that, progressively over time, the litter and shrub covers were recovering in oak and pine stands, thanks to the vegetation regeneration over the burned soils. In contrast, in chestnut soils, litter cover was still limited after one year, as in the soil condition immediately before and after the prescribed burning. Vegetation recovery and litter accumulation during the growing season reduced the impacts of heavy storms during the wet season in forests, preventing high soil loss (Klimas et al., 2020). Herbaceous and shrub vegetation, and leaf litter covers reduce runoff and erosion rates thanks to rainfall interception, soil surface protection, and evapo-transpiration (DeBano et al., 1998; Sayer, 2006; Stoof et al., 2011; Vega et al., 2005; Walsh and Voigt, 1977). Increases in surface roughness due to vegetation and litter on soil determine longer time for overland flow takes to begin during a storm (Cawson et al., 2012;

Johansen et al., 2001; Leighton-Boyce et al., 2007; Pierson et al., 2009; Stoof et al., 2011; Vega et al., 2005).

Overall, regarding the effects of prescribed fire on soil hydrology, our study confirms the “classic” post-fire erosion curve characterised by an early single peak immediately after burning, theoretically reported by (Shakesby et al., 2015; Shakesby and Doerr, 2006; Swanson, 1981), with erosion strongly declining in the subsequent period (Klimas et al., 2020). According to the literature, the effects of an individual prescribed burn lasts for a short period, from three months (Stephens et al., 2004) to one year (Bêche et al., 2005; Cawson et al., 2012). Soil loss then decline in subsequent months after a fire (Neary et al., 2005; Neary and Leonard, 2021), and is extensive in area but small in magnitude (Morris et al., 2014)

4.2. Effects of mulching on runoff and erosion

Soil treatment with fern mulch provided an effective soil protection, which was able to improve the hydrological response of burned soils. Mulching is effective in reducing runoff and erosion rates, since the mulch layer provides a cover that reduces raindrop impact and prevents soil sealing (Lucas-Borja, 2021), promotes infiltration (Bombino et al., 2021, 2019), and decrease runoff velocity (Lal, 1976); (Prats et al., 2016). Moreover, mulch cover synergistically acts with the remaining litter after burning (Vega et al., 2005) towards a reduction in the hydrological response of the burned soils to heavy seasonal storms.

However, it should be noticed that the response of the experimental soils was different among the studied forest species in the short term, but very similar between the two monitored hydrological variables. More specifically, fern mulching was particularly effective in reducing the runoff generation capacity immediately after the prescribed fire in both pine and chestnut plots. Here, reductions in runoff coefficients and erosion by 70-80% were achieved compared to burned soils. Conversely, this reduction was much lower (25-30%) in oak plots. The effectiveness of fern mulching in our study is higher compared results with Prats et al., (2015, 2014, 2013, 2012), who reported runoff reductions between 40 and 60% produced by mulching with forest residues or hydro-mulching during the first year, and (Groen and Woods, 2008) and (Robichaud et al., 2013), who achieved decreases in runoff between 30 and 60% using straw mulch under rainfall simulations and small paired catchments, respectively.

In our study, the beneficial effects of mulching treatment in the short term were not generally due to changes in the hydraulic properties of the soils (namely IR and SWR), and this contrasts the statement by (Lal, 1976), who reports that a mulch layer increases water infiltration and surface storage, and improves soil structure and porosity (Prats et al., 2015). This is confirmed by (Carrà et al., 2021), who reported that in the same forest stands the mean IR slightly increased in chestnut and oak soils, but did not vary in pine forests. SWR was not affected by mulching, in accordance with (Prats et al., 2015). This result is expected, since the vegetal residues require time to be incorporated into the soil and to play effects on soil hydrology (Bombino et al., 2021, 2019). Instead, mulching played its effectiveness providing soil with a cover of vegetal residues, as shown by the decreases in bare soil percentage and progressive establishment of litter (except in chestnut) and shrubs compared to burned soils.

The improvement of hydrological response of burned forests due to mulching was losing importance over time, since pre-fire soil hydrology (runoff coefficients and erosion rates) was restored just some months after burning. However, in pine and chestnut soils, the runoff generation capacity was even lower compared to unburned plots, and the same was observed for erosion in chestnut forest. This means that soil treatment with mulching may be effective also several months after fire, since the vegetal residues have been incorporated into the soil, where organic matter increases and play beneficial effects on soil macroporosity and infiltration capacity (Bombino et al., 2021, 2019; Lucas-Borja et al., 2019b; Shabanpour et al., 2020). As a matter of fact, one year after fire, the study by (Carrà et al., 2021) demonstrated that the infiltration capacity of soils mulched with fern noticeably increased over time, particularly in chestnut and oak, and less in pine soils among all soil conditions. However, the incomplete recovery of the pre-fire values of IR did not significantly alter the runoff and erosion rates compared to the unburned soils, and it may be presumably that this recovery will complete in the short term (Carrà et al., 2021).

One year after fire, the litter cover was restored in the oak and pine forests, but the area with bare soil was higher compared to the soil condition immediately after the prescribed burning, since the mulch cover has progressively disappeared due to wind and degradation of the vegetal material. A comparative analysis of the organic matter content among the different soil conditions - not carried out in this study, since it was beyond its hydrological focus - could have quantified the amount of degraded mulch residues incorporated into the soil over time.

5. Conclusions

This study has evaluated runoff and erosion in soils of three tree species of Mediterranean forest after a prescribed fire and mulching treatment, and the results help in replying to the three research questions supporting the investigation.

First, immediately after the prescribed fire, runoff and erosion significantly increase in all forest plots compared to the unburned soils, but these increases (150% to 375% for the runoff coefficients, and 100% to 800% for the soil losses) are much lower compared to the highest values reported by some studies.

Secondly, the window of disturbance after fire is limited to three-four months after fire, and after five months the pre-fire runoff generation and erosion rates of the soils are practically restored, and, if the runoff and erosion are still increased compared to the unburned soils, these changes are not significant.

Thirdly, the mulch application using fern residues, which is widely available in Mediterranean forest and is more advisable compared to the most common use of straw, is effective to limit the increase in the hydrological response observed in the burned soils. This has been demonstrated by reductions in runoff coefficients and erosion by 70-80% (except for oak soils, -25-30% for both runoff and erosion) in the experimental sites.

The changes in soil hydrology due to the prescribed fire are due to the reductions in IR, occurrence of SWR (particularly in pine and oak soils), and litter and vegetation removal. The soil cover due to mulching is effective and its influence on water infiltration and repellency is very limited on the hydrological response of the burned soils, while the increases in these hydraulic properties gain importance over time and become beneficial one year after fire, even determining in some cases higher infiltration and lower runoff and erosion compared to the unburned soils.

Further research is needed (i) to validate the results of this study achieved at the plot scale through upscaling at the hillslope or better catchments scale and (ii) to explore the influence of the physico-chemical properties (particularly for organic matter content) on soil hydrology under burned (with and without treatments) conditions.

Overall, the results of this investigation can support the tasks of landscape managers to identify proper fuel management practices for wildfire risk reduction (such as the prescribed fire), and of hydrologists to identify cheap and effective techniques of ecological engineering (such as the mulching with fern) in forests.

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CHAPTER 3

Short-term changes in soil properties after prescribed fire and mulching with fern in Mediterranean forests

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Abstract

Prescribed fire, although having low intensity and being able to reduce the risk of wildfire may modify soil properties in the short term, with possible increases in runoff and erosion risk. Soil mulching with vegetation residues is one of the most common post-fire management strategies. Residues of fern, which is abundant on the Mediterranean forest floor, may be used to replace straw for mulching fire-affected areas. However, the effects of prescribed fires are not completely understood, and there is no data regarding the use of fern to protect soil after fire in the literature. To fill this gap, selected soil chemical parameters were analysed, on a comparative basis, in three Mediterranean forests (pine, oak and chestnut) in Calabria (Southern Italy). These parameters were measured immediately and one year after fire in unburned, burned and not treated, and burned and mulched soils. Changes in soil chemical properties among the different treatments were significant, and the effects of the prescribed fire and mulching were dependent on the time elapsed from their application and forest species. In general, mulching was not effective in limiting the changes in the monitored soil properties compared to the pre-fire values. Each forest species showed different temporal trends in changes of soil properties.

Keywords: organic matter; nutrients; macro-elements; vegetation cover; post-fire management.

1. Introduction

Fire is a natural phenomenon with positive and negative impacts on forest ecosystems (Pereira et al., 2021). High-severity fires heavily impact the forest areas of the Mediterranean Basin, where the semi-arid climate conditions are favourable for wildfires (Cerdá and Mataix-Solera, 2009; Hueso-González et al., 2018; Inbar et al., 2014).

Fire induces important changes in almost all soil properties (Badía et al., 2017; Zavala et al., 2014), such as organic matter, nutrient contents, changes in hydraulic conductivity, and microbial communities (Certini, 2005; Inbar et al., 2014; Shakesby, 2011). These changes in soil properties alter many processes in forest ecosystems, such as hydrology, plant regeneration, carbon and nutrient cycles (Certini, 2005; Neary, 2009). Every year local administrations make great efforts to limit the risk of wildfire, adopting diverse prevention strategies with varying effectiveness. The use of the prescribed fire - the planned use of low-intensity fire to eliminate the fuel that can generate high-intensity fires (Alcañiz et al., 2018; Fernandes et al., 2013) - has been increasingly gaining popularity among forest managers (Neary and Leonard, 2021; Úbeda et al., 2005), mainly in USA, Australia and the Iberian Peninsula (Klimas et al., 2020). Many regions in these countries are prone to wildfire, and authorities have prioritized prescribed fire due to the numerous experiments that have demonstrated its feasibility to limit the wildfire risk (Alcañiz et al., 2018; Lucas-Borja, 2021). In general, the changes in soil properties after a prescribed fire are limited (Lucas-Borja et al. 2019). Moreover, prescribed fire is thought to be beneficial for many ecosystem services, such as natural hazard regulation and pest and disease control (Pereira et al., 2021). However, although being of low intensity with lower impacts compared to wildfires, prescribed fire can also exert negative effects on soil properties (Alcañiz et al., 2018; Francos and Úbeda, 2021). Some studies which analyzed soil changes after prescribed fires detected increases in pH, electrical conductivity, total carbon and nitrogen, magnesium, calcium and potassium in burned soils (e.g., Úbeda et al. 2005; Scharenbroch et al. 2012; Alcañiz et al. 2020). However, how these compounds or elements vary after low-intensity fires differs. For instance, total carbon content may decrease or its changes may be extremely small or not significant (Alcañiz et al., 2018; Úbeda et al., 2005). Likewise, the changes in nitrogen may increase (Alcañiz et al., 2016; Kennard and Gholz, 2001; Shakesby et al., 2015), be stable (Lavoie et al., 2010; Moghaddas and Stephens, 2007; Neill et al., 2007) or even decrease (Alcañiz et al., 2020; Badía et al., 2017; San Emeterio et al., 2016).

The duration of the effects of prescribed fire on soil properties is also debated in the literature. Some authors report unaltered pH values and ephemeral variations in electrical conductivity (Alcañiz et al., 2020; Badía et al., 2017), while, one year after the prescribed fire, other research shows marked decreases in nitrogen (Blankenship and Arthur, 1999; Muqaddas et al., 2015) as well as the recovery to pre-fire contents of magnesium and calcium (Alcañiz et al., 2020). It has been found that the recovery of pre-fire soil properties may take place over short (Zhao et al., 2015) or long (Alcañiz et al., 2016) time spans, depending on fire temperature and residence time, topography of the burned area, rainfall, and degree of vegetation regeneration (Girona-García et al., 2019; Úbeda et al., 2018). Therefore, further research is needed to better understand the effects of prescribed fires in environments with contrasting characteristics (Hubbert et al., 2006; Hueso-González et al., 2018).

One of the most detrimental effects of prescribed fire is the possible increase in runoff and erosion in the short term (Lucas-Borja et al., 2019), due to vegetation removal and sudden changes in the properties of soils (Alcañiz et al., 2018; Cawson et al., 2012). This is an important concern when risk of wildfire endangers forests on steep slopes with highly erodible soils, such as in many environments of Southern Europe (e.g., (Bombino et al., 2021b; Fortugno et al., 2017) in Southern Italy). To the authors' best knowledge, so far no studies have evaluated the soil changes after prescribed fire in these environments, where the wildfire and erosion risks are high, due to climatic and geomorphological characteristics.

Soil mulching with vegetation residues is one of the most common post-fire management strategies to limit runoff and erosion in the short term (Lucas-Borja et al., 2019; Zema, 2021). Mulching protects soils with cover (Zituni et al., 2019), and can increase water infiltration and soil quality over time, thanks to the supplied organic matter and nutrients (Prosdocimi et al., 2016). However, post-fire mulching is not always beneficial for soil properties. For instance, in pine forests of Central Eastern Spain, (Lucas-Borja et al., 2018) measured lower infiltration rates when soil is not saturated compared to unburned and untreated forest areas, especially in the dry seasons. (Fernández-Fernández et al., 2016) stated that mulching is not effective in reducing soil erosion and subsequent loss of nutrients; in an experiment carried out in North-Western Spain, these authors showed that the reduction in soil erosion was due to moderate precipitation rather than mulching, and the concentration of some elements in sediments led to problems for vegetation growth and soil health. Moreover, the use of straw, which is commonly used as mulch material, is not always suitable in forest areas, since biomass transport from agricultural sites may be expensive; moreover, these vegetal residues often contain agro-chemicals and parasites, and thus development of non-native vegetation and diseases to forest plants are possible (Bento-Gonçalves et al., 2012). Fern (*Pteridium aquilinum* (L.) Kuhn) is a vascular plant that can grow in semi-arid climates, where the water competition among plants is high (Bombino et al., 2019). Since fern is abundant on the Mediterranean forest floor, its residues may replace straw for mulching fire-affected areas. Fern can be directly cut on the forest floor (without being transported from distant sites) and does not bring non-native seeds or chemical residues into the forest ecosystem. Fern can be cut in adjacent forests, where the fire and erosion risks are lower, and applied as it is (that is, without desiccation, as usually carried out for straw) on soil. Moreover, fresh fern may be more rapidly and easily degraded and incorporated in soil compared to other dried residues.

However, the changes in the properties of soils mulched with fern residues should be properly evaluated after the prescribed fire, particularly in the short term, when the impact of the fire is higher (Lucas-Borja, 2021; Zema, 2021). Unfortunately, no data on using fern to protect soil after fire is present in literature. Due to this gap in research, little is known about the effective of fern mulching in limiting soil changes after fire. To fill this gap, the present study evaluates short-term changes in soil properties and covers after a prescribed fire and mulching treatment with fern in three Mediterranean forests (pine, oak and chestnut) in Calabria (Southern Italy). The selected properties have been analysed on a comparative basis in unburned, burned and not treated, and burned and mulched soils immediately and one year after fire. More specifically, the research questions of this study are the following: (i) does prescribed fire modify the soil properties in the short term after burning? (ii) is mulching with fern able to reduce these changes? The answers to these questions aim to give more insight to land managers about the possible changes in burned soils due to low-intensity fire and post-fire management.

2. Material and methods

2.1. Study area

Three forest sites in an area in the municipality of Samo (Calabria, Southern Italy) were selected for this investigation (Figure 1). This area exhibits the typical climatic characteristics of semi-arid zones (“Csa” class, “Hot-summer Mediterranean” climate, according to Koppen (Kottek et al., 2006), with mild and rainy winters, and warm and dry summers. The mean annual precipitation and temperature are 1102 mm and 17.4 °C, respectively. The minimum and maximum temperatures are - 4.3 and 43.1 °C, respectively (weather station of Sant'Agata del Bianco, UTM coordinates 4217548 N, 595159 E, period 2000-2020).

The soils of the experimental sites - Haplic Cambisols, according to the World Reference Base for Soil Resources classification dated 2014 - were homogenous, with a loamy sand texture (10.6 ± 2.57% of silt, 8.76 ± 0.61% of clay, and 80.7 ± 2.68% of sand), except for an unburned area of the pine forest, which was sandy loam (10.1 ± 1.01% of silt, 9.0 ± 0.01% of clay, and 81.0 ± 0.99% of sand).

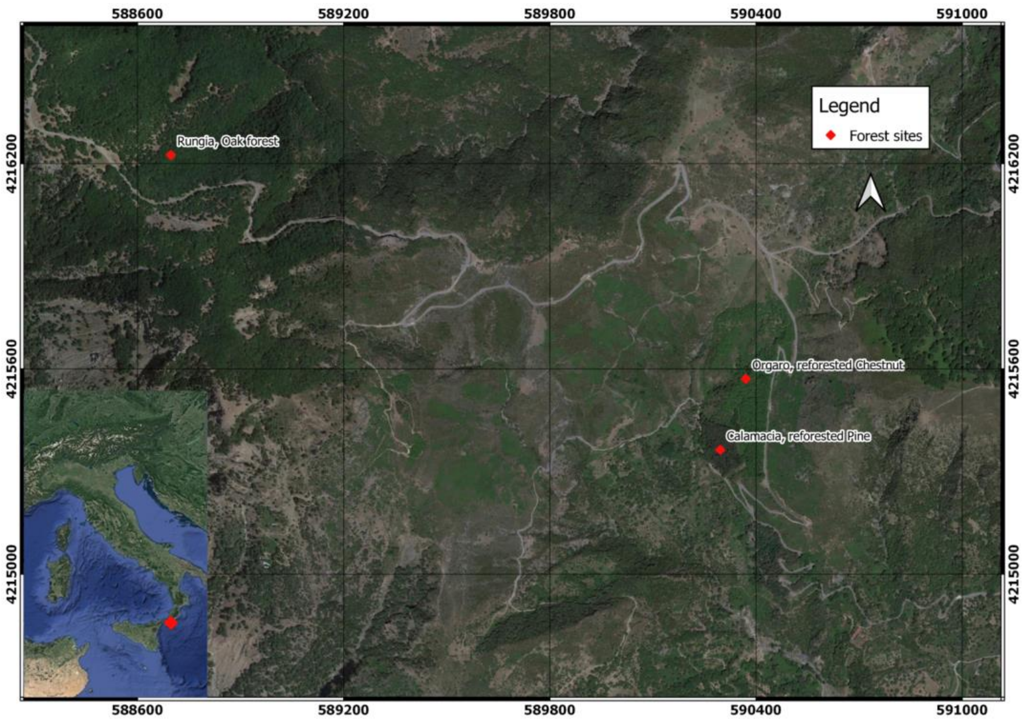


Figure 1 – Geographical location (Southern Calabria, Italy) and aerial view of study area with the experimental forest sites (Samo).

Of the three forest sites, one was a pine (*Pinus pinaster* Aiton) in the locality of Calamacia, and one a chestnut stand (*Castanea sativa* Mill.) in the locality of Orgaro, both reforested between 20 and 30 years ago, respectively. The third site was a natural stand of oak (*Quercus frainetto* Ten.) in the locality of Rungia. Site elevation was in the range 650 to 950 m a.s.l., while their mean slope was about 20% for all stands (Table 1).

Table 1 - Main characteristics of the experimental forest sites (Samo, Calabria, Southern Italy).

Characteristics		**Site**					
		Rungia		*Calamacia*		*Orgaro*	
UTM coordinates		588635 E	4216172 N	590293 E	4215327 N	590389 E	4215530 N
Aspect		North-East		South-West		West	
Altitude (m a.s.l.)		900-950		650-700		700-750	
Slope (%)		19.1 ± 1.65		20.0 ± 0.82		20.3 ± 0.96	
Tree layer	species	Oak (*Quercus frainetto* Ten.)		Pine (*Pinus pinaster* Aiton)		Chestnut (*Castanea sativa* Mill.)	
	density (n/ha)	225 ± 44.7		950 ± 86.4		725 ± 89.1	
	diameter at breast height (cm)	40.7 ± 8.9		28.3 ± 9.4		20.2 ± 5.6	
	height (m)	18.2 ± 1.9		20.5 ± 1.4		9.6 ± 1.2	
Shrub layer	species	*Cyclamen hederifolium, Bellis perennis L.*		*Quercus ilex* L., *Rubus ulmifolius* S., *Bellis perennis L.*		*Rubus ulmifolius* S., *Pteridium aquilinum L.*, *Bellis perennis L.*	

Tree density was higher in the pine (about 950 trees/ha) and chestnut (725 trees/ha) stands compared to the oak forest (225 trees/ha). The chestnut forest showed the lowest mean values of tree height (10 m) and diameter (20 cm), while the pine and oak trees were taller (18 - 20 m) and had a larger mean diameter (28 and 40 cm, respectively). Shrub formations mainly consisted of *Quercus ilex L., Rubus ulmifolius S., Bellis perennis L.* (pine forest), *Cyclamen hederifolium, Bellis perennis L.* (oak) and *Rubus ulmifolius S., Pteridium aquilinum L., Bellis perennis L.* (chestnut) (Table 1).

2.2. Prescribed fire operations and mulching application

The prescribed fire was carried out in three areas (each of about 250 m^2) of the three forest stands on 10 June 2019 with the support of the Forest Regional Agency (Calabria Verde) and the surveillance of the National Corp of Firefighters (Figure 2a). Wind was practically absent and air humidity was between 50 and 60%. The same ignition technique was used for all forest sites, which furthermore had the same fuel characteristics. The main conditions during the fire (temperatures of flame, air and soil) were measured at a soil depth of 2.5 cm - the mid-point of the surface layer (0-5 cm), where the main effects of the prescribed fire occur (considering that soil is a poor heat conductor, (Certini, 2005; Pereira et al., 2018)) - by a thermocouple connected to a datalogger (Table 2).

The burn severity of soils after the prescribed fire was evaluated according to the classification by (Parson et al., 2010). Accordingly, one day after the prescribed fire, the ground surface of all sites was visually checked, observing ash colour and fine roots. Following (Parson et al., 2010), the burn severity of all sites were low, that is, with small change from pre-fire status, black ground surface with recognizable fine fuels remaining on surface, and fine roots unchanged.

Table 2 – Main conditions during prescribed fire operations in the experimental forest sites (Samo, Calabria, Southern Italy).

Characteristics			**Site**		
			Rungia	*Calamacia*	*Orgaro*
Main forest species			Oak	Pine	Chestnut
Temperature (°C)	fire flame	mean	98	88.3	75
		max	720	712	645
	air	mean	43	25.7	29.1
		max	180	102	139
	soil	mean	21	21.9	24.7
		max	26.9	22.7	28.8
Residence time (% on total fire duration)	mean temperature		44.4	18.0	69.7
	max temperature		17.2	1.6	9.3

In these burned areas, three small portions (about 9 m^2) in each site were mulched with a cover (more or less 3 to 5 cm of thickness) of fern residues, cut from an adjacent zone, and distributed over the ground at a dose of 500 g/m^2 of fresh weight (equivalent to 200 g/m^2 of straw dry matter, usually applied in burned and mulched areas after fire (Lucas-Borja et al., 2018; Vega et al., 2014) (Figure 2b).

A third area, unburned and at a distance of less than 10 m from the burned areas, in each site was selected as a “control”.

(a)

(b)

Figure 2 - Prescribed fire operations (a) and fern mulch application to burned soil immediately after fire (b) in the experimental forest site (Samo, Calabria, Southern Italy).

2.3. Experimental design

The experimental design consisted of three forest stands (pine, oak and chestnut) × three soil conditions (unburned, burned and not treated, and burned and mulched, the latter two hereafter simply indicated as “burned” and “mulched”, respectively) × two survey dates (one day and one year after fire) × three sampling points (Figure 3). In these forests, measurements of soil properties and covers were carried out according this experimental design on both survey dates. The collection of samples within a very short reciprocal distance (lower than 20-25 metres) should have minimized the effect of spatial heterogeneity of soil, and allowed the attribution of the measured changes due to the treatments applied (prescribed burning and mulching).

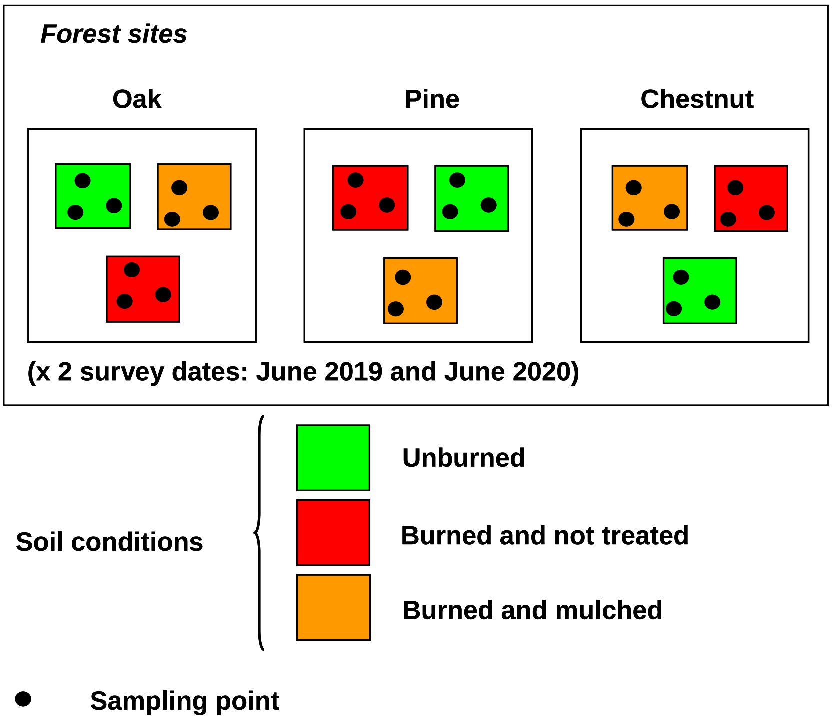


Figure 3 – Scheme of the experimental design (Samo, Calabria, Southern Italy).

2.4. Monitoring of the soil parameters

From each forest site and soil conditions, and immediately after the prescribed fire (one day), samples of soil were collected from between 0 and 5 cm below the soil surface (since the prescribed fire only affects the surface layer of soil (Alcañiz et al., 2018; Pereira et al., 2018). The sampling points were randomly chosen in three small plots (each of 3 m^2), surrounding the point in which the temperature was measured (max distance of 50 cm).

One year later (June 2020), the soil was sampled in the same points as in the surveys carried out immediately after fire. After collection, the soil samples were transported to the laboratory, where the main chemical properties (pH, electrical conductivity, contents of organic carbon, nitrogen, ammonium, potassium, magnesium, calcium, fluorides, chlorides, nitrates, phosphates, and sulphates) were determined. These properties were identified from the relevant literature (Certini, 2005; Cawson et al., 2011; Alcaniz et al., 2018) as the main soil properties that prescribed fire usually modifies. Moreover, we measured in field a set of soil covers (shrub, litter, ash and bare soil). All sampling operations were carried out in three replications.

2.4.1. Measurement of soil properties

Samples of soils were analysed to determine the size distribution of the mineral particles (texture). Soil texture was detected by using the hydrometer method with sodium hexametaphosphate as a dispersant (Bouyoucos, 1962). The final classification was done using the USDA triangle method. The values of pH were measured using a soil:water suspension ratio of 1:2.5 (w/v) with a glass electrode. Electrical conductivity (EC) was determined in distilled water by using a 1:5 residue/water suspension, mechanically shaken at 15 rpm for 1 h to dissolve soluble salts and then detected by Hanna instrument conductivity meter. Organic carbon (OC) was determined by dichromate oxidation method, according to (Walkley and Black, 1934), and titration with iron-sulphate ($FeSO_4$, 0.2 N). Total nitrogen (N) was measured using the method by (Kjeldahl, 1883). The concentration of water-extractable ions, ammonium (NH_4^+), potassium (K^+), magnesium (Mg^{2+}), calcium (Ca^{2+}), fluorides (F^-), chlorides $(Cl)^-$, nitrates (NO_3^-), phosphates (PO_4^{3-}), and sulphates (SO_4^-) was determined by ionic chromatography after extraction of the samples with bidistilled water (soil: water 1:10) for 24 h at 25 °C (Wang et al., 2013) to detect ion concentration (mg/g dry soil) by using a chromatography system (Dionex ICS-1100).

2.4.2. Measurement of soil covers

To evaluate whether the changes in soil surface properties (henceforth “covers”) had impacts on the changes of subsurface soil layer, the shrub and litter cover, and the bare soil, stoniness in percent over the total surveyed area were also measured at the same dates as the chemical properties. Moreover, we also measured the area (as a percentage) covered by ash, which was not removed, to consider its effect on soil changes. The measurements were carried out in nine adjacent areas (each 5 m long x 5 m wide, at a maximum distance of 3 m from the areas where soil properties were measured). The grid method (Vogel and Masters, 2001) for shrub cover, and the photographic method for the remaining variables (litter cover, bare soil, stoniness, and ash) were used. The grid method was applied, using a 0.50 x 0.50-m grid square on the sampling areas (upstream, in the middle, and downstream of each plot).

2.5. Statistical analyses

MANOVA (Multivariate ANalysis Of VAriance) was applied to all soil parameters (response variables) separately for the three forest species, assuming as factors the soil condition (unburned, burned, and mulched) and measurement date (one day and one year after fire). The pairwise comparison by Tukey's test (at $p < 0.05$) was also used to evaluate the statistical significance of the differences in the response variables. In order to satisfy the assumptions of the statistical tests (equality of variance and normal distribution), the data were subjected to normality Anderson-Darling's test or were square root-transformed whenever necessary.

Following this, a Principal Component Analysis (PCA) was applied, in order to identify the existence of meaningful derivative variables (Principal Components, PCs) (Rodgers & Nicewander, 1988) and simplify the analysis of the large number of soil properties and conditions, losing as little information as possible. In this study, PCA was carried out by standardizing the original variables (expressed by different measuring units) and using Pearson's method to compute the correlation matrix. This matrix allowed the identification of relationships among the soil properties analysed. The first three PCs, explaining at least at least a percentage of 70% of the original variance, have been retained.

Finally, the soil samples were grouped in clusters using Agglomerative Hierarchical Cluster Analysis (AHCA), a distribution-free ordination technique to group samples with similar characteristics by considering an original group of variables. As similarity-dissimilarity measure the Euclidean distance was used (Zema et al., 2015).

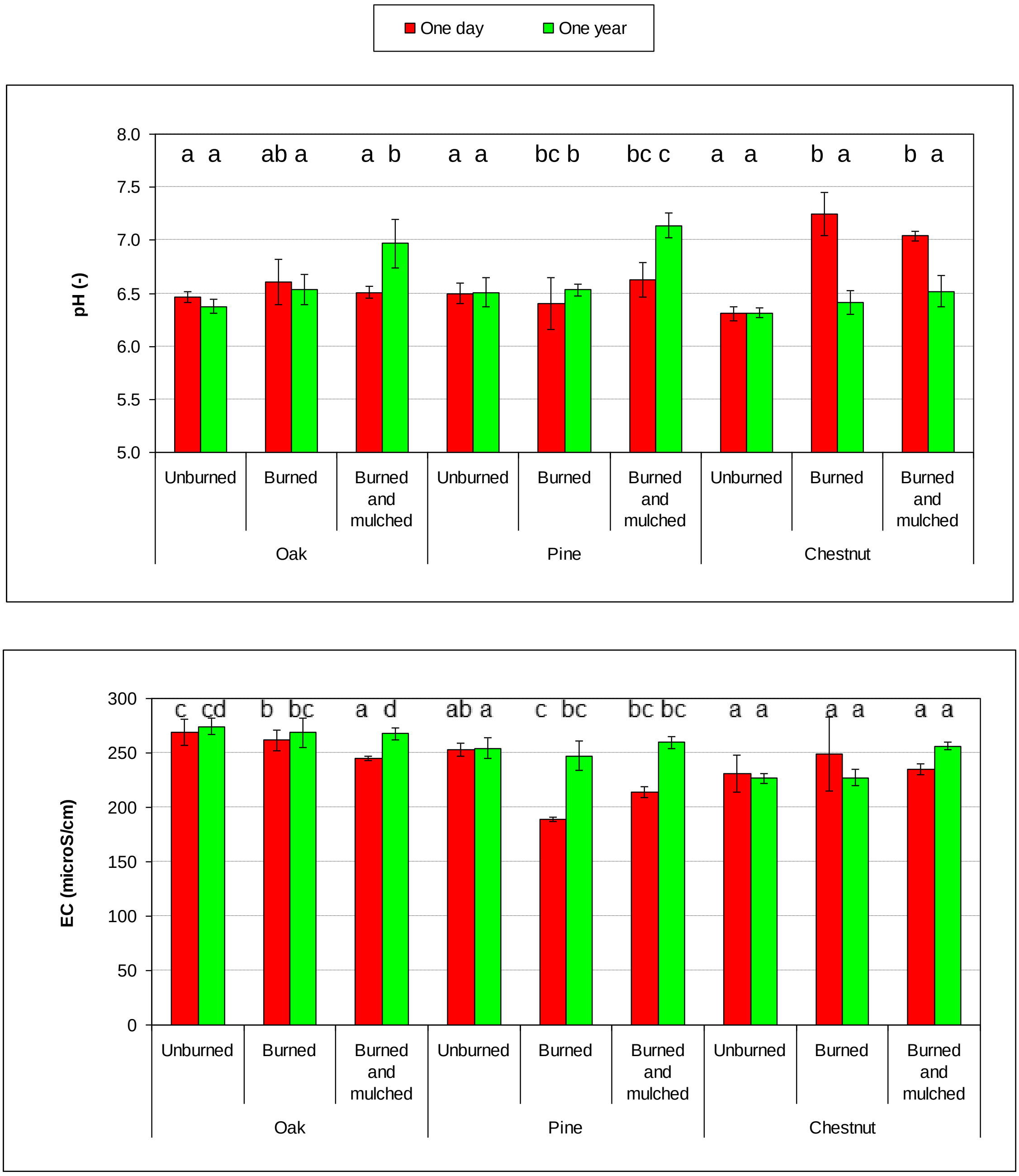
The statistical analysis was carried out using the XLSTAT release 2019 software.

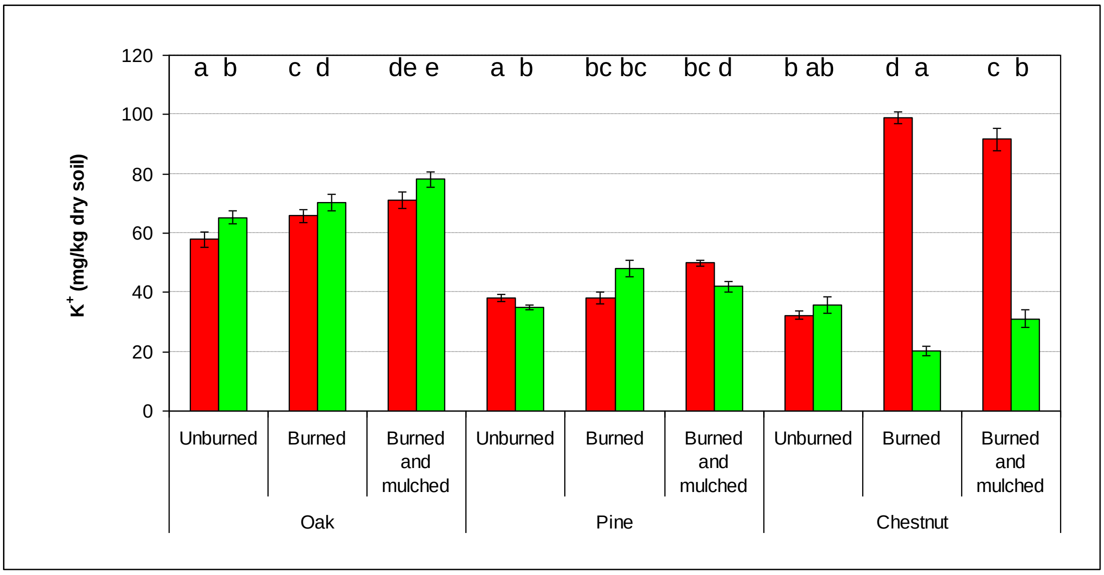
3. Results

3.1. Chemical properties of soils

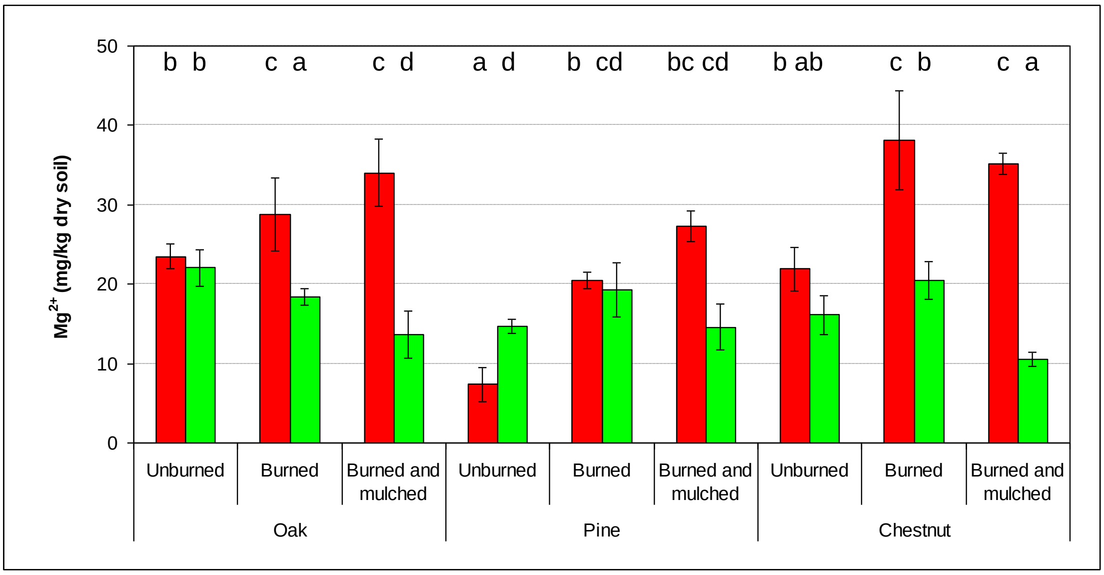
Burning and mulching noticeably altered soil pH and EC in the pine stand one year after the prescribed fire with an increase of one pH unit. Conversely, compared to the unburned soils, significant increases (by 11.6% in mulched areas and 14.8% in burned sites) were measured in the chestnut forest immediately after the fire, while a noticeable decrease (-11.6% and -7.39%, respectively) was observed in burned and mulched sites one year after the fire (Figure 4). No significant pH variations (less than 2%) were observed in soils in the oak forest. One year after the fire, EC generally increased in all forest soils, and particularly in pine (+30.9%, burned soils, and +21.4%, mulched sites) (Figure 4).

The OC content of soils, which showed a limited variability in the unburned sites, significantly increased immediately after the fire (Figure 4). The greatest increase was observed in chestnut forest (+40.2% for both burned and mulched soil conditions). In the oak forest, OC decreased one year after the fire (-18.2%, burned soils, -35.9%, mulched sites), and a very high decrease (-73%) was observed in mulched chestnut site (Figure 4).

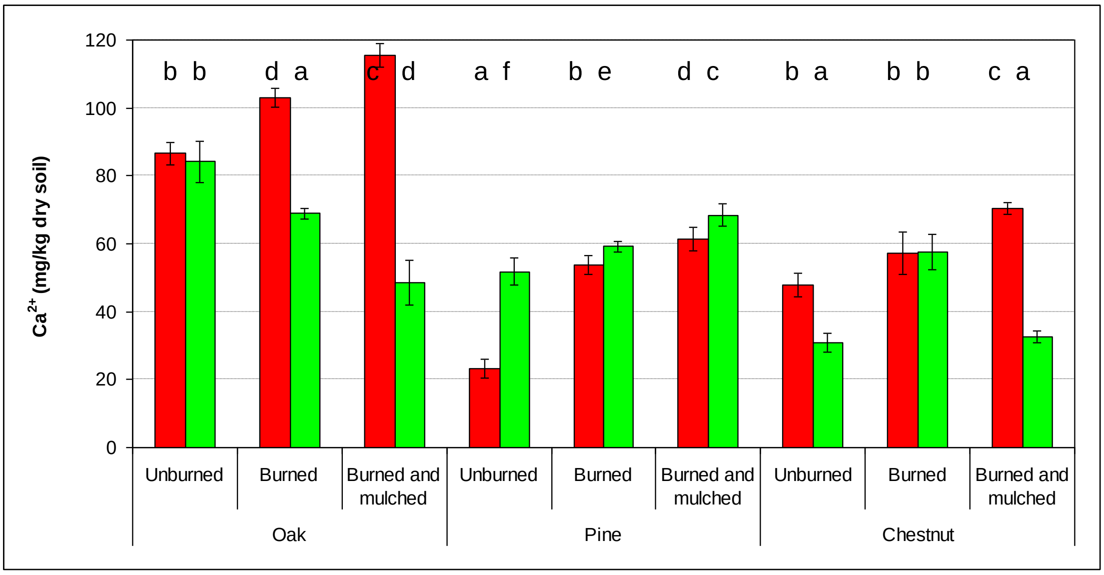




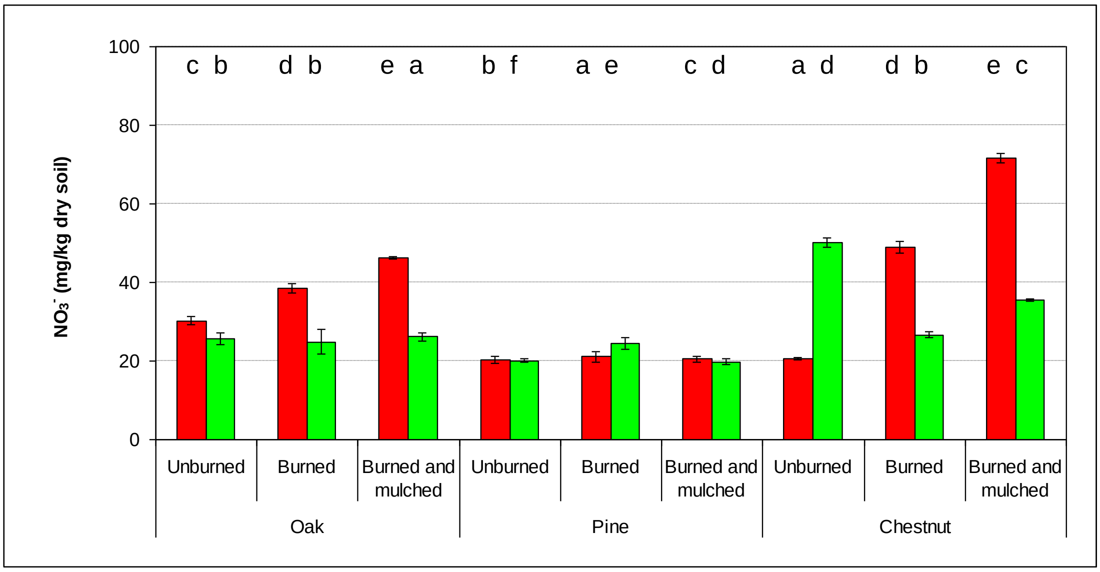
K+ (mg/kg dry soil)
120
100
80
60
40
20
0
a b
c d
de e
a b
bc bc
bc d
b ab
d a
c b
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut

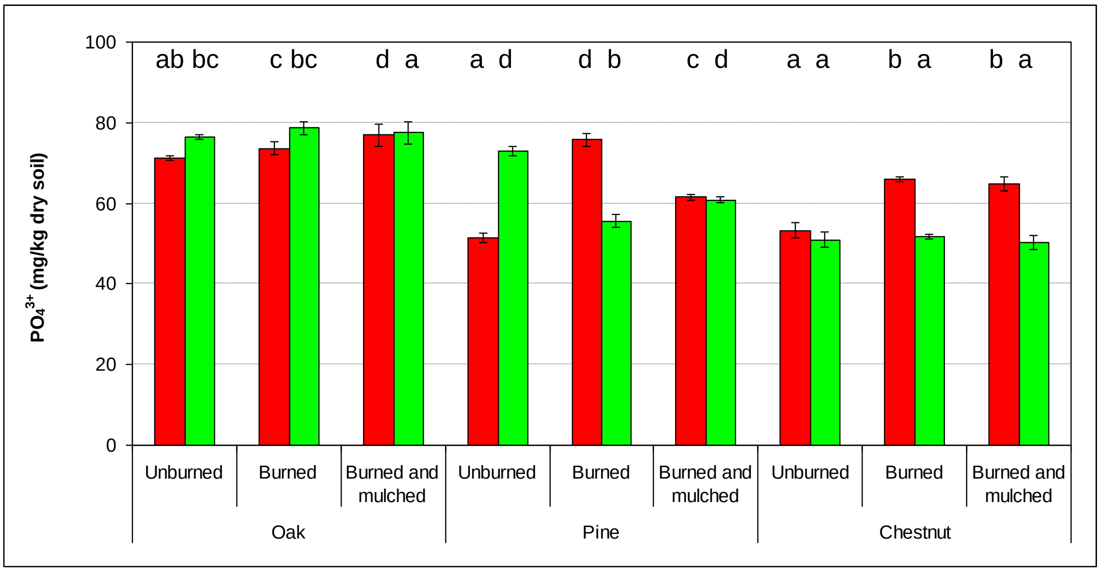


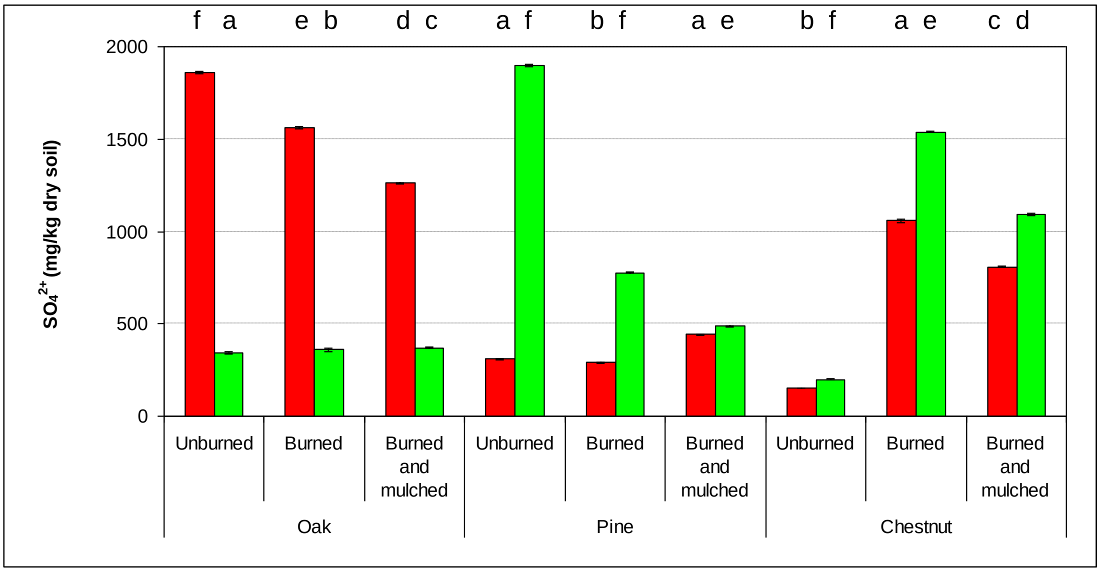
Mg2+ (mg/kg dry soil)
50
40
30
20
10
0
b b
c a
c d
a d
b cd
bc cd
b ab
c b
c a
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut

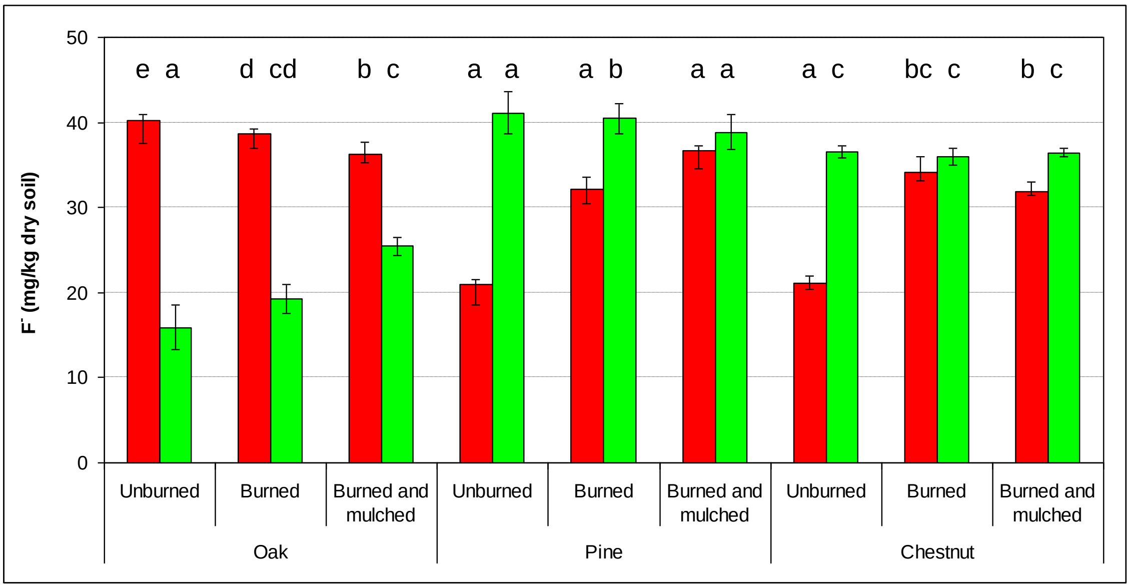


Ca2+ (mg/kg dry soil)
120
100
80
60
40
20
0
b b
d a
c d
a f
b e
d c
b a
b b
c a
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut


NO3- (mg/kg dry soil)
100
80
60
40
20
0
c b
d b
e a
b f
a e
c d
a d
d b
e c
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut


PO4 3+ (mg/kg dry soil)
100
80
60
40
20
0
ab bc
c bc
d a
a d
d b
c d
a a
b a
b a
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut


SO4 2+ (mg/kg dry soil)
2000
1500
1000
500
0
f a
e b
d c
a f
b f
a e
b f
a e
c d
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut



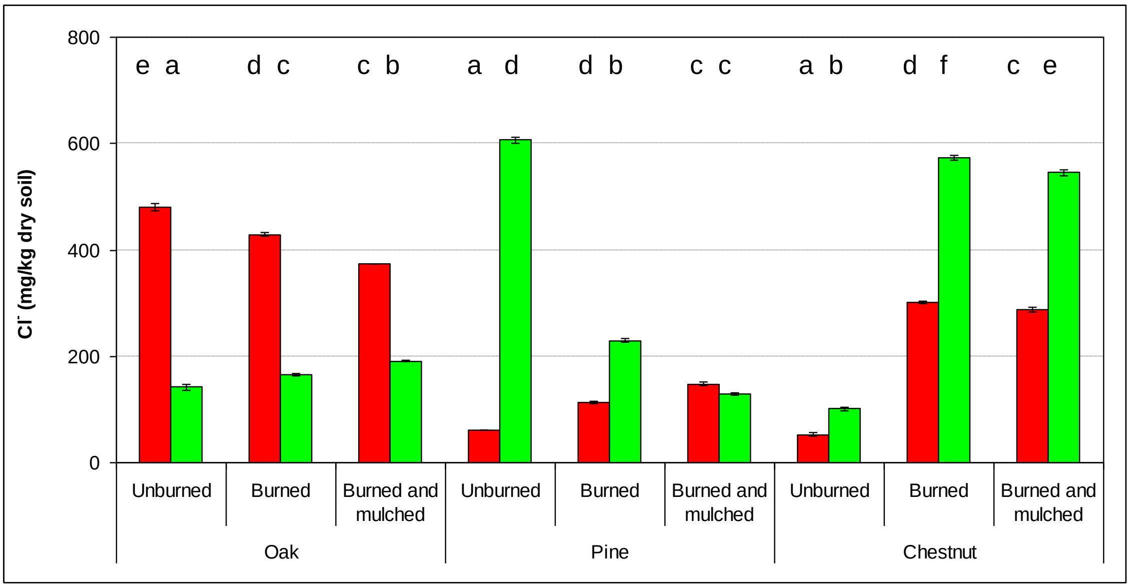


Figure 4 – Soil physico-chemical properties (mean ± standard deviation) measured after the prescribed fire and soil mulching with fern at two survey dates (one day and one year after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

Notes: EC = electrical conductivity; OC = organic carbon; N = nitrogen; NH_4^+ = ammonium; K^+ = potassium; Mg_2^+ = magnesium; Ca_2^+ = calcium; NO_3^- = nitrates; PO_4^{3-} = phosphates; SO_4^{2-} = sulphates; F^- = fluorides; Cl^- = chlorides. Different letters indicate significant differences among soil conditions of each forest site at $p < 0.05$.

The differences in N content of soils due to prescribed fire and mulching were not significant in the oak forest (-6.8%, burned sites, and -17.4%, mulched soils), while it significantly increased in pine (+30.3%, burned, and +29.8%, mulched sites) and chestnut

(+14.4% and +5.6%, respectively) soils (Figure 4). Also, for this parameter, the variability was significant over time, except for pine forest. One year after the fire, the N content of burned soils significantly decreased compared to the unburned sites, particularly when the soils were mulched, with a maximum variation of -58.3% in chestnut forest. Only for pine forest, the interaction of soil condition × survey date was not significant (Figure 4).

Soil ions (NH_4^+, Mg_2^+, Ca_2^+, F^- and Cl^-) significantly changed among conditions and survey date. The interactions between these factors were significant. Compared to the unburned sites, significant increases in Mg^{2+} and Ca^{2+} were observed in burned (up to 176% for Mg^{2+}, and 131% for Ca^{2+}, in pine forest) and mulched (up to 268% for Mg^{2+} , and 164% for Ca^{2+}, in pine forest) soils for all species immediately after the fire, while, one year after, a general decrease in these ion concentrations were noticed (on average -44.1% for Mg^{2+}, and –20.3% for Ca^{2+}) (Figure 4). The NH_4^+ concentration was practically the same immediately after the fire (except in the mulched soils of chestnut forest, +93.8%), and was subject to noticeable increases (up to 172% in mulched soils of pine) over time (Figure 4).

The contents of K^+, NO_3^- and PO_4^{3-}in soils were significantly different among the soil conditions in all forest sites, but for all these properties the time was not significant in pine forests. The same was found for the interaction of soil condition × survey date for K^+ (Figure 4). The latter element, as also found for Mg^{2+} and Ca^{2+}, increased immediately after the fire (on average by about 75%), and particularly in chestnut forest, +206% in burned soils). In the chestnut forest, a marked decrease (on average by 73%) was noticed one year after the fire. A similar trend was detected for NO_3^- and PO_4^{3-} (an increase immediately after the fire, on average + 78.1% for NO_3^- and +20.6% for PO_4^{3-}, and a decrease one year after the fire, by 27% for NO_3^- and 10.6% for PO_4^{3-}) compared to the unburned sites), with some exceptions (NO_3^- in pine forests, and PO_4^{3-} in oak soils) (Figure 4).

The analysis of Pearson's matrix reveals interesting correlations among the soil properties (Table 3). It is worth highlighting that the most significant correlation (r over 0.90) was detected between SO_4^{2-} and Cl^- on both the survey dates, and the correlations between EC and ion concentrations are not significant immediately after the fire (Table 3).

PCA separately applied on the two survey dates provided three principal components, which together explain 80.2% (one day after the fire) and 76.2% (one year after the fire) of the original variance; the first two components explain 68.42% and 59.6% of this variance.

In more detail, according to the PCA related to the surveys carried out immediately after the fire, K^+, Mg^{2+}, Ca^{2+}, NO_3^-, $PO_4^{3-,}$ SO_4^{2-}, Cl^- and had high loadings (> 0.5) on PC1, while EC, C, N, and NH_4^{+-} were effective on the PC2, and pH on the third PC (Figure 5a and Table 4).

For the surveys carried out one year after the fire, PC1 was associated with high EC, NH_4^+ K^+, Mg^{2+}, Ca^{2+} and F^-, while C, N and their ratio weighed on PC2 (loadings over 0.5), and NO_3^-, SO_4^{2-}, Cl^- and on PC3 (Figure 5b and Table 4).

AHCA allowed the soil samples to be clustered according to the soil conditions and forest species. More specifically, immediately after the fire, three similar clusters of soil samples were evident: (i) unburned soils of all forest species; (ii) burned and mulched soils of oak and chestnut; (iii) burned and mulched soils of pine (Figures 5c ad 6a). One year after the fire, the clusters of soil were characterized by a higher level of similarity, 0.30, compared to the survey one day after the fire, 0.21; the following three clusters were depicted by AHCA: (i) oak soils (unburned and mulched) and burned soils of chestnut; (ii) unburned soils of pine and chestnut; (iii) burned soils of oak and pine, and mulched soils of chestnut and pine (Figures 5d and 6b).

Table 3 - Correlation matrix of the soil parameters measured after the prescribed fire and mulching with fern at two survey dates (one day and one year after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

Soil parameter	**pH**	**EC**	**OC**	**N**	**NH_4^+**	**K^+**	**Mg^{2+}**	**Ca^{2+}**	**NO_3^-**	**PO_4^{3-}**	**SO_4^{2-}**	**F^-**	**Cl^-**
	One day after the fire												
pH	**1**	0.245	0.306	0.366	0.056	**0.723**	**0.506**	0.018	**0.471**	-0.064	0.279	0.292	0.350
EC		**1**	**-0.550**	**-0.512**	**0.448**	0.162	-0.115	-0.025	-0.014	-0.357	**0.572**	0.171	**0.473**
OC			**1**	**0.671**	-0.128	0.346	**0.416**	0.287	0.276	**0.440**	-0.088	0.380	0.053
N				**1**	-0.308	**0.498**	**0.613**	0.318	**0.472**	**0.482**	-0.200	0.013	-0.048
NH_4^+					**1**	0.290	0.276	**0.556**	0.325	0.151	**0.772**	**0.541**	**0.814**
K^+						**1**	**0.822**	**0.544**	**0.889**	**0.491**	**0.481**	0.221	**0.607**
Mg^{2+}							**1**	**0.719**	**0.758**	**0.540**	0.355	0.322	**0.528**
Ca^{2+}								**1**	**0.639**	**0.763**	**0.481**	0.274	**0.618**
NO_3^-									**1**	**0.649**	0.295	-0.063	**0.450**
PO_4^{3-}										**1**	0.242	-0.015	0.346
SO_4^{2-}											**1**	**0.660**	**0.972**
F^-												**1**	**0.691**
Cl^-													**1**
	One year after the fire												
pH	**1**	**0.543**	0.325	0.062	0.231	0.309	**0.404**	**0.505**	**-0.520**	0.109	-0.168	0.107	-0.264
EC		**1**	-0.045	-0.337	**0.708**	**0.752**	**0.628**	**0.705**	**-0.519**	**0.557**	-0.258	**-0.519**	-0.271
C			**1**	**0.896**	0.068	-0.083	0.031	0.306	**-0.409**	0.138	-0.070	0.175	-0.375

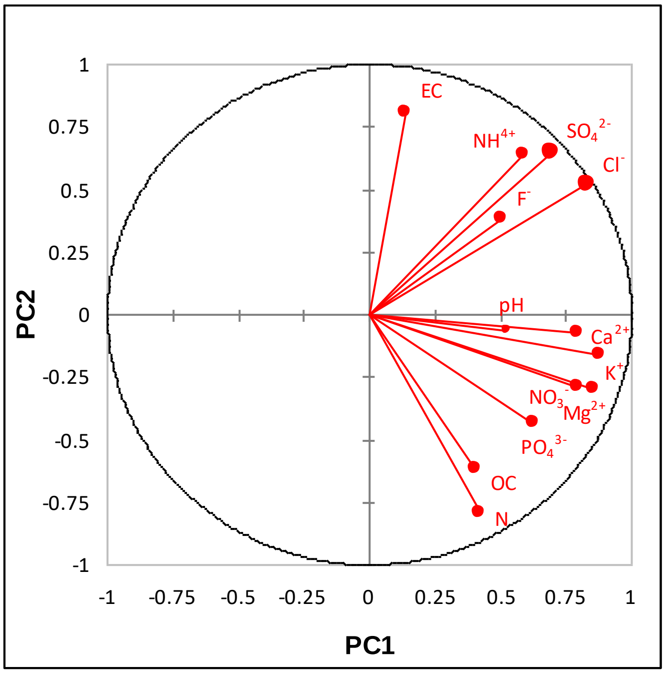
N				**1**	-0.033	-0.259	-0.013	0.150	-0.101	-0.056	-0.047	0.212	-0.325
NH_4^+					**1**	**0.660**	**0.911**	**0.883**	-0.352	0.325	-0.210	**-0.523**	-0.292
K^+						**1**	**0.567**	**0.516**	-0.340	**0.645**	**-0.562**	**-0.582**	**-0.554**
Mg^{2+}							**1**	**0.885**	-0.295	0.127	-0.173	-0.345	-0.247
Ca^{2+}								**1**	**-0.557**	0.294	-0.148	**-0.456**	-0.285
NO_3^-									**1**	**-0.549**	-0.302	0.084	-0.137
PO_4^{3-}										**1**	-0.108	**-0.542**	-0.188
SO_4^{2-}											**1**	**0.479**	**0.943**
F^-												**1**	**0.401**
Cl^-													**1**

Notes: EC = electrical conductivity; OC = organic carbon; N = nitrogen; NH_4^+ = ammonium; K^+ = potassium; Mg^{2+} = magnesium; Ca^{2+} = calcium; NO_3^- = nitrates; PO_4^{3-} = phosphates; SO_4^- = sulphates; F^- = fluorides; Cl^- = chlorides; for each variable values in bold correspond to the factor for which the factor loading is the largest values in bold are significant at $p < 0.05$.

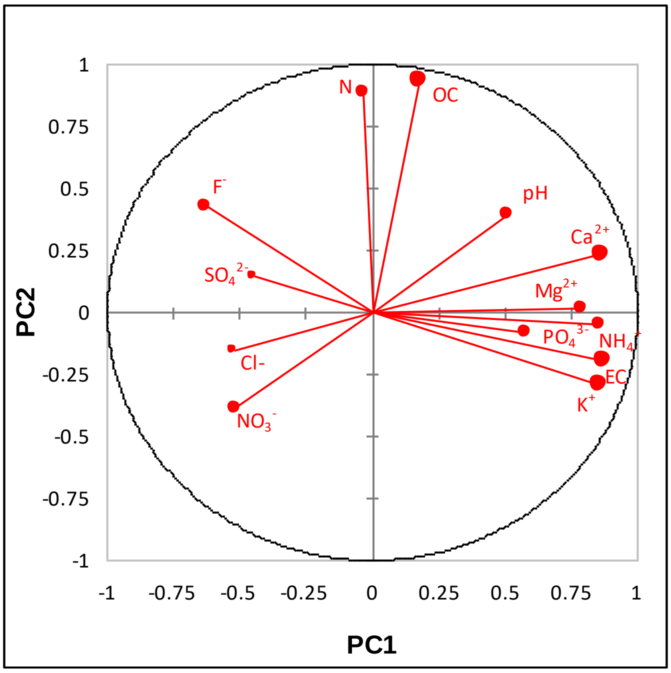
Table 4 - Factor loadings of the soil properties on the first three Principal Components provided by the PCA after the prescribed fire and soil mulching with fern at two survey dates (one day and one year after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

Soil parameter	**Survey time**					
	One day after the fire			**One year after the fire**		
	PC1	PC2	PC3	PC1	PC2	PC3
pH	0.274	0.004	**0.668**	0.259	0.153	0.032
EC	0.018	**0.654**	0.087	**0.762**	0.036	0.053
C	0.164	**0.383**	0.006	0.032	**0.879**	0.031
N	0.178	**0.626**	0.028	0.001	**0.781**	0.096
NH_4^+	0.344	**0.414**	0.077	**0.734**	0.003	0.020
K^+	**0.773**	0.026	0.101	**0.728**	0.082	0.022
Mg^{2+}	**0.735**	0.090	0.006	**0.621**	0.000	0.032
Ca^{2+}	**0.635**	0.005	0.256	**0.748**	0.056	0.048
NO_3^-	**0.632**	0.083	0.000	0.267	0.153	**0.357**
PO_4^{3-}	**0.389**	0.184	0.296	0.335	0.007	0.012
SO_4^{2-}	**0.487**	0.424	0.002	0.200	0.020	**0.724**
F^-	0.256	0.147	0.013	**0.402**	0.184	0.059
Cl^-	**0.691**	0.273	0.003	0.279	0.024	**0.674**

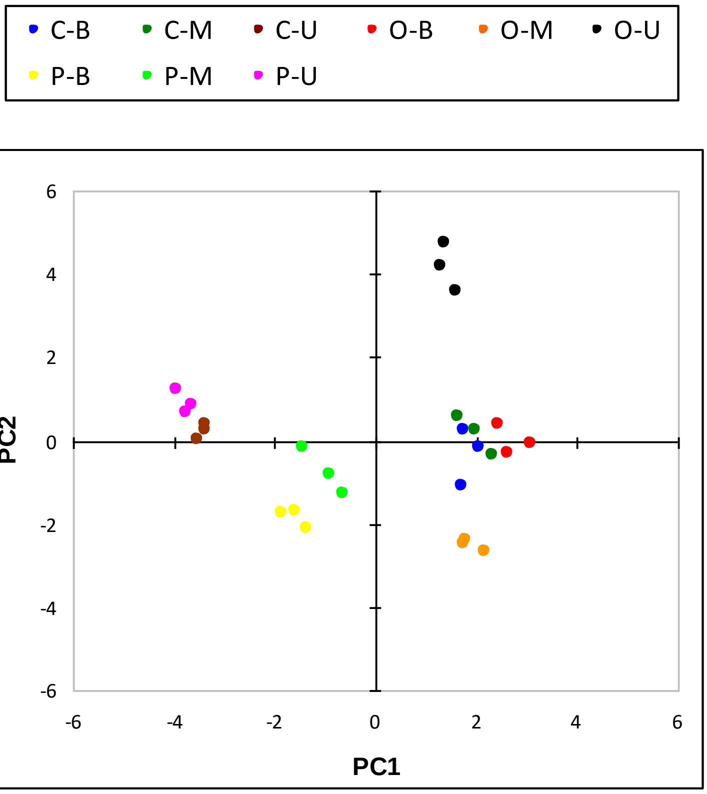
Note: EC = electrical conductivity; OC = organic carbon; N = nitrogen; NH_4^+ = ammonium; K^+ = potassium; Mg^{2+} = magnesium; Ca^{2+} = calcium; NO_3^- = nitrates; PO_4^{3-} = phosphates; SO_4^- = sulphates; F^- = fluorides; Cl^- = chlorides; for each variable values in bold correspond to the factor for which the factor loading is the largest.


EC
NH4+
SO4 2-
Cl-
F-
pH
Ca 2+
K+
NO3-
Mg2+
PO4 3-
OC
N
PC2
PC1
1
0.75
0.5
0.25
0
-0.25
-0.5
-0.75
-1
-1
-0.75
-0.5
-0.25
0
0.25
0.5
0.75
1

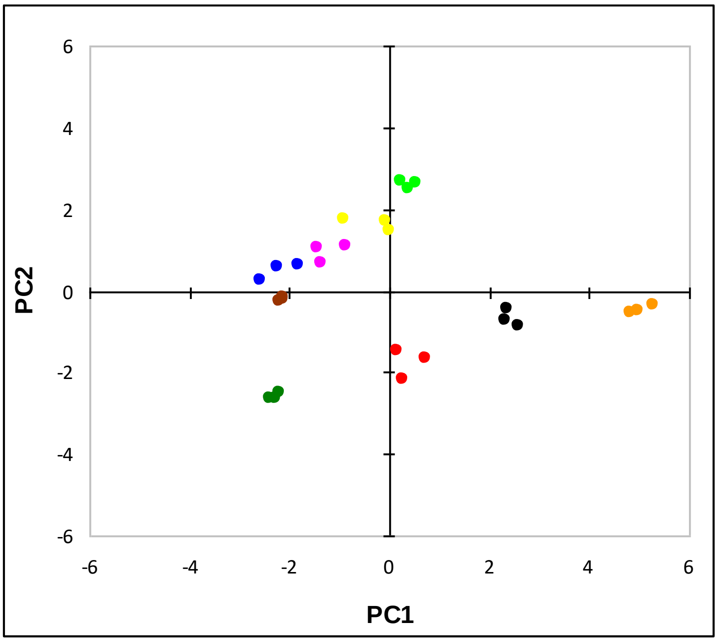
(a)


N
OC
F-
pH
Ca 2+
SO4 2-
Mg2+
PO4 3-
NH4+
Cl-
EC
K+
NO3-
PC2
PC1
1
0.75
0.5
0.25
0
-0.25
-0.5
-0.75
-1
-1
-0.75
-0.5
-0.25
0
0.25
0.5
0.75
1

(b)



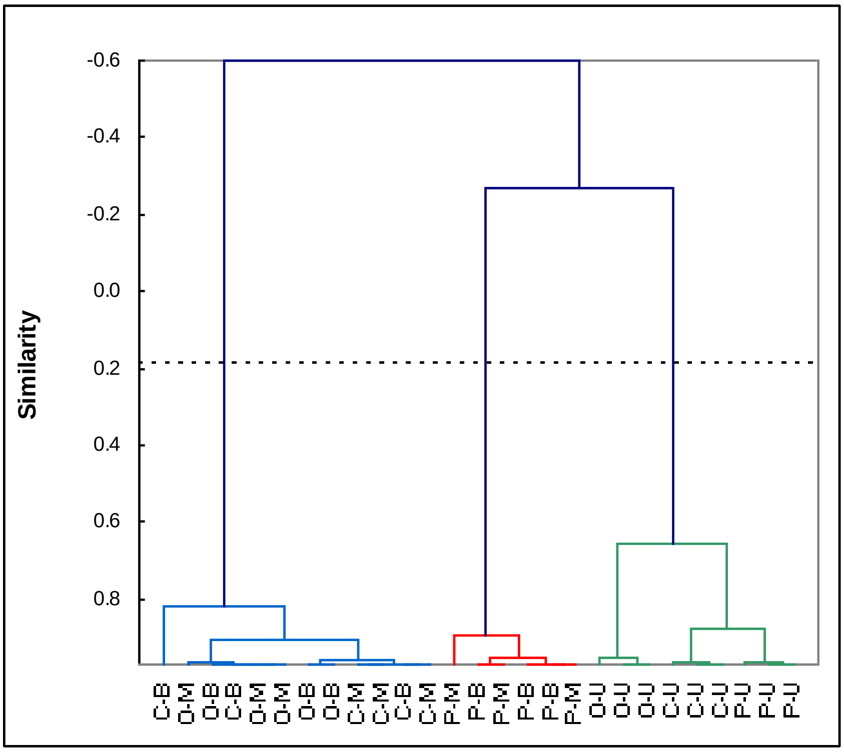
(c)



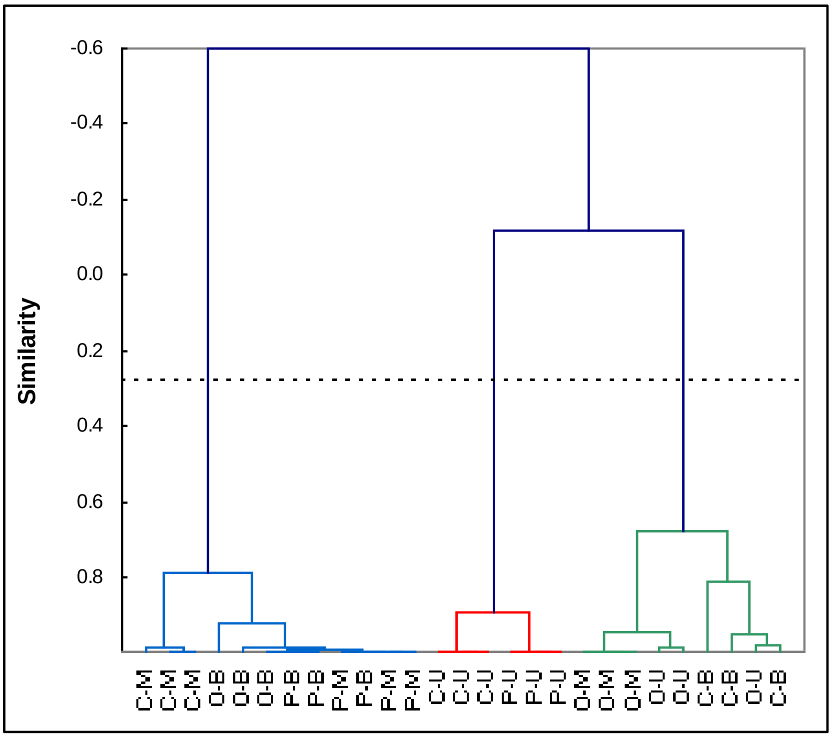
(d)

Figure 5 - Loadings of the soil properties (a) and scores of the soil samples (b) on the first two Principal Components (PC1 and PC2) provided by the PCA after the prescribed fire and soil mulching with fern at two survey dates (one day, a and c, and one year, b and d, after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

Notes: EC = electrical conductivity; OC = organic carbon; N = nitrogen; NH_4^+ = ammonium; K^+ = potassium; Mg^{2+} = magnesium; Ca^{2+} = calcium; NO_3^- = nitrates; PO_4^{3-} = phosphates; SO_4^- = sulphates; F^- = fluorides; Cl^- = chlorides; O = oak; P = pine; C = chestnut; U = unburned; B = burned; M = burned + mulching; for each variable values in bold correspond to the factor for which the factor loading is the largest.



(a)



(b)

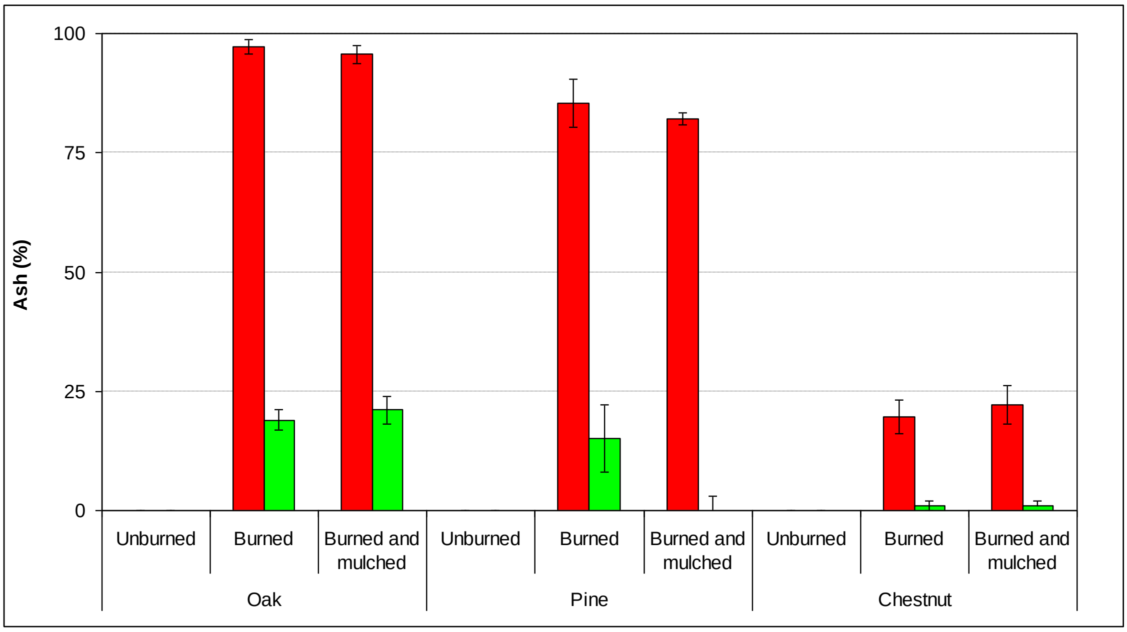
Figure 6 - Dendrogram provided by the Agglomerative Hierarchical Cluster Analysis (AHCA) of the soil properties after the prescribed fire and soil mulching with fern at two survey dates (one day, a, and one year, b, after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

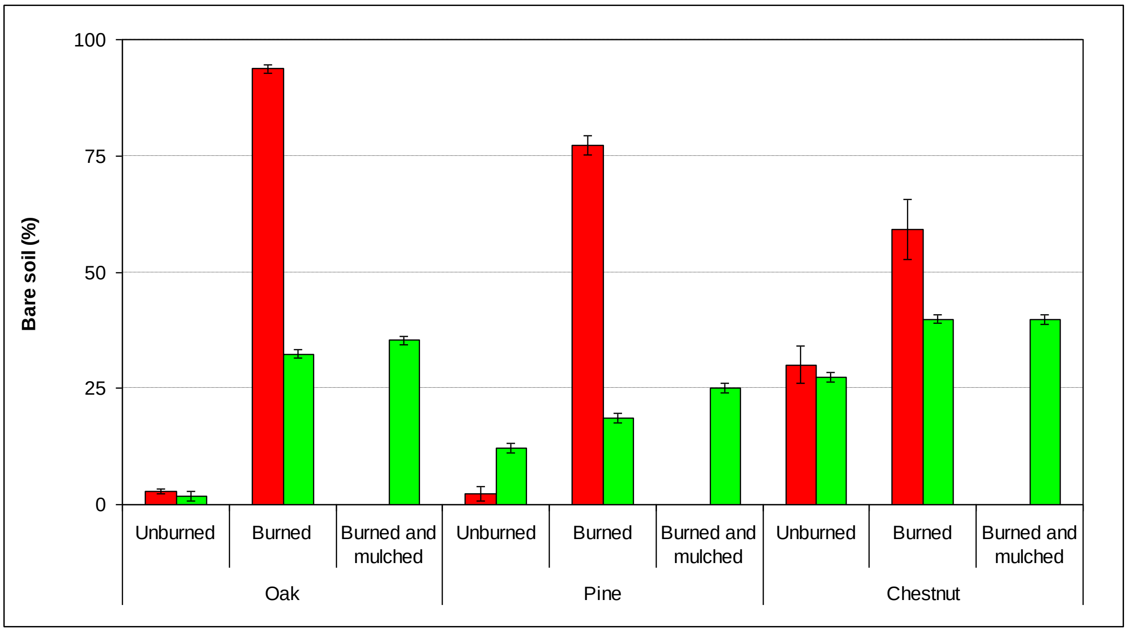
Notes: the y-axis reports the similarity level, while the dotted line the clustering level; O = oak; P = pine; C = chestnut; U = unburned; B = burned; M = burned + mulching; the dashed line is the level

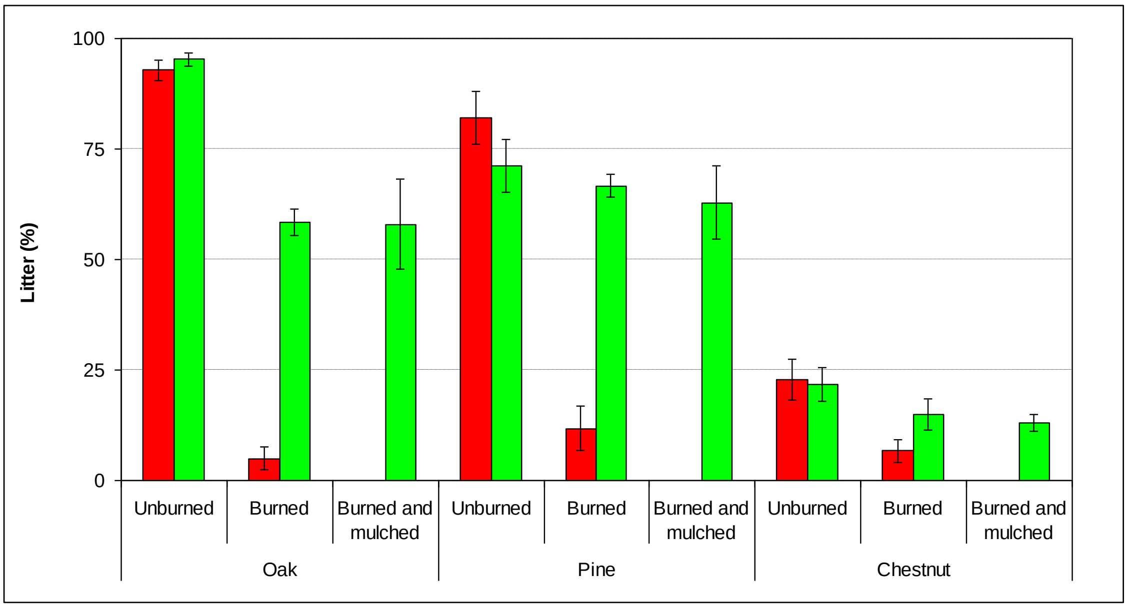
3.2. Soil covers

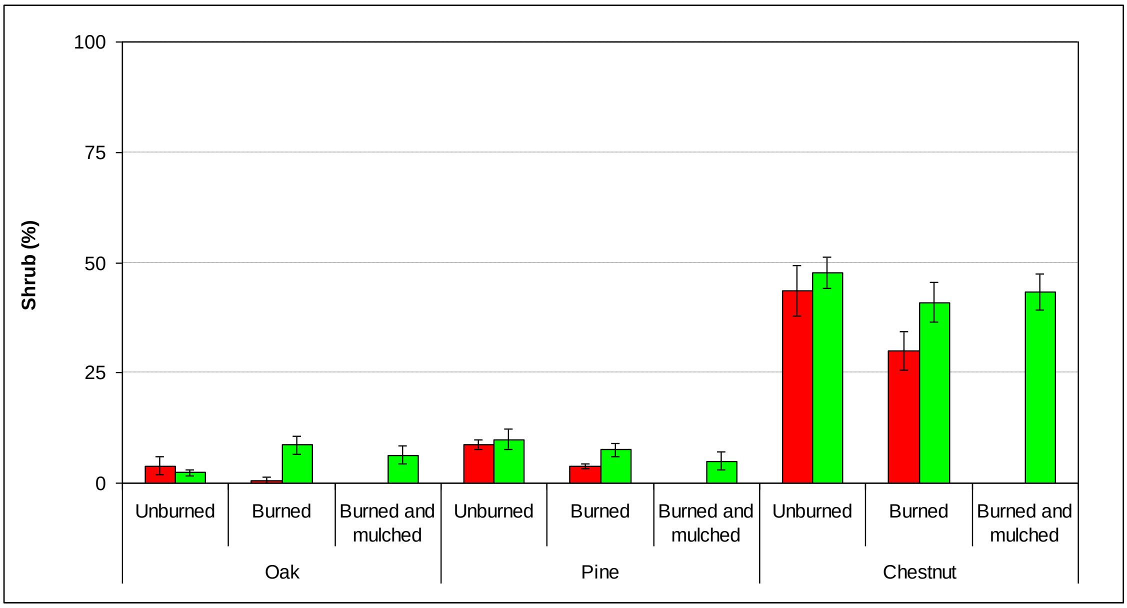
Immediately after the fire, the soils of the three forest stands were covered in ash, in particular in oak and pine rather than chestnut (higher than 85% for oak and pine, and only 20% in chestnut). Fire also removed shrubs and herbaceous vegetation, and burned litter (especially in pine and oak, where it was more abundant compared to chestnut), leaving from 63% (chestnut) to 94% (pine) of bare soil (Figure 7).

One year after the fire, the ash had almost completely disappeared, except in pine (burned sites) and oak (burned sites, mulched or not), where the maximum ash cover was lower than 25%. Shrub vegetation had regenerated, particularly in the mulched sites (up to 40% in chestnut forest), and litter covered the soils, especially in pine (up to 67%) and oak (95%), but less in chestnut forest (maximum litter cover equal to 15%). Vegetation recovery and litter covering over time caused reductions in the areas with bare soil, which decreased to 43% in chestnut (mulched sites), 36% in oak (burned soils, with or without mulching), and 32% in pine (mulched sites) (Figure 7). It should be highlighted that the differences in vegetation recovery between burned and mulched soils were limited (not higher than 2-3%) and never significant, and the same was noticed for litter cover (< 5%) and ash (< 15% in pine forest) (Figure 7).






Litter (%)
100
75
50
25
0
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut


Shrub (%)
100
75
50
25
0
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Unburned
Burned
Burned and mulched
Oak
Pine
Chestnut

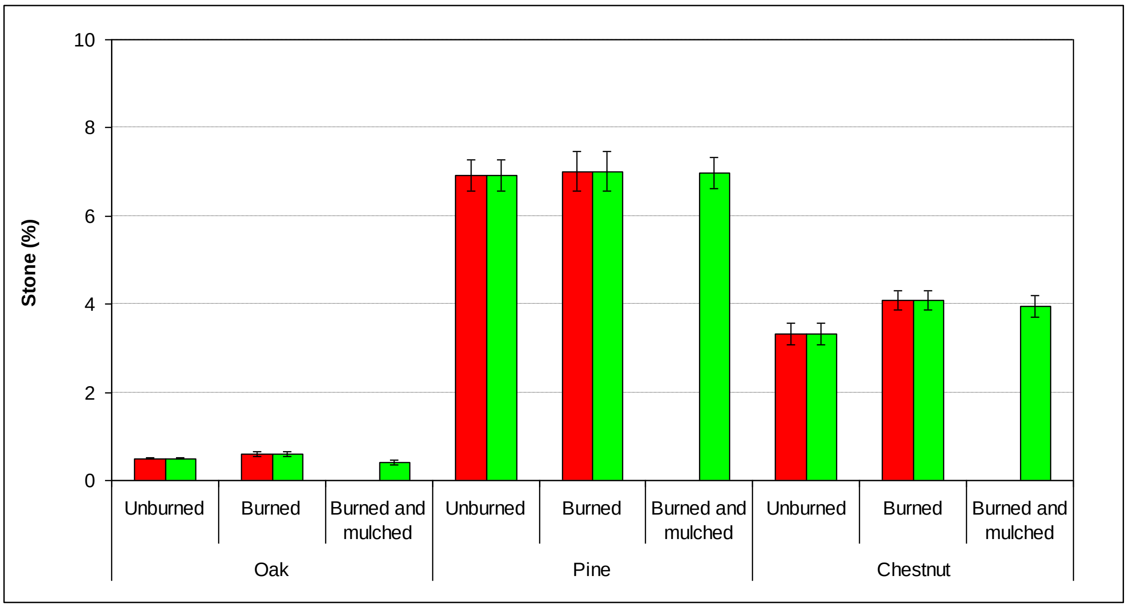


Figure 7 - Soil covers (mean ± standard deviation) after the prescribed fire and soil mulching with fern at two survey dates (one day and one year after the fire) in the experimental forest sites (Samo, Calabria, Southern Italy).

Note: no surveys were carried out in mulched soils, since soil cover with fern mulch was 100% and ash cover was the same as the burned soils.

4. Discussion

4.1. Changes in soil properties in burned soils

In this study, the soil pH was affected by burning immediately after the fire, although these changes were significant for two forest stands (pine and chestnut). One year after burning, these effects were significant in oak and chestnut soils. Although the individual processes that could explain these variations were not directly measured, the increase in pH could be due to consumption of organic acids during the oxidation of litter (Binkley and Fisher, 2019; Cawson et al., 2012; Úbeda et al., 2005) and complete oxidation of organic matter (Bodí et al., 2014; Hueso-González et al., 2018; Mataix-Solera et al., 2009). Other authors ascribe the pH increase in burned soils to the incorporation of ash into the soil (Alcañiz et al., 2020; Pereira et al., 2016; Sherman et al., 2005), which usually releases carbonates, base cations and oxides in the ash cations (Certini, 2005; Ekinci, 2006; Fonseca et al., 2017), but this may be excluded in our study, since the ash cover in chestnut and pine soils (where the pH increases were significant) was higher compared to oak sites (where this increase was less noticeable).

The increases in pH detected in our study are in close accordance with the majority of studies of soil changes after low-intensity fires (Badía et al., 2017; Kennard and Gholz, 2001; Scharenbroch et al., 2012). However, some other studies showed that pH values can remain unaltered, due to the low intensity of prescribed fires (Badía et al., 2017; Hueso-González et al., 2018; Marcos et al., 2009), except when fire was periodically repeated (Alcañiz et al., 2018). In general, the literature states that prescribed fire does not have a long-term effect on the soil pH, since this parameter depends on the time that ashes remain in the soil (Fonseca et al., 2017; Mataix-Solera and Guerrero, 2007).

Also, EC significantly increased after burning, and this effect lasted for at least one year. EC rises in fire-affected soils (also in the case of low-intensity fire) due to the incorporation of ash (Fonseca et al., 2017; Scharenbroch et al., 2012; Úbeda and Outeiro, 2009), release of soluble ions during the combustion of organic matter (Alcañiz et al., 2018; Certini, 2005), and formation of black C (Alcañiz et al., 2020; Certini, 2005). The increase in EC detected in this study may be surprising, since this effect is not reflected in the ion content. It is likely that this increase may be due to other ions that were not measured in this study (e.g., carbonates, sodium). An indirect confirmation of this effect may be the lack of significance of correlations between EC and concentrations of ions measured in this study.

The significant increases in EC detected in this study agrees with the results of many other authors (Alcañiz et al., 2020; Granged et al., 2011; Scharenbroch et al., 2012). However, the literature is not in agreement about the duration of these effects, since some studies report that EC is ephemerally increased by fire (Cawson et al., 2012; Hernández et al., 1997; Naidu and Srivasuki, 1994), while other authors stated that one year after the prescribed fire EC values decrease, but never below their pre-fire levels (Alcañiz et al., 2020).

Immediately after the prescribed fire, the burned soils showed significantly higher OC contents compared to the unburned sites. These significant variations were present one year in pine forest, while, in burned soils of oak and chestnut, C contents significantly decreased. After a fire it is normal to find increased soil OC (Alcañiz et al., 2018), and this is presumably due to the incorporation of unburned or partially unburned slash fragments into the soil or to the incomplete combustion of the organic matter (Alcañiz et al., 2020; Soto and Diaz-Fierros, 1993; Úbeda et al., 2005). Our results agree with (Alcañiz et al., 2016), (Binkley et al., 1992) and (Armas-Herrera et al., 2016), who reported increased soil C immediately after a prescribed fire, and decreases in this soil property one year after the fire compared to the unburned soils (Alcañiz et al., 2018; Hueso-González et al., 2018). However, we should bear in mind that the dynamic of OC content in soil is highly variable also after a fire of low intensity, since this variability depends on various factors, such as fire characteristics, ecosystem type, and land topography (Alcañiz et al., 2018).

The same variations among the analysed soil conditions (unburned vs. burned soils) and survey dates, detected for OC was observed for N content of soils. Increase in N was recorded immediately after the fire in burned soils, followed by significant reductions one year after. The similar trends detected for OC and N contents are again in close accordance with (Alcañiz et al., 2020), who found increases in N in soils subject to burning at low intensity, due to ash incorporation and forest floor decomposition (Alcañiz et al., 2020; Girona-García et al., 2018; Scharenbroch et al., 2012). The low temperatures of prescribed fire facilitates the incorporation of N that is present in large amounts in ash (Khouri and Prendes, 2006; Úbeda et al., 2005), and in the forest floor, which decomposing releases a substantial amount of N (Certini, 2005; Schoch and Binkley, 1986). The decrease in N one year after the fire agrees with the results of (Muqaddas et al., 2015) and (Blankenship and Arthur, 1999), who reported a marked loss of total N two years after a fire (Alcañiz et al., 2018). In any case, the N variability among soil conditions and time supports the suggestion of (Dimitrakopoulos and Martin, 1994) and (Úbeda et al., 2005) that also fires with low intensity are able to produce noticeable changes in N concentrations. Another reason for the N variability may be the sensitivity of N to fire, due to its multiple forms in the soil, since N can be volatilised, added and rapidly mobilised by plants (Úbeda et al., 2005).

The increases in the other soil properties immediately after the fire, such as Mg^{2+}, Ca^{2+}, K, NO_3^-, PO_4^{3-}, showed the key role of ash due to burning (Pereira et al., 2018a), which likely released these ions, increasing their content in burned soils (Cawson et al., 2012). This result agrees with (Alcañiz et al., 2020), who measured significant increases in Mg^{2+}, Ca^{2+} and K^+ contents, followed by a recovery of original pre-fire levels of Mg^{2+} and Ca^{2+}, but not K^+. In contrast, (Hueso-González et al., 2018), (Arévalo et al., 2007) and (Brye, 2006) reported non-significant increases in many ions after a low intensity burning.

Vegetation and litter burning caused noticeable increases in P content only in pine and chestnut soils, although this enrichment declined in the short term, in close accordance with (Cawson et al., 2012). However, the effects of fire are often ephemeral, since the recovery of pre-fire concentrations is achieved some months after the fire, such as, for instance, for K^+ in chestnut, Ca^{2+} and NO_3^- in oak and chestnut, Mg^{2+} in all forest soils). In accordance with our results, (Khouri and Prendes, 2006) reported that K^+ and P can return to pre-fire values after one to three months. In general, the trends detected in this study agree with several studies (Kennard and Gholz, 2001; Shakesby et al., 2015; Switzer et al., 2012).

When the pre-fire values do not recover one year after the fire (e.g., K^+ and PO_4^{3-} in oak, Ca^{2+} and SO_4^- in pine), the soil changes due to fire can last years (Alcañiz et al., 2016; Lavoie et al., 2010; Scharenbroch et al., 2012). This recovery may be the result of leaching processes due to runoff (Úbeda et al., 2005; Úbeda and Sala, 2001), erosion (Fonseca et al., 2017, 2011), and plant consumption (Alcañiz et al., 2018). Moreover, this variability over time is not always due to the effect of fire, as shown by the variations detected for some properties also in the unburned soil (e.g., Mg^{2+} and Ca^{2+} in pine and chestnut, NO_3^- in chestnut, PO_4^{3-} in pine). Overall, the soil changes measured in this study strictly depend upon the analyzed property and tree species (Cawson et al., 2012; Kutiel and Shaviv, 1992).

4.2. Changes in soil properties in mulched soils

Immediately after the fire, the burned soils of pine and chestnut treated with mulching showed significant reductions in pH, which did not recover after one year in comparison to the unburned sites. In all forest soils, the OC and N contents were significantly higher one day after the fire, and these variations were observed also after one year in pine forest, while, in soils of oak and chestnut, OC and N significantly decreased. EC also increased throughout the observation period. Therefore, mulching was not effective in limiting the variations in these soil properties.

The lower OC content detected in this study in the mulched soils of oak and chestnut may appear surprising. Presumably, the incorporation of the dead material supplied with mulching was not sufficient to balance the decreases in OC detected one year after the fire in burned soils; it not could be excluded that part of the organic matter applied with fern (presumably the soluble form) was leached by the infiltrating precipitation, which was higher in mulched areas compared to the burned sites (Carrà et al., 2021). However, according to (Cawson et al., 2012), the recovery of soil organic matter after the fire is generally fast, because it starts with the natural vegetation or its artificial reintroduction, thanks to the high productivity of secondary ecological succession. The higher shrub cover measured in this study for chestnut soils may also be an indication of higher plant uptake of OC and N due to regeneration, although this explanation should be confirmed by specific ecological surveys.

The higher water infiltration in all forest sites, and plant uptake measured in the chestnut forest may again explain the reduced content of N in mulched soils. In general, variations in total N availability seem to be associated with herbaceous vegetation growth (De Lillis, M, 1993; Marcos et al., 1999), and the rapid drop in this soil property in the post-fire recovery period could be attributed to leaching, microbial immobilization and plant uptake (Antos, 2003) as well as volatilisation (Binkley and Fisher, 2019; Romeo et al., 2020; Úbeda et al., 2005).

It is interesting to note that the increase or decrease in any property was not influenced by mulching; thus, this treatment did not affect the properties of burned soils. This deserves more investigation, since, when vegetation residues are used as mulch material, organic matter content can increase (García-Orenes et al., 2012; Jordán et al., 2010; Prosdocimi et al., 2016). Presumably, the time elapsed from mulch application was too short in order to complete incorporation of vegetal residues into soil or part of the mulch cover was displaced by water, hampering soil incorporation (Bombino et al., 2021a).

4.3. Relationships among the soil properties

At both survey dates, among all the soil properties investigated in this study, PCA provided some key parameters, which clearly modify soil characteristics among the different conditions and tree species studied. At both dates, EC and ion concentrations exert a noticeable influence on the first PC, and furthermore OC and N contents together influenced the second PC. Therefore, it is the role of ash that determines changes in EC and ion concentrations, and burning and mulching that cause changes in OC and nutrients. This holds true for different tree species and variable conditions. It is interesting to note that, immediately after the fire, burning determined significant differences in soils of all forest species (grouped in the same cluster, while the mulched soils were differently clustered). After one year, the disturbance factors (burning and mulching) altered the similarity among all the soil conditions and forest species, and no evident clusters of soils with the same condition can be noted, thus proving the transient effects of prescribed burning and soil mulching on soils.

5. Conclusions

The study showed that prescribed fire and post-fire management with fern mulching had noticeable impacts on the soil of three Mediterranean forest species. The changes in the soil's chemical properties were often significant, but the effects of the prescribed fire and mulching were often transient, since the changes depend on the ash production and incorporation, seasonality and forest species. Significant increases in EC, total C and N content, and ion concentrations were detected in burned soils (mulched or not) immediately after the fire, while soil pH was less affected by these disturbance factors. For some properties the increases were ephemeral with reductions detected one year after the fire (e.g., OC and N), while, for other parameters (e.g., EC, K^+ and NH_4^+), the pre-fire values were not restored. It is important to highlight that, in general, mulching was not effective in limiting the changes in the monitored soil properties compared to the pre-fire values. Each forest species showed different temporal trends in changes of soil properties. This was confirmed by the combined Principal Component Analysis and Agglomerative Hierarchical Cluster Analysis, which clearly differentiated the diverse soil conditions and forest species in homogenous clusters.

Overall, the study confirmed that prescribed fire, although being a low-intensity fire, is able to induce significant changes in soil properties. These changes depend on both soil properties and forest species considered. Mulching with fern is unable to limit the changes in chemical properties of soils. Future research paths should validate the findings of this study, which are specific to the semi-arid Mediterranean climate. Forests and soils in different environments (due to weather, pedology, ecology) may produce other trends in soil changes due to both prescribed fire and mulching. Moreover, a specific hydrological study on its effectiveness on runoff and erosion on forest hillslope may justify its use to mitigate the hydrogeological risks in wildfire-affected areas, exploiting its economic convenience and naturalness compared to other mulch materials. Extending the analyses to the microbiological properties of soils (e.g., enzyme contents, microbial respiration) may clarify other effects, which were not studied in this investigation. Finally, the general aim of this study was the identification of the indirect effects of burning and post-fire treatment on forest soils regardless of the direct cause. A specific and deeper analysis of individual or homogenous types of soil properties (e.g., nutrients, ions, covers) may be beneficial to fully understand the processes that have determined the observed changes. This may also resolve any apparent disagreement in the literature.

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CHAPTER 4

Short-term effects of prescribed fire and soil mulching with fern on natural regeneration of *Quercus Frainetto* L.

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Key message

In Mediterranean oak stands prescribed burning increases acorn emergence and plant survival, while post-fire soil mulching with fern does not significantly enhance the initial recruitment of plants.

Abstract

To avoid the negative impacts of wildfire, prescribed fire is applied in several environments, often with post-fire soil mulching, to control wildfire hazard and erosion in burned areas. However, uncertainties remain about impacts of these forest management techniques on post-fire regeneration, especially for some forest species, such as oak, which is predominant in Mediterranean fire-prone areas. This study evaluates the effects of prescribed fire and post-fire soil mulching with fern on initial recruitment of an oak forest of Southern Italy. Acorn emergence and seedling survival as well as some important plant and root biometric characteristics (height, diameter, and dry weight) have been monitored in plots burned by prescribed fire with or without post-fire treatment with fern. The acorn rain among the experimental conditions was not statistically different, whereas the prescribed fire significantly increased acorn emergence compared to the unburned area. About 30% of acorns germinated, of which more than 70% survived. Soil mulching with fern did not significantly increase acorn emergence (35%) and plant survival (85%) compared to the burned and untreated sites, presumably due to the shadowing effect of the cut fern, which reduces the light availability for juvenile plants. However, this post-fire treatment significantly enhanced plant height (+44%) and root mass (+30%) but not its diameter (+6%) and root length (+1%). These contrasting effects in part support the initial hypothesis that soil mulching may be synergistic with the prescribed fire. Overall, the knowledge of the beneficial influences of prescribed fire and post-fire treatments on oak recruitment, thanks to the fire-tolerance character of this forest species, is useful to develop sustainable management plans for the delicate forest ecosystems of the semi-arid Mediterranean environment.

Keywords: initial recruitment; oak stands; burning; acorn rain; acorn emergence, seedling survival.

1. Introduction

A detailed understanding of processes driving the dynamics of forest ecosystems is essential to develop sustainable management plans. This understanding must include the process of natural regeneration, since the initial recruitment of forest species is fundamental to achieve stand persistence under the pressure of future climate change (Manuel E. Lucas-Borja et al., 2012), which forecasts reduced lower precipitation and higher temperatures (Collins et al., 2013). Acorn emergence and seedling survival stages, which determine the future forest structure, have been recognised as the most limiting stages for natural regeneration (Calama et al., 2017). In early stages of forest trees, abiotic factors, such as soil moisture and temperature, and sunlight, primarily control seedling establishment and growth (Lucas-Borja et al., 2011). Some biotic factors, such as canopy cover, acorn and seedling predation, and/or sapling herbivore, may also influence these stages (Manuel E. Lucas-Borja et al., 2012).

Among these factors, fire may favour or hinder early seedling recruitment, since several forest species depend on the impact of fire to germinate (Keeley and Pausas, 2019). However, fire, when at high severity, such as wildfires, may pose pervasive effects on human goods and assets (Wittenberg and Pereira, 2021). To avoid the negative impacts of wildfire, prescribed fires - the planned use of low-intensity fire to achieve very different goals given certain weather, fuel and topographic conditions (Alcañiz et al., 2018; Fernandes et al., 2013) - has been suggested and applied in several environments (Klimas et al., 2020), for management of fuel accumulation and future fire hazard in forests that are prone to the wildfire risk (Fernandes and Botelho, 2003). On an ecological approach, prescribed fire is thought to change the structure and composition of the forest (Romeo et al., 2020), increasing the heterogeneity of the landscape, and can be purposely used for restoration of fire-associated ecosystems (Ryan et al., 2013), promoting or regenerating fire-adapted species (Van Lear and Waldrop, 1991). Its controlled and rational use can improve forest habitats (Fontaine and Kennedy, 2012), facilitating the emergence and growth of understory vegetation (Hunt et al., 2014; Neary et al., 1999), discouraging establishment of some plant species, while favouring some others (Barnes and Van Lear, 1998; Galford, 1989). Post-fire recovery is generally accomplished by direct regeneration, i.e., the recovery of a plant community composed of the same pool of species that existed before the fire (Romeo et al., 2020).

Despite these beneficial effects of prescribed fires, uncertainties remain about its ecological impacts (Fuentes et al., 2018). The forest species may perform differently after prescribed fires (Lucas-Borja et al., 2016), and even the results of relevant research on the fire effect on tree regeneration are often contrasting. For some forest species, such as oak, which is predominant in Mediterranean fire-prone areas and has adapted to fire (Curt et al., 2009), studies of fire effects on regeneration have been shown varying results (Petersson et al., 2020). Oak woodlands rely on the ability of this species to survive fire, resprout efficiently after fire and regenerate from seeds (Arthur et al., 2012; Pausas, 2006). However, failure of oak regeneration from acorns is frequent in Mediterranean countries (Curt et al., 2009; Maltez-Mouro et al., 2007). Oak is fire-adapted and moderately shade-intolerant, and therefore prescribed fire can be used as a management tool to decrease competition and increase light levels, thus promoting oak regeneration in some ecosystems (Brose et al., 2013; Izbicki et al., 2020). Prescribed fire also eliminates excess fire-sensitive vegetation and reduces litter, favouring the emergence of acorns (Blankenship and Arthur, 2006). Many studies have shown that fire alone or in combination with other treatments, such as partial canopy removal, can improve the establishment and growth of regenerating oak trees (Hutchinson et al., 2005). However, several oak species have exhibited unsuccessful regeneration in recent years, and it is difficult to identify the reasons, due to a multitude of biotic and abiotic factors influencing this ecological processes (Curt et al., 2009; Pausas et al., 2004; Royse et al., 2010). In this context, the role of how prescribed burning affects initial seedling recruitment is still not understood and more research is needed with particular focus for Mediterranean mountainous areas, where the fire hazard is high and the post-fire recruitment of vegetation is limited by drought and some adverse soil characteristics (Moody et al., 2013; Shakesby, 2011).

In spite of these beneficial effects on forest ecology, prescribed fire shows negative impacts on burned ecosystems, especially on soil hydrology. More specifically, removing litter and understory vegetation and modifying some important hydrological properties of soil (such as reducing the hydraulic conductivity of soil and inducing water repellency (Lucas-Borja et al., 2018; Plaza-Álvarez et al., 2019, 2018; Zavala et al., 2014), prescribed fire can increase the runoff and erosion rates, also by some order of magnitude (Cawson et al., 2012; González-Pelayo et al., 2010; Vega et al., 2005). Soil mulching with vegetal material (commonly pruning residues or straw) may mitigate runoff and erosion in burned areas (Lucas-Borja, 2021; Zema, 2021), since this post-fire management technique protects soil from raindrop impacts and reduces the velocity of overland flow (Patil Shirish et al., 2013; Prosdocimi et al., 2016; Zituni et al., 2019). This mulch action may also increase moisture and reduce temperature of burned soils, which are key factors in initial seedling recruitment of forest species (Calama et al., 2017). However, post-fire mulching can also have negative effects, especially regarding the mulching material. For instance, the residues of straw can be displaced by wind in some areas and may contain seeds, chemicals and parasites, which can be the sources of non-native vegetation and plant diseases. On this regard, forest residues (e.g. wood strands, chips or shreds) or dead plants may be preferable to straw, because these substrates do not carry non-native seeds or chemical residues, and are more resistant to wind displacement (Robichaud et al., 2020). In Mediterranean forest floor, fern - *Pteridium aquilinum* (L.) Kuhn - is widely available, and this avoids transport from other locations. On an ecological approach, fern stands act as an ecological filter that influences tree regeneration and favours emergence of late-successional species (Ssali et al., 2019).

However, to the best authors' knowledge, very few evaluations about the effect of fern on early recruitment of oak species in burned soil are available in literature. It results that the question of how prescribed fire and post-fire mulching affects the early seedling recruitment of an important forest species, as oak, is still not understood in Mediterranean forest ecosystems. To fill these gaps, this study evaluates the effects of prescribed fire and post-fire soil mulching with fern on initial recruitment of an oak forest of Southern Italy. More specifically, acorn emergence and seedling survival as well as some important plant and root biometric characteristics (height, diameter, and dry weight) have been monitored in plots burned by prescribed fire with or without post-fire treatment with fern. Since oak is a forest species that is adapted to fire, we hypothesised that prescribed burning enhances the initial recruitment of plants, and this effect may be synergistic with mulching application.

2. Material and Methods

2.1. Study site

The study was carried out in a natural stand of oak (*Quercus frainetto* Ten.) in the locality of "Rungia", UTM coordinates 588635 E and 4216172 N, municipality of Samo, Calabria, Southern Italy). The climate of the area is semi-arid ("Csa" class, "Hot-summer Mediterranean" according to Koppen (Kottek et al., 2006) with mild and wet winters, and warm and dry summers. The mean annual precipitation and temperature are 1100 mm and 17.4 °C, respectively.

The oak stand under investigation is between 900 and 950 m above sea level and is exposed to North-East (Figure 1). The stand is an adult forest with a large contribution of natural mulching from the fall of the leaves and fern. No management actions have been accomplished in the three forest stands. Table 1 reports the main dimensional characteristics of the tree layer and the composition of the shrub layer.

The soil of the experimental site, with a mean slope of 19.1 ± 1.65%, are loamy sand with contents of silt, clay and sand of 11.5 ± 1.89%, 9.2 ± 0.58% and 79.1± 1.59%, respectively.

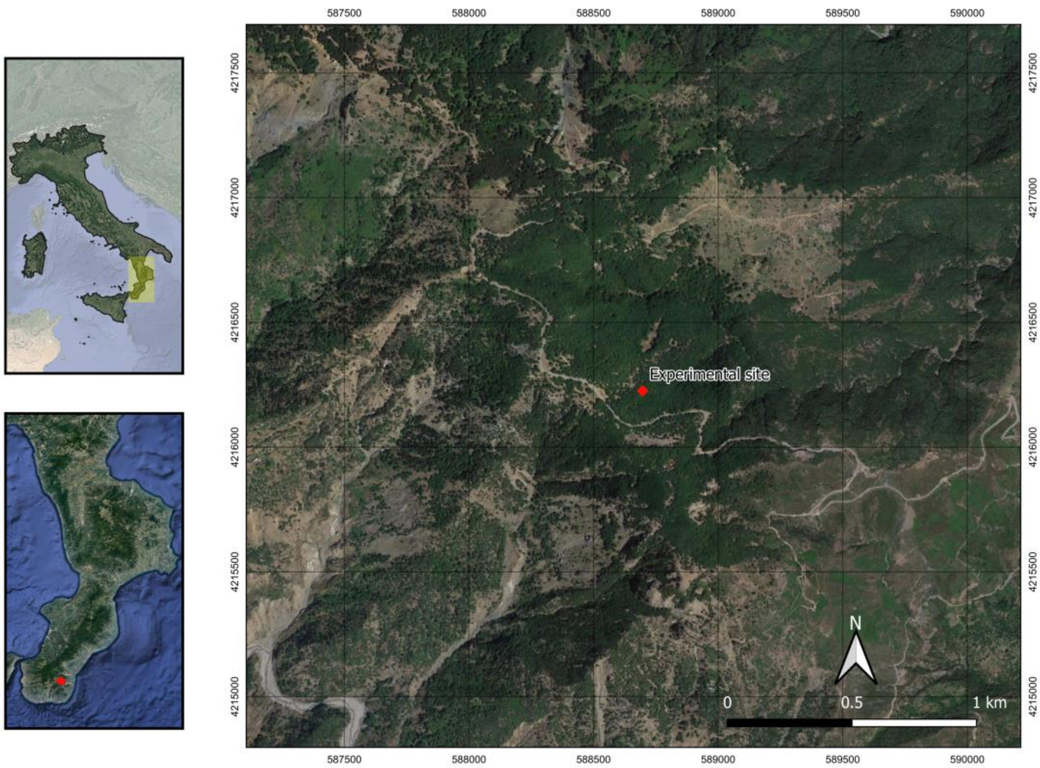

Figure 1 - Location of the experimental site (Samo, Calabria, Southern Italy).

Table 1 - Main characteristics of the oak forest stand in the experimental site (Rungia, municipality of Samo, Calabria, Southern Italy).

Oak forest characteristics		**Value**
Tree	density (n./ha)	225 ± 44.7
	diameter at the breast height (cm)	40.7 ± 8.9
	height (m)	18.2 ± 1.9
	basal area (m^2/ha)	31.1 ± 3.6
Litter	height (cm)	12.2 ± 3.9
Main shrub species		*Cyclamen hederifolium, Bellis perennis* L.

2.2. Experimental design

The prescribed fire was carried out in an area (about 250 m^2) of the oak forest in early June 2019 with the support of the Forest Regional Agency (Calabria Verde) and the surveillance of the National Corp of Firefighters. Wind was practically absent and air humidity was between 50 and 60%. The mean and maximum temperatures of soils, which were measured at a depth of 2.5 cm by a thermocouple connected to a datalogger, were 21 and 26.9 °C.

The burn severity of soils after the prescribed fire was evaluated according to the classification by (Parson et al., 2010). Accordingly, one day after prescribed fire, ground surface of all sites was visually checked, observing ash colour and fine roots. Following (Parson et al., 2010), the burn soil severity of all sites were low, that is, with small change from pre-fire status, black ground surface with recognizable fine fuels remaining on surface, and fine roots unchanged.

In the burned area, three small portions (about 9 m^2) was mulched with a cover (more or less 3 to 5 cm of thickness) of fern residues, cut from an adjacent zone, and distributed over ground at a dose of 500 g/m^2 of fresh weight (equivalent to 200 g/m^2 of straw dry matter, usually applied in burned and mulched areas after fire (Lucas-Borja et al., 2018; Vega et al., 2014).

A third area, which was not burned and was located less than 10 m from the burned areas, in each site was selected to be considered as "control".

Immediately after the prescribed fire and soil mulching, a total of nine small plots (three series of plots (area of 3 m^2), each series consisting of three replicated plots) were selected and delimited using 0.3-m high metallic sheets inserted up to 0.2 m below the ground surface. The plots were at a reciprocal distance between 1.5 and 2 m. Three plots were set up in the unburned soils (considered as “control”), while six plots were located in the burned area, of which three in the area without treatment (Figure 2a) and three in the mulched area (Figure 2b).

Overall, the experimental design consisted of three soil conditions (unburned, burned and untreated, and burned and mulched) × three replicated plots, for a total of nine plots.

(a) (b)

(c)

(d)

Figure 2 - Photos of the experimental plots (a, burned and untreated soils; b, burned and mulched soils), acorn emergence (c) and measurement of seedling height and diameter (d) in oak forest under different soil conditions (Samo, Calabria, Southern Italy).

2.3. Vegetation sampling and analysis

Acorn rain was estimated in November 2019 (five months after the prescribed fire application and soil treatment with fern mulch) in each plot of the three soil conditions. Seed fall was calculated at each experimental treatment as a sum of acorns found at each plot. The plots were protected with wire netting (1 × 1 cm mesh size) to avoid acorn predation.

Acorn emergence (Figure 2c) was measured one year (June 2020) after the prescribed fire application and soil treatment with fern mulch, while seedling survival was surveyed at the end of the experiment in June 2021 (24 months after fire and mulching).

Moreover, the total length of the main stem and root-collar diameter of all surviving seedlings was measured in June of 2020 and 2021 (Figure 2d). Then, in June 2021, the soil supporting seedling growth was gently excavated and the aerial part of the plant was separated from the root system. On the latter, the root length and dry weight (at a temperature of 60 °C for 48 hours) were determined.

2.4. Statistical analyses

One-way ANOVA was applied to the rain, acorn emergence and seeding survival as well as plant root length and dry weight, assuming as factor the soil condition (that is, unburned, burned and not treated, burned and mulched). Two-way ANOVA was instead applied to evaluate the statistical significance of the differences in plant height and diameter among soil conditions and survey dates (June 2020 and June 2021), the latter considered as independent factors.

The pairwise comparison by Tukey’s test (at $p < 0.05$) was also used to evaluate the statistical significance of the differences in the response variables. To satisfy the assumptions of the statistical tests (equality of variance and normal distribution), the data were subjected to normality test or were square root-transformed whenever necessary. All the statistical tests were carried out by with the XLSTAT software (release 2019).

3. Results

Compared to the unburned sites, where acorn rain was on average 181 ± 35, 218 ± 56 and 214 ± 17 acorns were counted in burned, and burned and mulched plots (Figure 3), although the differences among the different soil conditions were not significant ($p = 0.497$).

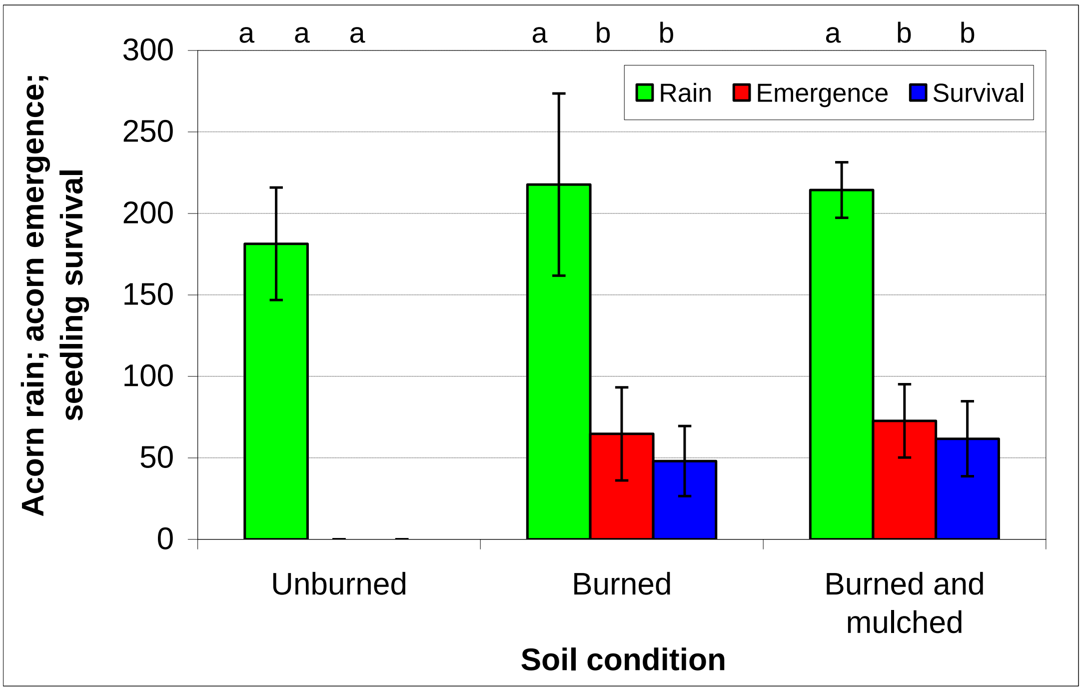

Figure 3 - Acorn rain and emergence, and seedling survival (mean ± standard deviation) counted in oak forest under different soil conditions (Samo, Calabria, Southern Italy). Different letters indicate significant differences ($p < 0.05$).

While no acorn emergence was noticed in unburned sites, 65 ± 29 acorns germinated on burned soils and 73 ± 23 on burned and mulched plots (Figure 3). Only the difference between unburned, and burned soils (mulched or not) was significant ($p < 0.01$).

Of the emerged acorns, 48 ± 22 and 62 ± 23 seedlings survived on burned, and burned and mulched soils (Figure 3), but the difference between this soil conditions was not significant ($p = 0.068$).

At both survey dates (June 2020 and 2021), plant height was higher in burned and mulched plots (92.8 ± 31.2 mm, June 2020, and 110.5 ± 40.3 mm, June 2021) compared to the burned and untreated soils (64.6 ± 35.3 mm, June 2020, and 85.3 ± 41.9 mm, June 2021) (Figure 4). According the 2-way ANOVA, both soil condition and survey date were significant factors ($p < 0.0001$ and 0.01, respectively) on plant height, but not its interaction ($p = 0.630$).

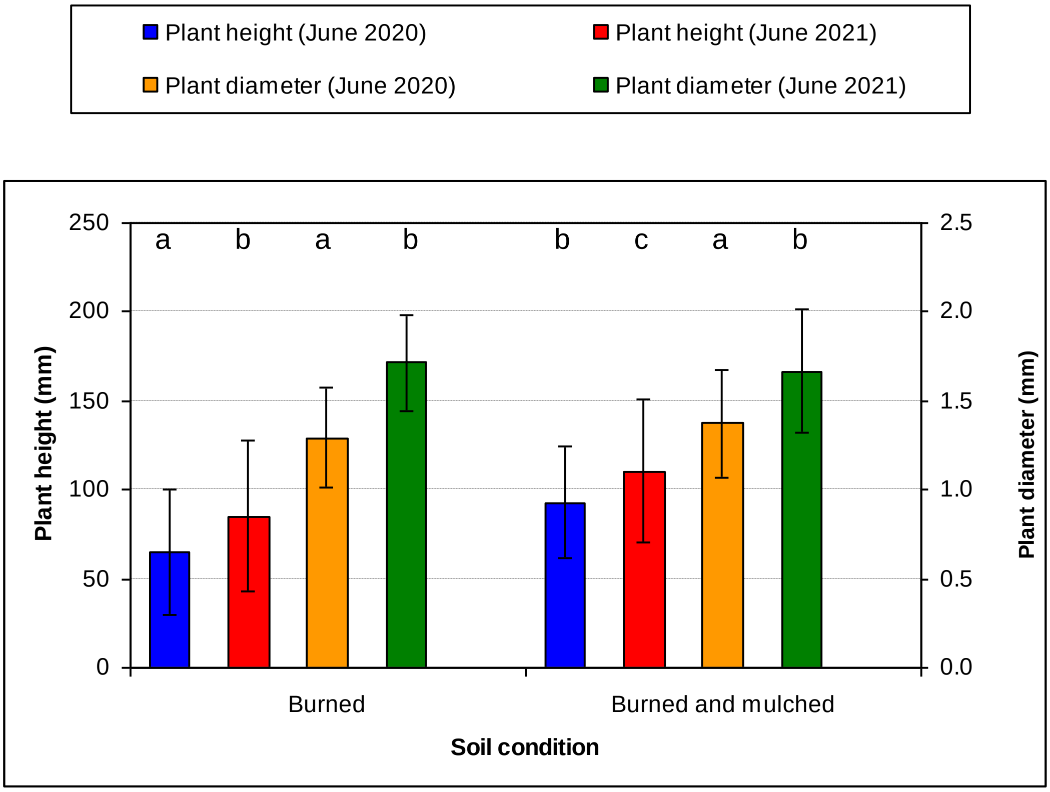

Figure 4 - Plant height and diameter (mean ± standard deviation) surveyed in oak forest under different soil conditions at two survey dates (Samo, Calabria, Southern Italy). Different letters indicate significant differences ($p < 0.05$).

In June 2020, seedling diameter was lower in the burned and untreated plots (1.29 0.28 mm) compared to the burned and mulched soils (1.38 ± 0.30 mm). In contrast, in June 2021, the latter soils showed seedlings with lower diameter (1.67 ± 0.35 mm vs. 1.71 ± 0.27 mm of burned and untreated plots) (Figure 4). For seedling characteristic, both the date and its interaction with the soil condition determined significant differences in plant diameter ($p <$ 0.0001 and 0.01, respectively. In contrast, soil condition did not statistically influence the seedling diameter ($p = 0.485$).

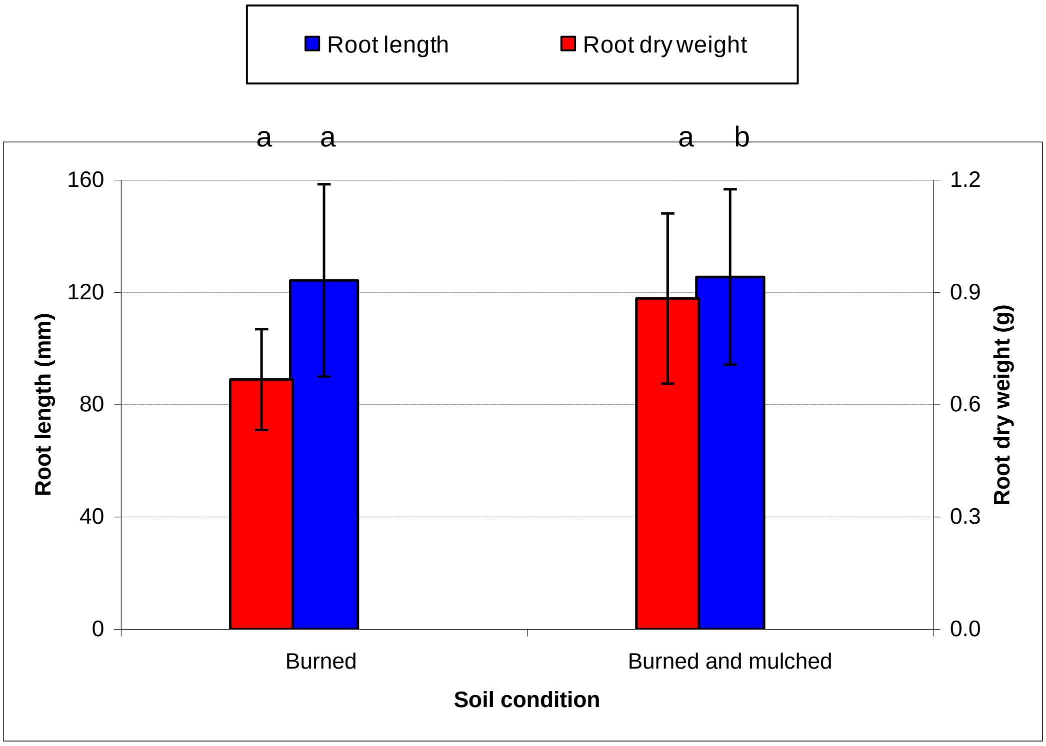

Figure 5 - Root length and dry weight (mean ± standard deviation) of oak seedlings under different soil conditions at two survey dates (Samo, Calabria, Southern Italy). Different letters indicate significant differences ($p < 0.05$).

Root length (125 ± 34.3 mm and 126 ± 34.0 mm, respectively) was practically the same in burned, and burned and mulched soils. In contrast, the latter soils showed a higher root dry weight (0.88 ± 0.23 g) compared to the burned and untreated plots (0.67 ± 0.14 g) (Figure 5). These differences were significant for root dry weight ($p < 0.001$), but not for root length ($p = 0.905$).

4. Discussions

Since seedling recruitment is ultimately limited by the quantity of viable acorn fall, an evaluation of seedling patterns is necessary to understand the dynamics of recruitment (Lucas-Borja and Vacchiano, 2018). Oak acorns are dispersed across a wide variety of sites covering a range of abiotic and biotic conditions. Given that a similar forest structure of oak forest among all plots, we assumed that acorns fall was similar at all plots. Furthermore, since parent rock material, topography, climate and forest structure were similar among selected plots, differences with respect to seedling emergence, survival and initial growth can be associated with the influence of fire and mulch.

While none of the fallen acorns sprouted in the unburned area, about 30% of acorns germinated in the burned areas, of which more than 70% survived. Higher regeneration in burned areas is in agreement with other studies dealing with oak recruitment, which reported higher survival rates (Brose et al., 2013; Petersson et al., 2020; Royse et al., 2010), the latter in combination with other treatments (e.g., tree canopy openness or ungulate exclosure) in burned oak stands, while (Hutchinson et al., 2005) found reduced sapling density in areas with burn treatments. Many reasons may explain the greater early recruitment of oak in the burned sites. First, the reduction in tree density and canopy cover increases light availability on the forest floor, which is very important for regeneration of oak species, which are considered light demanding for survival and growth once energy reserves of the cotyledons are exhausted (Annighöfer et al., 2015; Petersson et al., 2020). Second, the lower competition with herbaceous vegetation presumably enhances vegetation growth in this semi-arid environment, having a limited water availability (Garcia-Fayos et al., 2020; Petersson et al., 2020); in contrast, the presence of herb layer found in unburned plots could have delayed or impeded acorn emergence development (Caccia and Ballaré, 1998). Third, low-intensity fire creates more favourable seedbed conditions to oak establishment, promoting the contact with mineral soil and thus favouring acorn emergence (Facelli and Pickett, 1991; Hutchinson et al., 2005). Fourth, the increases in organic carbon, total nitrogen and available phosphorous measured in the same experimental site immediately after the prescribed fire compared to the unburned sites, followed by significant decreases one year after fire, when the plant regeneration was higher (Carrà et al., in press). A higher availability of organic matter and nitrogen contents of burned soils, which have been well documented after prescribed fires OC (Alcañiz et al., 2020, 2018; Hueso-González et al., 2018; Úbeda et al., 2005), may have supported the recruitment of oak seedlings detected in this study. These changes in soil properties also improve microbial diversity, which may increase soil fertility (Manuel Esteban Lucas-Borja et al., 2012; Nannipieri et al., 2003). The latter effect together with the low damage to canopy acorn bank and the low impact on forest floor conditions after prescribed fires might generate better conditions for initial seedling recruitment. In line with this, (Madrigal et al., 2010) have also shown a positive influence of the soil organic layer remaining after fire on seedling recruitment, which has been related to a higher water content of soils with higher organic matter. However, since several studies have indicated that oak regeneration may be more effective after a single prescribed fire when combined with other treatments (Izbicki et al., 2020; Petersson et al., 2020; Ssali et al., 2019), the opening of tree canopy by mechanized operations and replicated applications of prescribed fire can be suggested, in order to remove midstory competition, increasing understory light, and increasing oak seedling growth and density (Brose et al., 2013; Green et al., 2010; Izbicki et al., 2020).

Mulching determined increases - although not being significant - in these percentages (about 35% of germinated acorns, and 85% of survived seedlings) that can be quantified in 14% and 12% for emergence and survival, respectively. (Lucas-Borja et al., 2021) have demonstrated that seedling survival but not emergence rates is largely controlled by post-fire treatments enhancing the establishment of vegetation cover, due to the beneficial effect of shrubs on the recruitment of seedlings located under their canopies (Emborg, 1998; Heydari et al., 2017). Moreover, the mulch cover spread over the ground of burned soils acts as a barrier that reduces drought by lowering solar radiation and soil temperature and increasing its water content (Castro et al., 2003; Lucas-Borja et al., 2021). Also (Ssali et al., 2019) reported higher seedling survival on soils of tropical forests subjected to post-fire treatment with fern.

Many studies have stated that growth of oak seedlings in burned areas is higher compared to fire-excluded sites, since larger diameters (Royse et al., 2010), faster biomass growth (Wang et al., 2005), and greater heights (Petersson et al., 2020) were observed in oak seedlings after prescribed fires.

The lack of significance of the differences in acorn emergence and seedling survival detected in mulched areas compared to the burned and untreated sites may be due to the shadowing effect of the cut fern, which reduces the light availability for juvenile plants. It is well documented in literature that light can significantly impact seedling and sapling survival (e.g., (Hutchinson et al., 2005; Izbicki et al., 2020; Royse et al., 2010).

In spite of the non-significant difference in the initial recruitment of oak seedlings between mulched and untreated areas, this post-fire treatment enhanced plant growth at both survey dates, as shown by the significantly greater height (+44% one year after the treatment) compared to the plants grown on burned and untreated plots. In contrast, the plant diameter was not significantly influenced by the treatment (only +6% after one year). These enhancements of the plant morphology lost weight over time (+30% in height two years after the treatment) and even became negative (-3% in diameter) over time. Higher growth in soils burned and mulched with fern was reported also by (Ssali et al., 2019).

Regarding the root system, mulching did not significantly influence the length (only +1% in mulched areas compared to the burned and untreated plots), but determined a significant increase (by more than 30%) of the root mass.

The faster growth of seedlings in mulched areas compared to the untreated sites may be due to the higher water content, since it is well known that mulching enhances soil moisture which is a key point in natural regeneration process, especially in forest ecosystems of the semi-arid environments with limited water availability (Lucas-Borja et al., 2018; Prosdocimi et al., 2016). (Bautista et al., 2009) theorized that the main advantage of mulch application is the immediate increase in vegetal cover of soil, which result in an effective protection during the first rain events after fire. Moreover, the mulch material can be fast incorporated into the soil, thus increasing the its content of organic matter and nutrients (Bombino et al., 2019).

5. Conclusions

This study has demonstrated that the application of prescribed burning to an oak stand of the Mediterranean environment play significant effects on plant regeneration compared to the unburned sites. In relation to emergence, in burned areas, about 30% of acorns emerged, of which more than 70% survived. This may be due to the higher light availability on the forest floor, lower competition with herbaceous vegetation, more favourable seedbed conditions to oak establishment, and increases in soil contents of organic carbon, total nitrogen and available phosphorous. This result confirms the first working hypothesis that prescribed burning enhances the initial recruitment of plants, since oak is a forest species that is adapted to fire.

Soil mulching with fern applied as anti-erosive post-fire treatment did not significantly increase acorn emergence and plant survival compared to the burned and untreated sites, presumably due to the shadowing effect of the cut fern, which reduces the light availability for juvenile plants. However, this post-fire treatment significantly enhanced plant height and root mass but not its diameter and root length. These contrasting effects, which may be attributed to the higher water content of mulched area of this water-limited in forest ecosystem of the semi-arid environments, in part support the initial hypothesis that soil mulching may be synergistic with the prescribed fire.

Overall, the knowledge of the beneficial influences of prescribed fire and post-fire treatments on oak recruitment, thanks to the fire-tolerance character of this forest species, is useful to develop sustainable management plans for the delicate forest ecosystems of the semi-arid Mediterranean environment. More research is needed in order to validate the outcomes of this study on different forest species and to evaluate the effects of prescribed fire and mulching with other forest operations.

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Author contributions

Conceptualization, B.G.C., G.B., M.E.L.B. and D.A.Z.; methodology, B.G.C., A.L., P.A.P.A., M.E.L.B. and D.A.Z.; validation, G.B., M.E.L.B. and D.A.Z.; formal analysis, G.B., M.E.L.B. and D.A.Z.; investigation, B.G.C., A.L., and P.A.P.A.; data curation, B.G.C., A.L., P.A.P.A., and D.A.Z.; writing - original draft preparation, B.G.C., and D.A.Z.; writing - review and editing, B.G.C., G.B., M.E.L.B and D.A.Z.; supervision, M.E.L.B and D.A.Z.; project administration, D.A.Z.; funding acquisition, D.A.Z. All authors have read and agreed to the published version of the manuscript.

Conflicts of interest statement

The authors declare that they have no conflict of interest.

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CHAPTER 5

Modelling the Event-Based Hydrological Response of Mediterranean Forests to Prescribed Fire and Soil Mulching with Fern Using the Curve Number, Horton and USLE-Family (Universal Soil Loss Equation) Models

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Abstract: The SCS-CN, Horton, and USLE-family models are widely used to predict and control runoff and erosion in forest ecosystems. However, in the literature there is no evidence of their use in Mediterranean forests subjected to prescribed fire and soil mulching. To fill this gap, this study evaluates the prediction capability for runoff and soil loss of the SCS-CN, Horton, MUSLE, and USLE-M models in three forests (pine, chestnut, and oak) in Southern Italy. The investigation was carried out at plot and event scales throughout one year, after a prescribed fire and post-fire soil mulching with fern. The SCS-CN and USLE-M models were accurate in predicting runoff volume and soil loss, respectively. In contrast, poor predictions of the modelled hydrological variables were provided by the models in unburned plots, and by the Horton and MUSLE models for all soil conditions. This inaccuracy may have been due to the fact that the runoff and erosion generation mechanisms were saturation-excess and rainsplash, while the Horton and MUSLE models better simulate infiltration-excess and overland flow processes, respectively. For the SCS-CN and USLE-M models, calibration was needed to obtain accurate predictions of surface runoff and soil loss; furthermore, different CNs and C factors must be input throughout the year to simulate the variability of the hydrological response of soil after fire. After calibration, two sets of CNs and C-factor values were suggested for applications of the SCS-CN and USLE-M models, after prescribed fire and fern mulching in Mediterranean forests. Once validated in a wider range of environmental contexts, these models may support land managers in controlling the hydrology of Mediterranean forests that are prone to wildfire risks.

Keywords: runoff; erosion; soil loss; hydrological modeling; oak; chestnut; pine; calibration

1. Introduction

Wildfire is one of the most dangerous threats to forest ecosystems, since it impacts almost all components (air, soil, plants, fauna, surface water (Kozlowski, 2012)). Fire alters the soil properties and removes vegetation, leaving the soil bare and, thus, exposed to flooding and erosion (Moody et al., 2013; Zavala et al., 2014; Zema, 2021). The hydrological impacts of fire are particularly important in Mediterranean forests. Here, the frequency, extent, and intensity of wildfires have been associated with an increase in climate warming in the last three decades (Pausas and Fernández-Muñoz, 2012), due to specific weather conditions (e.g., low humidity, high temperature, and strong winds (Morán-Ordóñez et al., 2020)), and hydrological regimes (extreme and flash storm events with heavy and erosive rainfalls (Diodato and Bellocchi, 2010; Giorgi and Lionello, 2008)).

To control fire severity and frequency, and at the same time mitigate the hydrological impacts in fire-affected areas, several pre- and post-fire management actions have been proposed. Prescribed fire applications, as well as soil mulching, have been used in several environments with satisfactory results (Lucas-Borja, 2021; Neary and Leonard, 2021). Many studies have demonstrated how low-intensity fires are effective for removing the fuel that can generate a high-intensity fire, and therefore for reducing the wildfire risk in treated forests (Alcañiz et al., 2018; Vega et al., 2005). Soil mulching with vegetation residues is one of the most common post-fire management strategies to limit runoff and erosion in the short term, since mulch protects soil from rainsplash prior to vegetation regrowth (Kozlowski, 2012; Prosdocimi et al., 2016).

The hydrological processes in burned soils are very complex, since several factors (weather, fire severity, vegetation cover, soil properties, morphology, and land management) influence the hydrological response of soil (Moody et al., 2013; Nunes et al., 2018; Zema, 2021). Computer-based hydrological models are essential tools to better understand and predict the hydrological processes in a cost-effective and time-efficient way (Filianoti et al., 2020), including in burned soils. However, validation under different environmental conditions is required, to ensure model applicability and reliability for predicting post-fire hydrology (Bezak et al., 2021; Fernández and Vega, 2016).

Many models have been developed to forecast runoff, erosion, and transport of polluting compounds under a large variety of climatic and geomorphological conditions (Bezak et al., 2021; Borrelli et al., 2021). Empirical models are of easier and quicker to apply compared to the most sophisticated physically-based routines. Their prediction capability may be considered acceptable for many needs and uses (Aksoy and Kavvas, 2005; Lucas-Borja et al., 2020). The use of empirical models is practically compulsory in data-poor environments, where input parameters are scarce or are difficult to collect, so that more complex models cannot be implemented (Zema et al., 2020b, 2020a). The Soil Conservation Service (SCS)-curve number (CN) (henceforth "SCS-CN model"), Horton, and USLE-family (universal soil loss equation) models are the most common empirical methods adopted in various environments to predict runoff and erosion, respectively (Bezak et al., 2021; Borrelli et al., 2021; Mishra and Singh, 2013). Thanks to the advantages of these empirical models, the SCS-CN, Horton, and USLE methods have been used as the hydrological and erosion components of many hydrological models applied at catchment scale (e.g., AnnAGNPS and SWAT models).

Regarding runoff and erosion predictions after fire using these models, several authors have evaluated the performance of hydrological models in burned forest areas, with or without post-fire treatments (Zema, 2021). Regarding runoff prediction, the SCS-CN method is commonly used in fire-affected forests, but the CN values are still not well-known for burned conditions (Soulis, 2018; Springer and Hawkins, 2005). According to the latter authors, the literature on SCS-CN model accuracy and CN values for simulating runoff in burned forests is limited. Moreover, the variability of the hydrological predictions of different models, using the SCS-CN method as sub-component, highlights the difficulties in setting proper CN values for post-fire conditions using observed data (Soulis, 2018).

The USLE-family models (USLE, RUSLE, MUSLE, USLE-M, HUSLE, etc.) have been developed to estimate soil erosion in agricultural lands, and their applicability in burned forest is not predictable, since fire impacts on vegetation and soil properties are different, and complicated by several factors (e.g., soil water repellency, ash effects), especially in the Mediterranean environment (Lopes et al., 2021). Moreover, these equations have mainly been applied for multi-year predictions, while modeling tests after the so-called 'window of disturbance' (Fernández et al., 2010) (occurring immediately after a fire and throughout about one year, when the soil is bare and the pre-fire levels of soil properties have not yet been recovered) have been mainly carried out using RUSLE in soils burned by wildfires (e.g., (Karamesouti et al., 2016; Larsen and MacDonald, 2007)). To the authors' best knowledge, no applications are available in the literature about the use of the MUSLE and USLE-M models after prescribed fires. As such, the optimal values of CNs and C factors, which are key parameters for the accurate estimation of runoff and erosion (Panagos et al., 2015; Shrestha et al., 2006), have not yet been identified in burned conditions and with post-fire management. Moreover, the validation of soil erosion models in soils subjected to post-fire management treatments, such as mulching, is particularly scarce globally (Fernández et al., 2010; Robichaud et al., 2007). Therefore, the applicability of the SCS-CN method and USLE-family models may remain questionable, without targeted modeling evaluations.

These literature gaps require studies that should assess the prediction capability of the SCS-CN, Horton, and USLE models in burned forests in Mediterranean areas under pre-fire and post-fire management, such as prescribed fire and soil mulching.

To satisfy this need, this study evaluates the prediction capability for runoff and soil loss of the SCS-CN, Horton, and USLE-family models (MUSLE and USLE-M) in three forests (pine, chestnut, and oak) of Southern Italy. The investigation was carried out at the plot and event scales throughout one year after a prescribed fire and with post-fire soil mulching with fern residues. The research questions which this study aims to answer are two: (i) Are the tested models reliable and accurate for predicting surface runoff and soil erosion in Mediterranean burned forests? (ii) Which are the optimal values of the input parameters of the tested models?

2. Materials and Methods

2.1. Study Area

The investigation was carried out in three of the most dominant forests of Calabria (Southern Italy), whose climate is semi-arid ('Csa' class, 'hot-summer Mediterranean' climate, according to Koppen) (Kottek et al., 2006). The mean annual precipitation and temperature are 1102 mm and 17.4 °C, respectively (weather station of Sant'Agata del Bianco, UTM coordinates 4217548 N, 595159 E, period 2000–2020).

Close to the municipality of Samo, three forest sites were identified to collect the hydrological observations used for the model evaluation (Figure 1):

- a pine (*Pinus pinaster* Aiton, "Calamacia" site, UTM coordinates 4215334 N; 590300 E) stand reforested in 1984 over an area between 650 and 700 m a.s.l.
- a natural oak (*Quercus frainetto* Ten., "Rungia" site, UTM coordinates 4216191 N; 588655 E) stand (900–950 m a.s.l.)

- a chestnut stand (*Castanea sativa* Mill., "Orgaro" site, UTM coordinates 4215622 N; 590410" E) about 30 years-old, between 700 and 750 m.

The tree density was about 950 (pine), 225 (oak), and 725 (chestnut) trees/ha. The tree height was 21 (pine), 10 (chestnut), and 18 (oak) m, while the breast diameter was 28, 20, and 41 cm, respectively. Shrub formations mainly consisted of *Quercus ilex L., Rubus ulmifolius S.,* and *Bellis perennis L.* (pine forest); *Cyclamen hederifolium* and *Bellis perennis L.* (oak); and *Rubus ulmifolius S., Pteridium aquilinum L.,* and *Bellis perennis L.* (chestnut). All forest stands had not been subject to management actions after planting or in the last fifty years for the natural stand.

The soils of the experimental sites (Cambisols, according to the World Reference Base for soil resources classification) were homogenous. The mean slope of soils was about 20% for all stands, and the texture was loamy sand (10.6 ± 2.57% of silt, 8.76 ± 0.61% of clay, and 80.7 ± 2.68% of sand). The unburned area of the pine forest in Calamacia instead showed a sandy loam texture (10.1 ± 1.01% of silt, 9.0 ± 0.01% of clay, and 81.0 ± 0.99% of sand).

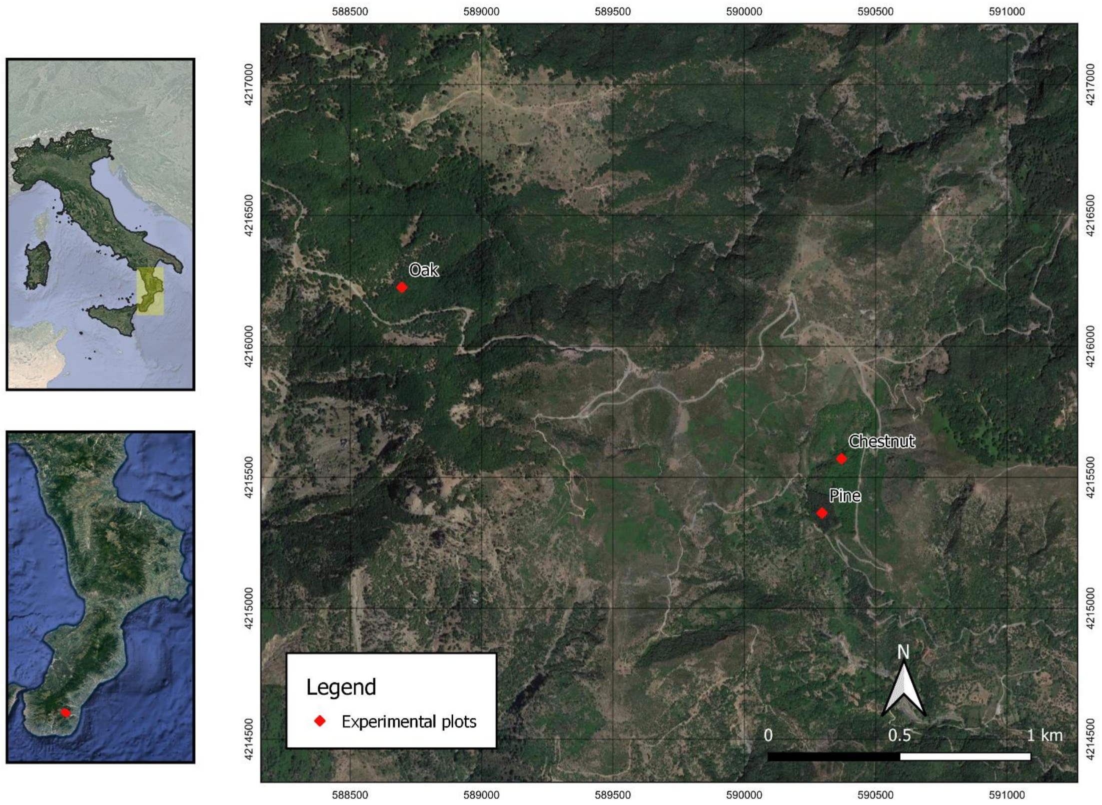

Figure 1. Location of the experimental sites (Samo, Calabria, Southern Italy).

2.2. Prescribed Fire Operations and Mulching Application

The prescribed fire was carried out in early June 2019, following the national and regional rules, and taking care that the main conditions during fire application at the experimental site (absence of wind and air humidity between 50 and 60%) were ideal, to avoid an uncontrolled wildfire. The burn severity of soils after the prescribed fire was low, according to the classification by (Parson et al., 2010).

Straw is commonly used as a mulch material, mainly in croplands, and this mulch is not always suitable in forest areas. Biomass transport from agricultural sites may be expensive, and these vegetal residues often contain agro-chemicals and parasites, with the possible development of non-native vegetation and diseases to forest plants (Bento-Gonçalves et al., 2012). Woodchips are sometime used as a mulch material in forests (particularly in young stands), but production may be expensive and difficult, due to the need of big machinery and a large amount of wood biomass (Klimek et al., 2020). Fern (*Pteridium aquilinum* (L.) Kuhn), which is abundant on the Mediterranean forest floor and does not bring non-native seeds or chemical residues into the forest ecosystem, may replace straw for the mulching of fire-affected areas. However, no studies prior to the present investigation have evaluated its suitability as a mulch cover for burned soils.

At the experimental sites, one day after the fire, small pieces (maximum length of 5 cm) of fern stems were applied to the soil as a mulch material in a part of the burned area. The fern was supplied from the same forest and the fresh residues were spread on the ground at a dose of 500 g/m^2 of fresh weight (200 g/m^2 of dry matter, as suggested for straw mulching by (Lucas-Borja et al., 2018; Vega et al., 2014)).

2.3. Hydrological Monitoring

In each forest site, three series of plots (each one with three replications) were delimited at a reciprocal distance, between 1.5 and 20 m (Figure 2).

(**a**)

(**b**)

(**c**)

Figure 2. Images of the experimental plots installed for hydrological monitoring of the experimental sites: (**a**) pine; (**b**), oak, and **c**), chestnut forests.

The plots (3 m in length × 1 m in width, for a total area of 3 m^2) were hydraulically isolated, in order to prevent the inflow of surface water, using 0.3-m high metallic sheets inserted up to 0.2 m below the soil surface. Downstream of each plot, a transverse channel intercepted the water and sediment flows, which were collected in a 100-litre tank.

The hydrological monitoring started immediately after site installation and was carried out until September 2020, over 15 months (Table 1). Precipitation height and intensity were measured in 15-min steps by a tipping bucket rain gauge at a maximum distance of 1 km from the experimental sites. Surface runoff and sediment concentration after the monitored events were measured according to (Lucas-Borja et al., 2019). In short, after mixing the water in the tank, three separate samples were collected for each rainfall–runoff event (total of 0.5 L). The samples were oven-dried at 105 °C for 24 h in the laboratory. Then, the dried sediments were weighed, and the weight was divided by the sample volume, to calculate the sediment concentration. The product of the latter by the runoff volume gave the soil loss.

Table 1. Main hydrological variables of rainfall events monitored at the experimental sites (Samo, Calabria, Southern Italy).

Date	**Height (mm)**	**Net Height (mm) ***			**Duration (h)**	**Intensity (mm/h)**	
		Pine	**Oak**	**Chestnut**		**Max**	**Mean**
15 July 2019	65	61.8	59.8	60.5	36	22.2	1.99
9 October 2019	49.9	45.4	43.9	44.9	26	14.6	1.85
11 November 2019	142.8	135.7	132.8	132.8	41	26.2	3.49
23 November 2019	87.1	82.7	81.0	81.9	19	24.7	4.58
5 December 2019	147.2	141.3	138.4	139.8	30	19	4.90
24 March 2020	155.9	149.7	146.5	149.7	32	13.8	2.86
14 July 2020	22.4	20.6	19.7	20.4	7	12.8	2.58

Note: * recorded at the rain gauge station under a tree canopy in each forest.

2.4. Short Description of the Models

Some brief information about the tested models is provided below, while more details are available in the works by the cited authors.

2.4.1. SCS-CN Model

The Soil Conservation Service-curve number (SCS-CN) (USDA, 1969) was developed by the United States Department of Agriculture in the 1950s. This empirical model derives some assumptions from physically-based infiltration equations and requires only a few data points to estimate runoff for a given rainfall event.

The SCS-CN method assumes:

$$\frac{V}{P_n} = \frac{W}{S} \tag{1}$$

where V is the runoff volume, P_n is the net rainfall, W is the soil potential retention, and S is the maximum soil potential retention (all values are in mm).

V is calculated by the following equation:

$$V = \frac{{P_n}^2}{P_n + S} \tag{2}$$

where P_n is the difference between the rainfall depth P and the initial abstraction I_a (both in mm). The latter is the amount of rainfall retained in soil storage as interception, infiltration, and surface storage before runoff begins (Ponce and Hawkins, 1996). By convention, I_a is equal to the product of a coefficient λ (generally equal to 0.2) by S.

Therefore, V becomes:

$$V = \frac{(P - 0.2S)^2}{P_n + S} \tag{3}$$

S is a function of the dimensionless 'curve number' (CN) parameter:

$$S = 25.4\left(\frac{1000}{CN} - 10\right) \tag{4}$$

CN describes the antecedent potential water retention of a soil (Hawkins, 1982). Theoretically, CN varies between 0 and 100, but the usual values of CNs are in the range 40–98 (Ponce and Hawkins, 1996).

The CN of agro-forest soils depends on the soil hydrological class, vegetal cover, hydrological condition (good, medium, poor), and cultivation practice; moreover, for CN calculation the antecedent moisture condition (AMC) of the soil must be determined. The soil hydrological class (*A* to *D*) is related to the soil's capability to produce runoff, which in turn is due to the soil infiltration capability. The actual AMC of the soil subject to a rainfall/runoff event is estimated as a function of the total height of precipitation in the five days before the event in the two different conditions of crop dormancy or growing season. In this regard, three AMCs are identified:

- AMC_I: dry condition and minimum surface runoff
- AMC_{II}: average condition and surface runoff

- AMC_{III}: wet condition and maximum surface runoff.

The SCS-CN guidelines report tables to calculate the *CN* values for soils of a given hydrological class and condition, vegetal cover, cultivation practice, and average *AMC* (AMC_{II}). The values of *CN*s related to AMC_I (CN_I) or AMC_{III} (CN_{III}) can be calculated with the following equations:

$$CN_I = \frac{4.2CN_{II}}{10 - 0.058CN_{II}} \tag{5}$$

$$CN_{III} = \frac{23CN_{II}}{10 + 0.13CN_{II}} \tag{6}$$

2.4.2. Horton Equation

Horton's method was formulated by Robert E. Horton in 1939 as an infiltration model to describe the physical process of infiltration in a quantitative manner.

The runoff rate *q* (in mm h^{-1}) at a given time *t* is given by:

$$q(t) = i(t) - f(t) \tag{7}$$

where *i(t)* and *f(t)* (both in mm h^{-1}) are the rainfall intensity and infiltration rate at time *t*, respectively.

The infiltration rate *f(t)* is calculated as:

$$f(t) = f_c + (f_0 - f_c)\, e^{-kt} \tag{8}$$

During a storm, *f(t)* generally declines from the maximum rate f_0 to the minimum value f_c through the parameter *k*. Equation (7) gives *q(t)* when *i(t)* exceeds *f(t)*. The runoff volume is the integral of Equation (7), when *q(t)* is positive, between the start and the end of the runoff event.

2.4.3. MUSLE Equation

The 'universal soil loss equation' (USLE) was first established in the USA to model erosion in small agricultural catchments. USLE has a mathematical form that depends on six input parameters linked to climate, soil cover and properties, topography, and human activities; the six so-called "USLE-factors" (*R, K, L, S, C*, and *P*).

The USLE equation has been modified and updated over several versions and has been replaced by the revised USLE (RUSLE) (Dabney et al., 2012; Renard et al., 1991). Reference (Williams, 1975) developed a modified version, called MUSLE, which is the acronym modified USLE. The MUSLE model replaces the USLE rainfall factor (*R*) by a runoff factor, to consider the effect of flow on sediment transport. Therefore, the expression of the MUSLE equation has the following general form:

$$Y = a\,(Q'q_p)^b\,K\,L\,S\,C\,P \tag{9}$$

where Y is the soil loss (tons ha^{-1}) on a storm basis, Q is the runoff volume (m^3), q_p is the peak flow rate (m^3 s^{-1}), K is the soil erodibility factor (tons h MJ^{-1} mm^{-1}), L and S are the slope-length and steepness factor, respectively, C is the cover management factor, and P is the conservation practice factor. The a and b coefficients are site-specific empirical factors for calculating the runoff factor.

2.4.4. USLE-M Equation

Kinnell and Risse (1998) (Kinnell and Risse, 1998) proposed the USLE-M model based on the hypothesis that the sediment concentration in the runoff is affected by the event rainfall erosivity index (R_e, (Wischmeier and Smith, 1978) per unit quantity of rain (P_e, mm). According to the USLE-M, Y is calculated as:

$$\mathrm{Y} = Q_R\,R_e\,K\,LS\,C\,P \tag{10}$$

where Q_R and R_e are the runoff coefficient and the erosivity index for the modelled event, respectively. The other factors of the USLE-M have the same meaning as the USLE and MUSLE equations, but the values of K and C factors are calculated using different expressions (see Section 2.4.3 and 2.4.4) (Kinnell and Risse, 1998).

2.5. Model Implementation in the Experimental Plots

2.5.1. SCS-CN Model

The sub-hourly precipitation records collected at the rain gauge stations were aggregated in daily values and supplied as input to the SCS-CN model. The *AMC* was derived according to the antecedent rainfall depths of each precipitation event. The soil hydrological group was identified using the data of the soil map of Calabria (ARSSA, 2003) and according to (Carrà et al., 2021), who measured the hydraulic conductivity of the same sites.
The default values of *CN* were assumed, following the standard procedure by the USDA Soil Conservation Service (USDA, 1969) (Table 2).

2.5.2. Horton Equation

In the same experimental sites, Carrà et al., (2021) determined the water infiltration curves for the three soil conditions using a rainfall simulator (Eijkelkamp®, https://en.eijkelkamp.com/), following the methods reported by (Bombino et al., 2021). In short, for each forest stand and soil condition, rainfall simulations were carried out in three randomly chosen points. Rainfall of 3.0 mm, at an intensity of 37.8 mm/h, was generated over a surface area of 0.305 m × 0.305 m. Throughout the simulated rainfall, the surface runoff volume was collected and measured in a small graduated bucket at a time scale of 30 s. The infiltration curves were determined by subtracting the runoff from the rainfall at each time interval. The infiltration test stopped when three equal time measurements of instantaneous infiltration had been recorded.

For Equation (8), we interpolated these infiltration curves using Equation (13), which has the following mathematical structure:

$$f(t) = me^{-nt} \tag{11}$$

where *m* and *n* are the two constant coefficients and *t* is expressed in seconds. The goodness-of-fit of this equation was measured by the coefficient of determination (r^2) (Table 2).

For the modeled events, the hyetograph *i(t)* was derived from the rainfall records and the difference between *i(t)* and *f(t)* at a given *t* gave the runoff rate *q(t)* every five minutes. Given the very short time of concentration (less than one minute) of the plot, the surface runoff stop was considered the same as the rainfall end.

2.5.3. MUSLE Equation

The MUSLE model is usually applied at the catchment scale; however, in some studies, it has been implemented at the plot scale (e.g., (McConkey et al., 1997; Pongsai et al., 2010)). *Q* of Equation (9) was the runoff volume predicted by the SCS-CN method. The parameter q_p was calculated using the following formula:

$$q_p = z\frac{AQ}{t_p} \tag{12}$$

where *A* is the plot area (m^2), *z* is a conversion coefficient, and t_p (0.01 h in this study) is the plot concentration time, which was experimentally measured at the plots using a surface tracer.

We deliberately adopted the values of *Q* and q_p as predicted by the SCS-CN method or calculated using Equation (12), rather than using the observed values, since these observations are never available in practical applications, and, therefore, the modeler is forced to use estimations from models. The site-specific factors, *a* and *b*, in Equation (8) were 0.87 and 0.56, according to the suggestions by (Williams, 1982).

The K-factor was estimated using the nomograph in (Wischmeier and Smith, 1978). The C-factor was calculated using an empirical equation based on canopy cover and aboveground biomass proposed by (Bombino et al., 2002). Usually, the C-factor is calculated as the product of some sub-factors, depending on the previous land use, canopy cover, surface cover, surface roughness, and soil moisture content (e.g., in (Vieira et al., 2018)). In this study, we preferred to use a simpler equation, based on soil cover, since often these sub-factors are not easy to measure or determine. The P-factor was always set to one (Table 2).

2.5.4. USLE-M Equation

The runoff coefficient Q_R of Equation (10) was calculated as:

$$Q_R = Q/P_e \tag{13}$$

where Q and P_e are the runoff volume (again estimated using the SCS-CN method) and rainfall depth (both in mm), respectively. Following Renard et al. (1991) (Renard et al., 1991), the rainfall R-factor (R_e, MJ mm ha^{-1} h^{-1}) factor was calculated using the following equation:

$$R_e = EI_{30}/1735 \tag{14}$$

where NSE is the rainfall kinetic energy (tons m ha^{-1}) and I_{30} is the maximum rainfall intensity in 30 min (mm h^{-1}) for each event.

The correction of K proposed by Kinnel et al. (1998) was adopted to calculate the K-factor of USLE-M (K_{UM}). The C- and P-factors (C_{UM}) were calculated as for the MUSLE equation (Table 2).

2.6. Model Calibration

The SCS-CN, MUSLE, and USLE-M models were initially run with default parameters, estimated from the guidelines of the three models. Since the models provided poor predictions (see Section 3), we chose the most sensitive input parameters (*CN* for the SCS-CN model (Baginska et al., 2003; Yuan et al., 2001), and the C-factor for the MUSLE and USLE-M equations (Biddoccu et al., 2020; Hammad et al., 2004)) and calibrated the models. In our study, the hydrological effects of prescribed fire and soil mulching were considered by tuning these parameters. The models were first calibrated using constant *CN*s and C-factors over time. In a further calibration trial, the *CN*s and C-factors were increased for the first two rainfall events, to take into account the variability of the soil hydrological response throughout the first year after fire, as demonstrated by several authors (e.g., (Cawson et al., 2012; Vieira et al., 2015; Zema et al., 2020b)).

Calibration was carried out manually using a trial-and-error procedure, and was considered optimal when the coefficient of efficiency (*NSE*, see Section 2.6) was the highest and the error between the mean values of the observations and simulations of runoff (SCS-CN and Horton models) or soil loss (MUSLE and USLE-M models) was the lowest among the runs. The Horton equation was not calibrated, since all the relevant input parameters come from measurements from the infiltration tests. Table 2 reports the default and calibrated parameters, with their sources, for the four models.

2.7. Model Performance Evaluation

The runoff/erosion prediction capability of the four models was evaluated using qualitative and quantitative approaches. First, the observed and simulated values were visually compared around the line of perfect agreement in scatter plots. Then, we adopted a combination of the following criteria for model quantitative evaluation: (i) the main statistics (i.e., maximum, minimum, mean, and standard deviation of both observed and simulated values); and (ii) a set of indexes, commonly used in hydrological modeling. These indexes consisted of the determination coefficient (r^2), the efficiency coefficient (*NSE*, (Nash and Sutcliffe, 1970)), and percentage bias (*PBIAS*, (Willmott, 1982)), adopting a p-level of 0.05. In more detail, r^2 measures the dispersion of the 'observations vs. predictions' points around the interpolating line; values over 0.5 are deemed acceptable (Santhi et al., 2001; Van Liew et al., 2003; Vieira et al., 2018). *NSE*, which is the 'goodness of fit' of the model predictions, is optimal if $NSE = 1$, good if $NSE \geq 0.75$, satisfactory if $0.36 \leq NSE \leq 0.75$, and unsatisfactory if $NSE \leq 0.36$ (Van Liew et al., 2003). *PBIAS* indicates whether the model over-predicts (if negative) or under-predicts (if positive) the output variable. A *PBIAS* below 0.25 and 0.55 for runoff and erosion, respectively, are considered fair (Gupta et al., 1999; Moriasi et al., 2007).

Table 2. Values of input parameters adopted to simulate surface runoff volumes and soil loss using the SCS, Horton, MUSLE, and USLE-M models applied in the experimental plots.

	Model	Input Parameter	Measuring Unit	Soil Conditions					
				Unburned		Burned		Burned and Mulched	
				Default Model	Calibrated Model	Default Model	Calibrated Model	Default Model	Calibrated Model
Chestnut	*SCS-CN*	CN	-	46	43	70	80 * 45	50	65 * 32
		λ	-	0.2					
	Horton	m	mm h^{-1}	33.65	-	30.51	-	37.61	-
		n	s^{-1}	0.006	-	0.004	-	0.004	-
		r^2	-	0.90	-	0.95	-	0.99	-
	MUSLE	a	-	89.6					
		b	-	0.56					
		K-factor	tons h MJ^{-1} mm^{-1}	0.03					
		C-factor	-	0.009	1	0.390	1	0.011	1
		P-factor	-	1					
	USLE-M	Q_r	- max	0.17		0.34		0.10	
			min	0.07		0.09		0.02	
		R_e-factor	MJ mm ha^{-1} h^{-1} max	69.5		84.8		54.6	
			min	2.86		3.26		1.05	
		K_{UM}-factor	tons h MJ^{-1} mm^{-1}	0.043		0.024		0.102	
		C_{UM}-factor	-	0.004	0.021	0.203	0.288 * 0.038	0.008	0.019 * 0.004
		P-factor	-	1					
Oak	SCS-CN	CN	-	46	45	80	83 * 47	50	76 * 45
		λ	-	0.2					
	Horton	m	mm h^{-1}	17.95	-	16.38	-	22.93	-
		n	s^{-1}	0.007	-	0.005	-	0.005	-
		r^2	-	0.67	-	0.95	-	0.90	-
	MUSLE	a	-	89.6					
		b	-	0.56					

		K-factor	tons h MJ^{-1} mm^{-1}		0.03							
		C-factor	-		0.001	1	0.356	1		0.011	1	
		P-factor	-		1							
	USLE-M	Q_r	-	max	0.19		0.48			0.35		
				min	0		0.14			0.07		
		R_e-factor	MJ mm ha^{-1} h^{-1}	max	62.7		84.8			72		
				min	0		24.9			8.16		
		K_{UM}-factor	tons h MJ^{-1} mm^{-1}		0.052		0.023			0.034		
		C_{UM}-factor	-		0.001	0.020	0.356	0.104 *	0.056	0.011	0.056 *	0.045
		P-factor	-		1							
Pine	SCS-CN	CN	-		39	40	90	79 *	41	55	65*	39
		λ	-		0.2							
	Horton	m	mm h^{-1}		34.69	-	39.90	-		32.44	-	
		n	s^{-1}		0.003	-	0.00	-		0.004	-	
		r^2	-		0.95	-	0.98	-		0.94	-	
	MUSLE	a	-		89.6							
		b	-		0.56							
		K-factor	tons h MJ^{-1} mm^{-1}		0.03							
		C-factor	-		0.001	1	0.36	1		0.003	1	
		P-factor	-		1							
	USLE-M	Q_r	-	max	0.08		0.34			0.10		
				min	0		0.06			0.06		
		R_e-factor	MJ mm ha^{-1} h^{-1}	max	50.1		75.8			33.8		
				min	0		15.6			10.9		
		K_{UM}-factor	tons h MJ^{-1} mm^{-1}		0.085		0.033			0.068		
		C_{UM}-factor	-		0.004	0.008	0.203	0.208 *	0.006	0.008	0.055 *	0.003
		P-factor	-		1							

Notes: *CN* = curve number; *λ* = initial abstraction ratio; *m*, *n* = coefficients of Equation (13); r^2 = coefficient of determination; *a*, *b* = site-specific factors; R_e = RUSLE rainfall R-factor; *K*, *C* and *P* = factors of the MUSLE model; K_{UM} and C_{UM} = factors of the USLE-M model; * first two modeled events.

3. Results and Discussions

3.1. Hydrological Characterization

Throughout the monitoring period, 516 rainfall events with a total depth of 1120 mm were recorded at the rain gauging stations. Of these events, only seven were classified as erosive events (that is, with a depth over 13 mm), according to (Wischmeier and Smith, 1978) (Table 1).

In the unburned plots, the maximum runoff (up to 18.1 ± 12.9 mm in chestnut forest) was observed after the highest rainfall (156 mm). In two events (9 October 2019 and 14 July 2020), no runoff was collected in the unburned chestnut and oak forests and in all soil conditions of the pine and oak sites (Figure 3).

The highest runoff volume in the burned plots was always collected after the first post-fire event. One month after the prescribed fire, the runoff was from 22.3 ± 1.35 mm (chestnut) to 31.3 ± 2.29 (oak) (Figure 3).

The first rainfall event also produced the highest runoff in the burned and mulched plots of the chestnut and oak forests (6.61 ± 1.16 mm and 23 ± 3.69 mm, respectively). Conversely, in the pine forest, the maximum runoff (10.4 ± 0.80 mm) was measured after the second event (Figure 3).

The soil loss in the unburned plots was in the range 5.31 ± 1.40 g/m^2 (pine forest) to 15.34 ± 3.21 g/m^2 (chestnut), and these values were measured after the first and second post-fire rainfalls (Figure 4). For these two events, erosion increased very much in the burned soils of all forests, and particularly in the pine and chestnut soils (soil loss equal to 51.61 ± 6.92 and 52.26 ± 13.67 g/m^2, respectively). In oak soils, the soil loss was noticeably lower (15.12 ± 2.87 g/m^2), but much higher compared to the unburned plots. Soil mulching with fern was effective for reducing erosion, and, under this soil condition, the maximum soil losses were between 10.62 ± 0.99 g/m^2 (pine forest) and 14.58 ± 4.80 g/m^2; again recorded after the first event. After the first two events, the soil loss showed a low variability in the unburned soils, while, in the burned and not treated soils, erosion decreased over time (Figure 4).

Soil mulching with fern mainly reduced the erosion in pine and chestnut forests compared to the fire-affected plots. The maximum soil losses were equal to 1.87 ± 0.33 and 0.81 ± 0.16 g/m^2 (both surveyed in the third event), respectively. In these plots, the estimated soil losses were even lower compared to unburned soils, while the pre-fire erosion rates were only restored in oak forests for two events (Figure 4).

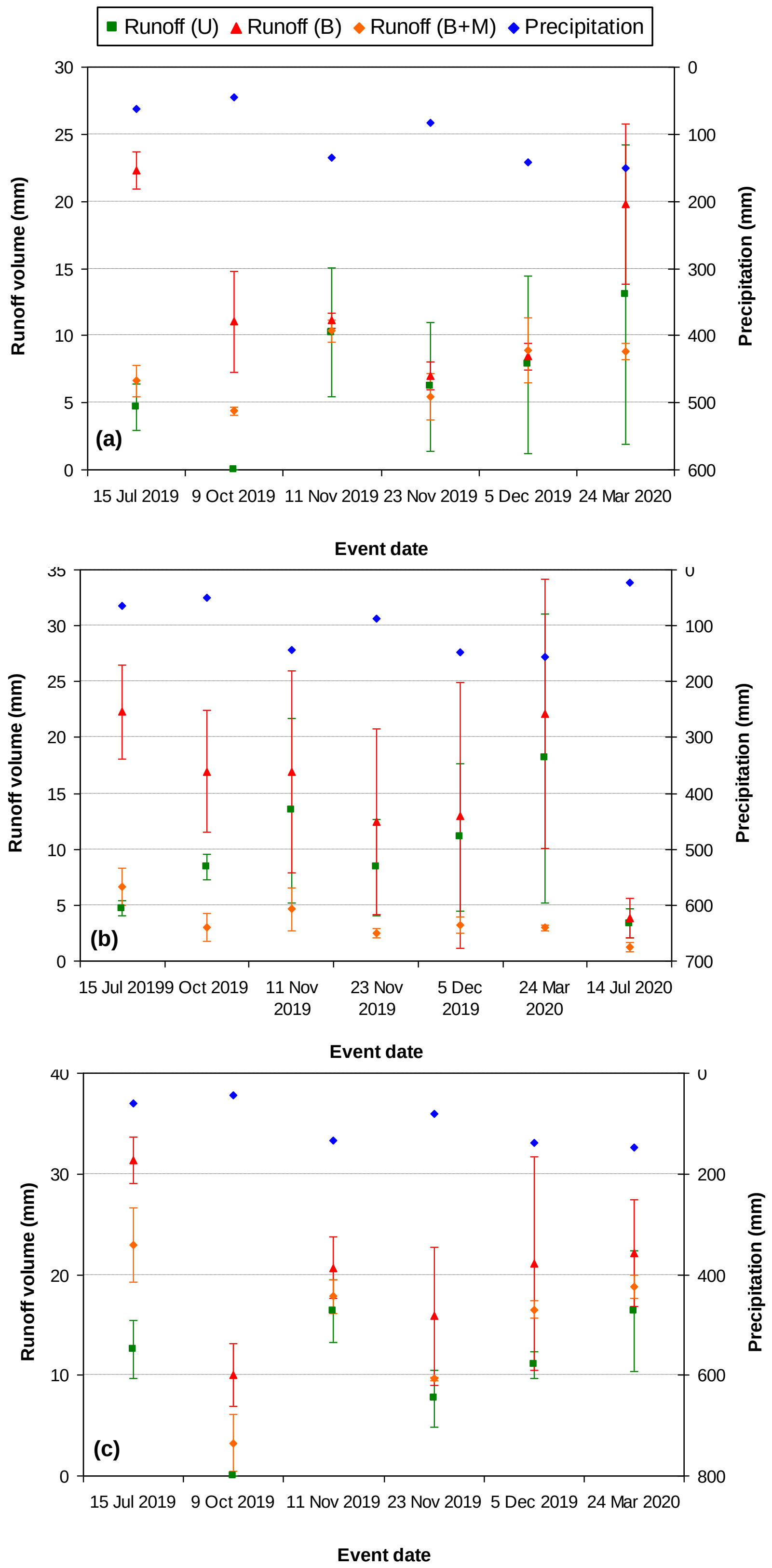

Figure 3. Precipitation and runoff volume (mean ± std. dev., *n* = 3 plots) measured in plots ((**a**), pine; (**b**), chestnut; (**c**), oak) after prescribed fire and soil mulching using fern. Notes: U = unburned soils; B = burned and not treated soils; B + M = burned and mulched soils; no runoff was observed in pine and oak plots on 14 July 2020.

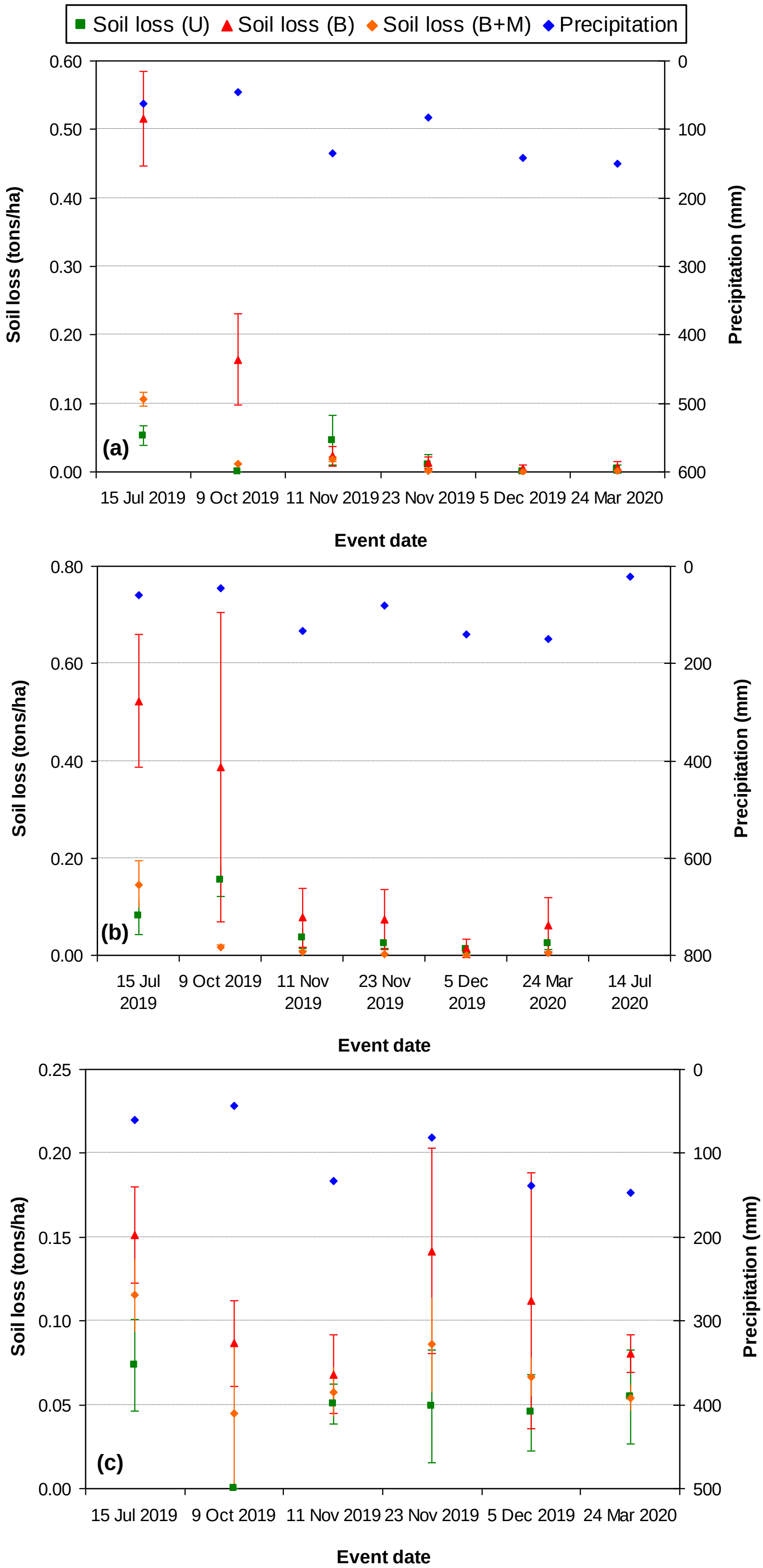

Figure 4. Precipitation and soil loss (mean ± std. dev., n = 3 plots) measured in plots ((**a**), pine; (**b**), chestnut; (**c**), oak) after prescribed fire and soil mulching using fern. Notes: U = unburned soils; B = burned and not treated soils; B + M = burned and mulched soils; no soil loss was observed in pine and oak plots on 14 July 2020.

3.2. Hydrological Modeling

3.2.1. SCS-CN Model

The SCS-CN model, running with default input *CN*s, always gave poor predictions of surface runoff, as shown by the great scattering of the observations/simulations around the line of perfect agreement (Figure 5). This low accuracy is confirmed by the poor values of the evaluation indexes (Table 3). In more detail, r^2 was much lower than 0.5 (with two exceptions, unburned soils in pine and chestnut forests, r^2 of 0.73 and 0.79), and *NSE* was below 0.35 (except for unburned soils in pine forest, *NSE* = 0.36). *PBIAS*, which was positive in some soil conditions and negative in others, indicates a high underprediction or overestimation for a observation, respectively. Moreover, the statistics calculated for the observations and predictions were highly different (mean error of up to 500%).

These model's poor performances indicate that the literature values that were adopted as defaults for the input *CN*s were not suitable for simulating the runoff volume. This especially holds true under burned soil conditions, when the soil's hydrological response was increased by fire. The *CN*s of fire-affected areas are usually estimated by increasing the post-fire values, depending on the fire severity (e.g., (Papathanasiou et al., 2015)). This statement agrees with the findings of (Springer and Hawkins, 2005), who highlighted the need to increase the *CN* values of burned soils by about 25 units. The worsening of the hydrological response of the burned soil after fires with different intensities has been shown by various studies (e.g., (Keizer et al., 2018; Wilson et al., 2018)]), and this is also true in the case of prescribed fire (e.g., (Kozlowski, 2012)). This increase is mainly due to the soil hydrophobicity and removal of vegetation due to fire. However, these effects vanish some months after a fire. The mulching treatment 'smooths' the increased hydrological response of burned soils, and this effect requires a lower increase in *CN* values (Zema et al., 2020b).

However, in our experience, model runs with higher but constant CNs increase the runoff prediction capability of the SCS-CN model, but this calibration effort did not significantly improve the model performance in all soil conditions (data not shown). Regarding the unburned plots, the model predictions of runoff were also disappointing. Although the values of r^2 were satisfying (over 0.49), the *NSE* was negative in chestnut and oak forests, and lower than 0.36 in pine soils, while *PBIAS* (> 0.08) showed a noticeable tendency to model underprediction (Table 3). The difference between the mean values of the observed and predicted runoff was 8.3% to 28%. In contrast to our results and those by (Springer and

Hawkins, 2005), references (Canfield et al., 2005; Lopes et al., 2021) observed no apparent increase of *CN*s in post-fire conditions.

In order to simulate the effects of a repellent and bare soil on surface runoff, there is a need to increase the *CN*s in the window-of-disturbance (Fernández et al., 2010; Moody et al., 2013). The authors of (Zema et al., 2020b), working in burned pine forests under semi-arid Mediterranean conditions, demonstrated that the SCS-CN model performs better for simulating surface runoff, if the *CN*s are increased in the few months after the prescribed fire. Accordingly, when the *CN*s were increased for the first two modeled events after fire in our study, the performances of the calibrated model were always satisfactory for burned soils (mulched or not), except for the mulched soils in pine forest. The scattering of observations/simulations around the line of perfect agreement was reduced (Figure 5), and the evaluation indexes were generally over the acceptance limits for model predictions (Table 3). The SCS-CN model performance was good in burned soil for pine (r^2 = 0.69, *NSE* = 0.81 and *PBIAS* = 0.07), and satisfactory, both in burned and not treated, and burned and mulched soils, of chestnut (r^2 > 0.72, *NSE* > 0.65 and *PBIAS* < 0.17) and oak (r^2 > 0.52, *NSE* > 0.61 and *PBIAS* < 0.06) forests (Table 3). The difference between the mean observed and predicted runoff was lower than 17.5%. The model prediction capability of runoff was excellent in the burned and mulched soils of chestnut, where the r^2 and *NSE* were 0.72 and 0.94, respectively (Table 3).

The worse performance of the SCS-CN model in unburned soils compared to burned conditions was quite surprising. This low accuracy could be explained by the low generation capacity of the unburned soils of the experimental sites, which was not well simulated by the SCS-CN model, unless unrealistic *CN*s are input (runoff of three rainfall events simulated as zero) (Figure 5). The hydrological models generally tend to overestimate the lower events and underestimate the most intense flows (Kinnell, 2003; Tian et al., 2014). A modified hydrological response due to fire increases the runoff generation capacity, which is better reproduced by this method. Overall, since the SCS-CN method does not optimally simulate the changes in soil properties due to management or other factors, without calibrating the input *CN*s, further studies should improve the model simulation of the temporal evolution of soil properties (Romero et al., 2007). This could be done, for instance, by tuning the *CN*s proposed in the SCS guidelines using correction factors that should take into account the effects of soil water repellency and changes in hydraulic conductivity (Plaza-Álvarez et al., 2019, 2018). Until then, our results indicate that the suggested values of *CN* should be used instead of the standard SCS values for runoff predictions in soils burned by prescribed fires and treated with mulching under similar properties, climate, and management conditions as our experimental sites.

Table 3. Statistics and indexes evaluating the runoff prediction capability of the SCS-CN model in forest plots subject to prescribed fire and soil mulching with fern.

	Runoff Volume	Mean (mm)	Standard Deviation (mm)	Minimum (mm)	Maximum (mm)	r^2	*NSE*	*PBIAS*
Pine				Unburned				
	Observed	7.00	4.54	0.00	13.06	-	-	-
	Simulated (default)	5.18	5.74	0.00	12.34	0.73	0.36	0.26
	Simulated (calibrated)	5.86	6.43	0.00	13.79	0.73	0.34	0.16
				Burned				
	Observed	13.28	6.26	7.01	22.31	-	-	-
	Simulated (default)	81.00	43.61	27.02	126.50	0.00	−85.36	−5.10
	Simulated (calibrated)	12.41	7.06	0.52	22.28	0.69	0.81	0.07
				Burned and mulched				
	Observed	7.41	2.31	4.37	10.35	-	-	-
	Simulated (default)	20.04	18.32	0.32	40.58	−1.70	−81.26	0.79
	Simulated (calibrated)	7.06	4.53	0.14	12.34	0.62	−0.70	0.05
Chestnut				Unburned				
	Observed	9.65	5.09	3.37	18.13	-	-	-
	Simulated (default)	9.13	10.78	0.00	23.49	0.79	−0.72	0.05
	Simulated (calibrated)	6.95	8.45	0.00	18.44	0.79	−0.13	0.28
				Burned				
	Observed	15.39	6.39	3.85	22.31	-	-	-
	Simulated (default)	35.35	31.85	0.00	74.04	0.14	−16.25	−1.30
	Simulated (calibrated)	13.71	9.32	0.00	23.62	0.75	0.65	0.11
				Burned and mulched				
	Observed	3.46	1.72	1.25	6.63	-	-	-
	Simulated (default)	12.43	14.07	0.00	30.76	0.00	−5.14	−2.59
	Simulated (calibrated)	2.86	2.77	0.00	8.13	0.72	0.94	0.17
Oak				Unburned				
	Observed	10.66	6.18	0.00	16.36	-	-	-
	Simulated (default)	10.65	10.96	0.00	23.49	0.49	−0.66	0.00
	Simulated (calibrated)	9.77	10.17	0.00	21.77	0.49	−0.43	0.08
				Burned				
	Observed	20.18	7.09	10.00	31.34	-	-	-
	Simulated (default)	59.25	37.76	13.74	99.21	0.04	−19.13	−1.94
	Simulated (calibrated)	19.04	8.85	2.81	27.96	0.52	0.70	0.06
				Burned and mulched				
	Observed	14.84	7.13	3.27	22.98	-	-	-
	Simulated (default)	14.51	14.19	0.00	30.76	0.19	−1.28	0.02
	Simulated (calibrated)	14.53	7.32	1.86	21.77	0.54	0.61	0.02

Notes: r^2 = coefficient of determination; *NSE* = coefficient of efficiency; *PBIAS* = coefficient of residual mass.

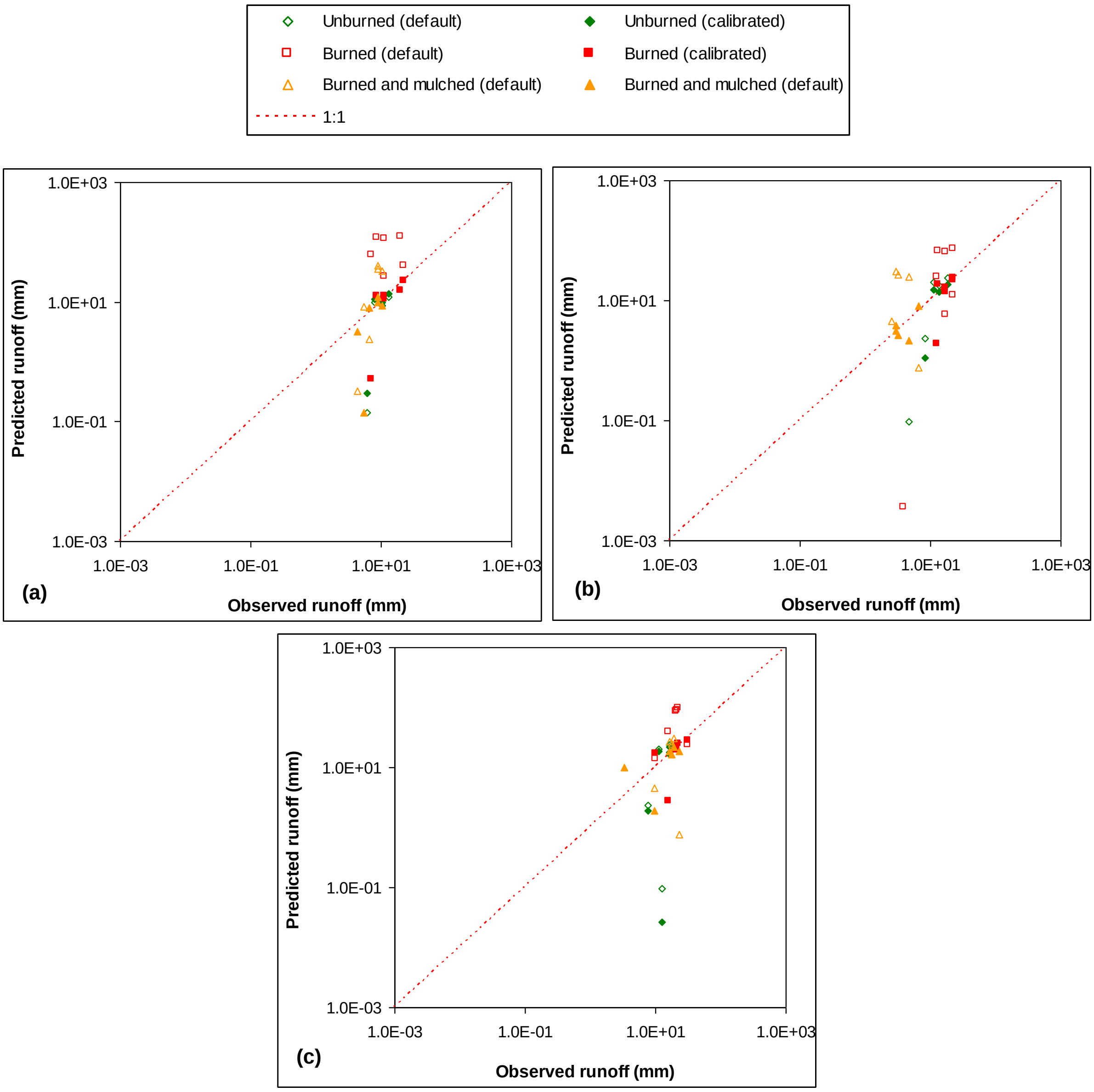

Figure 5. Scatter plots of runoff volumes observed in forest sites ((**a**), pine; (**b**), chestnut; (**c**), oak) subject to prescribed fire and soil mulching with fern vs. predicted using the SCS-CN model. Values are reported on logarithmic scales.

3.2.2. Horton Model

The runoff prediction capability of the Horton model was inaccurate under all soil conditions and forest species. In more detail, despite the satisfactory coefficients of determination calculated in the unburned soils of the three forest species ($r^2 > 0.65$), the r^2 was always lower than 0.14 in the other soil conditions. The differences between the mean observed and predicted runoff volumes were over 50%, with peaks of up to 677%. Moreover, *NSE* and *PBIAS* were negative for all modeled soil conditions at the three sites (Table 4). These coefficients indicate the wide scattering of the observations/simulations around the line of perfect agreement (Figure 6), and a noticeable overestimation of the modeled runoff volumes (Table 4).

The inaccuracy detected in this study for the Horton model was basically due to the noticeable overestimation of the observed runoff volumes, since the model was not calibrated, and, furthermore, the infiltration rate curves were not updated to account for the variability of infiltration rates over time. Furthermore, it cannot be excluded that the worse runoff prediction capability shown by the Horton model in comparison to the SCS-CN model may be due to the fact that the dominant runoff generation mechanism of the experimental soils is soil saturation during a storm (which is better simulated by the SCS-CN model) rather than infiltration excess (on which the Horton equation is based); as would be expected in soils in semi-arid environments (Vega et al., 2014).

Table 4. Statistics and indexes evaluating the runoff prediction capability of the Horton model in forest plots subject to prescribed fire and soil mulching with fern.

	Runoff Volume	**Mean (mm)**	**Standard Deviation (mm)**	**Minimum (mm)**	**Maximum (mm)**	r^2	***NSE***	***PBIAS***
				Unburned				
Pine	Observed	10.70	4.68	4.69	18.13	-	-	-
	Simulated	24.83	17.05	6.12	41.63	0.65	−18.32	−1.32
				Burned				
	Observed	17.31	4.24	12.49	22.31	-	-	-
	Simulated	26.96	17.23	6.91	44.10	0.03	−5.43	−0.56
				Burned and mulched				
	Observed	3.83	1.55	2.51	6.63	-	-	-
	Simulated	26.51	18.49	1.71	44.48	0.14	−15.67	−5.92
				Unburned				
Chestnut	Observed	9.65	5.09	3.37	18.13	-	-	-
	Simulated	25.03	18.30	1.92	45.49	0.76	−17.30	−1.59
				Burned				
	Observed	15.39	6.39	3.85	22.31	-	-	-
	Simulated	23.99	18.28	1.51	44.57	0.12	−3.81	−0.56
				Burned and mulched				
	Observed	3.46	1.72	1.25	6.63	-	-	-
	Simulated	24.06	17.92	0.87	44.21	0.04	−15.94	−5.95
				Unburned				
Oak	Observed	10.70	4.68	4.69	18.13	-	-	-
	Simulated	31.51	15.43	14.95	46.50	0.68	−29.15	−1.95
				Burned				
	Observed	17.31	4.24	12.49	22.31	-	-	-
	Simulated	30.60	15.70	13.12	46.03	0.02	−6.06	−0.77
				Burned and mulched				
	Observed	3.83	1.55	2.51	6.63	-	-	-
	Simulated	29.77	16.17	11.30	45.80	0.08	−17.39	−6.77

Notes: r^2 = coefficient of determination; *NSE* = coefficient of efficiency; *PBIAS* = coefficient of residual mass.

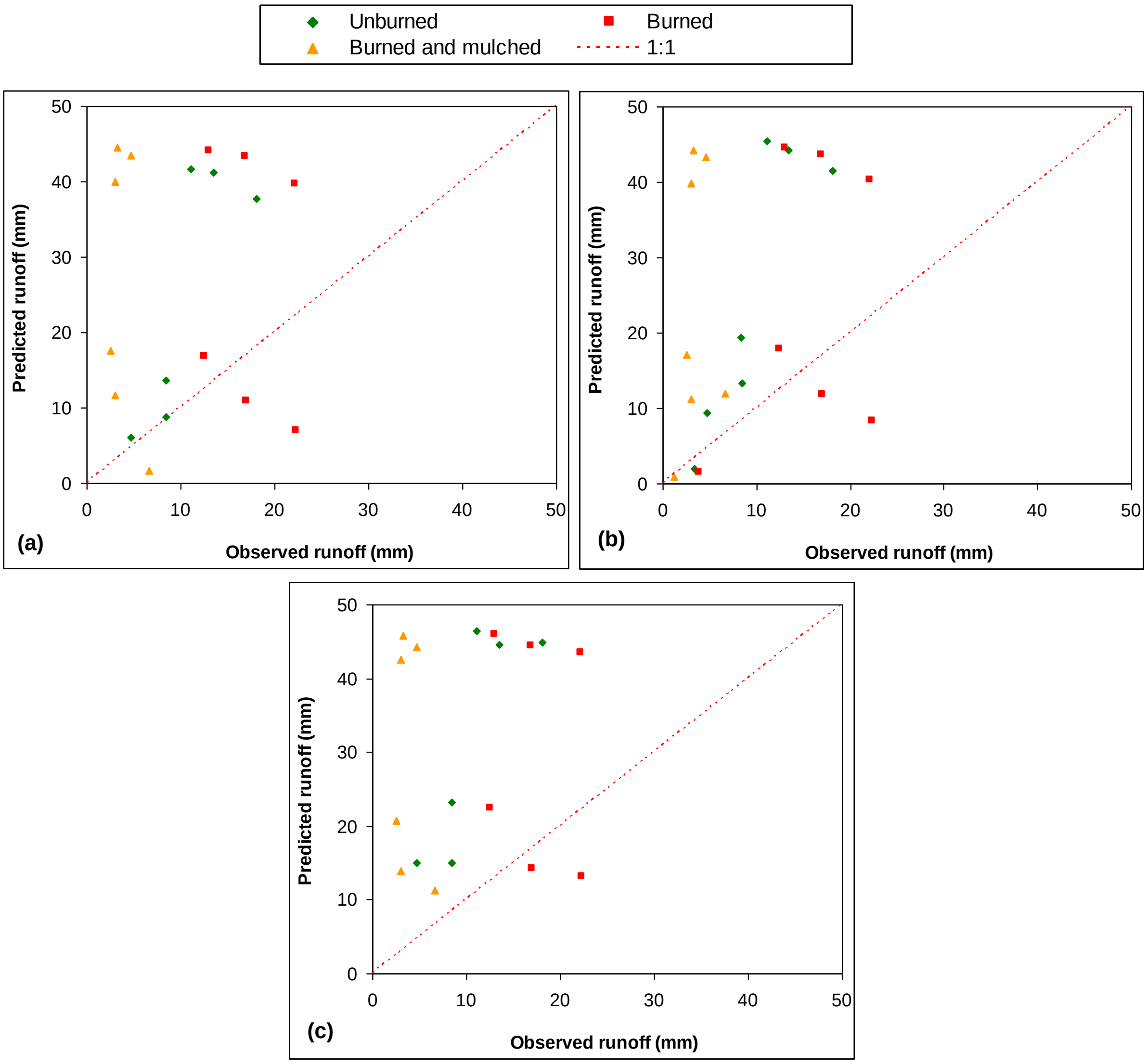

Figure 6. Scatter plots of runoff volumes observed in forest sites ((**a**), pine; (**b**), chestnut; (**c**), oak) subject to prescribed fire and soil mulching with fern vs. predicted using the Horton model.

3.2.3. MUSLE Model

Before calibration, the erosion predictions provided by the MUSLE equation were poor. The simulated soil losses were lower by at least one order of magnitude compared to the corresponding observations. The model strongly underestimated erosion in all soil conditions and forests (see the *PBIAS* close to one), which caused a great scattering of observations/simulations around the line of perfect agreement (Figure 7). These poor predictions were confirmed by the very low *NSE* and r^2 (< 0 and < 0.20, the latter except for burned and mulched soils of chestnut forests, r^2 = 0.79) (Table 5).

This inaccuracy of the MUSLE equation for simulating soil loss suggested the need to calibrate the model. We were forced to input C-factors equal to one, trying to reduce the model underestimation. This attempt was however disappointing, since the reliability of the calibrated MUSLE remained unsatisfactory. In addition, after calibration, the differences between predictions and observations were high, over 76% (Figure 7), and the evaluation indexes were poor. The values of *NSE* and r^2 were negative and lower than 0.20 in all forests (except r^2 = 0.79 in burned and mulched soils of chestnut), and the underestimation of soil loss was always high (as shown by the positive *PBIAS*, > 0.76) (Table 5).

The unsatisfactory performance of the MUSLE model was quite surprising, despite calibration. We verified whether the applications of the Q and q_p simulated by the SCS-CN method rather than the observed values had influenced the erosion prediction capability of the MUSLE model. In modelling soil loss, the use of measured runoff volume and peak flow is suggested to improve the model accuracy, although this practice is often impossible in ungauged plots without runoff monitoring devices. However, the erosion prediction capability of the MUSLE model did not noticeably improve using the observed values of the hydrological variables in the runoff factor, since r^2 and *NSE* were lower than 0.50 and negative, respectively, and *PBIAS* was always over the acceptance limit of 0.55 for erosion prediction, as suggested in the literature.

The over-prediction of the MUSLE model is common in some studies carried out in different environments and soil conditions (Chen and Mackay, 2004; Shen et al., 2009). These authors reported that the low prediction capability could often be due to the fact that the model is applied in contexts that are different from the environments where the MUSLE was developed. More generally, the authors of (Flanagan and Nearing, 1995; Nearing, 2000) highlighted that small soil losses are usually over-predicted by USLE-family models.

Table 5. Statistics and indexes evaluating the runoff prediction capability of MUSLE model in forest plots subject to prescribed fire and soil mulching with fern.

	Soil Loss	**Mean (tons/ha)**	**Standard Deviation (tons/ha)**	**Minimum (tons/ha)**	**Maximum (tons/ha)**	r^2	***NSE***	***PBIAS***
Pine	Unburned							
	Observed	0.02	0.02	0.00	0.05	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.04	−0.73	1.00
	Simulated (calibrated)	0.01	0.01	0.00	0.01	0.04	−0.55	0.76
	Burned							
	Observed	0.12	0.20	0.01	0.52	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.01	0.53	−0.06	0.97
	Simulated (calibrated)	0.01	0.01	0.00	0.02	0.53	−0.01	0.92
	Burned and mulched							
	Observed	0.02	0.04	0.00	0.11	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.01	−0.37	1.00
	Simulated (calibrated)	0.01	0.00	0.00	0.01	0.01	−0.21	0.77
Chestnut	Unburned							
	Observed	0.05	0.05	0.00	0.15	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.18	−0.90	1.00
	Simulated (calibrated)	0.00	0.00	0.00	0.01	0.18	−0.86	0.95
	Burned							
	Observed	0.16	0.21	0.00	0.52	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.20	−0.25	0.99
	Simulated (calibrated)	0.01	0.00	0.00	0.01	0.20	−0.22	0.97
	Burned and mulched							
	Observed	0.03	0.05	0.00	0.15	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.79	−0.05	1.00
	Simulated (calibrated)	0.00	0.00	0.00	0.00	0.79	−0.02	0.97
Oak	Unburned							
	Observed	0.05	0.02	0.00	0.07	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.05	−4.15	1.00
	Simulated (calibrated)	0.01	0.01	0.00	0.02	0.05	−2.86	0.83
	Burned							
	Observed	0.11	0.03	0.07	0.15	-	-	-
	Simulated (default)	0.01	0.00	0.00	0.01	0.03	−1.35	0.94
	Simulated (calibrated)	0.02	0.01	0.00	0.02	0.03	−1.01	0.86
	Burned and mulched							
	Observed	0.07	0.03	0.04	0.12	-	-	-
	Simulated (default)	0.00	0.00	0.00	0.00	0.00	−3.63	1.00
	Simulated (calibrated)	0.01	0.01	0.00	0.02	0.00	−2.47	0.84

Notes: r^2 = coefficient of determination; *NSE* = coefficient of efficiency; *PBIAS* = coefficient of residual mass.

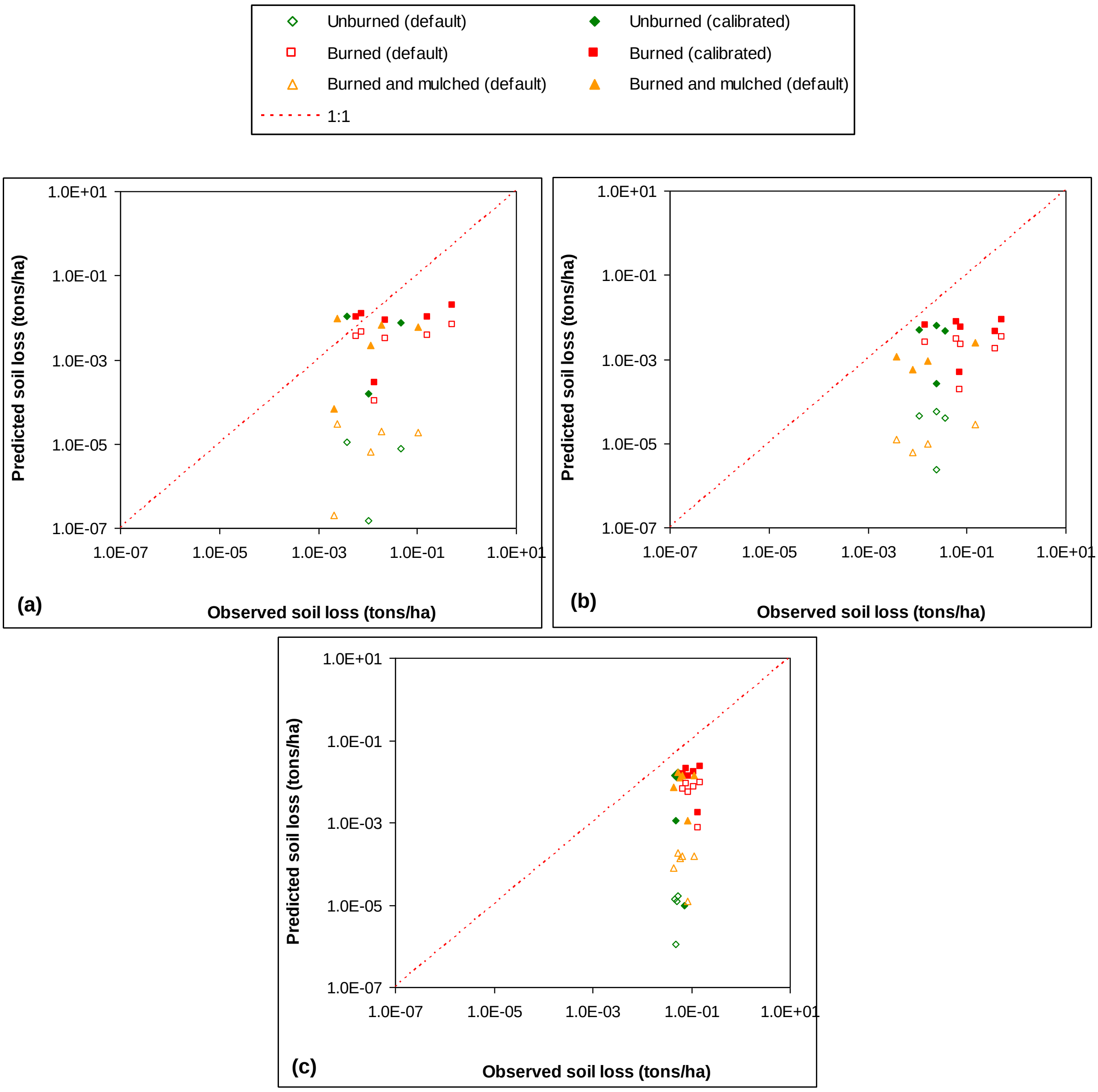

Figure 7. Scatter plots of soil losses observed in forest sites ((**a**), pine; (**b**), chestnut; (**c**), oak) subject to prescribed fire and soil mulching with fern vs. predicted using the MUSLE model. Values are reported on logarithmic scales.

3.2.4. USLE-M Model

As found for the MUSLE equation, the erosion predictions using the uncalibrated USLE-M were inaccurate, as visually shown in the relevant scatter plots (Figure 8). All the values of the evaluation indexes were unsatisfactory, since r^2 was lower than 0.41, *NSE* was negative, and *PBIAS* indicated strong model underprediction or overprediction ($|PBIAS| > 0.74$; except for unburned, as well as burned and mulched, soils of pine, with *PBIAS* equal to 0.43–0.44, and therefore acceptable). Moreover, the differences between the mean or maximum values of predicted soil losses and the corresponding observations were always higher than 40% (with one exception, unburned soils of chestnut, 29%) (Table 6). Moreover, for this erosion model, we ascribed this poor performance to the tendency of hydrological models to overestimate and underestimate the lower and higher soil losses, respectively (Bezak et al., 2021; Kinnell, 2003; Nearing, 2000). According to (Bezak et al., 2021), the tendency for USLE-family models to overpredict low soil losses could be improved by incorporating an erosivity threshold in precipitation that must be exceeded before any sediment is generated.

The USLE-M model inaccuracy was removed thanks to calibration for the conditions of burned, and burned and mulched soils of all the forest species, while the erosion predictions provided by the calibrated USLE-M equation were still unsatisfactory for the unburned plots. For the latter soil condition, r^2 was lower than 0.14 and the *NSE* was negative (Table 6). In contrast, these evaluation indexes were over 0.56 (r^2) (except in the burned soil of oak, r^2 = 0.23) and 0.67 (*NSE*) in burned soils (mulched or not) of all forests, and the |*PBIAS*| was lower than 0.17. The latter index reveals that in some soil conditions and forest species the model generally underpredicted erosion (burned soils, treated or not, of oak, and burned plots of chestnut), while, in the other cases, a slight tendency for the overestimation of soil loss was found). Moreover, the values of *PBIAS* were well below the acceptance limit of 0.55 stated in the literature ((Gupta et al., 1999; Moriasi et al., 2007), see also Section 2.6). In addition, for burned soils of oak, the erosion prediction capability of the USLE-M equation can be considered as satisfactory, although the r^2 was low (0.23). As a matter of fact, both the *NSE* and *PBIAS* indexes complied with the acceptance limits (*NSE* > 0.36 and *PBIAS* < 0.55), and the differences between the mean or maximum values of the observations and predictions was only 8.5%. This statement is a proof that sometimes r^2 may be misleading in model evaluation (Legates and McCabe Jr, 1999; Willmott, 1982), since it measures the scattering of values around the regression line and not around the line of perfect agreement.

The contrasting performances of the USLE-M model in predicting erosion between unburned and burned soils contrasts with the findings of (Thompson et al., 2019), who reported insignificant impacts on erosion estimates between burned and non-burned forests.

Overall, for the USLE-family models, a calibration process has been considered necessary by several authors for improving their prediction accuracy. For instance, (Bagarello et al.,

2015) and (Di Stefano et al., 2016), applying the USLE-M in plots in Western Sicily (Italy) and under different soil conditions, highlighted the importance of the calibration process to allow its adaption to the different climatic and edaphic conditions.

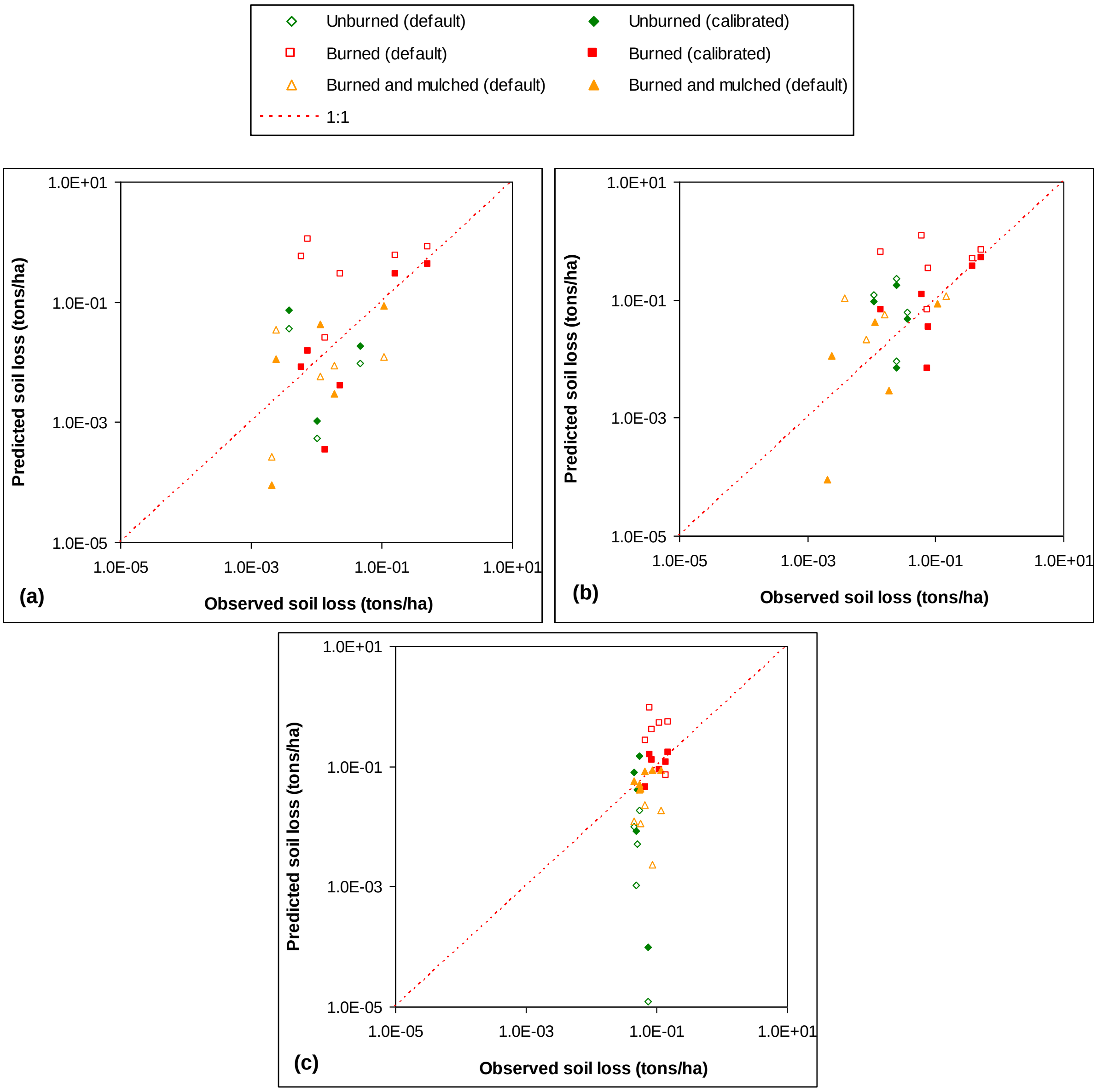

Figure 8. Scatter plots of soil losses observed in forest sites ((**a**), pine; (**b**), chestnut; (**c**), oak) subject to prescribed fire and soil mulching with fern vs. predicted using the USLE-M model. Values are reported on logarithmic scales.

Table 6. Statistics and indexes to evaluate the runoff prediction capability of the USLE-M model in forest plots subject to prescribed fire and soil mulching with fern.

	Soil Loss	Mean (tons/ha)	Standard Deviation (tons/ha)	Minimum (tons/ha)	Maximum (tons/ha)	r^2	*NSE*	*PBIAS*
Pine	Unburned							
	Observed	0.02	0.02	0.00	0.05	-	-	-
	Simulated (default)	0.01	0.01	0.00	0.04	0.14	−0.93	0.43
	Simulated (calibrated)	0.02	0.03	0.00	0.07	0.14	−2.33	−0.15
	Burned							
	Observed	0.12	0.20	0.01	0.52	-	-	-
	Simulated (default)	0.57	0.38	0.02	1.08	0.12	−5.84	−3.65
	Simulated (calibrated)	0.12	0.19	0.00	0.42	0.87	0.90	−0.01
	Burned and mulched							
	Observed	0.02	0.04	0.00	0.11	-	-	-
	Simulated (default)	0.01	0.01	0.00	0.03	0.01	−0.20	0.44
	Simulated (calibrated)	0.03	0.03	0.00	0.09	0.81	0.81	−0.08
Chestnut	Unburned							
	Observed	0.05	0.05	0.00	0.15	-	-	-
	Simulated (default)	0.06	0.09	0.00	0.23	0.14	−3.97	−0.29
	Simulated (calibrated)	0.05	0.07	0.00	0.18	0.14	−2.57	0.00
	Burned							
	Observed	0.16	0.21	0.00	0.52	-	-	-
	Simulated (default)	0.49	0.41	0.00	1.18	0.05	−4.06	−1.98
	Simulated (calibrated)	0.16	0.21	0.00	0.53	0.95	0.96	0.01
	Burned and mulched							
	Observed	0.03	0.05	0.00	0.15	-	-	-
	Simulated (default)	0.05	0.05	0.00	0.12	0.41	0.25	−0.98
	Simulated (calibrated)	0.03	0.05	0.00	0.13	0.86	0.88	−0.17
Oak	Unburned							
	Observed	0.05	0.02	0.00	0.07	-	-	-
	Simulated (default)	0.01	0.01	0.00	0.02	0.05	−3.13	0.87
	Simulated (calibrated)	0.05	0.06	0.00	0.15	0.05	−4.67	−0.02
	Burned							
	Observed	0.11	0.03	0.07	0.15	-	-	-
	Simulated (default)	0.45	0.29	0.07	0.93	0.07	−40.70	−3.26
	Simulated (calibrated)	0.12	0.05	0.04	0.17	0.23	0.67	0.08
	Burned and mulched							
	Observed	0.07	0.03	0.04	0.12	-	-	-
	Simulated (default)	0.02	0.01	0.00	0.04	0.05	−2.00	0.74
	Simulated (calibrated)	0.07	0.02	0.04	0.09	0.56	0.78	0.05

Notes: r^2 = coefficient of determination; *NSE* = coefficient of efficiency; *PBIAS* = coefficient of residual mass.

Since five (*K*, *L*, *S*, *C*, and *P*) of the six USLE-factors are common in the two models under each soil condition, it is possible to compare the effects of the R-factor on the predicted soil losses. This indicates that, under the experimental conditions, the soil loss is mainly due to rainsplash erosion. This fact derives from the better performance of the USLE-M model, whose rainfall erosivity is based on the R-factor, compared to the MUSLE model. In contrast, the lower accuracy in simulating erosion shown by the latter equation, which includes parameters related to surface runoff, indicates the minor role of the soil loss produced in our plots by overland flow, which determines particle detachment. However, these statements depend strictly on the small scale of the experimental plots (only a few square meters), as well as the low number of plots, and thus must be verified at larger scales. As carried out for the MUSLE model, we ran the USLE-M model using the observed Q_R rather than the value calculated using the runoff volume predicted by the SCS-CN model, but the erosion prediction capability of USLE-M did not appreciably improve (data not shown). This means that the model can be applied to ungauged plots (that is, without equipment to measure runoff and peak flow) without losing accuracy when predicting soil losses in burned soils.

As previously mentioned, the MUSLE and USLE-M models had not been applied to predict erosion in fire-affected areas (neither by wildfires nor low-severity fires) before this study. Therefore, the results of the present study cannot be compared to similar experiments in the same or other environmental contexts. Comparisons with published studies that have evaluated the erosion prediction capability of USLE-family models are thus possible with reference to the RUSLE model. The authors of (Vieira et al., 2018) found that the latter equation is ideal for fast and simple applications (i.e., prioritization of areas-at-risk) in zones burned by wildfires in Portuguese pine forests; since, in this application, an *NSE* between 0.63 and 0.70, and *PBIAS* in the range −12% to −20% were achieved. In contrast, in burned forests of pine and eucalyptus, and in lands with post-fire management in Galicia (NW Spain), references (Karamesouti et al., 2016; Larsen and MacDonald, 2007) and (Fernández and Vega, 2018) found poor predictions of soil loss. Negative or very low positive *NSE* and high *PBIAS* were detected by these authors, since the RUSLE model overestimated the soil losses. These authors attributed the poor performance of the model to three factors: (i) the use of an inadequate kinetic energy equation of rainfall for the Mediterranean climate; (ii) the fact that soil water repellency, a key factor in post-fire soil hydrology (Nunes et al., 2018; Pereira et al., 2018), is not explicitly considered in the RUSLE model; and (iii) the poor ability of C- and K-factors to reflect changes in soil properties induced by fire. The attempts at calibration by tuning the R- and C-factors (Karamesouti et al., 2016) and by introducing a reduction factor for soil erodibility (to take into account stone cover of the experimental soils) (Larsen and MacDonald, 2007), or accounting for high contents of soil organic matter (Fernández and Vega, 2018), did not noticeably improved the RUSLE accuracy for predicting soil loss. These statements agree with those reported by (Bezak et al., 2021), who achieved a low reliability with the RUSLE equation in modeling sediment yields throughout the first year after fires of different severity in Colorado (USA). The same authors reported that the model accuracy was not improved by increasing the K-factor of RUSLE (which should simulate a decrease in aggregate stability after fire). Moreover, they advised that the use of RUSLE in unburned forests may be troublesome (as was found in our study, given the basically poor performance of the MUSLE and USLE-M models in the unburned plots), because overland flow is not common (Bezak et al., 2021; Dunne and Leopold, 1978). Since the main effects of fire are the changes in soil properties and surface covers (Alcañiz et al., 2018; Moody et al., 2013; Shakesby, 2011), which may induce noticeable changes in the K- and C-factors, all these studies concluded that these USLE-factors do not adequately describe soil modifications after fire. Our study has instead demonstrated that satisfactory predictions of erosion in soils burned by prescribed fire (with or without mulch treatment) can be provided by the USLE-M model when using a simple calibration of the C-factor and regardless of the specific process (e.g., soil water repellency, decrease in aggregate stability, depletion of organic matter content) that affects the post-fire soil hydrology. A limitation of this procedure is the need to calibrate the C-factor in each environmental context (that is, in soils of different types and textures) before applying the USLE-M model. However, this requirement seems to be an easy task, considering the low time and money requirements of a small plot installation, as well as the low monitoring period (less than one year), as in the present study. This consideration is in close accordance with (Vieira et al., 2018) and (Thompson et al., 2019), who suggested that estimations of C-factors for burned areas should be determined for each context and fire type, although previous works reported several indicative values (Karamesouti et al., 2016; Vieira et al., 2014). Therefore, the estimation of the site-specific C-factors using locally measured data, as was the aim of our study, increases the erosion prediction accuracy of USLE-family models (Kebede et al., 2021).

A limitation of our study is that the C-factors proposed in this investigation for the modeled soil conditions were calibrated using observations collected at only one study area. Furthermore, the low number of plots (due to budget limitations) in this study may explain the low accuracy of the tested methods for modeling runoff and soil losses, especially in the unburned plots. In some cases, this procedure can be misleading and must be validated in other environmental conditions or supported by external parameters. Bearing in mind the limitations of our study, the C factors proposed for the experimental soil conditions could be reliable, at least for burned soils of Mediterranean forests (treated with mulching or not), and fill the lack of similar values for USLE-M applications in the literature.

Overall, the investigation enriched the literature about post-fire hydrological modeling, which is not homogeneously distributed worldwide and is still far from being exhaustive (Hosseini et al., 2018; Prosser and Williams, 1998). The results of our study confirm the applicability of two very common hydrological models (SCS-CN and USLE-M). Moreover, the unsuitability of two other methods (Horton equation and MUSLE) was demonstrated, at least in various forests in Mediterranean areas, whose intrinsic conditions (e.g., very shallow soils, strong soil water repellency, and peculiar hydrologic regime) often make the available hydrological models unsuitable. These models were developed in other climatic contexts and not in fire-affected areas, and therefore may find limited applicability without targeted modifications (Vieira et al., 2018). This study supports the efforts of modelers in choosing the most suitable hydrological models after prescribed fires and mulching treatments in Mediterranean forests that are prone to risk wildfire. Moreover, our modeling experiment proposed important input parameters (*CN*s and C-factors) for loamy sand soils supporting broadleaf (oak and chestnut) and conifer (pine) forest species, and the proposed values can be used under the evaluated soil and climatic conditions.

4 Conclusions

This study has demonstrated the feasibility of the SCS-CN and USLE-M models for predicting surface runoff and erosion, respectively, in plots burned by prescribed fire and mulched with fern residues in three Mediterranean forests of pine, chestnut, and oak. In contrast, poor predictions of the modelled hydrological variables were provided by the models in unburned plots, and by the Horton and MUSLE models for all the soil conditions. This result answers the first research question about the feasibility of the four models for hydrological applications in Mediterranean burned forests.

Regarding the second research question about the optimal values of the input parameters for the tested models, the study has proved that a calibration process is a prerequisite for the tested models for accurate predictions of surface runoff and soil loss. After calibration, we suggest the following two sets of *CN*s and C-factors for applications of the SCS-CN and USLE-M models after prescribed fire and fern mulching in Mediterranean forests:

for runoff modelling *CN*s of 80, 83, and 79 (burned soils) and 65, 76, and 65 (burned and mulched soils) throughout the two to three months after fire, and 45, 47, and 41 (burned soils) and 32, 45, and 39 (burned and mulched soils) in the following period, for chestnut,

oak, and pine forests, respectively; for soil loss predictions C-factors of 0.521, 0.085, and 0.339 (burned soils) and 0.069, 0.060, and 0.040 (burned and mulched soils) immediately after a fire, and 0.052, 0.059, and 0.008 (burned soils) and 0.007, 0.050, and 0.001 (burned and mulched soils) in the following period, for chestnut, oak, and pine forests, respectively.
The different *CN*s and C-factors consider the variability of the hydrological response of soil after fire.
Despite the satisfactory accuracy shown by the SCS-CN and USLE-M models in this modeling experiment, these results are limited to the experimental conditions; however, they are encouraging for further applications of these conceptually simple and widely used models in analogous climatic and geomorphologic conditions. Further modeling studies should also enlarge the spatial scale from plots to watersheds. Once validated in a wider range of environmental contexts, these models may support land managers in controlling runoff and erosion in forest soils that are prone to hydrogeological risks.

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Future perspectives and research paths, and conclusive remarks

In spite of the ample literature about the use of prescribed fire and post-fire management of forests to reduce the wildfire risk and control the soil's hydrological response after fire, the preliminary analysis of the state-of-the-art has identified important research needs (with a special focus to the forests of Southern Italy) that this Ph.D. thesis aimed at addressing.

Regarding the hydrological aspects, in the first study we have demonstrated under simulated rainfalls that prescribed fire may worsen the soil's hydrological response to burning, but mulching is particularly effective in reducing the runoff volume and peak flow immediately after fire mainly in broadleaves species and less in conifers. The hydrological effects of both burning and mulching decrease over time until being not significant or even negligible some months after fire, and, in conifers, sometime detrimental.

A possible limitation of the first study was the point scale and the simulated rainfall in the experiment, and the erosion response of burned soils (treated or not) was not evaluated using the portable rainfall simulator. Therefore, in the following study the spatial scale of the hydrological analysis was enlarged to a plot experiment and to erosion monitoring. This investigation has demonstrated that, immediately after the prescribed fire, runoff and erosion significantly increase in all forest stands compared to the unburned soils, but these increases are much lower compared to the highest values reported by some studies. This window of disturbance after fire is limited to three-four months, and the pre-fire runoff and erosion rates of the soils are practically restored after five months. Soil mulching with fern is effective to limit the increase in the hydrological response observed in the burned soils, as shown by reductions in runoff coefficients and erosion by 30 to 80% detected in the experimental sites.

About the soil properties and covers evaluated in the third study, we have demonstrated that prescribed fire, although being a low-intensity fire, is able to induce significant changes in soil chemistry and surface, and that the magnitude of these changes depends on the soil property and forest species. The effects of the prescribed fire and mulching were often transient. Mulching with fern has been shown to be unable to limit the changes in chemical properties of soils.

About the ecological impacts of the prescribed fire and soil mulching, the fourth study has demonstrated that low-intensity burning enhances the initial recruitment of plants, since oak is a forest species that is adapted to fire, and that soil mulching may be partly synergistic with the prescribed fire. This post-fire technique does not increase acorn emergence and plant survival compared to the burned and untreated sites, but enhances plant height and root mass.

The modelling approach applied in the fifth study has confirmed the applicability of the SCS-CN and USLE-M models to predict surface runoff and erosion, respectively, in forests burned by prescribed fire and treated with fern under Mediterranean conditions. In contrast, poor predictions of the modelled hydrological variables were provided by the models in unburned plots, and by the Horton and MUSLE models for all the soil conditions. Moreover, the study has proposed optimal values of the input parameters of the tested models, and proved that the calibration process is a prerequisite for accurate predictions of surface runoff and soil loss.

Overall, this PhD thesis has shown by experimental investigations that, in forest sites with typical species of Southern Italy (as pine, chestnut and oak) and under Mediterranean semi-arid conditions, the use of prescribed fire is promising to reduce the wildfire risk, although this practice is surprisingly of uncommon use. However, we must pay caution to the worsening of the hydrological response of burned soils, which are left bare due to vegetation removal after fire and undergo changes of some important properties (e.g., contents of organic matter and nutrients, water infiltration and repellency). To control the soil's response to fire and avoid the hydrogeological risks in the delicate forest environments of Southern Italy, post-fire management actions are required, and this study has demonstrated the hydrological effectiveness of a cheap mulch material, such as the fern residues.

Overall, despite these results, we think that this study, although being useful to consolidate the use of prescribed fire and soil mulching as pre- and post-fire management actions on a broader scale, is not thoroughly exhaustive, since some research gaps are still open.

First, this study was limited to soils of a given type and validations in other sites with soils of different texture are needed to verify whether the same or similar effects are achieved.

Secondly, for budget limitations, the plot size of the experimental site was small, and therefore we suggest upscaling the experiments to the hillslope or better catchment scale. This extension would allow to consider the patchy nature of prescribed fire and the effects of mulching on hydraulic connectivity, and to give indications about the most suitable distribution of mulch material over soil (e.g., homogenous cover or patchy distribution).

Third, some important soil properties have not been considered (again for budget limitations, which have forced the authors to limit the analysed parameters). For instance, extending the attention to the microbiological properties of soils (e.g., enzyme contents, microbial respiration) may clarify other important effects on soil functionality, which were not studied in this investigation.

Fourth, the satisfactory capacity to simulate soil hydrology by the SCS-CN and USLE-M models in this study is limited to the experimental conditions, although being encouraging in view of further applications in analogous climatic and geomorphological conditions. Also in this case, we suggest enlarging the spatial scale from plots to hillslopes or catchments as well as extending the modeling exercise in other soil types, in order to achieve a large applicability in a wider range of environmental contexts.

Fifth, we have evaluated the effects of the prescribed fire and soil mulching only on three forest species, and, therefore, more research is needed in order to validate the outcomes of this study on different species and with other forest operations.

Sixth, since the effects of the prescribed fire on the wildfire prevention is limited in this study to one or two years, there is the need to verify the impacts of repeated prescribed fires (and, perhaps, of post-fire soil mulching) on the analysed environmental variables.

Finally, we think that the knowledge of the beneficial influences of prescribed fire and post-fire treatments on forest ecosystem that this Ph.D. thesis has studied is useful to forest managers and hydrologists, in order to consolidate sustainable fuel management practices for wildfire risk reduction (such as the prescribed fire), and cheap and effective ecological engineering techniques (such as the mulching with fern). Therefore, we wish that the results of this study may be of help for the crucial tasks of landscape management to develop sustainable plans for protection of the Mediterranean forest ecosystems.

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