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Fire and Soils in a Changing World

Issue Editors

Jorge Mataix-Solera
Miguel Hernández University of Elche, Spain

Diana Vieira
Joint Research Centre, Italy

Nicasio Jiménez-Morillo
University of Évora, Portugal

Xavier Úbeda
University of Barcelona, Spain

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Fire and Soils in a Changing World

Fire is a natural ecological process that shapes many ecosystems, however, the distortion of the natural fire regimes can provoke significant additional impacts. Land use changes in the last decades driven by human activity, and changes in climate due to global warming have led to projections of increased fire recurrence and corresponding socioeconomic and environmental impacts. During the last decades, an impressive bulk of research has been produced addressing fire impacts on soils, but the most recent shifts indicate more severe wildfires, as well as novel occurrences in non-fire-prone areas, which are less adapted and more vulnerable to this perturbation, highlights the need of translating the current knowledge to other conditions.

FuegoRED is a research network created in 2007 aiming for a more direct knowledge transfer between scientists and society. The X FuegoRED 2022 International Congress was held in Évora (Portugal), in October 2022. This Special Issue targeted this specific FuegoRED event but was open to additional research contributions on the effect of fires on the soil system and post-fire management.

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Editorial: Fire and Soils in a Changing World

Jorge Mataix-Solera[1], Nicasio T. Jiménez-Morillo[2], Xavier Úbeda[3] and Diana Vieira[4]*

[1]GETECMA – Soil Science and Environmental Technologies Group, Department of Agrochemistry and Environment, University Miguel Hernández, Elche, Alicante, Spain, [2]Instituto de Recursos Naturales y Agrobiologia de Sevilla (IRNAS-CSIC), Seville, Spain, [3]Department of Geography, University of Barcelona, Barcelona, Spain, [4]European Commission, Joint Research Centre (JRC), Ispra, Italy

Keywords: wildfires, megafires, prescribed fires, soil system, post-fire managements

Editorial on the Special Issue

Fire and Soils in a Changing World

Fire is a natural ecological process that shapes many ecosystems (Pausas and Keeley, 2009); however, the distortion of the natural fire regimes can lead to significant additional impacts in soils (Úbeda et al., 2023) but also to alternative geomorphic states (McGuire et al., 2024). Land use changes in the last decades driven by human activity, and changes in climate due to global warming have led to projections of increased fire recurrence and corresponding socioeconomic and environmental impacts (Rogers et al., 2020). During the last decades, an impressive bulk of research has been produced addressing fire impacts on soils (Almendros and González-Vila, 2012; Santín and Doerr, 2016), but the most recent shifts indicate more severe wildfires, as well as novel occurrences in non-fire-prone areas, which are less adapted and more vulnerable to this perturbation (Mataix-Solera et al., 2021), highlights the need of translating the current knowledge to other conditions. This Special Issue aims to contribute with a new series of five articles related to this topic.

García-Carmona et al. make an interesting summary of the main results of the role of biocrust and soil microbial communities in the recovery of Mediterranean soils after different post-fire management, guiding post-fire interventions such as burnt trees management, soil protection, and practices aimed at ecosystem restoration. The authors focus this mini-review in the colonization of biocrust-forming mosses in early successional stages after fire, and how different post-fire management treatments can affect their efficiency and soil microbial communities, evidencing the importance of these organisms and how we have to pay more attention in the short- and mid-term after fires.

Moreno-Rosso et al. assess the effects of prescribed burns of different burnt severity but covering the gap in understanding their effects at the micro-scale level. Prescribed fires are expected to cause low soil burn severity (SBS), but their effects vary due to numerous factors. The study was carried out in managed pine forest in western Mexico. The authors found that generally the top centimetres of soil structure are impacted by low SBS, while high SBS is restricted to the top 2 cm, evidencing disturbed soil structure and reddish aggregates. Immediate post-burn actions are needed to prevent soil erosion before rain even for prescribed fires.

Olivares-Martínez et al. explore the effects of surface and ground fires on the infiltration capacity of volcanic forest soil in pine-oak forests in central Mexico. Five sites with fires in the past 20 years were analysed. Tension-infiltration tests measured hydraulic conductivity and active macropores, revealing moderately high conductivity, with burned plots showing lower

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Avelino Núñez-Delgado, University of Santiago de Compostela, Spain

****Correspondence***
Jorge Mataix-Solera, jorge.mataix@umh.es

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infiltration capacity than control plots. A non-linear relationship was found between fire recurrence and soil properties, such as water repellency and pore concentration. While changes in soil water repellency and conductivity were observed, they do not necessarily indicate exceeded infiltration capacity. The authors conclude that more research is needed to assess if increasing fire frequency, driven by agricultural activities, could reduce soil resilience and lead to land degradation.

García-Braga et al. question what researchers understand by the long-term effects of fire on the soil. A review of the literature that exposes the impact of fire and residence time in the soil concludes that there are external variables, such as climate or substrate, and internal variables, such as soil type and its properties, that extends such effects through time. One variable that depends on the fire itself is its intensity, which is expressed in the severity of burning of the elements such as vegetation, fauna and soil. Forest management, suitable for each location, can prevent high intensity fires and thus improve the recovery time, understood as a natural system, is shorter and the soil is less negatively affected.

García-Redondo et al. analyse the wildfire-landscape dynamics in Baixa Limia Serra do Xurés Natural Park in Galicia from 2000 until 2020. Due to a change in land use resulting in a change in forest species and because of climate change, there has been a change in the fire regime. This translates into an increase in severity and a de-seasonalisation, that is, a potential extension or change of the fire season. Using available statistical and remote sensing data, authors have verified how there has been soil degradation and potential desertification in areas affected by recurrent and severe fires. The study provides valuable insights into the impacts of wildfires, changes in land cover, and post-fire soil-vegetation dynamics, which can inform management and conservation efforts in fire-prone mountainous regions.

In conclusion, this Special Issue contributes with knowledge about fire and soil and identifying issues that are important to address in future research. Climate change has already modified the fire regime, which translates into an increase in the intensity of fires, which are more severe, which implies a more serious impact on soil properties among others. It has also been proven that the abandonment of agroforestry activities has also induced this change in the fire regime. Given this scenario, it is important to advance in the knowledge of the type of management both pre- and post-fire to achieve less severe fires that in turn produce less drastic changes in soil properties.

AUTHOR CONTRIBUTIONS

This editorial has been drafted by lead guest editor JM-S. All authors contributed to the article and approved the submitted version.

CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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ORIGINAL RESEARCH
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Long-Term Cumulative Effects of Wildfires on Soil-Vegetation Dynamics in the "Baixa Limia–Serra do Xurés" Natural Park

Concepción García-Redondo[1], Montserrat Díaz-Raviña[2] and Adrián Regos[2,3,4]†*

[1]Area de Edafoloxía e Química Agrícola, Departamento de Bioloxía Vexetal e Ciencia do Solo, Facultade de Ciencias, Universidade de Vigo, Ourense, Spain, [2]Misión Biológica de Galicia del Consejo Superior de Investigaciones Científicas (MBG-CSIC), Santiago de Compostela, Spain, [3]Centre de Ciência i Tecnologia Forestal de Catalunya, Solsona, Spain, [4]BIOPOLIS Program in Genomics, Biodiversity and Land Planning, CIBIO/InBIO (Research Centre in Biodiversity and Genetic Resources), Vairão, Portugal

Wildfires are recognized as major contributors to forest loss and soil degradation on a global scale. Understanding the cumulative effects of fire regimes on forest ecosystems and soil dynamics necessitates a deeper exploration of wildfire-vegetation-soil interactions over the long term. This study delves into the wildfire-landscape dynamics within the "Baixa Limia Serra do Xurés" Natural Park, a region prone to fires in Galicia, Spain. By analyzing available statistical and remote sensing data, we identified significant shifts in fire regimes and landscape dynamics between the periods of 2000–2010 and 2010–2020. Our findings indicate a potential extension of the fire season, reflecting the impacts of climate change. Despite improvements in firefighting capabilities, the occurrence of large fires is on the rise in the Natural Park, underscoring the need for proactive management strategies in such areas. Notably, significant fire events in 2011, 2016, 2017, and 2020 extensively affected wooded areas, constituting the majority of the burned area. Shrubs and forests emerged as particularly vulnerable, with varying degrees of burn severity influencing post-fire vegetation recovery rates. While shrublands expanded their coverage between 2000 and 2010, rocky areas with sparse vegetation showed an increase over the subsequent decade (2010–2020), indicating soil degradation and potential desertification in areas affected by recurrent and severe fires, especially within zones designated for the highest levels of protection (with fire rotation periods of less than 1 year). In conclusion, this study provides valuable insights into the impacts of wildfires, changes in land cover, and post-fire soil-vegetation dynamics, which can inform management and conservation efforts in fire-prone mountainous regions. Leveraging advanced remote sensing techniques enables the monitoring of cumulative soil degradation resulting from repeated wildfires over extended periods.

Keywords: burn severity, fire recurrence, protected areas, remote sensing, soil degradation

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Diana Vieira,
Joint Research Centre, Italy

****Correspondence***
Adrián Regos,
aregos@mbg.csic.es

†ORCID:
Adrián Regos
orcid.org/0000-0003-1983-936X

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INTRODUCTION

Wildfires are known to be one of the main drivers of forest loss and soil degradation worldwide (Pausas and Fernández-Muñoz, 2012; Díaz-Raviña et al., 2021; Tyukavina et al., 2022). High-intensity and recurrence wildfires strongly impact vegetation cover, increasing soil erosion and loss (Chandler et al., 1983; Shakesby, 2011; Díaz-Raviña et al., 2012; Regos and Díaz-Raviña, 2023). In Europe, four Mediterranean countries—Italy, France, Portugal, and Spain—rank among the top six nations grappling with the highest numbers of wildfires (San-Miguel-Ayanz et al., 2021). Spain and Portugal, in particular, stand out as the countries most affected by both the frequency of fires and burned areas. Official European statistics underscore the persistence of severe fire seasons in the last years—three of the most catastrophic fire seasons on record occurred within the past 6 years. The 2022 fire season is the second worst for EU, speaking about fire burnt area (San-Miguel-Ayanz et al., 2023). Spain and Portugal, respectively showing 267,947 ha and 110,097 ha, and Romania presenting 1,599 as the percentage of average, were the most affected regions. Moreover, large wildfires (those larger than 500 hectares) began occurring as early as June–July, preceding the traditional fire season (Rodrigues et al., 2023). Longer fire seasons and increased burnt area in the autumn season have been also recorded over the past decade. These deviations from the norm were correlated with unprecedented levels of fuel dryness, atmospheric water demand, and pyrometeorological conditions. Large wildfires accounted for 82% of the total burnt area, 47% of which took place within protected areas (Rodrigues et al., 2023).

Since 2017, approximately 35% of the total burnt area, exceeding 11,600 km^2, has occurred within the Natura 2000 network (San-Miguel-Ayanz et al., 2021), highlighting the vulnerability of these conservation zones. Socioeconomic factors, including the traditional use of fire for land management and the abandonment of traditional agricultural practices, together with more adverse fire-weather conditions due to climate warming have exacerbated wildfire hazards in many mountainous regions (Moreira et al., 2011). This escalating wildfire hazard underscores the pressing need to comprehend the spatial interactions among various factors driving wildfire-landscape dynamics in fire-prone regions, particularly in protected areas that serve as “living labs” for sustainability.

Forest ecosystems consist of two interconnected components: vegetation and soil. Wildfires impact both the aboveground vegetation and the belowground soil components, which are strongly interdependent (Zhang and Biswas, 2017; Pressler et al., 2019; Yuan et al., 2020; Dove et al., 2022). Forests play a crucial role in protecting soils from degradation and enhancing their functions, such as carbon sequestration, provision of food, fiber and fuel, water purification and soil contaminant reduction, climate regulation, nutrient cycling, habitat for organisms, flood regulation among others (Pereira et al., 2018). Additionally, forests provide a multitude of services that are directly or indirectly dependent on soil, including the provision of food products such as wild berries and mushrooms, timber and biomass for fuel, medicinal plants, support for pollination, and the regulation of oxygen and clean water (Pereira et al., 2018). Wildfires have the potential to induce a range of physical, chemical, and biological degradation processes in soil, leading to significant modifications in soil functions and the overall quality, quantity, and sustainability of burned forest areas. These alterations can have far-reaching implications for ecosystem health and resilience, as well as the climate change mitigation potential of affected landscapes (Neary et al., 1999; Certini, 2005; Martín et al., 2012; Lombao et al., 2015a; Zhang and Biswas, 2017). The alterations caused by fires to soil quality, microbial biodiversity, water availability, and post-fire erosion create a cycle of environmental degradation. This cycle jeopardizes the natural values that protected areas aim to preserve (Martín et al., 2009; Díaz-Raviña et al., 2012; Barreiro and Díaz-Raviña, 2021; Girona-García et al., 2021). Soil microorganisms, involved in over 95% of soil processes, play a crucial role for ecosystem recovery (Villar et al., 2004; Mataix-Solera et al., 2009; Barreiro and Díaz-Raviña, 2021; Certini et al., 2021). While these effects primarily impact the uppermost layers of the soil profile, high-frequency fires lead to the progressive thinning of soils. In extreme cases, this can result in complete soil destruction, leading to desertification and the formation of bare soils and rocky landscapes (Shakesby, 2011; Perez-Rey et al., 2023).

Numerous studies have demonstrated that the impact of wildfires on soils and the subsequent recovery of burned areas depend on various factors, including soil type, vegetation, topography, meteorological conditions during and after the fire, and the characteristics of the fire regime (such as severity, duration, and recurrence) (Pausas and Fernández-Muñoz, 2012; Lombao et al., 2015b; Francos et al., 2018; Dove et al., 2020; Lombao et al., 2020; Agbeshie et al., 2022). The impact of fire regimes on forest ecosystems is site-specific, meaning that conclusions drawn from studies conducted in one area may not necessarily apply to other locations with different conditions. Many studies focusing on the effects of wildfires on soils primarily examine short- or medium-term impacts following the most recent fire event. However, these studies often do not monitor the long-term evolution of post-fire properties or record the severity of the fires to which ecosystems are exposed, which are critical aspects for understanding and predicting ecosystem recovery. Thus, due to the limitations of time, data availability, logistics and investment, research concerning the long-term impact of fire regime on forest ecosystems is scarce (see, e.g., Francos et al., 2018). It is necessary to better understand the long-term effects of wildfires on soil properties, especially in fire-prone ecosystems, in order to develop greater insights into their resilience and capacity to respond to such repeated perturbations. These studies are especially relevant considering the climate change impacts on global fire activity (Moritz et al., 2012).

To better understand the long-term impact of fire regimes on forest ecosystems, particularly regarding soil and vegetation dynamics, it is essential to characterize the historic fire regime. Integrating this historical context with updated information gathered through remote sensing technologies holds significant promise for elucidating the long-term consequences of wildfire

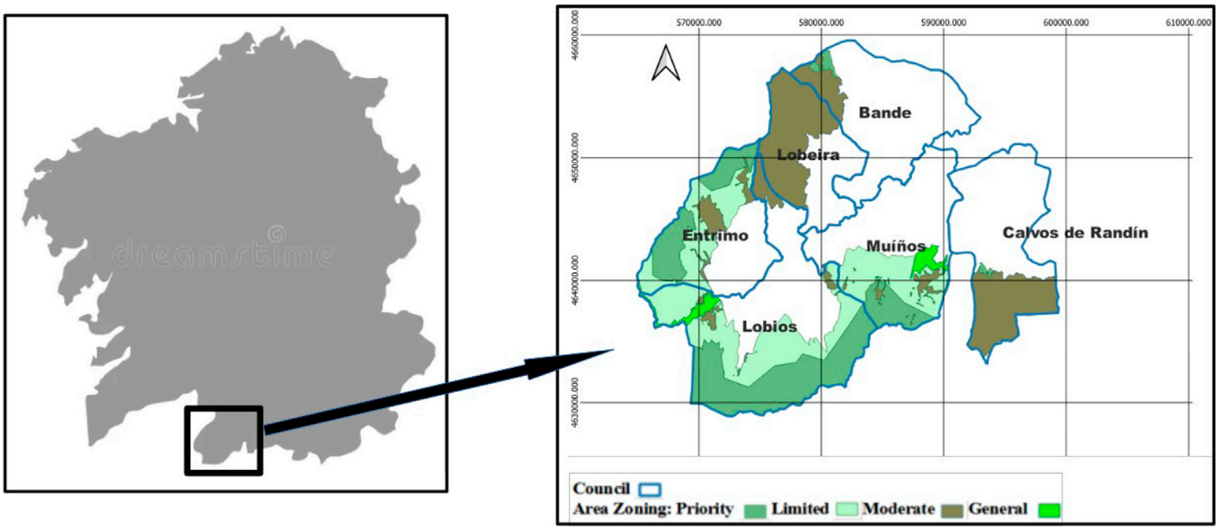

FIGURE 1 | Map illustrating the zoning of the different protection levels within the "Baixa Limia–Serra do Xurés" Natural Park, spanning six municipalities: Bande, Calvos de Randin, Entrimo, Lobeira, Lobios, and Muíños. Green colors depicts the level of protection according to the natural resources management plan of the Natural Park.

disturbances on soil degradation. This study focuses on assessing the cumulative impact of wildfires on soil-vegetation dynamics within the Natural Park "Baixa Limia–Serra do Xurés," a region known for its high susceptibility to fires in Galicia (Northwest Spain). Our approach considers the interactions and changes between soil properties and vegetation over time within an ecosystem (i.e., soil-vegetation dynamics). This concept encompasses the various processes and feedback mechanisms that occur between the soil and plant communities, directly affected by wildfires. To achieve this, we integrated statistical data with remotely sensed analyses on burn severity and land-cover changes to identify: 1) the vegetation types predominantly impacted by fires, 2) the primary land-cover transitions occurring in recently burned areas, and 3) the spatial distribution of fires occurring between 2010 and 2020, considering the administrative and protective designations of the natural park. We particularly focus on areas designated as having a high level of protection, characterized by heathlands and grasslands that are highly susceptible to frequent wildfires and potential soil erosion.

MATERIAL AND METHODS

Study Area

The Natural Park "Baixa Limia-Serra do Xurés" (hereafter referred to as BL-SXNP) stands as the largest Natural Park in Galicia, covering nearly 30,000 hectares spread across six municipalities: Bande, Calvos de Randín, Entrimo, Lobeira, Lobios, and Muíños (see **Figure 1**).

Designated under DECREE 64/2009 on February 19, this Natural Park boasts a multifaceted conservation status. It forms an integral part of the Natura 2000 Network, earning recognition as a special conservation area (ZEC ES1130001 "Baixa Limia") and a special protection area for birds (ZEPA ES0000376 "Baixa Limia – Serra do Xurés"). Notably, UNESCO declared it a Transboundary Biosphere Reserve, together with the "Peneda–Gerês" National Park on 27 May 2009 (Macedo et al., 2009). This designation underscores the area's commitment to fostering a sustainable relationship between its inhabitants and the natural environment, aiming to balance the conservation of biological and cultural diversity with economic and social development.

The climate in the BL-SXN is temperate oceanic sub-Mediterranean, with a mean annual temperature of 8°C–12°C and a mean annual precipitation of 1,200–1,600 mm, which involves a significant water shortage in summer. According to the Köpen-Geiger classification system, it corresponds to the "Csb" class. The soil, developed over granitic rocks, had a sandy texture and the range values of all physical and chemical properties are associated with poor soils (acid pH and low nutrient availability and organic matter content) (Rodríguez-Lado et al., 2018). The study area encompasses climatic conditions characteristic of both the Mediterranean and Atlantic regions, boasting a diverse array of 26 distinct habitat types, six of which are designated as priority habitats. Of particular significance are the shrub habitats, notably EU priority habitat 4020 "Atlantic wet heaths of *Erica ciliaris* and *Erica tetralix*," which are particularly vulnerable to wildfires and are therefore prioritized for conservation efforts. The preservation of shrubs and rocky formations from an ecological standpoint carries considerable weight, establishing this area as one of Galicia's most compelling regions of interest (Decree 401/2009 of October 22) (Macedo et al., 2009).

Wildfires pose a significant threat across all municipalities within the park. To mitigate the impact of fires throughout Galicia, zoning based on spatial fire risk has been implemented since 18 April 2007. This zoning classifies territories into areas of low, medium, and high fire risk, with heightened measures in high-risk zones. Additionally, Law 3/2007 of 9 April defines high fire risk areas and parishes of high incendiary activity, where extraordinary measures are warranted to prevent fires and protect forests from their impacts. In alignment with the Prevention and Defense Plan against Forest Fires of 2023 (de Galicia, 2023), 40 parishes of high incendiary activity have been identified, including four within the natural park and one in close proximity.

For effective management of activities within the Natural Park, the territory is delineated into various zones with differing levels of protection, operationalized through zoning. These zones, categorized based on the significance of natural values, serve as a pivotal management tool, regulating permissible and prohibited uses:

- Zone I. *Priority Conservation Interest* (Reserve): These are areas that require a high degree of protection due to harboring the highest natural, scientific, and landscape values, as well as the uniqueness of their habitats, species, and communities.
- Zone II. *Limited Use*: It consists of areas that have a high degree of naturalness and can support a certain level of public use, oriented towards research, education, environmental interpretation, and controlled nature viewing. The conservation of its resources and values is guaranteed while allowing certain primary uses.
- Zone III. *Moderate Use*: This category includes lands where natural formations, generally of medium quality and uniqueness, have undergone a higher degree of humanization, or have good capacity to support more intense public use. In these areas, agricultural and livestock uses, and traditional exploitation is allowed.
- Zone IV. *General Use*: These are areas that have lower quality within the protected natural space. They may be used for locating facilities for public use.

Fire Regime Characterization

The available fire perimeter dataset (collected on the ground via GPS) for the entire Autonomous Community of Galicia starts from the year 2010. The numerical datasets utilized in this study, spanning from years 1983–2020, are sourced from two distinct origins. Data spanning from 1983 to 2010 are derived from the official statistics of the Ministry for the Ecological Transition and Demographic Challenge at the municipal level (MITECO, 2023). Fire data statistics for the period 2010–2020 are gathered from the Department of Rural Environment of the Xunta de Galicia (hereafter, the DGDM database).

The dataset from the DGDM comprises annual records of fire incidents for each municipality. Each entry includes information such as the onset date, duration, parish where the fire originated, burned forest area, and total affected area, all expressed in hectares, for each year between 2010 and 2020. The DGDM database encompasses fire events affecting areas larger than 0.01 hectares. However, incidents where the burned area is less than 1 hectare are not categorized as fires unless they affect more than 0.5 hectares of wooded terrain.

In our study, we classified the fire events based on their size, considering the fire-weather conditions prevailing during ignition and development:

1) Fire events smaller than 1 hectare typically denote burning of agricultural or forestry debris that did not get out of control.
2) Fires spanning 1–10 hectares often indicate incidents that momentarily escaped control but were swiftly contained due to favorable environmental conditions.
3) Incidents ranging from 10 to 100 hectares typically signify fires that unfolded under challenging meteorological conditions and prolonged previous drought, presenting very difficult-to-control fire-weather situations.
4) Fires surpassing 100 hectares, and up to 500 hectares, are characterized by adverse environmental conditions and several secondary fire outbreaks. Large Forest Fires, exceeding 500 hectares, manifest under exceptionally unfavorable meteorological conditions, often taxing firefighting resources due to their potential threat to inhabited areas.

To characterize the fire regime, we computed several statistics for the forest and shrubland land cover types, number of fires and burnt areas throughout the specified period across municipalities, and different protection levels. We also estimated fire recurrence, defined as the number of times an area is burned, and fire rotation, which is the time required to burn the equivalent of a specified area. Fire rotation was calculated as follows (Bond and Keeley, 2005; Agee, 2016):

$$Fire\,rotation = \frac{Period\ (in\ years)}{Proportion\,of\ area\,burned\,in\,that\ period}$$

In addition, we used burn severity maps derived from García-Redondo et al. (2023). In particular, Landsat 5 imagery was used for 2010 and 2011, while Landsat 7 was used for 2012 due to cloud cover rendering Landsat 5 imagery unusable. From 2013 onwards, Landsat 8 and later Sentinel 2 data were employed. All images were processed to correct for geometric distortions and atmospheric effects using the "Dark Object Subtraction" methodology. This correction enhanced the reliability of the data by converting top-of-atmosphere reflectance to surface reflectance.

The NBR is a widely used index derived from satellite imagery that measures the difference in reflectance between pre- and post-fire conditions, specifically capturing changes in vegetation and soil characteristics. In our study, burn severity maps were produced using the NBR calculated from pre- and post-fire satellite images. The NBR effectively highlights areas of high burn severity where vegetation and organic matter have been significantly altered, thus providing insights into the impact on both soil and canopy levels. High NBR values indicate areas with severe canopy damage and soil exposure, while lower values correspond to less affected regions. Normalized Burn Ratio (NBR) was calculated for each year, which measures the ratio between near-infrared (NIR) and short-wave infrared (SWIR) radiation. NBR helps identify burned areas as NIR and SWIR bands of satellite sensors respond differently to burned vegetation (Eq. **1**):

$$NBR = (NIR - SWIR) / (NIR + SWIR) \qquad (1)$$

The NBR captures changes in vegetation and soil characteristics, indicating the degree of ecological change caused by fire. High NBR values correspond to areas with significant canopy damage and soil exposure, while lower values indicate less affected areas (all details in García-Redondo et al., 2023). The fire recurrence (defined as the number of fire events per spatial unit), was estimated by overlapping the annual burnt area maps. Thus, for each grid cell, we counted the number of times a fire event has occurred within the defined time frame.

Land-Cover Change Analysis

To analyze landscape changes, we used the Land Use and Land Cover (LC) maps derived from Cánibe et al. (2022). These maps were obtained from satellite images captured by Landsat 5 TM, Landsat 7 ETM+ and Landsat 8 OLI/TIRS sensors. These images were sourced from the USGS (United States Geological Survey) Earth Explorer database. Landsat Collection 1 Level-1 imagery was chosen since they are already geometrically corrected, ensuring consistent geo-registration with image-to-image tolerances of ≤12 m radial root mean square error (RMSE). All images were then calibrated and radiometrically corrected by using the "Dark Object Subtraction" methodology (see Cánibe et al., 2022).

Supervised classification methods were applied to generate LC maps. Training and validation areas were established for six LC classes, including 1) water bodies, 2) deciduous forests, 3) evergreen forests, 4) shrublands, 5) croplands and grasslands, and 6) bare soil with sparsely vegetated areas. Training and validation areas for each habitat class were established by on-screen digitizing in QGIS software, and consisted of a set of pixels identified over well-known homogeneous areas in each Landsat image, thus providing a reference spectral signature for each class. Two images per year were utilized to enhance spectral separability among LC classes, especially for deciduous forests. Four different classification algorithms were employed: Random Forest, Least Squared Support Vector Machines with Radial Basis Function Kernel, Monotone Multi-Layer Perceptron Neural Network, and Adaptative Boosting. Accuracy assessment was conducted using confusion matrices to calculate sensitivity and positive predictive power.

We analyzed the changes in the extent of each LC class for the period 2000–2010, and for the period 2010–2020, both inside and outside the burnt areas, to infer land cover conversions from one LC class to another caused by wildfires that took place in last 10 years. In particular, we quantified the spatial extent (in ha) of each LC class per year (2000, 2010, and 2020) from the different LC maps obtained from each classification algorithm. Boxplots were constructed using the R package "ggplot2" (Wickham, 2009). The contribution of each LC class to the change (i.e., conversion from one LC class to another) was shown through a transition matrix obtained by cross-tabulation of an ensemble LCC map (i.e., "majority vote" rule across the methods used for the supervised classification). Transition matrices were computed with the R package "lulcc" v.1.0.2 (Moulds, 2017). Statistical analyses and graphical representations were conducted using RStudio. Additionally, maps were generated utilizing both the RStudio software application and the QGIS application.

RESULTS

Fire Regime Characterization

The analysis of fire occurrences spanning from the 1980s to the 2020s reveals distinct trends (**Figure 2**). Initially, there were few fires in the 1980s, but this number increased exponentially in the 1990s, reaching up to 700 fires in 1997, before decreasing throughout the 2000s, 2010s, and 2020s, eventually returning to levels comparable to or even lower than those seen in the 1980s (**Figure 2**). This fire regime shift is attributed to the implementation of the Prevention and Defense Plan against Forest Fires by Xunta de Galicia in 1999. Interestingly, the number of fires does not correlate with the burnt area, with many years showing relatively low burnt areas despite varying fire frequencies, mainly due to the prevalence of small fires (<100 hectares) that are effectively contained. Surveillance and management strategies are crucial in combating fires. The implementation of the Prevention and Defense Plan against Forest Fires, which encompasses both aspects, has been the primary factor in reducing the number of fires and the area burned. This reduction has been consistently maintained over time.

Exceptions to this pattern occurred in 1985, 1989, 1998, 2005, 2011, 2015, 2016, and 2020, with 1998 showing the most significant deviation, where over 8,000 hectares were burned (**Figure 2**). Over the period from 1983 to 2020, there was an overall increase in fire size, with small and medium fires dominating during the 1980s and 1990s, aside from 1998, which showed a large fire. However, the contribution of medium-sized fires increased during the 2000s, and by the 2010s, medium and large fires became more prevalent, leading to larger burnt areas, particularly evident in 2011, 2015, 2016, and 2020.

In terms of fire seasonality, the number of fires remained relatively stable, with spring consistently recording the highest values, followed by autumn and summer (see **Figure 3**). However, statistics indicate a concerning trend: an increase in the burnt area during the autumn season over the past decade. This observation suggests an extension of the fire season (**Figure 3**).

Additionally, fire occurrence also varied across municipalities, with "Muíños" experiencing the highest number of fires following by "Entrimo," "Lobeira," and "Lobios" with the largest burnt areas due to large fires (see **Supplementary Figures S1, S2** for fire prevalence at the municipal and parish level, respectively). Although the proportion of fires remains relatively consistent

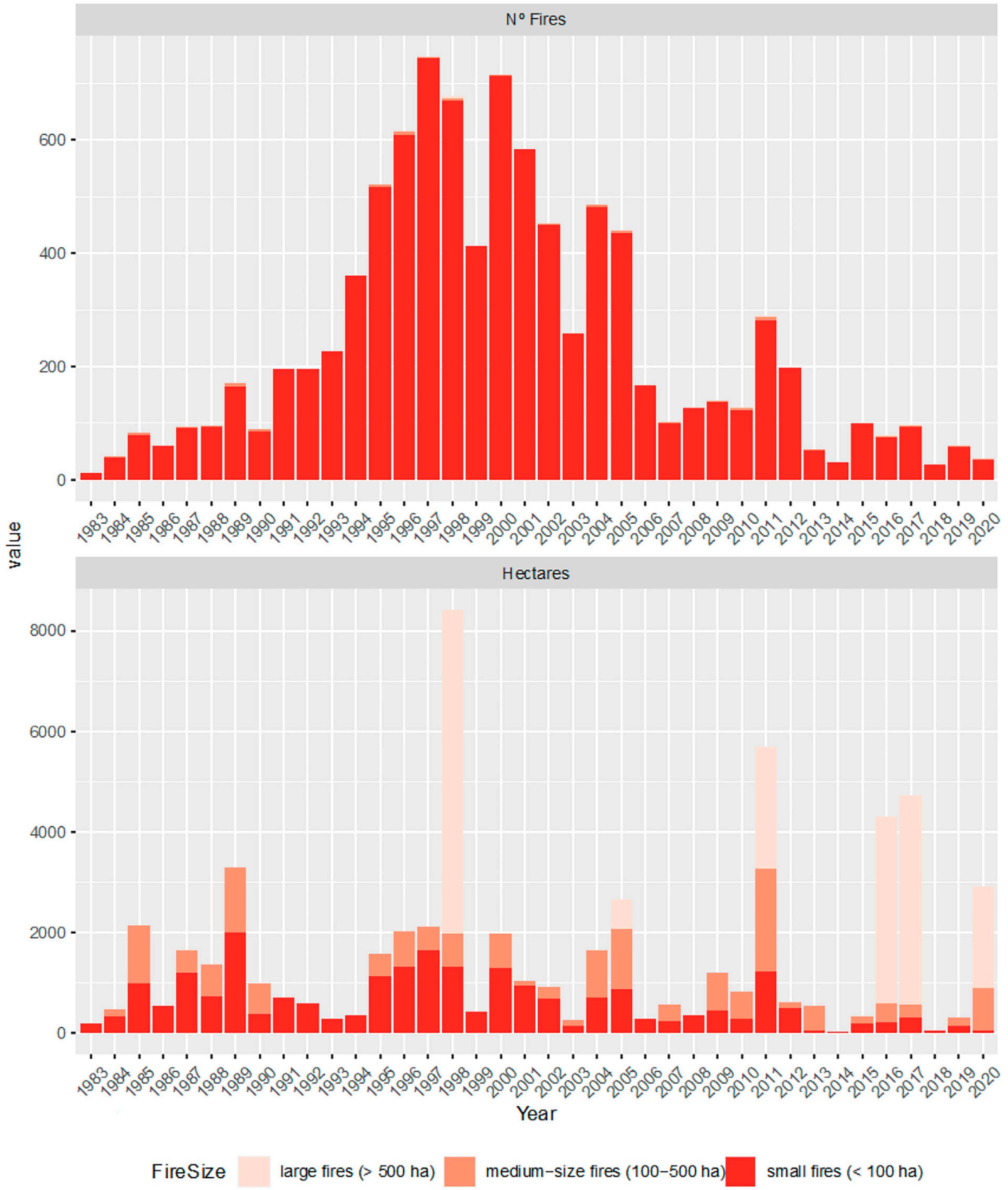

FIGURE 2 | Fire frequency (number of fires) and burnt area (hectares) in the municipalities within the Natural Park from 1983 to 2020. The burnt area is represented by fire sizes. Additionally, a dotted black line indicates the fire regime observed in the last decade, which was subjected to more detailed analysis.

across seasons, the burnt area is significantly larger during the high-risk season, typically from July to October. During the period from 2010 to 2020, the burnt area in the high-risk season is nearly five times greater than the area burned throughout the rest of the year, despite a similar total number of fires (**Table 1**). The most notable difference is observed in fire-size group G5, with the number of fires in G4 close to doubling (**Table 1**).

FIGURE 3 | Fire frequency (number of fires) and burnt area (hectares) in the municipalities within the Natural Park from 1983 to 2020, categorized by each season.

TABLE 1 | Number of fires produced according to the size of the fire surface, and total surface area affected by each group, in the six municipalities in the period 2010–2020 during the months of maximum risk (July–October) and the rest of the year (November–June).

2010–2020	Remaining year			High-fire-risk season		
	Num	Area (ha)	Area/N°	N°	Area (ha)	Area/N°
G1 (events < 1 ha)	331	80.36	0.24	386	76.93	0.20
G2 (Fires 1–9.9 ha)	178	544.05	3.06	110	365.02	3.32
G3 (Fires 10–99.9 ha)	35	1,012.97	28.94	42	1,208.19	28.77
G4 (Fires 100–499.9 ha)	7	1,888.38	269.77	12	2,925.85	243.82
G5 (Large Fires, >500 ha)	0	0	—	6	12,338.65	2056.44
Total	**551**	**3,525.76**	**6.40**	**556**	**16,914.64**	**30.42**

Bold values highlight the total values.

Fire Impacts on Vegetation

Official fire statistics indicate that wildfires have primarily affected shrubland covers within the six municipalities of the Natural Park (**Figure 4**). Exceptions to this trend were observed only in the years 2016 and 2017, during which the extent of forested areas affected by wildfires exceeded 1,500 hectares (**Figure 4**).

In the last 10 years (2010–2020), wildfires in the Natural Park affected over 14,642 hectares dominated by shrublands (representing 71.63% of the municipality) and 5,798 hectares of forested areas (approximately 28.37%, see **Table 2**).

In the past 5 years, wooded areas have shown increased vulnerability within the Natural Park. Of particular significance are the years 2016 and 2017, during which burnt wooded areas experienced the most significant degradation compared to shrubland areas during that period (**Figure 4**). Regarding fire recurrence, a large proportion of the Natural Park has burned at least once, with some areas burning up to

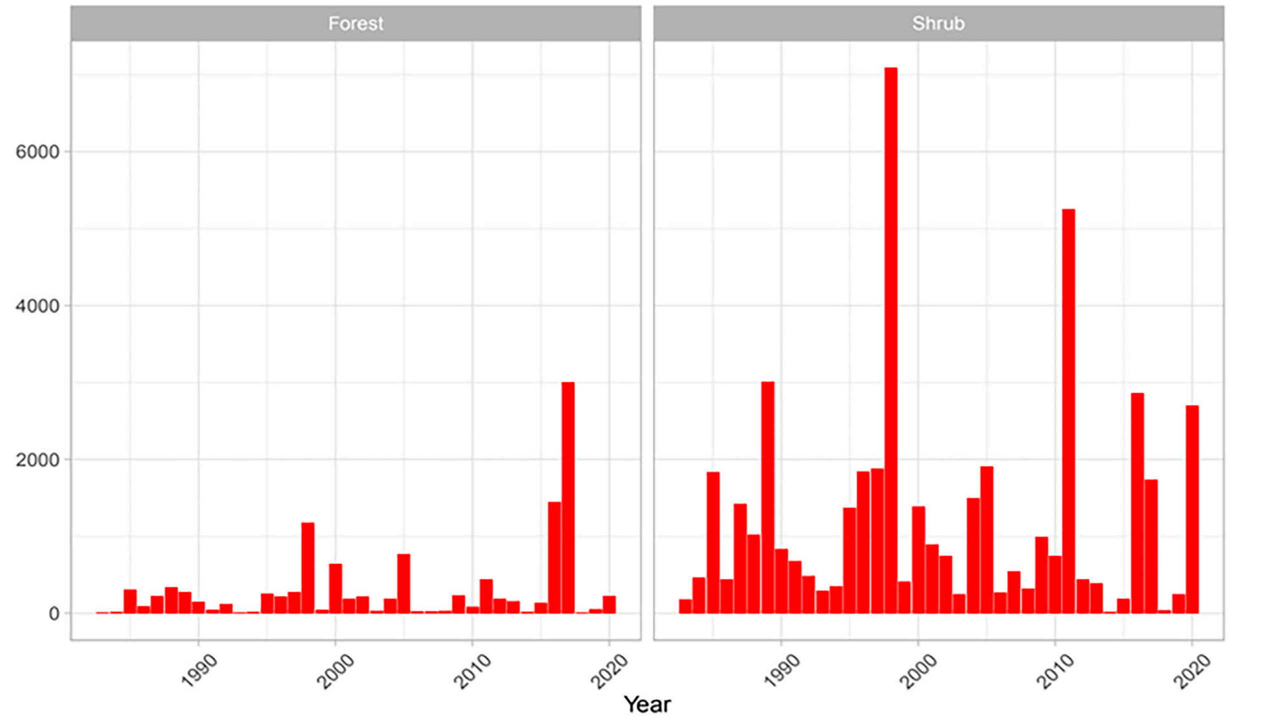

FIGURE 4 | Area burnt (in hectares) over the period 1983–2020 in forests and shrublands in the six municipalities included in the Natural Park "Baixa Limia Serra do Xurés."

TABLE 2 | Number of fires by fire-size group and total area, wooded and open, affected by group, in the six municipalities during the years 2010–2020.

Municipality	No of fires	Burnt forest area	Burnt shrub area	G1	G2	G3	G4	G5
Bande	108	189.25	577.8	71	31	5	1	0
Calvos	246	210.26	1,199.31	146	81	15	4	0
Entrimo	91	1,414.3	1,990.09	50	32	7	1	1
Lobeira	115	124.34	1,210.08	77	19	14	5	0
Lobios	298	3,514.58	7,284.72	207	64	18	5	4
Muíños	249	345.49	2,380.18	166	62	17	3	1
Total	**1,107**	**5,798.22**	**14,642.18**	**717**	**289**	**76**	**19**	**6**

G1: affected area <1 ha; G2: affected area 1–9.99 ha; G3: affected area 10–99.99 ha; G4: affected area 100–499.99 ha; G5: Large forest fires >500 ha. Bold values highlight the total values.

five times in the last 10 years (approximately every 2 years) (**Figure 7**).

Land-Cover Changes in Burnt Areas

The analysis of land cover change reveals significant shifts between the periods 2000–2010 and 2010–2020 (see **Figure 5**). During the latter period, there was a notable increase in shrubland coverage, accompanied by a decrease in shallow soils with sparse vegetation (hereafter: "rocky areas"). However, this trend reversed over the subsequent decade, with a decrease in shrublands in favor of rocky areas. Similar patterns were observed in evergreen forest cover, albeit to a lesser extent than shrubland. The observed pattern was consistent within the areas affected by wildfire between 2010 and 2020 compared to the rest of the Natural Park.

The intra-box plot variability illustrates that the land cover type most affected by the algorithm used for classifying satellite images was "shrubland." Despite this uncertainty, the period from 2000 to 2020 showed a predominant transition from rocky areas (i.e., shallow soils with sparse vegetation) to shrublands (see **Figure 6**). However, the subsequent decade a reversal in this trend was found, marked by a conversion from shrubland to rocky areas, and to a lesser extent, from evergreen forest (predominantly pine plantations) to shrubland. Notably, a significant portion of rocky areas in 2000 and 2010 remained unchanged after 10 years, despite natural successional processes that would typically favor vegetation encroachment in those areas.

The primary land cover transitions are spatially linked with areas previously affected by varying degrees of burn severity.

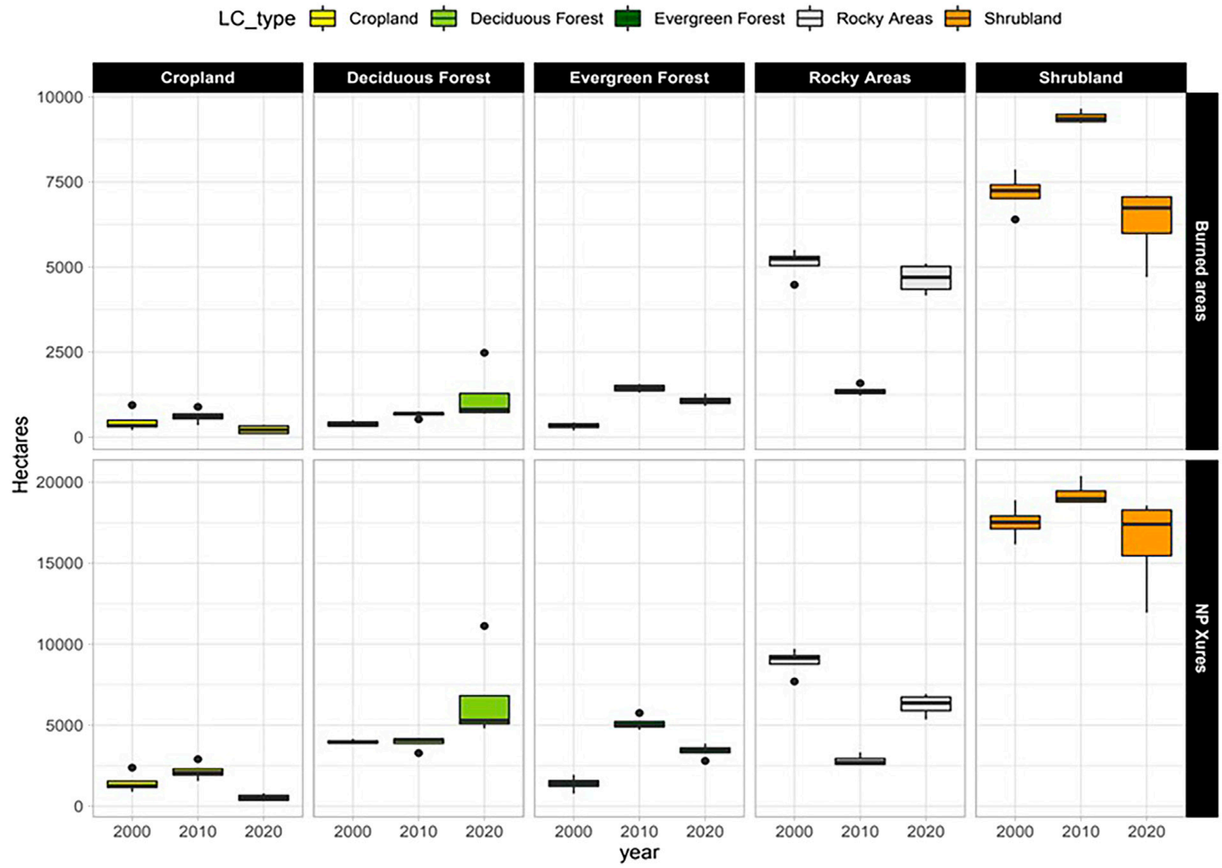

FIGURE 5 | Extent (in hectares) of each land cover class per year. The boxplots display the median, the 50th percentile (box), and 95th percentile (whiskers) confidence intervals. The intra-plot variability arises from the different estimations obtained from each classification algorithm.

There is a heterogeneous distribution of burn severity, which determines an uneven impact on vegetation, as reflected in the land cover composition of the year 2020 (see **Figure 7**). In fact, some of these areas have experienced multiple fire events, although the high rates of post-fire vegetation recovery might partially mask the spatial patterns of vegetation.

Fire Impacts in Protected Areas

The analysis of the spatial overlap between the areas burned during the period 2010–2020 and the zoning with different levels of protection revealed considerable variability in the burned area across different municipalities. Remarkably, wildfires impacted the areas with the highest levels of protection in the municipality of "Lobios" (approximately 3,423 and 3,097 hectares in priority and limited use zones, respectively; **Figure 8**). These areas, characterized by the highest levels of protection and restricted use, were also heavily affected by wildfires in the municipality of "Muíños" (burning approximately 646 and 2,213 hectares in priority and limited use zones, respectively; **Figure 8**). In "Entrimo," the priority areas experienced significant burning, totaling around 520 hectares, while those with limited use burned approximately 1,200 hectares (**Figure 8**). All these areas exhibited a fire rotation of less than 1 year, indicating that their entire extent burns annually (**Figure 8**). Conversely, Bande was the municipality least affected by fire, particularly in areas with the lowest protection levels, with fire rotations exceeding 200 years (**Figure 8**).

DISCUSSION

Overall, our findings underscore the complex interplay between wildfire occurrences, burn severity, land cover transitions, soil degradation and post-fire vegetation recovery rates within the Natural Park "Baixa Limia Serra do Xurés." Understanding these dynamics is crucial for effective wildfire management and conservation efforts in the Natural Park.

During the 2010–2020 period, six large wildfires (i.e., larger than 500 ha) occurred in 2011, 2016, 2017 and 2020. The largest among the six large wildfires occurred during 2017 and affected 3,485.70 hectares, of which 2,100.00 were wooded areas and 1,385.50 were open areas. The results show us that the majority of the burned area (60.08%, **Table 1**) is caused by

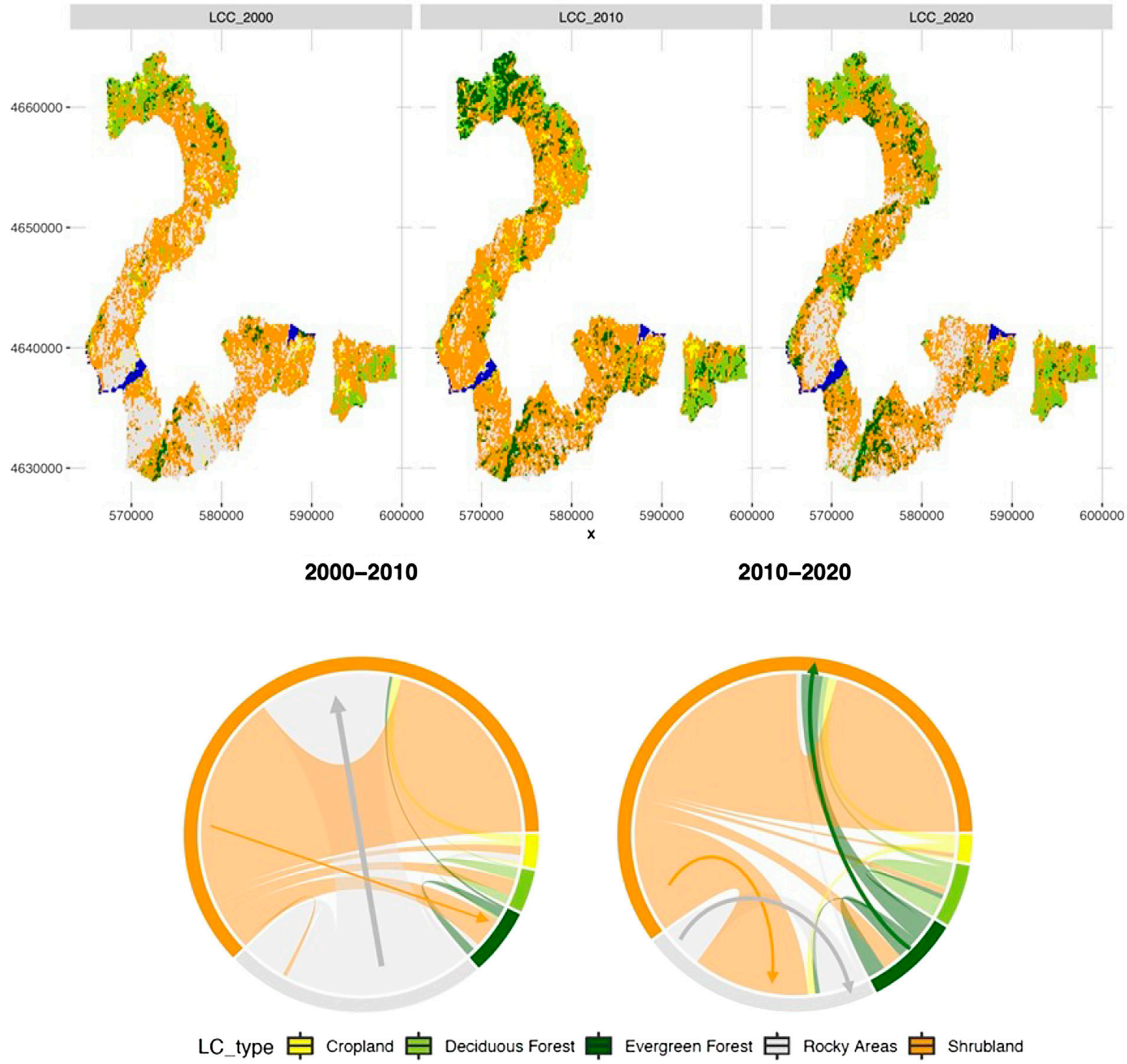

FIGURE 6 | Land cover maps derived from the ensemble of all supervised classifications for each year. Circular plots illustrating the changes in land cover between 2000 and 2010 and between 2010 and 2020 within the areas burnt between 2010 and 2020. The width of the lines is proportional to the contribution of each land cover type to the change. Different colors represent different types of land cover, and arrows indicate the direction of the change.

these large fire events. These results are in line with the proportion of 75%–80% of the burned areas that can be attribute to large fires in the Mediterranean area (San-Miguel-Ayanz and Camia, 2009). Our data showed that over the 1983–2000 period, the size and the extent of wildfires have been increasing in the Natural Park, and thereafter, during the 2001–2020 period, the trend shows a progressive reduction in fires and area burned (see **Figure 1**). Despite this reduction, the risk of having large and intense fires in the Natural Park increased notably, which is consistent with wildfire patterns observed in Europe (Fernandez-Anez et al., 2021) and United States (Iglesias et al., 2022a).

These large wildfires are often characterized by being developed under in high wind and dry vegetation conditions, related to the increase in available fuel and climate change (Bowman et al., 2011). They are considered to be increasingly frequent and virulent, being very difficult to control and affecting large areas (Hinojosa et al., 2016). This behavior is observed in the Natural Park, in some areas and certain years: although they cannot be considered as “mega-fires” due to their extension, they can be considered as extreme events due to its behavior (Tedim et al., 2013; (Linley et al., 2022). At the level of extinction, a fire can affect large areas since, when the flame front advances in a fire, not only does the length of the front that needs control

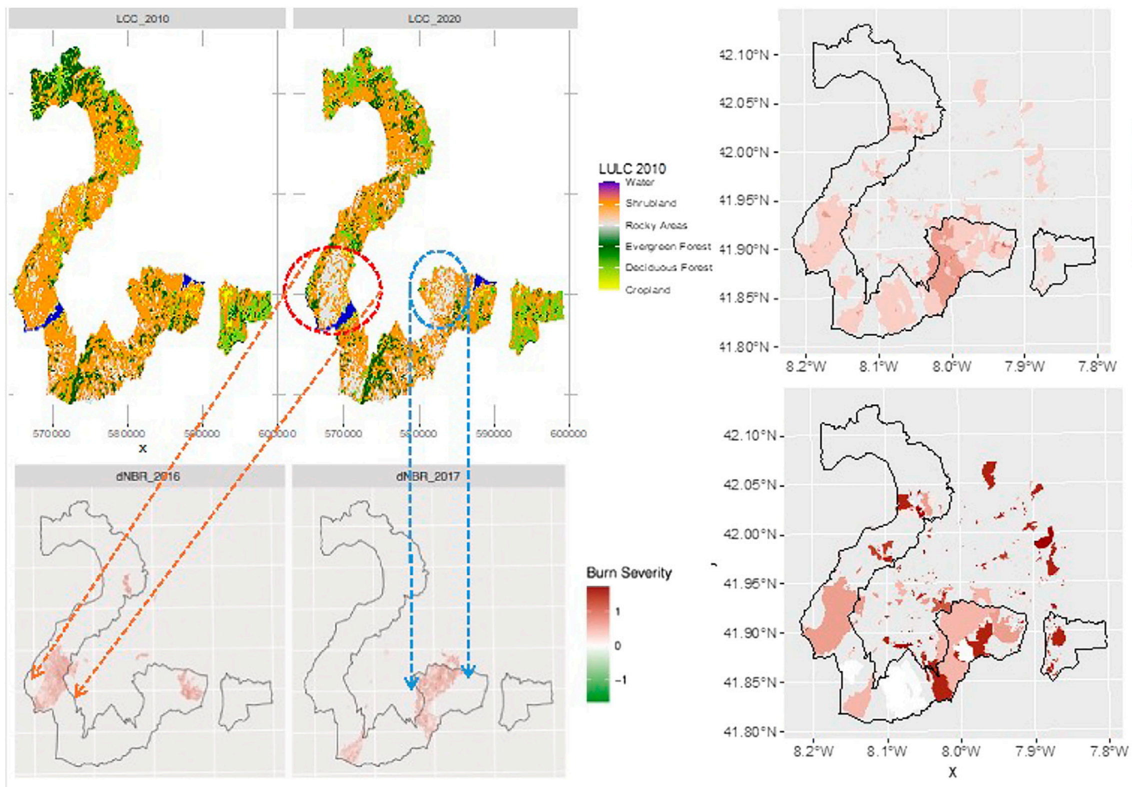

FIGURE 7 | Land cover maps for year 2000 and 2010. Circular plots illustrate rocky areas previously affected by wildfires. Burn severity: maps derived from the dNBR, for years 2016 and 2017. FireR, Fire Recurrence map; TSF, Time Since last Fire, from year 2020 (year 10) to 2010 (year 0).

increase, but it also increases the possibility of threat to population centers and the generation of secondary outbreaks. This requires an increase in fire-fighting resources to address these new situations, so the efforts of the resources that are fighting fire in the forests must abandon their defense to be used in the defense of the built structures. Our findings revealed that the areas designated with the highest levels of protection (reserve and limited use) experienced the most significant extent of burning (see **Figure 8**). This trend can be attributed to firefighting priorities, which are often hindered by challenging topography and limited accessibility in these regions. Steep slopes and difficult access can impede effective fire suppression efforts. Besides, the areas with the highest levels of protection are more vulnerable to wildfires because they are zones with minimal intervention. This approach aims to minimize disturbances to endangered habitat and species. Consequently, these protected areas often have denser vegetation and higher fuel loads, increasing their susceptibility to wildfires.

The spatial distribution of wildfires within the park demonstrates a pronounced impact on shrubland covers, with exceptions noted in the years 2016 and 2017 where forested areas experienced significant degradation, exceeding 1,500 hectares (see **Figure 4**). This underscores the vulnerability of both shrublands and forests to wildfire events within the park. These results are in line with other regions in NW Iberia. In north of Portugal, shrublands are also the land cover type most affected by wildfires, following by forest types (Calheiros et al., 2022). Furthermore, the examination of land cover transitions highlights a clear association between areas previously affected by wildfires and subsequent changes in land cover composition (see **Figures 6**, **7**). The observed heterogeneity in burn severity across these areas plays a crucial role in shaping the post-fire vegetation recovery rates, which has been previously found across different fire-prone ecosystems (Chu et al., 2017; Viana-Soto et al., 2017; Guz et al., 2022). While some regions have undergone multiple fires, the rapid recovery of vegetation following wildfires suggests a resilient ecosystem within the park (Torres et al., 2018). However, it is worth noting that the high rates of post-fire vegetation recovery may partially mask the spatial patterns of vegetation, potentially complicating efforts to accurately assess the long-term impacts of wildfires on soils. Previous studies suggested that an indicator-based approach based on satellite time-series of vegetation indices can effectively cover various facets of post-fire recovery (Torres et al., 2018). Therefore, the application of these new advances in remote sensing could improve the monitoring and prediction of post-fire recovery dynamics, with valuable applications in fire hazard management and post-fire ecosystem restoration and soil monitoring in the natural park.

Moreover, the analysis of fire seasonality reveals a notable increase in the area burnt during the autumn season over the last

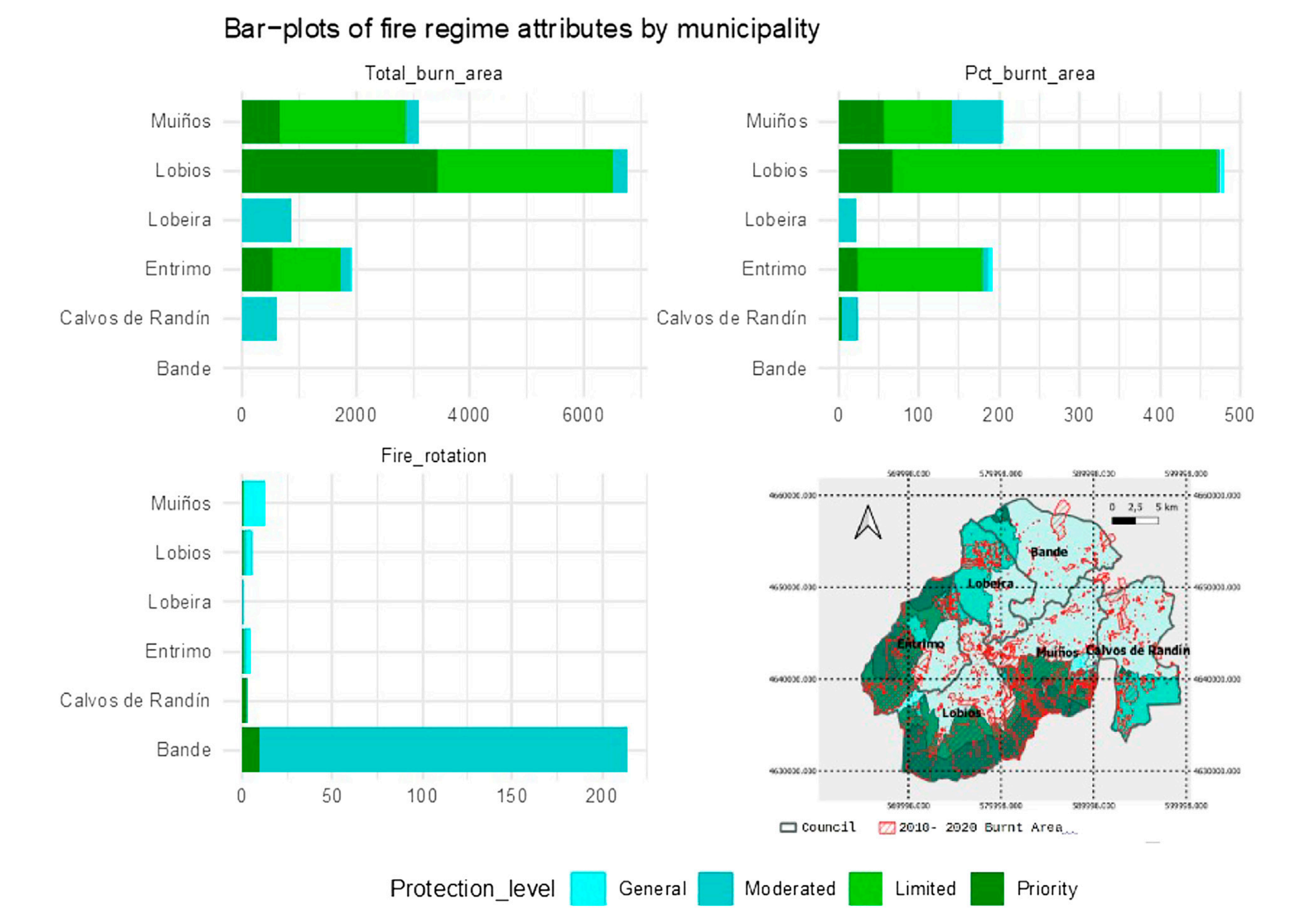

FIGURE 8 | Fire rotation, total burned area, the percentage of the area burnt in each zone of the natural resources managing plan of the Baixa Limia Serra do Xurés Natural Park by municipality.

decade (see **Figure 3**), which suggests a potential lengthening of the fire season within the park. This observation aligns with the widely recognized transformation of the fire regime due to climate change, impacting not only the size and intensity of fires but also the duration of the fire season (Flannigan et al., 2013; Marlon et al., 2013; Moreno et al., 2019; Keeley and Pausas, 2022). Traditionally, the fire season was confined to the summer months. However, the extended duration of the fire season, attributed to climate change, has prompted revisions in fire management strategies in Galicia. This is evidenced by Law 3/2007, enacted on 9 April, which focuses on the prevention and defense against forest fires in Galicia. Notably, the fire season was extended from late June, July, August, and September to include early October (de Galicia, 2018). These regulatory changes reflect the need to adapt to fire regime shifts and highlight the importance of proactive measures to mitigate wildfire risks in fire-prone regions like the Natural Park "Baixa Limia Serra do Xurés." In addition, and despite the increase in budgets and technology for extinction, the area burnt by large fires is increasing in the natural park (**Figure 2**), as reported in other areas of the Mediterranean climate (Seijo, 2005). Many authors place the drastic change in fire regime in Spain around the 1970s due to the land abandonment produced by the rural exodus (Salis et al., 2013; Moreno et al., 2014; Pausas and Fernández-Muñoz, 2012). In the Natural Park, we found a steep decrease in the number of fires from 2000 onward, but an increase in the area burnt by large fire especially in the last decade (see **Figure 2**).

Considering the ratio of the total burned area in each season divided by the total number of fires in that season, we observed a ratio of 6.40 for July to October and 30.42 for November to June. In other words, interpreting this figure as the average surface area per fire during these periods, the maximum risk area would be approximately 4.75 times greater. However, if we discount the total surface area affected by fires from groups G4 and G5 during these periods, considering that firefighting resources were diverted to protecting populations rather than combating fires as usual, and then add the average suppressed area (those with a burned area greater than 100 hectares), we find a different perspective. These results reveal that firefighting resources operate with great effectiveness under normal conditions, as evidenced by similar ratios during low and high-risk times (when fire-size groups G4 and G5 are excluded). Even when faced with harsh conditions to contain fires, their effectiveness remains high, with a ratio of 3.07, close to the baseline ratio of 3.01 (see **Table 1**). The significant drop in the ratio during high-fire-risk situations (from 30.42 to 3.07) is attributed to the distribution of the available firefighting resources, both personnel and materials. It is only when these resources are diverted to attend to more urgent needs that the forest suffers the consequences, resulting in a higher ratio of 30.42. This underscores the crucial importance of managing fuel quantity and continuity as key strategies for addressing the heightened wildfire risk resulting from climate change and rural abandonment (Moreno et al., 2014).

Throughout the period from 2010 to 2020, a significant portion of the 29,379.4 hectares within the Natural Park has endured multiple fire incidents, evident from the cumulative area burned, totaling 13,261.46 hectares. This extensive area represents almost half of the size of the Natural Park itself. While this calculation suggests that, theoretically, the entirety of study area could burn enterally over the next decade, it is essential to acknowledge that fires do not spread uniformly across the landscape. As depicted in **Figure 8**, certain areas have been subjected to repeated burns, exacerbating the impact on soils. The majority of the affected area within the six municipalities studied, except for the year 2016 y 2017, predominantly comprises herbaceous and shrubland areas (as illustrated in **Figure 4**). Within the shrublands, the area affected by fires approximates 14,642 hectares, contrasting with approximately 5,798 hectares forest areas burned within the municipalities. When firefighting efforts are deployed in open fields, priority is often given to safeguarding wooded areas, not only due to their intrinsic ecological value but also recognizing the challenge in preventing these regions from becoming engulfed in flames, particularly given the practice of using controlled burns to clear brush. The vegetation composition reflects the adaptation to the region's fire history, predominantly composed of pyrophytic shrubs (Buján, 2010). However, these species, while resilient to fire, neither impede fire spread nor facilitate the movement of ground-based firefighting resources.

The analysis of land cover change provides valuable insights into the dynamics of vegetation within the study area. The observed shifts between the periods 2000–2010 and 2010–2020 indicate significant alterations in land cover composition, with implications for ecosystem resilience and fire susceptibility. During the first period, there was a noteworthy increase in shrubland coverage, accompanied by a reduction in rocky areas (**Figures 5**, **6**). This trend suggests a potential encroachment of vegetation into previously bare soil or sparsely vegetated areas, likely influenced by factors such as post-fire conditions promoting plant growth (increased nutrient availability, favorable moisture and temperature conditions, etc.) (Chungu et al., 2020; Fernández-García et al., 2021; Blanco-Rodríguez et al., 2023). However, the subsequent decade showed a reversal of this trend, with a decrease in shrublands and a corresponding increase in rocky areas (**Figure 6**). In fact, some rocky areas were not able to evolve to shrubland, which might be suggesting a long-term loss of soil. Similar patterns were observed in evergreen forest cover, albeit to a lesser extent than shrubland. This indicates that forested areas within the study region are also undergoing dynamic changes, albeit at a slower pace compared to shrublands. The observed patterns were consistent within areas affected by wildfire between 2010 and 2020, suggesting that fire events may play a role in shaping land cover dynamics over time.

Fires effects on vegetation cover are coupled with changes belowground systems, in other words, soils. Soil quality and depth is essential for supporting a forest with herbaceous, shrub and tree vegetation. Besides destruction of vegetation cover, wildfires provoke physical, chemical and biological soil degradation as well as soil, C and nutrient losses by lixiviation and post-fire erosion (Dove et al., 2020; Agbeshie et al., 2022). Under unfavorable conditions (high severity fire, area highly susceptible to post-fire erosion due to high slope and abundant precipitations), wildfire can cause the loss of 2–2.5 cm of soil and repeated fire can lead to progressive thinning of soil and even to an irreversible total soil loss (Díaz-Raviña et al., 2012). These accumulative soil degradation processes provoked by altered fire regime could partially explain the observed land cover changes. During the 2000–2010 period, an increase in the bare soils with sparse vegetation was observed in the burnt areas, potentially reducing soil quality. In addition, during the 2010–2020 period, the intensification of fire severity and frequency has exposed the soil to progressive thinning and degradation. Consequently, the landscape can be modified by the increase of rockfalls associated with the occurrence of wildfires (Cristóbal et al., 2024). This issue is of particular concern in areas with the highest levels of protection, which are often exposed to large wildfires, and frequented by tourists, over the fire season.

The analysis also emphasized the impact of burn severity on land cover transitions. Regions experiencing higher burn severity exhibited a greater propensity for vegetation loss, resulting in shifts in land cover composition (see **Figure 7**). Nevertheless, the substantial rates of post-fire vegetation recovery observed in certain areas underscored the resilience of natural ecosystems to fire disturbances. Conversely, in other areas, the presence of shallow soil and sparse vegetation indicated severe soil degradation and even desertification attributed to the altered fire regime of the last decades.

CONCLUSION

Overall, our findings highlight the intricate relationships among wildfires and post-fire soil-vegetation dynamics within the Natural Park "Baixa Limia Serra do Xurés." These dynamics are critical for effective wildfire management and conservation efforts, particularly in fire-prone regions.

Large fire events, particularly notable in 2011, 2016, 2017, and 2020, significantly impacted the landscape within the Natural Park, with the largest fire occurring in 2017 and affecting predominantly wooded areas. Large fires accounted for the majority of the burned area, emphasizing their significance in shaping landscape dynamics and posing challenges for fire suppression efforts. Spatial analysis revealed a substantial impact on shrubland covers, with exceptions noted in 2016 and 2017 where forested areas were significantly affected, highlighting the vulnerability of both shrublands and forests to wildfire events. Land cover transitions showed a clear association between areas previously affected by wildfires and subsequent changes in land cover composition, with burn severity influencing post-fire vegetation recovery rates. The surface area of rocky areas with sparse vegetation tends to increase, and the cover of shrublands and forests tends to decrease over time in areas affected by recurrent fires. This pattern suggests that erosion and soil degradation are key factor influencing post-fire vegetation composition. Advanced remote sensing techniques hold promise for improving the monitoring and prediction of post-fire recovery dynamics, aiding in fire hazard management and ecosystem restoration efforts. Analysis of fire seasonality suggests a potential lengthening of the fire season within the park, reflecting broader shifts in fire regimes attributed to climate change. Despite increased budgets and technology for fire suppression, the areas burnt by large fires are increasing in the natural park, underscoring the need for effective fire management strategies and landscape control to mitigate wildfire risks.

Our study provides a strong long-term and comprehensive analysis of the effects of fires on soil-vegetation dynamics, and their significance for protected areas. Such insights could prove invaluable for policymakers and stakeholders, guiding the development of more effective management strategies in fire-prone regions like the Natural Park “Baixa Limia Serra do Xurés.”

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

AR and MD-R designed the study; CG-R and AR conducted the analysis; CG-R, AR, and MD-R interpreted the results; CG-R wrote the first version of the manuscript later revised by MD-R and AR. All authors contributed to the article and approved the submitted version.

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CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontierspartnerships.org/articles/10.3389/sjss.2024.13103/full#supplementary-material

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How Long Is Long? A Bibliographic Review of What Is Meant by the *Long-Term* Effects of Fire on Soil Properties

Eduardo Garcia-Braga[1], *Antonio Peñalver-Alcalá*[1], *Joaquim Farguell*[1], *Marcos Francos*[2] and *Xavier Úbeda*[1]*

[1]*Grup de Recerca Ambiental Mediterrània (GRAM), Department of Geography, University of Barcelona, Barcelona, Spain,* [2]*Department of Geography, University of Salamanca, Salamanca, Spain*

Wildfires pose one of the greatest threats to the world's forests soils. After exposure to fire, forests lose many of their ecological functions; moreover, the repercussions can extend well beyond the forest itself, as the erosive processes attributable to the combustion of vegetation and the soil's lack of protection against rainfall are likely to impact any areas of a catchment, contaminating reservoirs, estuaries and aquifers. A forest fire is not solely, therefore, an environmental issue, but also a social and economic problem. The recovery of a forest is heavily dependent on just how the soil has been affected and how rapidly the latter can be restored. Fire intensity is critical in understanding the temporal evolution of the forest, while its location—a clear determinant of its climate and the ecosystem it occupies—can undermine the functionality of the forest system and is critical in determining the duration of the effects of the fire episode. This paper undertakes a review of the literature with the aim of understanding what might be understood when studies speak of the *long-term* effects of fire on the soil and when a soil might be considered to have recovered from these effects. What is evident is that many variables have a role to play and that not all soil properties recover at the same rate; indeed, some may never be restored to pre-fire levels.

Keywords: fire intensity, climate change, soil recovery, prescribed fire, forest management

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****Correspondence***
Xavier Úbeda, xubeda@ub.edu

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STATE-OF-ART

What exactly is understood by *the long-term* effects of wildfire on forest soils is worthy of some consideration. One interpretation is that the main effects of the fire will still be observed after a period of some years; however, this begs the question as to how many years these effects can remain evident. When does the forest recover its pre-fire characteristics? Just how long is *long term* when we refer to the modifications that have been suffered by the forest ecosystem and its soil characteristics? While the literature seems to have a fairly clear idea of what it means when it speaks of the *long-term* impacts of a fire, here we seek to highlight some of the uncertainties that inevitably underpin any attempt at definition.

The impact of a forest wildfire is conditioned by a wide range of variables and, importantly, varies across different time scales. The bio-climatic region, the type of ecosystem, the plant species, soil type, fire regime (including the season), topography, intensity and severity of the fire, recurrent fires, and post-fire meteorological conditions, especially as regards timing, intensity, and duration (of, for example, rainfall and wind) are all variables that determine the response of ecosystem components and processes to a fire episode (Pereira et al., 2018). A heavy rainfall event, for instance, in the aftermath of a wildfire may cause severe erosion and lead to the most lasting effects over time, simply because of massive soil loss (Francos et al., 2016). Despite a considerable body of research that has examined these variables, there are insufficient long-term data to understand the consequences of climate change on the recovery of soil properties and the recovery of a soil's ecological functions (Halofsky et al., 2020).

Studies have been published in which the authors assume that *long-term* post-fire effects can persist for a different number of years (e.g., 11, 27 or more), but they fail to consider all the variables that might influence the trajectory of this post-fire recovery. Yet, many stakeholders, including land managers and water providers, need an answer to the question as regards the time frame in which the *long-term* effects of fire have to be considered. Clearly, there is no one answer that can be applied to all fire-affected areas and ecosystems of the world. Moreover, we also need to determine just when a forest can be considered to have fully recovered from a fire disturbance—assuming that the forest soil and vegetation have the necessary resilience—and, more relevantly, how we can measure the recovery of variables to their pre-fire state, especially under variable climate conditions (McGee et al., 2022). As DeBano (1991) pointed out towards the end of the 20th century: The key here is to ensure the sustained productivity of ecosystems despite fire-induced soil alterations.

The forest should be seen as a unit comprising a huge set of variables (each of its soil properties, for example,), which may or may not have suffered the same fire-induced effects (Úbeda and Outeiro, 2009). Indeed, the *long-term* effects of the same fire will only be *long term* in the case of certain variables: at some point, some properties will have recovered and others not. The *long-term* recovery from the effects of a fire, be it a wildfire or a prescribed fire, cannot be defined in the same number of specific years given the multiplicity of variables.

OBJECTIVES AND METHODS

The objective of the current paper is to undertake a review of studies that have purported to analyse the effects of fire on soil properties in a timeframe stated as *long term*. In so doing, we have separated these studies in two groups: those that focus on the most influential external variables related to the intensity of the fire episode and those that concern themselves with the soil's internal properties. In performing this review, we first conducted a literature search for scientific articles published, primarily in English, over the last two decades (2004–2023) in the Scopus database. In some instances, however, earlier studies were included because of the explanations and discussion they provide of given dynamics.

The search terms used were the combination of Long-term effects + Wildfire effects on soil; Fire-induced changes in soil properties; Soil heating by fire; Soil erosion after wildfires; Soil water repellency after wildfires; Soil compaction after wildfires; Soil nutrient losses after wildfires; Soil rehabilitation after wildfires; Soil carbon losses after wildfires; Soil microbial activity after wildfires; Soil organic matter decomposition after wildfires; Wildfires; Fire; Wildfire management; Wildfire prevention; Fire ecology; Fire effects; Fire severity; Fire behaviour; Fire regime; Fire disturbance; Burn severity; Burned area emergency response; Fire-adapted ecosystems; Fire-suppression; Fire-fighting; Soil structure; Soil texture; Soil acidity; Soil nutrients; Soil pH; Soil organic matter; Soil biology; Post-fire recovery; Soil carbon sequestration; Soil water repellency; Soil aggregate stability; Soil compaction; Soil microbial diversity; Soil microbial ecology; Soil microbial biomass; Soil erosion control; Soil erosion prevention; Soil remediation; Soil restoration techniques; Prescribed fire effects; Soil heating; Nutrient cycling in soil; Soil nutrient availability; Soil nutrient management.

In all, a total of 102 references were identified; however, we opted only to include those that specified exactly the number of years that had elapsed after the wildfire or the prescribed fire (**Table 1**).

EXTERNAL VARIABLES: FIRE INTENSITY, FIRE SEVERITY AND CLIMATE TYPE

Fire intensity has been identified as being of critical importance when seeking to understand the effects of fire on soil properties. Hurteau and Brooks (2011) examine carbon sequestration in the temperate forests of the western United States and conclude that the key factor in determining whether the carbon sequestered is subsequently restored is the intensity at which the forest burns. They argue that, in a high-intensity fire, more carbon is lost and the recovery of the forest is slower; indeed, in many instances, it is unlikely to recover its pre-fire carbon stock as the vegetation structure may well have changed, becoming bushy and even more herbaceous. Thus, these authors do not propose a specific time period as corresponding to the expression *long term*, but rather report that the effects of a fire can be drawn out indefinitely over a long period of time. They conclude that avoiding the fire risk in forests altogether is largely futile and that the best strategy has to be forest management practices that mitigate the risk of high-intensity fires, whose *long-term* impact can be great. Yet, to do so requires, for example, prescribed burning that will release carbon periodically into the atmosphere. In short, carbon stocks should not be seen as the most important aspect, but rather they should be considered as just one more element in the suite of ecosystem services.

Ibañez et al. (2022) investigate the effect of fires on soil nitrogen (N) concentrations in a Swedish boreal forest in which post-fire management takes the form of salvage logging. They claim that while much is known about what happens in the immediate aftermath of a fire, less is understood about the *long-term* effects. The authors report that, in response to high fire severity, gross N mineralization and consumption rates per unit carbon (C) increased by 81% and 85%, respectively, and that nitrification rates per unit C basis fell by 69%, while net N mineralization was unresponsive. They also found that,

TABLE 1 | Summary of the studies developed in this work, arranged chronologically.

Authors	Type of fire and location	Dominant vegetation	Time after fire	Parameters analysed	Overall effects
Ffolliot and Guertin (1990)	Prescribed fire in Arizona (United States)	*Pinus ponderosa*	22 years	Litter, duff and humus layers	Decrease of litter and duff
Slaughter et al. (1998)	Wildfire in Minnesota (United States)	*Taiga*	23 years	Cmass	Almost total recovery
Roscoe et al. (2000)	Wildfire in Southeast Brasil	Cerrado	20 years	Stocks of C and N	Similar in soil but less stocks of C and N in litter
Johnson et al. (2005)	Wildfire in Sierra Nevada, California (United States)	*Pinus jeffreyii*	20 years	C, N, P, K, Ca, Mg, S	P, K and S has not differences. N, Ca, Mg increase and C decrease in the burnt area
De Luca et al. (2006)	Wildfire in Montana (United States)	*Pinus ponderosa*	17 years	Charcoal, N, microbialactivity	Charcoal avoid a higher decrease of microbial activity
Yermakov and Rothstein (2006)	Wildfire in Michigan (United States)	*Jack pines*	72 years	N, organic horizons	Depending of climate there is recovery or not after a long term sequence
Capogna et al. (2009)	Wildfire in Castel Volturno (Italy)	*Phillyrea angustifolia L.*	3 years	C, N, P_2O_4, Ca, Mg, Na,K, microfungal variables	Increase in photosynthetic activity, particularly in the high-intensity fire plots
Fflolliot et al. (2009)	Prescribed fire in Arizona (United States)	*Pinus ponderosa*	43 years	Litter, duff and humus layers	Recovery of these layers
Kaye et al. (2010)	Wildfire in Garraf massif, (Spain)	*Quercus coccifera* and *Pinus halepensis*	30 years	Cstock	Increase just after fire, decreased with time and not recovery after 30 years
LeDuc and Rothsein (2010)	Wildfire in Canada	*Jack pines*	46 years	N	Recovery in 22 years and increase after 46 years
Hurteau and Brooks (2011)	Wildfires in Oregon (United States)	*Pinus ponderosa*	35, 100 and 200 years	Cstock	Decrease immediately after wildfire, recovering with time, although the non-managed forest result in high intensity wildfire than reduce again the Cstock
Longo et al. (2011)	Wildfire in Patagonia (Argentina)	*Nothofagus pumilio*	10 years	pH, C, N, P	Increase of pH and decrease of C, N, P
Johnson et al. (2012)	Wildfire in Sierra Nevada, California (United States)	*Ceanothus velutinus* *Pinus jeffreyii*	46 years	pH, C, N, P, K, Ca, Mg, S	C and N are similar in the burnt and unburnt soil. P similar in *Pinus jeffreyii* plot. Ca, Mg, K and S are increasing while P is decreasing in *Ceanothus velutinus* plot
Bennett et al. (2014)	Prescribed fire in Victoria (Australia)	*Eucalyptus obliqua* L'Her.	27 years (3 and 10 years frequency of burning)	Cstock in above-ground biomass, dead wood, Litter and soil	Decrease of the Cstock in all the analysed organic variables and frequency of burning
Alcañiz et al. (2016)	Prescribed fire in Montgrí massif, (Spain)	*Pinus halepensis*	9 years	EC, P_2O_4, pH, C, N, Ca, Mg, K	EC, P_2O_4 increase; pH, C, N, Ca, Mg, K decrease
Muñoz-Rojas et al. (2016)	Wildfire in Western Australia	Grassland	14 years	C, N, Soil microbiology	Partial recovery
Francos et al. (2018)	Wildfire in Cadiretes massif, (Spain)	*Pinus pinaster* and *Quercus suber*	18 years	C, N, C/N, SOM, Ca, Mg, Na, K	Similar to control for N, Ca, Mg, Na. Decrease of C, SOM in Low severity. Similar to control for N, Na, K. Decrease of C, SOM, Ca, Mg in High severity
Francos et al. (2019)	Prescribed fire in Tarragona (Spain)	*Pinus pinea*	13 years	EC, P_2O_4, pH, C, N, Ca, Mg, K, SOM	Decrease in N, SOM, P_2O_4
Francos et al. (2020)	Wildfire in Ódena, Barcelona (Spain)	*Pinus halepensis*	30 years	EC, P_2O_4, pH, C, N, Ca, Mg, K, SOM	Recovery in areas with forest management
Robichaud et al. (2020)	Wildfire in Colorado (United States)	*Pinus ponderosa*	10 years	Erosion	No differences with unburned after 4 years
Sadeghifar et al. (2020)	Wildfire in Zagros monuntains (Iran)	*Quercus brantii* Lindl.	10 years	qCO_2, microbal biomass, C	Increase of qCO_2 and Cmic:Corg decrease of C
Li et al. (2021)	Meta-analysis		10 years	C, N, PyC, PyC/TOC	Recovery of C and N after 10 years. Increases in PyC and PyC/TOC
Pérez-Quesada et al. (2021)	Wildfire in Chiloé (Chile)	*Drimys winteri* Jordan Forst.	50 years	Soil CO_2	The unburned forest is still a source of CO_2
Dove et al. (2022)	Wildfire in Central Sierra Nevada (United States)	*Pinus ponderosa*	More tan 25 years	Different soil microbes, C, N	Not recovery of microbial community, affecting C and N cycles after high severity fires
Follmi et al. (2022)	Wildfires in Águeda catchment (Portugal)	Majoritary Eucaliptus, pines and shrub vegetation	41 years	Soil erosion	Decreasing with time and equal than non-burnt areas in some places after 41 years. Erosion increases after wildfires and its higher depending fire severity and fire recurrence

(Continued on following page)

TABLE 1 | (*Continued*) Summary of the studies developed in this work, arranged chronologically.

Authors	Type of fire and location	Dominant vegetation	Time after fire	Parameters analysed	Overall effects
Ibañez et al. (2022)	Wildfire in Central Sweden	Conifer forest	4 years	Different forms of N and C	Net N was unresponsive. Gross N mineralization and consumption rates per unit carbon (C) increased by 81% and 85%
Kastridis et al. (2022)	Wildfire in Thessaloniki (Greece)	*Pinus brutia*	25 years	Erosion	Increasing after fire and decreasing with time but remaining higher than in control
Margiorou et al. (2022)	Wildfire in Thessaloniki (Greece)	*Pinus brutia*	30 years	Erosion	Not recovery
McGee et al. (2022)	Wildfire in Utah (United States)	*Temperate coniferous forest*	20 years	Soil fungal communities	Recovery
Orumaa et al. (2022)	Wildfire in Estonia	Scots pine	From 12 to 181 years	Fungal community, C and N	Similar quantity after 181 years but differences in the species composition over the years
Voropay et al. (2022)	Wildfire in Baykal region (Russia)	Pine forest	10 years	Soil temperature	Increase until the 8 years
Liu et al. (2023)	Meta-analysis		Different years considered "long term"	SOM	Recovery of SOM at long term
Taylor et al. (2023)	Prescribed fire in Florida (United States)	*Pinus palustris* Mill.	60 years	Soil horizons	Increase of the A horizon

regardless of burn severity, the rate of N immobilization exceeded the rates of N nitrification and, as such, immobilization was the dominant pathway of gross N consumption. The study shows that soil N transformation rates were more strongly affected by changes in fire severity than by salvage logging, and that 4 years after the fire many aspects of the N cycle in burned and unburned stands did not differ, suggesting substantial resilience of the N cycle to fire and salvage logging.

In the tropical climate of Brazil, Roscoe et al. (2000) study the differences in three areas of forest—two of them burned at low intensity and one at high intensity—after 20 years. They find that fire intensity is responsible for the difference in the recovery of C and N content. Thus, in the two low intensity plots the stock of C in the litter was lower than that in the unburned area, but that this was not the case in the mineral soil. In the most heavily burned area, the stocks of C and N were lower than those of the control area while significant differences were seen with the other two plots.

In the Mediterranean, the most intense erosion is a short-term phenomenon, that is, it occurs immediately after a fire, given that the many plant species that colonize the burned area protect the soil from erosive processes. And yet, Follmi et al. (2022) show that although erosion in a Portuguese forest is reduced over time, it remains more important in the *long term* in areas that have been burned than in those that avoided the effects of fire. To achieve this goal, the authors build a landscape evolution model and find that over a 41-year timespan erosion rates in the burned area were 5.95 ton compared to 0.58 ton ha^{-1} $year^{-1}$ in the non-burned area. The authors conclude that burn severity is the most important variable here, but that the topography should not be underestimated given that it can concentrate erosion hotspots, coinciding with the creation of rills and gullies. Likewise, Kastridis et al. (2022) report that, 25 years after a fire in Greece, erosion is greater in a burned area; however, the infestation of the forest by the bark beetle *Tomicus piniperda* impedes soil recovery because of the logging and removal of infected trees. The authors show that these variables are determining factors of the rates of erosion in disturbed and undisturbed areas of the forest. Margiorou et al. (2022) study erosion rates in Greece in basins in which check-dams were built some 20 years ago to prevent soil loss, some 3 years after the fire. The authors find that the average annual erosion rate for the pre-fire period was 0.0419 t/ha/year, rising to 0.998 t/ha/year 3 years after the fire but remaining at 0.08 t/ha/year 20 years afterwards, that is, twice as high as in the unburned control. As such, the forest has yet to fully recover. The authors attribute this to a very thin soil depth, the high intensity fire and the geomorphology of the basin, which favours erosion.

In the continental climate of the state of Colorado in the United States, Robichaud et al. (2020) found that the burned control plots of a forest presented high sediment flux rates until post-fire year 3, when they fell significantly to a level that was statistically no longer higher than those of the unburned reference plots in post-fire years 4 and 10. The authors stress the importance of the climate and its relationship with the rates at which existing vegetation are allowed to regenerate and, hence, to determine the duration of the erosive processes. In this instance, the authors identify the need to reduce erosion in the immediate aftermath of the fire using mulch treatments, such as the application of straw and wood.

In Spain, in the Cadiretes Massif, Francos et al. (2018) studied the long-term (18-year) impact of a wildfire on two areas affected by low and high fire severity regimes, comparing outcomes with the characteristics of an unburned control (**Figure 1**). The authors concluded that 18 years on, many of the respective soil properties differed and that the intensity of the episode could be considered largely responsible. Vegetation regrowth was rapid although it differed in density and species type, factors that appeared to explain the evolution of the soil parameters. After 18 years the decrease in the soil organic

FIGURE 1 | Recent sample from the high severity of the Cadiretes Massif study area in 1994. It is clear how the organic layer was burned. After the first rains, this layer was eroded. Picture by Xavier Úbeda.

FIGURE 2 | A prescribed fire in Montgrí Massif in 2022. Picture by Eduardo García-Braga.

matter (SOM), C and N values in the high severity area was more accentuated than that in the low severity area. In the same Mediterranean ecosystem, Francos et al. (2020) observed the influence of long-term forest density after a wildfire. The authors compared high and low density areas with a control forest and detected the positive effects of a management action conducted almost 20 years after a wildfire. However, they identified problems in the recovery of certain physicochemical soil properties in high density areas, stressing the need to study forests in a context of global change.

Not all studies dedicated to the analysis of the long-term effects of burning focus their attention on wildfires; some interest themselves in the impact of prescribed burning (**Figure 2**). This is the case of Bennett et al. (2014) who examine the effects on total soil C of a prescribed fire in a temperate climate zone of Australia. In general, the authors report a decline in C stocks following such an episode, not only in the soil, but also in the vegetation. They identify fire intensity as well as frequency as the most important variables in this regard; moreover, the effects are evident 27 years after the first burning, being greater when the frequency factor (every 3 or 7 years, depending on the fire intensity) is added. Another factor that the authors show to be important is the time of year when the fire treatment is administered, with carbon stock decreases being greater in the dry autumn season than in the wet spring season when greater burning intensities can be achieved. Such studies serve to demonstrate how frequently fire should be used as a management tool to avoid negative soil effects.

In other studies of the use of fire as a tool for managing forests, Alcañiz et al. (2016) found that pH levels, C, N and available phosphorus (P) were significantly lower 9 years after a prescribed fire in a *Pinus halepensis* forest than their pre-fire values, while the rest of the variables analysed did not present a statistically significant difference. In this forest, the vegetation burned was the *Quercus coccifera* L. shrubland, but the fact that the fire intensity achieved during the prescribed burning was not great explains, according to the authors, why the effects were not greater after 9 years. Francos et al. (2019), among others, have evaluated the impact of prescribed fire on soil chemical properties, in this particular instance in a wildland-urban interface in the Mediterranean environment. The authors report that the burning sought specifically to eliminate the continuity of plant fuel, and, therefore, the prescribed fire was of a higher intensity than in other cases; indeed, controlling the intensity of the fire during the execution of a prescribed fire is, they stress, critical. Thirteen years after the burn, they conclude that the inhabited area has been successfully protected from fire risk. However, even after this period of time, they report impacts on soil chemical properties that still need to be monitored. Ffolliott and Guertin (1990), in a study conducted in a forest of Ponderosa pine in Arizona, found that the forest floor (litter and duff) failed to recover its original depth even 22 years after a fire. In short, the objectives of a prescribed fire, and its consequences, must be very clearly determined, which means a good understanding of the effects of fire in each site is fundamental so as not to harm the soil and its functions (Alcañiz et al., 2018; Francos and Úbeda, 2021). Ffolliott et al. (2009), in line with others, point out that after 40 plus years the effects of a prescribed fire disappear, making it necessary to repeat the measure less than every four decades to maintain the effects and avoid a detrimental impact on the ecosystem.

Historically, fire, both natural and anthropogenic, has not been as recurrent in the easternmost forests of the United States. For this reason, Miesel et al. (2012), who undertake an analysis of the forests of the Lake States region, conclude that there have been few studies of post-fire dynamics in this area of the country. According to these authors, climate is decisive in understanding the effects and, in particular, the *long-term* effects—10 years being deemed sufficient, although they offer no concrete reasons as to why—on the composition and structure of vegetation and on soil properties. They also stress that the recovery of forest soils differs from one ecosystem to another, the characteristics of which are determined primarily by climate factors. The authors conclude that in the Lake States there have been insufficient *long-term* studies to draw any safe conclusions about the role played by the variables analysed and, moreover, call for the inclusion of more physical parameters in future studies. However, they are able to show that the *long-term* effects differ in the coldest forests of the Lake States from those in the westernmost forests of the United States In none of the cases reviewed are the authors able to verify that there are no longer term effects of fire on soils than those reported in a 10-year period after the fire episode.

A further example of the effects of fire in a temperate climate is provided by Longo et al. (2011) in a study of wildfires that occurred 6–10 years earlier in Argentine Patagonia. In what the authors describe as a *long-term* study (having stressed that most studies are conducted in the immediate aftermath and very few more than 5 years on), they show that in all study plots, the soils present different rates of decrease in their C, N and P concentrations and different degrees of increase in pH. They attribute these differences essentially to the time elapsed, which in all instances is insufficient for the parameters to have returned to their pre-fire values. In this Patagonian forest, the authors suggest that the evolution in the development of ectomycorrhizas in the soil can be directly related to such elements as N and P. The climate, they argue, is also important, given that at higher temperatures the recovery of certain parameters, especially biological, such as fungi, is faster than that in colder climates, and that this biological recovery is determined by the recovery in the accumulation of SOM.

McGee et al. (2022) study the *long-term* effects on soil fungal communities over a 20-year chronosequence in Utah. The authors report that at the end of this period the fungal community has recovered, but that the rate of recovery is dependent on the temperature—a difference of as little as just 2°C making a difference—and annual precipitation. In addressing climate change in alpine areas, the authors believe that as temperatures increase, forest fires are also increasing with all the ramifications this might have; yet, in these new scenarios, fungal recovery time should be cut.

Another example of the way in which climate and soil type can respond differently to wildfires is provided by Yermakov and Rothsein (2006). In more northerly ecosystems (in this instance, in Michigan), where the accumulation of N in surface organic layers can constrain plant productivity, wildfires induce ecosystem rejuvenation and increase N cycling rates. In contrast, N availability, in drier, sandy places with a very thin O horizon, a fire can actually consume the main nutrient pool. In soils of this type, the restoration of pre-fire N levels is much longer.

Some forests will never recover their pre-fire state. New climate situations are likely to prevent a forest from recovering its structure and composition in the way it would have done decades earlier when exposed to the same kind of fire event (Halofsky et al., 2020).

INTERNAL SOIL VARIABLES

Shifting the focus more specifically to a soil's internal variables, Johnson et al. (2012) conducted a study in California, in which they compared the evolution over time of two plots that had burned 46 years earlier. The authors detected differences in the respective forest soil parameters but the degree of these differences varied. They attributed the variations in certain soil chemicals to the vegetation that had emerged after the fire, with some species providing better fixation, for example, of P and N. They also attributed higher concentrations of major cations in burned areas to the greater accumulation of burned material above the soil surface. However, they stress that the substrate type had been decisive in producing these different soil parameters, specifically, in the *long term*, they recorded higher concentrations of P in areas of andic parent material.

Carbon is one of the most frequently measured elements in studies of this type, given the importance of carbon sequestration by forests for mitigating the effects of atmospheric emissions. In a study conducted in a southern boreal climate, it was found that in the five immediate post-fire years there was a general loss of C throughout the forest from the soil to the vegetation (Slaughter et al., 1998), with losses peaking in the fourth year; however, 23 years later, the forest had recovered 91% of its entire pre-fire C mass. Similar results were reported by Roscoe et al. (2000) in native *cerrado* (savannah) ecosystems in Brazil, where 21 years after an intense fire, the soils' C and N values had been restored, albeit the quantities of C and N were lower in the litter horizon than in the control forest. Likewise, Kaye et al. (2010) reported an increase in soil C after a fire in a Mediterranean forest, but concentrations had fallen to pre-fire levels after 10 years. The authors conclude that the type of vegetation to emerge after a fire can be decisive in determining soil C storage. If there is vegetation recovery, the authors determine that in the organic horizon (LF) there are no significant differences after 30 years. However, they find differences in the organic H horizon after 15 years and report that these can be maintained even after 30 years. In the mineral horizon, they found significant differences with the unburned area, with the C content being higher in this latter area.

Johnson et al. (2005) conducted a study in a Californian forest some 20 years after a fire and found that the soil contained less C and more N than that of the adjacent forest ecosystem. They attribute this to the fact that one species, *Ceanothus velutinus* Dougl, has helped N fixation not only in the organic horizon, but also in the mineral horizon. They found no differences in ecosystem P, K and S, while exchangeable K^+, Ca^{2+}, and Mg^{2+} were greater in the burned zone. The authors speculate that the large increase in the soil and ecosystem Ca content resulted from the release of these elements by the ash and the rapid absorption and recycling of Ca by vegetation after the fire. LeDuc and Rothstein (2010) found similar results for different forms of N availability in a jack pine forest in Canada. They reported that in the first 10 years after the fire the uptake of N forms beneficial for vegetation (amino-acid N) was lower than that of pre-fire values; however, a rapid increase was recorded 15–22 years post-fire, reaching values that exceeded those before the fire after 46 years. Li et al. (2021) carried out a meta-analysis focused on the loss and subsequent recovery of soil C and N concentrations in the wake of a fire, on the understanding that these elements determine soil health and ecosystem services at the global scale. After reviewing 3,173 publications, the authors conclude that although the intensity of burning determines the losses of soil C and N, geographical variables determine both wildfire severity and soil recovery. Thus, greater negative impacts on soil C and N were found in tropical and temperate climates than in Mediterranean and subtropical climates, while stronger effects were found in forest ecosystems than in non-forest ecosystems. Yet, on average, the authors conclude that pre-fire levels can be recovered after 10 years.

One of the main ecosystem services provided by forest soils is their ability to serve as C sinks and so minimize greenhouse gases (GHGs). Pérez-Quezada et al. (2021), in a study conducted on the island of Chiloé in Chile, compared CO_2 emissions in a forest that had not suffered a fire with an adjacent forest that burned more than 50 years earlier. Starting from the premise that the greatest quantities of CO_2 are released into the atmosphere during the combustion event, once the fire has burnt out, the time of year, as well as temperature and humidity can be deemed decisive in understanding annual GHG variations. The authors conclude that in the net ecosystem exchange of CO_2, the unburned forest is a sink, while the burned site is a source of GHGs, although more than 50 years have passed, these differences can still be observed. Pellegrini et al. (2022) make an important contribution regarding the complexity of soil carbon content after impacts such as wildfires, based on a review of studies conducted in different types of ecosystem. The authors consider that fire affects soil C content in quite a different fashion, given that most of this C is found in SOM. Fire can mean that soil C occurs in a much more stable form that persists over time and that the effect of fire on the decomposition of SOM would appear to be important for understanding the *long-term* changes in soil C storage and fluxes. They conclude that perhaps, by means of prescribed fires, climate change could be mitigated since the stability of this organic matter seems to increase.

It is clear, as discussed above, that not all ecosystems behave in the same fashion when exposed to fire, the response depending on the climate zone in which it is located and, arguably even more so, on its characteristic soil type. Taylor et al. (2023), in a study conducted at Spodosols in Florida, show that A-horizon thickness is greater in places that have been managed for more than 60 years with prescribed burning than in those that have not been exposed to any management practices. They attribute the greater thickness to the fact that in burned soils, with a thinner O horizon, there is likely to be more movement of particles towards the A horizon. The authors report differences in thickness of more than 2.5 cm between a soil that has been burned recurrently for 60 years and another that has not been burned. They also report that fire can cause changes to the bulk density of burned plots because of the inputs of ash. This means the A horizons tend to be less dense, thus influencing their thickness. The authors conclude that, following a fire, the C of the organic matter is better protected in this A horizon than in the organic horizon which is more prone to burning.

Closely related to horizon thickness and the effects of fire is the importance that the increase or decrease in temperature can have on altering a soil's physicochemical and biological processes. In a study undertaken in the south-western area of the Baykal region, Voropay et al. (2022) found that, 10 years after a fire, increases and decreases in soil temperature differed. The authors conclude that the burned forests need to be managed to ensure these temperature dynamics do not take so many years to recover. They report that differences in average monthly temperatures at the surface began to decrease 8 years after the fire; however, trends have yet to be studied at greater depths.

Other authors, most notably Orumaa et al. (2022), have studied the short- and long-term effects of fire on fungal communities and soil properties. These latter authors conducted a study in Scots pine stands in the hemiboreal climate of Estonia, where fires had occurred at various times between the last 12 and 183 years. In this chronosequence, soil saprotrophs and ectomycorrhizal fungi (EcM) were predominant. The authors report marked differences in the species composition of EcM fungi, with *Piloderma sphaerosporum*, *Pseudotomentella* sp. and *Clavulinaceae* sp. being the most abundant EcM operational taxonomic units in the most recently burned stand while *Clavulinaceae* sp. and *Cortinarius* sp. were the most abundant in the three oldest burned stands. Soil C and N stocks were lower in the most recently burned stand, but the differences with the other stands were not statistically significant. Soil pH had a significant effect on fungal species composition, with the older stands presenting a substantially lower pH than that of the more recently burned areas. It should be noted that the forests have not undergone any type of forest management. As discussed, it is evident that not all soil properties evolve in a similar fashion over time, nor are they affected in a similar fashion by fire. What is important—as many researchers in this field have been at pains to point out—is the relationship between the different variables. Capogna et al. (2009) highlight the close relationship between a soil's biotic and abiotic soil components, above and below the ground. The authors observe that the more intense the fire, the more important these relationships and interconnections are. They highlight the importance of soil fungal components and their relationship with the reserve and translocation of chemical elements in the soil in a Mediterranean forest 10 years after a fire. This also means that the plants have higher nutrient stocks and are able to emerge more vigorously after the fire.

Soil microbiology is critical for ensuring that degradation of organic matter and mineralization occur at rates that safeguard the proper functioning of the C and N cycles and provide sustenance for the vegetation and the structural stability of the soil's most superficial layers (Muñoz-Rojas et al., 2016). The same authors report that 14 years after a forest fire in a semi-arid grassland ecosystem of Western Australia, the recovery of soil properties continued to be only partial, but given their interconnections it was impossible to identify which were the most important. They also stress how a soil's physical and organic properties are critical for its capacity to retain water, the latter being a basic element for many of the functions of the soil system and its relationship with plants.

Dove et al. (2022) have studied high-intensity burned forests in the western United States and find that the recovery of soil microbial communities can take more than 25 years. They report, moreover, that not all communities recover at the same rate and that changes may occur in the microbiome composition. As wildfires tend these days to be more severe, the authors claim that post-fire management is necessary to speed up rates of recovery and so favour biogeochemical dynamics. Similarly, Liu et al. (2023), based on an analysis of 371 works studying the impact of fire on microbial properties, conclude that the effects are highly variable depending on the location and soil type, but given that the majority constitute short-term studies, what may occur in the *long term* is largely unknown. These authors report that their review of the literature suggests that SOM is not affected in the *long term*, although the biological properties may be impacted. However, they stress that there is a relationship between the effect and the intensity of the fire. In the short term, reported outcomes show that fires significantly increase microbial metabolic quotient (qCO_2) by an average of 19.45%, and reduce soil microbial and fungal biomass carbon by 8.41% and 27.17%, respectively.

In a similar vein, Sadeghifar et al. (2020), in a study conducted in the Zagros Mountains, report that not all microbial eco-physiological indices behave in the same way, both as regards their impact and their recovery time. They find an increase in some properties after the fire—the case, for example, of the eco-physiological indices, including the ratio of respiration to qCO_2 and the ratio of microbial biomass to soil organic carbon (Cmic: Corg); however, some parameters had not recovered 10 years after the fire—the case of acid phosphatase (ACP) activity and, unlike the previous study, the amount of Corg had not recovered, being up to 21% less than in the unburned control plot.

FOREST MANAGEMENT

Post-fire forest management (e.g., mulching, salvage logging, reforestation, etc.) can affect soil properties even in the long term. It is therefore essential to take into account the existence or absence of intervention in an area to determine its natural recovery or whether it has been intervened or favoured by humans. According to some studies (e.g., Mitchell et al., 2009), forest management can help some types of forest recover from the impact of a wildfire more quickly and without any detrimental effects to any of the ecosystem services. Such management initiatives might centre on clearing the post-fire vegetation combined with the felling of some individuals (Stephens and Ruth, 2005). These management practices are of benefit at the ground level, given that not as many nutrients and as much water are extracted and so recovery is likely to be quicker. They also have the benefit of minimizing the risk of a large forest fire in the event of a future outbreak. However, Mitchell et al. (2009) claim that in forests with little accumulation of forest mass—such as, monospecific forests or forests with a highly dominant species (e.g., the Douglas fir)—management strategies that promote logging can be counterproductive, since they undermine a forest's carbon sequestration capacity.

However, each ecosystem needs to be considered in isolation. For example, DeLuca et al. (2006), in a study carried out over a 17-year period, highlight that in fires in boreal forest ecosystems one of the few products produced is charcoal. Charcoal is of particular importance in environments of this type as a *long-term* driver of ecosystem processes and, specifically, of N cycling. The authors report that post-fire forest management and road construction can result in the disappearance of this passive form of C, leading to soil impoverishment and alterations to the nutrient cycles.

CONCLUSIONS

The authors of this review believe that a study might assume that the years that have elapsed since the fire episode is sufficient to offer conclusions, but this is often determined by the actual opportunity a team has to conduct its study (often dictated by funding or availability of team members). Thus, the literature is full of studies conducted after varying numbers of years simply because the authors believed the *long-term* effects of the fire might still be evident; indeed, knowing with any degree of certainty when the effects are no longer visible is largely impossible.

A review of the extant literature fails to reveal just how many years might be understood to constitute *long* when considering the *long-term* effects of fire on soil properties. Indeed, the variation in responses is high. However, the review does identify two key variables that seem to determine the prolonged impact of fire on soils: first, the intensity of the fire and the consequent severity of the burning, and, second, the climate, closely associated with the forest's ecosystem, insofar as prevailing temperatures and levels of precipitation, and the vegetation type that prospers in each place, are decisive.

The literature also highlights that not all soil properties recover at the same rate and that not all effects remain visible over time. Moreover, certain soil properties can impact others, thus modifying soil system dynamics.

The literature, likewise, stresses the importance of *long-term* studies because there has been a change in the global fire regime, characterised today by more severe, more recurrent fires that are likely to occur throughout longer periods of the year. These changing circumstances emphasise the need to conduct studies of these characteristics. Undoubtedly, the changes in the world's fire regimes—above all, their increasing severity—can be attributed to the effects of climate change, a phenomenon that the literature reviewed here constantly associates with the difficulties faced in recovering soils.

Clearly, both pre- and post-fire management practices need to be the object of very careful study: the former to prevent the proliferation of more severe fires, which as we have documented here are increasing in number, and the latter to minimize the effects of severe burning on a temporal scale. Disseminating the findings of this research to forest managers and administrators must be the ultimate goal of these studies.

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All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Ground Fire Legacy Effects on Water-Dynamics of Volcanic Tropical Soils

Luis D. Olivares-Martinez[1,2], Alberto Gomez-Tagle[3] and Jorge Mataix-Solera[1]*

[1]Soil Science and Environmental Technologies Group (GETECMA), Department of Agrochemistry and Environment, Miguel Hernández University, Elche, Spain, [2]Graduate Program in Integrative Ecology, Natural Resources Institute (INIRENA), Universidad Michoacana de San Nicolás de Hidalgo, Morelia, Mexico, [3]Earth Science Department, Natural Resources Institute (INIRENA), Universidad Michoacana de San Nicolás de Hidalgo, Morelia, Mexico

The forest floor is a critical component in maintaining the life cycles of forest ecosystems. It normally includes organic soil horizons, known as duff and litter, which are prone to be rapidly consumed after flaming and smoldering fires. This work aims to understand the legacy effects of surface and ground fires on the infiltration capacity of a volcanic forest soil. We studied five sites with fires recorded in the last 20 years. All of them are located in pine-oak forests of the volcanic mountain region in central Mexico with a temperate climate and Andic soil properties. Tension-infiltration tests were carried out to determine hydraulic conductivity and the number of active macropores. After each test, cores were taken to evaluate in a laboratory setting, where soil water repellency at different moisture concentrations and the integrative dynamic repellency index were determined. Field-saturated hydraulic conductivity was moderately high in all sites, with mean values of 13 and 42 mm·h^{-1} for burned and control plots, respectively. A non-linear relationship was found between recurrence and type of fires with the concentration of active pores and several dynamic water repellency parameters. This work confirmed the presence of latent combustion in these temperate neotropical forests. The changes in soil water repellency and hydraulic conductivity detected do not necessarily imply an exceeded soil infiltration capacity. However, many of the fires in this region are associated with increasing agricultural activities, so further studies are needed to determine if higher fire frequencies could exceed the resilience capacity of the soils triggering land degradation.

Keywords: soil water repellency, smoldering fire, resilience, mountain forest, field saturated hydraulic conductivity

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****Correspondence:***
Luis D. Olivares-Martinez
lolivares@umh.es

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INTRODUCTION

Tropical mountain ecosystems are characterized by significant biological and lithological diversity, which plays a critical role in maintaining ecosystem services and supporting a variety of life forms (Bruijnzeel et al., 2011; Poorter et al., 2015). Among the various components of these ecosystems, the forest floor plays a critical role in maintaining the life cycles of forest ecosystems (Descroix et al., 2001; Keith et al., 2010; Neris et al., 2013). This typically includes organic soil horizons, known as duff (O_e and O_a horizons) and litter (O_i horizons), and in some cases of abundant moisture, even peat (H_a, H_e, and H_i horizons). Most meso- and microfauna and ectomycorrhizal fungi concentrate their life cycles in these first centimeters of forest soils (Prescott and Vesterdal, 2021; Peng et al., 2022).

TABLE 1 | Summary of environmental features of the study sites.

Code	Type of fire	Land tenure	Fires[a]	YSLF[b]	Vegetation	Management
C(0)	None (Control plot)	Private	—	—	*Pinus-Quercus* Forest	Research activities
S(1)	Surface	*Ejidal* (Santa Rita)	2017	2	*Pinus-Quercus* Forest	Conservation
S(2)	Surface	*Ejidal* (Santa Rita)	2016, 2019	0	*Pinus-Quercus* Forest	Forestry
G(1)	Ground	Private	1998[c]	21	*Pinus-Quercus* Forest	Resin tapping
G(2)	Ground	Private	2012, 2016	3	*Pinus-Quercus* Forest	Conservation

—, Not applicable.
[a]Year of each fire in the period between 1998 and 2018.
[b]Years since last fire to the sampling year.
[c]Estimated year through conversations with the landowner.

Surface fires, which spread at a high speed, rapidly consume the understory without significantly changing the soil properties (Agee, 1993; Keeley, 2009). However, when the forest floor includes thick layers of peat, duff, or litter, slow smoldering combustion can take place after flame combustion, which can generate profound soil changes (Ogle and Schumacher, 1998; Rein, 2015; Santoso et al., 2022). These effects have been studied in tropical, boreal, and sub-boreal peatland ecosystems; however, there is a gap in this knowledge in neotropical mountain ecosystems (Santoso et al., 2019).

Different time intervals between fires could affect tree-seed sprouting (Busby et al., 2020) and, thus, the forest's community composition and ecological services (Miller et al., 2011). However, the effects on soil properties are still an emerging field where non-linear relationships have been observed (Moghli et al., 2022).

Moreover, while it is well established that low soil water content in fire-affected water-repellent soils can exacerbate the risk of erosion (Doerr et al., 2000; Shakesby and Doerr, 2006; Bento-Gonçalves et al., 2012), the emergence and severity of water-repellency do not occur in a linear progression as it dries. Rather, it occurs as a complex phenomenon where slightly moist soils can be even more repellent than completely dry soils (Wallis and Horne, 1992; Majid et al., 2023). Parameters such as IRDI (Integrative Repellency Dynamic Index), gravimetric water content at maximum water repellency levels (w.max), or gravimetric water content where the water repellency starts to be negligible (w.min) are alternatives to capture this dynamism (Regalado and Ritter, 2006; Regalado and Ritter, 2008).

In this sense, this work arises as a response to understanding the legacy effects of surface and ground fires on the infiltration capacity of mountain forest soils and to evaluate the degree of resistance of the Andosols to up to two recurring fires. We will also validate the presence of smoldering fire in two plots with duff in their forest floor. The central hypothesis of this research is that there will be a negative effect of the fire recurrence on soil hydraulic properties with a positive correlation of the number of fires with soil water repellency, and a negative relationship with hydraulic conductivity.

MATERIALS AND METHODS

Study Sites and Soil Description

We studied five sites, one of them an unburned control, and four affected by different fires between 2000 and 2018; showing blackening of the trunks up to a height of 1.5 m (Rodríguez Rodríguez, 2014; Olivares-Martínez, 2020). Their land tenure was collective (ejido) and private (**Table 1** and **Figure 1**), so all landowners of all sites were contacted to obtain their informed consent before fieldwork.

Due to the serious insecurity caused by drug cartels in the region, ensuring a minimum security condition necessary to carry out the fieldwork was also an important criterion. This limited the final number of sites and samples studied, in order to ensure the safety and physical integrity of the staff who worked in the field.

All the sites are within the physiographic province called *Neovolcanic Axis* with a semi-humid temperate climate, with annual average temperatures of 12°C–18°C and annual rainfall between 520 and 1,000 mm concentrated almost entirely during the summertime (INEGI, 1985a, 1985b, 2013). It corresponds with the Köppen climate classification Cwb, and with the modified Köppen climate classification C(m)(w) and C(w2)(w) (García, 2004; Peel et al., 2007).

The vegetation is composed by pine-oak forest over volcanic soils with slopes between 15° and 35° (INEGI, 2017; Olivares-Martinez et al., 2023). The forest has dominance of *Pinus pseudostrobus* and co-dominance of *Quercus rugosa*, *Quercus laurina,* and *Quercus crassipes* (Olivares-Martínez, 2020). Tree densities are between 170 and 620 trees per hectare, canopies of more than 80%, and in these sites there are some forestry activities like resinating pine trees or collecting firewood without logging.

The burned sites were affected by small fires of less than 5 ha and the burned trees had flame scars up to 2 m. In two plots, in addition to the evidence of flames, there were charcoal residues between the deeper duff and down to 10 cm in the mineral soil, which suggested the presence of latent combustion (Olivares-Martínez, 2020). They were assigned codes according to their fire history status: capital letters to express if they were Surface (S), Ground (G) or Control (C) plots, and number for the number of fires in the 18 years period.

The burned soils have loamy textures with clay percentages between 9% and 20% and very high organic matter contents (Olivares-Martínez, 2020); the first 5 cm of the mineral soil have soil organic matter levels between 7% and 19.5%. The sites affected by ground fires had duff forest floors up to 12 cm in depth, with assumed soil organic carbon levels of more than 20%, by definition (IUSS Working Group WRB, 2022). On the other hand, the reference control plot has a silty loam texture, with 6%

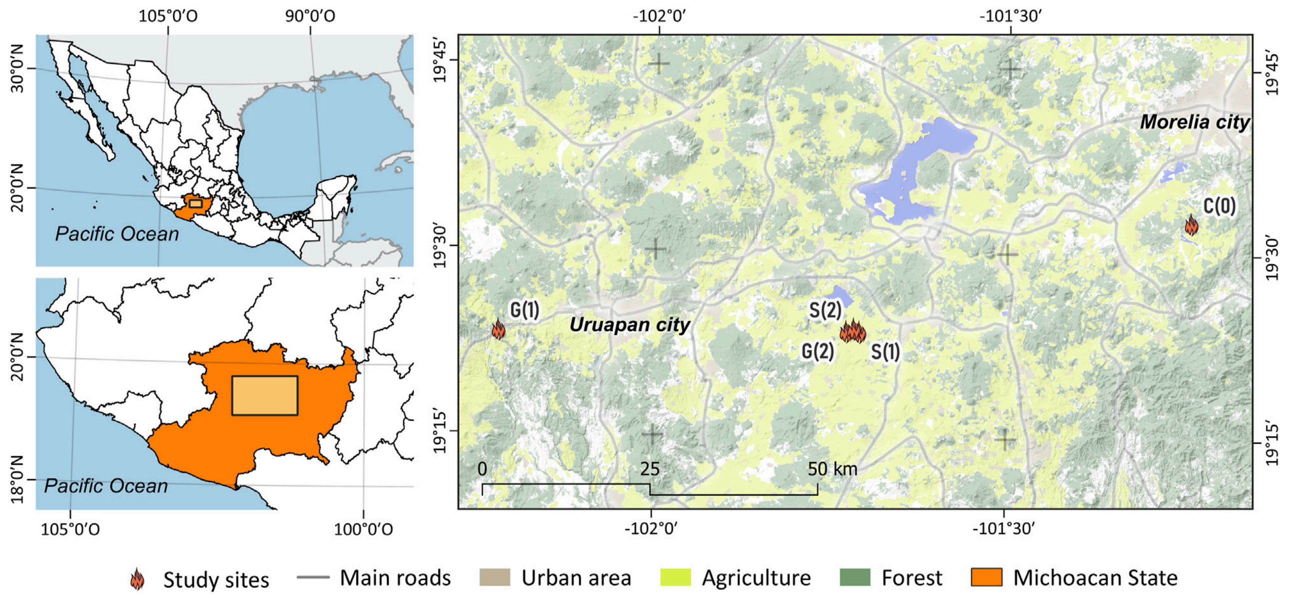

FIGURE 1 | Study sites inside the volcanic region of Central Mexico.

of clay, 7% of organic matter, thick organic layers of duff, and litter of 14 cm in depth (Rodríguez Rodríguez, 2014). All the soils have *Andic* properties and were classified as *Silandic Andosols* (IUSS Working Group WRB, 2022). At each site, the type of mulch was evaluated according to its thickness and mineralization/humification processes (Siebe et al., 1996), a soil pit was dug, and the genetic horizons were described (FAO, 2009).

Infiltration Tests

INDI tension infiltrometers were used to perform 15 to 20 infiltration tests per site to ensure adequate representativeness according to its log normal statistical distribution [Gómez-Tagle (Jr.) Ch. et al., 2011]. The tests were conducted on the forest floor, which comprised mainly mineral A horizons, and in some cases organic O_a horizons after removing the litter. The INDI is a low-cost self-made device to characterize the saturated & unsaturated field hydraulic conductivity and estimate the effective soil porosity (Eqs **A1–A3**) (Watson and Luxmoore, 1986; Gómez-Tagle Chávez et al., 2014). The INDI tension infiltrometer is inspired by designs from Špongrová et al. (2009) and Perroux and White (1988). Further details can be reviewed by the reader in Gómez-Tagle et al. (2011).

We applied −9.0, −3.0, and −0.5 cm tensions. We considered geometric corrections and the non-linear numerical approximation from Wooding's and Gardner's equations (Eqs **A1**, **A2**) to know the saturated and unsaturated hydraulic conductivity of the field (Logsdon and Jaynes, 1993). The number of active pores and the volumetric percentage were estimated from the unsaturated infiltration curves and the Jurin Law (Eqs **A3**, **A4**) (Watson and Luxmoore, 1986; Gómez-Tagle Chávez et al., 2014). We assumed the dynamic water viscosity as 1.002E-03 $kg \cdot m^{-1} \cdot s^{-1}$ (standard at 20°C), the water density as 998.2 $kg \cdot m^3$ (standard at 20°C), and the gravity acceleration as 9.81 $m \cdot s^{-2}$, and a macropore size of more than 0.5 mm (according to Luxmoore, 1981).

To ensure the complete contact of the base of the infiltrometer and the soil surface, we used a fine sandy marble contact bed (90% sand, 8% silt, and 2% clay; *"Marmolina comercial Tipo I, Fina"*) (Gómez-Tagle Chávez et al., 2014; Rodríguez Rodríguez, 2014). In the case of steep slopes (more than 35%), we made small terraces to level the infiltrometer, ensuring the measurement occurred within the target soil depth. This is a reliable method with no risk of losses by lateral flow when there are no abrupt textural changes (Miyazaki, 1988).

Soil Sampling and Lab Analysis

A core of about 1 cm in thickness and 5 cm in diameter was taken inside the area where each infiltration test was performed. The cores were carefully put in Petri dishes to measure water repellency at different moisture levels, according to the *Water Drop Penetration Time* test and the *Molarity of an Ethanol Droplet* (Woudt, 1959; Letey et al., 2000). In each case, the integrative dynamic repellency index (IRDI), maximum water repellency angle (a.max or α_{max}), water repellency angle after drying the samples at 105°C (a.105 or $\alpha_{105°C}$), the trapezoid integral area below the $\alpha(\theta_g)$ curve (S), and gravimetric water content where the water repellency starts to be neglectable (w.min or $\theta_{g.min}$), were calculated (Regalado and Ritter, 2005).

Equation Solving and Statistical Analysis

R v4.2.2 statistical software was used to solve the Wooding & Gardner non-linear equations and to perform statistical tests (R Core Team, 2022). One-way ANOVAs were done to assess the

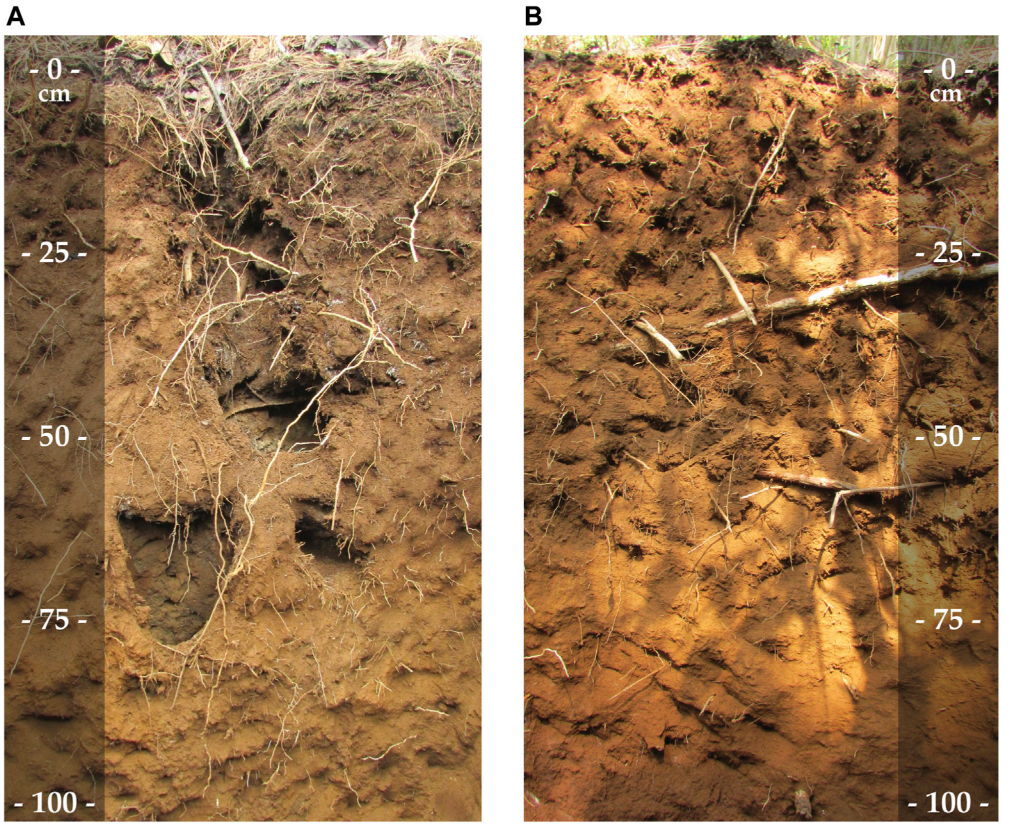

FIGURE 2 | Morphology comparison between plots with strong **(B)** and insignificant **(A)** presence of smoldering combustion. Note the galleries with root charcoal until 80 cm.

effects of type of fire and recurrence onto the hydrophysical soil variables comparing with control plot. Two-way ANOVAs were carried out, excluding the control plot to explore the effect of the interaction between the factors of type (surface, and ground fire), and recurrence (one, or two fires inside the period). Log-normal adjustments to compare maximum likelihood estimators were performed to the four hydraulic conductivity-derived variables, the water repellency angle after drying the samples at 105°C, and the maximum water repellency angle. A *post hoc* honestly-significant-difference test by means of Tukey (HSD) was done in the significant models.

RESULTS

Morphology of the Burned Volcanic Soils

The four burned sites showed fire scars with charcoal residues in the bark of their trees and soil water repellency. The volumetric field water contents of the sites, when sampled, were from 50% to 102%. The main burned trees were pines, but in the G(1) plot, scars were also found on some oaks.

The S(1) and S(2) plots presented a *Mull humus* (duff). On the other hand, the G(1) and G(2) plots had a *Mor humus* (duff), and there was evidence of smoldering combustion in both. There were visible charcoal residues found to a depth of more than 10 cm below the surface. A system of galleries and macropores with the rest of the root charcoal down to more than 50 cm was observed on the G(2) plot (**Figure 2**).

Hydro-Physical Effects From Ground and Surface Fire

A total of 85 water infiltration tests were performed. Nine tests failed due to high water repellency levels on the field, so only 76 tests were completed. The field-saturated hydraulic conductivity was from moderately high to high in all sites, the control plot had 42 mm per hour, much higher compared to the burned plots, with a mean hydraulic conductivity of 12.88 mm per hour (**Table 2**).

The water flux via macropores decreased after fires, like the field-saturated hydraulic conductivity. Burned soils could have a macropore flux 20 times less than the reference soil. Nevertheless, the number of active pores ($N_{M \cdot m^{-2}}$) had contrasting scenarios across the different types and fire recurrencies (**Tables 2**, **3**).

A total of 63 water repellency tests were performed throughout different levels of soil moisture content. Almost all the samples showed water repellency to some extent; only tree samples in G(2)

TABLE 2 | Hydraulic conductivity and active porosity summary.

Plot	*n*	Ks mm · hr^{-1}	$\alpha \cdot m^{-1}$	PF_M %	$N_M \cdot m^{-2}$
C(0)	20	42.14 ± 9.42 **a**	1.57 ± 0.35 **c**	92.6 ± 20.7 **a**	715 ± 160 **a**
S(1)	15	12.35 ± 3.19 **bc**	11.94 ± 3.08 **ab**	22.1 ± 5.7 **bc**	451 ± 116 **ab**
S(2)	20	13.68 ± 3.06 **b**	7.19 ± 1.61 **b**	18.7 ± 4.2 **bc**	466 ± 104 **ab**
G(1)	8	6.51 ± 2.3 **c**	10.24 ± 3.62 **b**	16.5 ± 5.8 **c**	171 ± 60 **b**
G(2)	13	18.99 ± 5.27 **b**	22.91 ± 6.35 **a**	33.7 ± 9.4 **b**	1757 ± 487 **a**

Mean values ± standard deviations, where n *is the number of water infiltration essays. Ks is the hydraulic conductivity with Gardner's α parameter. Bolded letters indicate the HSD groups for each variable (log-normal distribution in the four cases), same letter means no significant differences.*

TABLE 3 | Significative ANOVA models between fire histories.

Model	*p*	d.f.	F
Ks ~ Type + Recurrence	*	1	5.59
α ~ Type × Recurrence	**	1	11.28
PF_M ~ Type × Recurrence	**	1	10.70
$N_M \cdot m^{-2}$~ Type × Recurrence	**	1	7.57
IRDI ~ Type × Recurrence	**	1	11.06
a.max ~ Type + Recurrence	*	1	5.53
a.105 ~ Recurrence	***	2	9.41
a.105 ~ Type	***	2	15.84
S ~ Type	***	1	9.06
w.max ~ Type	*	2	4.03
w.min ~ Type	***	2	11.45

*Where *** means* p < 0.001, **p < 0.01, and *p < 0.05, *× means a significant interaction between the factors.*

plot were completely hydrophilic. The mean a. max throughout the samples was 100.7° with no significant differences when type and recurrence of the fire were considered ($p < 0.05$, *d. f.* = 1, $F = 5.53$), significant differences could be found only considering a one-way ANOVA model ($p > 0.001$, *d. f.* = 4, $F = 6.55$). The rest of the water-repellency parameters showed statistically significant differences (**Figure 3**).

The control site showed a higher a.105 dry-soil water repellency than the burned plots: 100.6° against a range of 90.7°–96.5°, with only one significant difference between C(0) and S(1). The a.max, and the S parameters showed similar patterns. The w.min showed significant differences after the first fire, according to the type of fire but not when compared to the control plot (**Figure 3**). On the other hand, the IRDI showed significant differences in the control plot, specifically after a second surface fire. However, a second ground fire reestablished the IRDI to reference values.

The hydraulic soil properties and the active porosity showed a consistent and significant relationship to the type and the cumulative number of fires (**Table 3**). On the other hand, the soil water repellency dynamic parameters showed differences mainly according to the type of fire, and just the integrated index IRDI showed a significant relationship with fire type and recurrence ($p > 0.01$, d.f. = 1, and F = 11.06).

After two fires, the water repellency could have contrasting dynamics depending on the type of fire and the type of forest floor. For example, the duff-covered G(2) soil showed a similar total water repellency to the control plot with the IRDI index but with higher maximums at a. max and even a lower dry-soil water repellency at a.105 values; on the other hand, the S(2) showed a decrease in the IRDI, similar values to the control plot on w. min and a. max, but a lower a.105 (**Figure 3**).

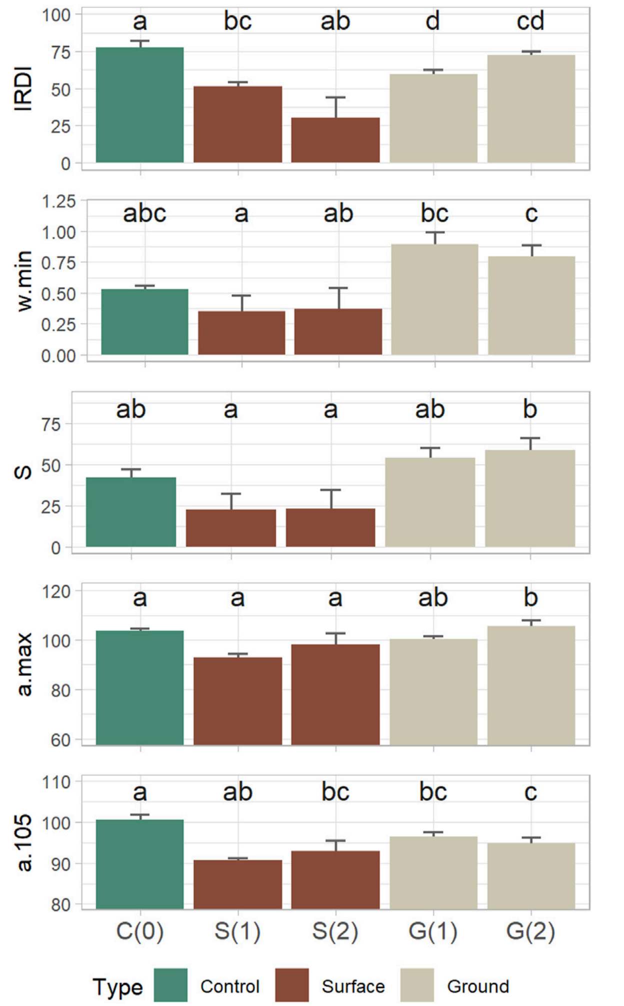

FIGURE 3 | Water repellency across different fire histories. IRDI as a dimensionless index, w.min as volumetric water content, a.105 and a.max as the estimated contact angles. The bars represent standard errors, and the letters indicate the HSD statistical groups. Codes in the horizontal axis according to **Table 1**.

DISCUSSION

Fire Legacies on Soil Memory

The presence of residues of organic duff layers (O_a and O_e genetic horizons) were found only on ground-fire-affected plots. When the soil has high organic matter (nearly 20%), it is highly likely to develop smoldering combustion after a surface fire (Rein, 2015). It is possible that the increased number of galleries and macropores on the G(2) plot after a first fire may have enhanced the oxygen availability so that a second fire could burn deeper layers and leave deep charcoal residues such as those found in the field.

Of all the components of a fire regime, the rate of spread of fire (and therefore the time of residence of the heat) is the one that can have the greatest effect on the soil (Agee, 1993; Keeley, 2009). In this sense, the two sites with clear evidence of smoldering combustion presented higher hydrophobicity at deeper layers (Olivares-Martínez, 2020). These plots had scars on pines and even on oaks (*Quercus* spp.), whose bark is less prone to be burned unless there are lower fire spread rates. The longer the time of residence, the deeper and greater the fire severity in the soil (Neary et al., 1999; Varner et al., 2005; Rein, 2015).

Smoldering fires can increase soil temperatures by more than 60°C down to 20 cm on soil mineral horizons (Varner et al., 2005) and more than 50 cm on peat layers (Wösten et al., 2006). So, it is logical that a ground fire would show lower standard errors than a surface fire in its water-repellency-related properties since the former has more homogeneous fires. The G(2) plot could have presented deep charcoal residues and big galleries due to the smoldering of whole trees' root systems adding to the organic soil smoldering combustion.

Organic layers can start to smolder even with gravimetric water contents up to 120% (Neary et al., 1999). When sampled, the field water contents of the ground-fire-burned sites were from 79% to 102%, so they could have burned at that time even when they seemed wet. Soil humidity is a factor that regulates both ease of burning and soil water repellency and water infiltration rates (Jordán et al., 2013; Mataix-Solera et al., 2014; Marín-Castro et al., 2016). It suggests that the humus type of the forest floor drives the water thresholds on smoldering to fire dynamics and the charcoal imprinted legacies.

To understand better why the control plot and G(2) plot presented the higher IRDI values is necessary to understand how the index is structured. The index is constructed with the S and the w.min variables to describe the dynamic of water repellency according to soil humidity, so its changes can be due to the size of the S parameter, which tells us about the total area under the water repellency curve, and in an inversely proportional way, the index changes with the w. min parameter (Regalado and Ritter, 2005).

Moreover, other indirect effects on the IRDI could be explained by the size of the differences between a.max and a.105 (difference known as α_{err}) (Regalado and Ritter, 2005). Although a. max did not show great differences between sites except for the Sup(1) plot, a.105 was sensitive to the number and type of fire. It is possible that the presence of resins and other tree exudates could increase the dry water repellency at the control plot even more than the burned plots (Rodríguez Rodríguez, 2014).

The humus type, porosity, and water repellency were noticeable components of the soil memory after different fire histories in addition to other traits such as recalcitrant organic matter (Jordán et al., 2013; Mataix-Solera et al., 2014), aggregation (Lal, 1997; Mataix-Solera et al., 2011) and ashes, which in turn alter the electrical conductivity, nutrient availability, and texture of the soil (Bodí et al., 2012; Francos et al., 2018; Olivares-Martínez, 2020).

Dynamic Water Repellency Remarks

The IRDI index was proposed as an attempt to understand better the non-linear dynamic of water repellency, and it should be interpreted as a shape index more than a total repellency index (e.g., S parameter), which tells where there are major differences between the hydrophobicity of dry and moist soil in narrow moisture gradients (Regalado and Ritter, 2005). A high IRDI value does not necessarily imply a stronger water repellency, it tells us about the relationship between the total water repellency and how easy it disappears with humidity.

Although no significant differences were seen in S and w.min, when combined in the IRDI it was possible to find significant differences generated by both recurrence and type of fire (**Figure 3**). The soils have imprinted different shapes in the repellency curve according to their fire history, so we can understand the IRDI as a "repellency signature." Moreover, although the plot two ground fires had increases in the persistence of soil water repellency, it reestablishes the form of this relationship, so the IRDI is very similar to the control plot. It would be interesting to further expand on this pattern by comparing with more sites that have a similar fire history.

It could be confusing when trying to consider the IRDI in hydrologic terms; the most prone sites to have problems with water infiltration and erosion will be those with the greatest water repellency area (S), a combination between high maximum water repellencies (a.max) and high minimum humidities (w.min) with low standard deviations. Water repellency is a patchy-distributed spatial variable (Woods et al., 2007), so greater standard errors such as those found at surface fires (bigger error bars at S and w.min on **Figure 3**) suggest a strong relationship with Hortonian water fluxes to a more hydrophilic matrix. Surface fires tend to have higher spread rates, so heat transmissions to soil are more scattered, and, consequently, the soil changes (Agee, 1993; Mataix-Solera et al., 2007).

Although without noticeable effects on the number of active macropores on these volcanic soils, the amount of water flowing through them was affected by the type and number of fires. These reductions may be linked to the persistence of water repellency without significant changes in organic matter content (Olivares-Martínez, 2020). The w.min, as a measure of the persistence of hydrophobicity in humid conditions, is important to show that even after the first rains, soil can still have water repellency. That could be important when modeling surface water runoff and soil erosion processes (Cerdà and Robichaud, 2009).

Resilience and Degradation

Although only the C(0) and the G(2) plots were above the global hydraulic conductivity average according to their textural class (Gupta et al., 2020), it could be common that Andosols have low hydraulic conductivities, especially when water repellency is present (Farahnak et al., 2019). Neris et al. (2013) have shown that Andosols beneath pine vegetation cover tend to have lower infiltration capacity due to the water repellency associated with the type of duff of its forest floor. This makes them more susceptible to erosion by runoff, despite the high resistance to degradation that these soils may have.

Average rainfall in the region can be up to 9 mm daily during the rainy season (CNA and SEMARNAT, 2020). If we assume that convective precipitation events could concentrate the day's rainfall in 1 h, all the rainwater can be infiltrated with the cumulative effect of one or two surface fires. The assumed runoff exceeded the water infiltration capacity only in the G(1) plot, which has a mean hydraulic conductivity of 6.5 mm per hour, the rest of the burnt plots can buffer this mean rainfall (**Table 2**). So, this could indicate that even with a reduced infiltration and enhanced water repellency, one or even two fires in 18 years have no serious repercussions for surface runoff and water erosion. In any case, further studies are necessary to confirm this result.

Despite the infiltration increase after a second ground fire, it should be considered that a higher standard deviation (patchiness), coupled with a low percentage of macropore flow and a higher repellency, could imply preferential finger flows (Ruthrof et al., 2019). This type of water movement may lead to higher risks of nutrient and contaminant leaching at recursive soil points (Ritsema and Dekker, 1994; Debano, 2000) but higher water infiltration potential at the landscape level (Doerr et al., 2000). Therefore, there may be an increase in the ecological function of water supply but not in water purification.

In addition to preferential flow pathways, humidity's non-linear effect on the soil hydraulic conductivity becomes negligible only when soil water content surpasses a certain threshold (Regalado and Ritter, 2005). Nevertheless, it is essential to note that a high soil water content is not always a safeguard against water repellency since certain soil types, including Andosols, can retain their hydrophobic properties even when moistened (Wallis and Horne, 1992; Regalado and Ritter, 2008; Jordán et al., 2013).

Landscape Concerns

As shown in the previous sections, sites with sporadic fires in these temperate neotropical forests of the avocado belt will not face major degradation problems, especially on surface fires, where scattered water repellency does not imply infiltration reductions at landscape levels (Doerr et al., 2000). Moreover, water repellency could enhance water storage in the soil by diminishing direct soil evaporation rates (Ruthrof et al., 2019). So, we suggest that inherited water repellency in the soil can be part of the dynamics of a healthy forest ecosystem when fire regimes do not exceed resilience thresholds on the soil.

Water repellency on soils is a common phenomenon that could be found even on reference sites (Weninger et al., 2019). The reference forest of this work could have water repellency even during the rainy seasons (Rodríguez Rodríguez, 2014). Activities like resining pine trees could increase the water repellency of the soils, decreasing their water infiltration potential; so recurrent low-severity fires could ameliorate this situation when they increase the patchiness of the soil water repellency distribution as long as they do not exceed land degradation thresholds.

However, sporadic fires tend to be an exception more than a rule, many of the fires in the forests of this region are associated with agricultural activities that are rapidly increasing their surface area (Barsimantov and Navia Antezana, 2012; Tejera Hernández et al., 2013; Olivares-Martinez et al., 2023). In this sense, it is pertinent to study whether areas with a higher fire frequency, enhanced by activities such as the international avocado trade, could exceed the soil's resilience, and trigger degrading processes.

CONCLUSION

In this work, we ascertained the presence of latent combustion in two temperate neotropical forests. The infiltration capacity of the Andosols showed a high decrease after ground or surface low-severity fires with high resistance to change in periods of up to 20 years. The IRDI soil water repellency parameter was sensible for fire histories with up to two recurrent fires. Parameters such as the percentage of charcoal, pore space structure, and water repellency were useful indicators for understanding the fire history of these soils.

Parameters such as moisture at minimum hydrophobicity w. min and area S under the hydrophobicity curve were found to be edaphic memory traits related to fire history in the Andosols of subtropical regions. Although reduced, the infiltration capacity of S(1), S(2), and G(2) plots is not diminished to such an extent that surface runoff is generated in an average rainfall.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusion of this article can be found at: https://github.com/OlivaresLD/fire-effects-on-soils.

AUTHOR CONTRIBUTIONS

LO-M performed the statistical tests, developed figures and tables, sampled the soils, and wrote the first versions of the manuscript. AG-T performed resource and logistical management, soil sample processing, and laboratory analysis. JM-S reviewed and contributed to the writing process, and collaborated on the financing of the manuscript. All authors contributed to the article and approved the submitted version.

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CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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APPENDIX

Equation 1: Gardner's equation (1958).

$$K_{(\psi)} = K_{(s)}e^{\alpha \cdot \psi} \tag{A1}$$

Where $\mathbf{K_{(\psi)}}$ is the unsaturated hydraulic conductivity at a specific water tension $\boldsymbol{\psi}$, $\mathbf{K_{(s)}}$ is the saturated hydraulic conductivity, and **Gardner's** $\boldsymbol{\alpha}$ is a shape parameter.

Equation 2: Logsdon & Jaynes' non-linear approximation.

$$\frac{q_\infty}{\pi r^2{}_{(\psi)}} = K_{(s)}e^{\alpha \cdot \psi} \cdot \left(1 + \frac{4}{\pi r \alpha}\right) \tag{A2}$$

where $\mathbf{q_\infty/\pi r^2{}_{(\psi)}}$ is the volumetric flux in steady state by a specific water tension $\boldsymbol{\psi}$, and **r** is the ratio of the infiltrometer circular base. The rest of the variables are the same as in Eq. **A1**.

Equation 3: Percentage of water flux by active macropores.

$$PF = \frac{K_s - K_s e^{\alpha h_M}}{K_s} \cdot 100 \tag{A3}$$

where **PF** is the flux percentage between the tension according to the macropore radius of 0.5 mm, corresponding to a water tension $\boldsymbol{h_M}$ of −0.0297 m (according to Jurin's Law), $\mathbf{K_s}$ is field saturated hydraulic conductivity ($m \cdot s^{-1}$) with its **Gardner's** $\boldsymbol{\alpha}$.

Equation 4: Number of active pores per unit area.

$$\frac{N}{A} = \frac{8\mu K_{(h_M)}}{\pi \rho g r^4_{(h_M)}} \cdot \frac{1}{\pi r^2_{(I)}} \tag{A4}$$

where **N** is the number of effective pores per unit area **A**, $\boldsymbol{\mu}$ is de dynamic water viscosity ($kg \cdot m^{-1} \cdot s^{-1}$), $\boldsymbol{\rho}$ is the density of water ($kg \cdot m^{-3}$), g is the gravity acceleration ($m \cdot s^{-2}$), $\mathbf{K_{(hM)}}$ is the field saturated hydraulic conductivity for the macropore water tension $\boldsymbol{h_M}$ ($m \cdot s^{-1}$), $\mathbf{r_{(hM)}}$ is the macropore radius (m), and $\mathbf{r_{(I)}}$ is the infiltrometer base radius (m).

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Soil Burn Severities Evaluation Using Micromorphology and Morphometry Traits After a Prescribed Burn in a Managed Forest

S. Moreno-Roso[1,2]*, *B. Chávez-Vergara*[2,3], *E. Solleiro-Rebolledo*[2], *S. Quintero-Gradilla*[4], *A. Merino*[5] *and M. Ruiz-Rojas*[1]

[1]*Posgrado en Ciencias de la Tierra, Universidad Nacional Autónoma de México, Ciudad de México, México,* [2]*Instituto de Geología, Departamento de Ciencias Ambientales y del Suelo, Universidad Nacional Autónoma de México, Ciudad de México, México,* [3]*Laboratorio Nacional de Geoquímica y Mineralogía, Ciudad de México, México,* [4]*Departamento de Ecología y Recursos Naturales del Centro Universitario de la Costa Sur, Universidad de Guadalajara, Guadalajara, Jalisco, México,* [5]*Departamento de Edafología y Química Agrícola, Escuela Politécnica Superior de Ingeniería, Universidad de Santiago de Compostela, Lugo, Spain*

Prescribed burn is a tool that must imply low soil burn severity (SBS) levels; however, a wide range of soil impacts have been demonstrated because of the influence of very variable factors. The effects on biological, physical, and chemical soil properties are well reported in numerous studies; nonetheless, there are still questions about the effect of prescribed burns on soils at the micro-scale. As a result, an analysis of the link between micromorphological features and SBS does not currently exist. Thus, the main aim of the present study is to perform a micro-scale evaluation for complementing the SBS visual examination after prescribed burning in a managed pine forest in western Mexico. Morphometry and micromorphology analyses of mineral soil revealed that at low SBS levels, only the soil structure in the first centimeter is affected by prescribed burns. While at high SBS, the prescribed burn affected the first 2 cm, showing soil structure disturbance, ash filling porous, and soil aggregates getting reddish. Therefore, immediate actions have to be made by land managers after applying prescribed burns before the first rain to prevent post-fire surface soil erosion, particularly in bare soil patches where the burned aggregates are more susceptible to rain splash and runoff.

Keywords: prescribed burn, fire, soil burn severity, micromorphology, morphometry

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****Correspondence:***
S. Moreno-Roso
soldjmoreno@gmail.com

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INTRODUCTION

The impacts of prescribed burns on soils are spatially variable due to the high heterogeneity of the environmental conditions (Hubbert et al., 2006; Girona-García et al., 2019). Although prescribed burns imply the application of low-intensity fire (Fernandes et al., 2013), different intensities and severities have been reported (Alcañiz et al., 2018). Therefore, their effects on soil properties can be very variable depending on the fire and vegetation type, the previous soil conditions, and their correct execution (Scharenbroch et al., 2012; Badía et al., 2017; Alcañiz et al., 2018; Girona-García et al., 2019). Additionally, the sampling method is another factor that can also influence the variations of soil fire impacts, such as the time elapsed after the fire, the mixture of charred O and Ah horizons, and even the depth of soil sampling with a dilution effect (Armas-Herrera et al., 2016; Lucas-Borja et al., 2019; Pereira et al., 2023). The variability of all these factors for each intervened area can produce distinctive soil impacts, resulting in patches with different soil burn severity levels (Hubbert et al., 2006; Alcañiz et al., 2018).

The degree of soil burn severity determines the modification of biological, chemical, and physical soil properties (Santín and Doerr, 2016). The effect of heating on soil properties such as pH, total organic carbon (TOC), total nitrogen (TN), and total phosphorous (TP) have been extensively studied (Alcañiz et al., 2018; García-Oliva et al., 2018; Merino et al., 2019). It is accepted that pH values, TOC, and TP contents increase after a low-severity prescribed burn (Ulery et al., 1993; Arocena and Opio, 2003; Úbeda et al., 2005; Afif and Oliveira, 2006; Scharenbroch et al., 2012), however, there is an unclear pattern in the case of TN (Alcañiz et al., 2018). After the fire, changes in soil physical properties such as aggregates stability, bulk density, and moisture have been described (Neary et al., 1999; Debano, 2000; Mataix-Solera et al., 2011; Badía et al., 2017). Some authors reported a decrease in soil structure stability because high temperatures disrupted aggregates cement, generating cracks and the disaggregation of soil peds (Badía and Martí, 2003a; Arocena and Opio, 2003; Chief et al., 2012) related to TOC losses and changes in clays or Fe contents (Mataix-Solera and Doerr, 2004; Mataix-Solera et al., 2011). Regarding bulk density, there are also controversial results because after a prescribed burn, it can increase (Hubbert et al., 2006), decrease (Chief et al., 2012), or remain unchangeable (Phillips et al., 2000). These discrepancies are likely due to the different soil burn severity (SBS). Likewise, soil moisture behavior can vary according to the soil moisture conditions and the prescribed burn type (Girona-García et al., 2019). Even depending on the previous soil moisture conditions, this property can protect the soil against the effects of fire (Badía et al., 2017). Soil moisture can have insignificant changes following a prescribed fire (Iverson and Hutchinson, 2002), and under laboratory conditions, soil moisture content can decrease dramatically in the first centimeters of soil depth (Badía et al., 2017).

For evaluating the impacts of burn areas, a classification index has been proposed to estimate the SBS with a five levels scale based on soil organic layer characteristics and mineral soil surface morphology (Ryan and Noste, 1985; Vega et al., 2013). This index is closely related to soil physico-chemical changes from unburned to high burn severity (Vega et al., 2013; Merino et al., 2018). For prescribed burns, this tool must imply low SBS levels (Scharenbroch et al., 2012) corresponding to levels without visible surface soil impact but with charred forest floor (Vega et al., 2013). Although the important contribution of the SBS index, there are still questions about the changes at micro-scale that are not visible to the naked eye, underestimating the fire effect on the soil. Micromorphology is a valuable technique to detect the combustion features (e.g., charcoal, charred plant material, ashes, and reddened aggregates) in soils and sediments, commonly used in archaeological contexts (Alperson-Afil, 2012; Berna et al., 2012; Mallol et al., 2017), making possible to interpret the fire history in a specific place. In the case of prescribed burns, micromorphology has already been used in soils of Australia, United States, and Spain (Greene et al., 1990; Phillips et al., 2000; Badía et al., 2020). Particularly, Badía et al. (2020) observed the formation of coatings of fine sand-size particles mixed with charcoal fragments present in the soil surface and infill pores. However, no studies still associate the micromorphological characteristics of burn soils with different levels of SBS.

In this study, we applied micromorphological and morphometric analyses to complement the information from the field visual examination and soil physico-chemical properties changes after a prescribed burn where different SBS levels occurred. Micromorphology and morphometry are novelty analyses that support the interpretations of the level of affection caused by the fire that can help in the future management of the forest and the understanding of post-fire soil processes.

MATERIALS AND METHODS

Study Area

The study was conducted in a managed pine forest in Ahuacapán Ejido (19° 36′–19° 41′ N and 104° 16′–104° 21′ W) of Autlán de Navarro, Jalisco, Mexico (**Figure 1**). This area is located within the upper zone of the Sierra de Manantlán Biosphere Reserve, at 1,900 m a.s.l., where the climate is temperate sub-humid Ca (w2) (w) (e) (g) (Martínez et al., 1991). The mean annual temperature, precipitation, and humidity, from 2011 to 2020, are 15.7°C ± 0.17°C, 1,771 ± 173 mm, and 80.7% ± 20.8%, respectively (Zuloaga-Aguilar, 2021). The soils are classified as Distric Cambisols, developed on Tertiary (Paleocene-Oligocene) intermediate extrusive igneous rocks (Jardel and Moreno, 2000). The forest vegetation is dominated by *Pinus douglasiana* under silvicultural practices, together with *Arbutus xalapensis*, *Fraxinus uhdei*, and *Carpinus tropicalis* in the underground (Jardel and Moreno, 2000). Because of human pressure and climatic conditions, this forest is considered a fire-prone ecosystem, subjected to moderate and low-intensity fires every 4.2–7.3 years (Rubio, 2007; Cerano-Paredes et al., 2015). Therefore, previous fires might mask part of the results of the present analysis.

Prescribed Burn Event Characteristics

The prescribed burning event was carried out on 21 March 2017, to reduce hazardous fuels and improve pine forest regeneration. Forests firefighters of the National Forestry Commission (CONAFOR), in conjunction with a local forest brigade of the Ahuacapán Ejido, conducted the burning supervised by fire management specialists from the University of Guadalajara (DERN-IMECBIO) and from the Sierra de Manantlán Biosphere Reserve.

The size of the burned area was 7.3 ha (**Figure 1**). The weather conditions were a relative humidity of 38%–76%, a temperature of 19°C–22°C, and a monthly accumulated precipitation of 3.81 mm from 1 to 21 March. The prescribed burning had a duration of 10 h, with the following observed characteristics: a flame height of 1.01–3.27 m; a rate of spread between 1.4–2.5 m min^{-1}; flame temperature between 439°C and 593°C; and total vertical consumption of 9 cm of fuel (81% of the pre-burn material, see **Figure 2**) (Pérez-Salicrup, 2018).

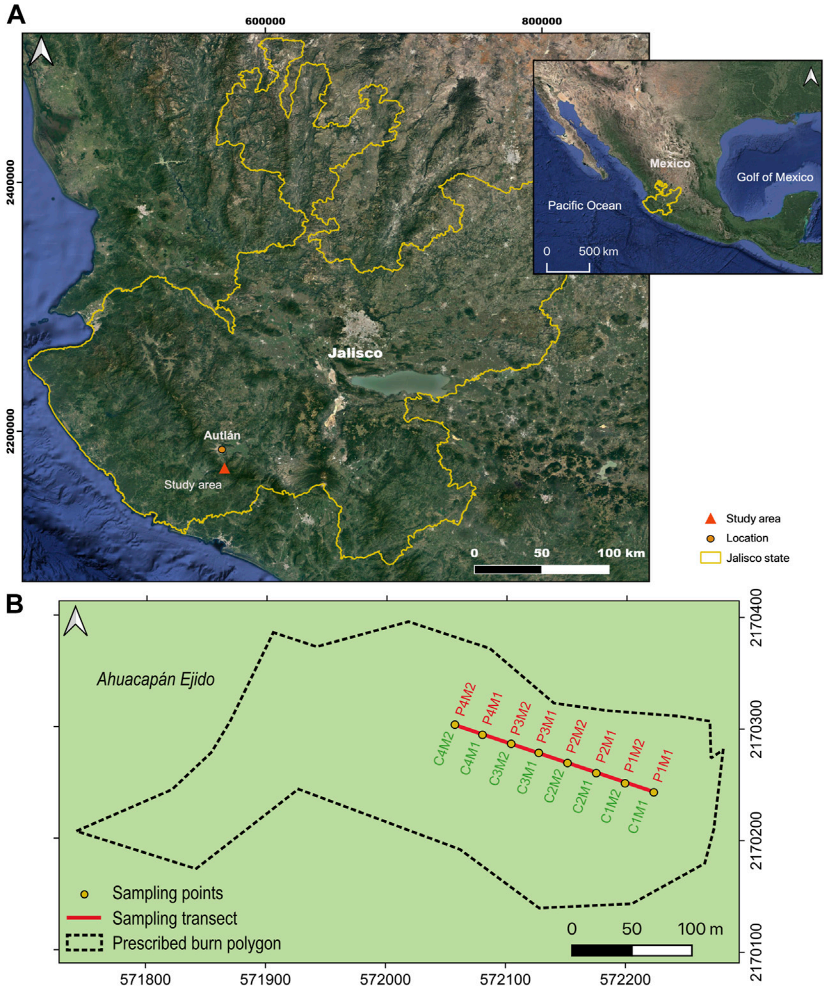

FIGURE 1 | Study area **(A)** location in the SW of Jalisco state, Mexico. **(B)** The prescribed burn area (7.3 ha) and the sampling design. The sampling was made in the same transect before and after the prescribed burn. Green names correspond to the unburned samples and red names to the burned samples.

Soil Sampling Design and Soil Burn Severity (SBS) Assessment

A transect randomly oriented from SE to NW 175 m long (**Figure 1B**) was established within the burned area. Before and after the prescribed burn, a sampling area of 1 m^2 every 25 m was delimited. After the fire event, on 22 March 2017, in the center of each square, the soil burn severity (SBS) was evaluated by a visual estimation using a 30 cm × 30 cm sampling metallic frame, following the procedure described by Ryan and Noste. (1985) modified by Vega et al. (2013). This SBS index considered the condition of the remaining soil organic layer, whether it was unburned, partially or wholly charred, or if it was completely consumed remaining only ash. Also, this post-fire assessment considered the visual characteristics of the mineral soil: if this was undisturbed, if the soil structure and the soil organic matter (SOM) were affected, and if there were soil color changes. Then, with these descriptions, a value from 0 to 5 was assigned for defining the SBS level in each sampling point, according to Vega et al. (2013) classification.

Before and after the fire, three types of samples were collected inside the 1 m^2 area from the first 5 cm of the mineral soil at each sampling point: 1) five samples integrated into one bulk sample for chemical analyses, stored in a plastic bag, 2) one unaltered sample was taken with a 118.8 cm^3 cylindrical steel core (5 cm depth), which was used for bulk density and moisture determinations, and 3) one unaltered sample was taken, from the organic layer and the upper 5 cm of the mineral soil, for micromorphological and morphometric analyses using cylindrical PVC cores.

Soil Analysis

Soil Physical Properties

The color measurements were triplicated with a colorimeter (COLORLITE Sph 870) considering the CIELAB color system (CIE, 1986). In this system, the color is represented by three scalar parameters: L*, luminosity, with values from 0 to 100 (from black to white, respectively); a*, related to changes from redness (a* > 0)

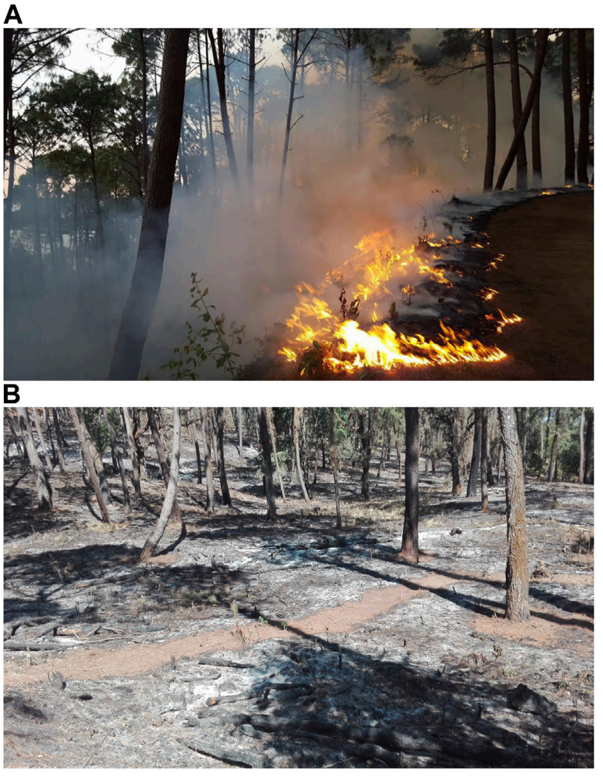

FIGURE 2 | (A) The prescribed burn event. **(B)** Features of the burned site the next day. Photos by Faviola Castillo.

to greenness ($a^* < 0$); and b^*, related to transitions from yellowness ($b^* > 0$) to blueness ($b^* < 0$) (Cancelo-González et al., 2014). Colors were obtained from sieved (2 mm) and homogenized samples.

Bulk density and moisture were evaluated using the cylinder method (Flores-Delgadillo and Alcalá-Martínez, 2010). Cores filled with soil were weighed fresh and dried at 105°C for 48 h until they reached a constant weight. The mass difference was used for moisture calculation, and the known volume's dry mass was used to calculate the bulk density.

Chemical Properties

Soil pH was measured in deionized water (1:10 W:V ratio) with a portable pH meter (Corning) in fresh samples. The total organic carbon (TOC), total nitrogen (TN), and total phosphorous (TP) were obtained after dry samples to 105°C until constant weight. The TOC was determined by combustion and coulometric detection (Huffman, 1977) in a UIC CM150 carbon analyzer. The TN and TP were determined in dry samples after acid digestion in H_2SO_4, H_2O_2, K_2SO_4, and $CuSO_4$ mixture at 360°C for 4 h. The TN was determined by the micro-Kjeldahl method (Bremner, 1996) and TP by the colorimetric method with molybdate after ascorbic acid reduction (Murphy and Riley, 1962); both elements were quantified by colorimetry in Seal AA3 segmented continue flow analyzer.

Morphometric and Micromorphological Analyses

The unaltered soil samples were previously dried and then impregnated with polyester resin. Thin sections (30 μm) were prepared at the Institute of Geology (UNAM). Morphometric analysis was done on high-resolution scanned images (2,400 dpi and 24 bits) of the thin sections using an EPSON scanner (Perfection V700 Photo). The Image-Pro Plus 7.0 software was used for processing the images. We selected a representative area (6 cm^2) of each thin section for developing the analysis. This analysis visually identified some burn features classified, such as charred elements, reddened zones, and ashes. The unburnt soil matrix and the porous space were also identified. The software separated and grouped the elements into five classes of different colors. Finally, we obtained a reclassified image, from which we calculated the proportion of each group per area.

The micromorphological analysis was made using a petrographic microscope (OLYMPUS). The analysis focused on identifying combustion features in thin sections such as charcoal, ashes, burned shells, charred aggregates, and reddened aggregates, according to Nicosia and Stoops. (2017) and Stoops et al. (2018).

Statistical Analysis

A correlation analysis (significance of $p \leq 0.05$) was performed among physical, chemical, and morphometric variables to avoid redundancy in the principal component analysis (PCA). The PCA was done to explain the changes due to the fire effect considering samples before and after the prescribed burn. The selection of the variables for the PCA considered the highest significance among them, demonstrated by their highest correlation. A JMP trial software was used for statistical analyses (JMP Statistical Discovery LLC, Cary, NC, United States).

RESULTS

Soil Burn Severity Levels Recorded After the Prescribed Burn

The prescribed burn event in the Ahuacapán Ejido (**Figure 1**) left a heterogeneous mosaic of areas with different SBS (**Figure 2**). Although most of the area was affected by low SBS (levels 2 and 1), high SBS levels (4 and 5) were only found in smaller areas (only in two sampling sites of eight, **Table 1**; **Figures 3C, D**). In level 1, the organic layer was partially burned; in the case of level 2, the Oa layer was completely charred, and gray ash was present. In these low SBS, the temperature did not affect the mineral soil and remained undisturbed (**Figures 3A, B**). On the contrary, in the highest SBS areas, the organic layer was entirely consumed (levels 4 and 5). In SBS 4 level, high temperatures also affected the uppermost mineral soil horizon in the 0–2 upper cm, showed a slightly reddish color, lack of fine roots, and degraded structure. Besides, in SBS 4, a thick layer of white ash covered the mineral soil; while in SBS 5 most of the surface soil was bare, with black and reddish colors (**Figures 3C, D**).

TABLE 1 | Visual description of the soil burn severity (SBS) levels found after the prescribed burn according to Vega et al. (2013).

SBS Level		Sampling sites	Visual characteristics
Low	1	P3M1 P4M4	The organic layer is partially intact and there is charred material. The forest floor can be distinguishable
	2	P1M1 P1M4 P2M1 P4M1	The organic layer was totally charred/Grey ash, and charcoal is covering the soil. The mineral soil is undisturbed
High	4	P2M4	The organic layer was completely consumed, and a thick layer of white ash is covering the soil. The uppermost mineral soil horizon (0–2 cm) is lighter and, without fine roots and lack of structure
	5	P3M4	The organic layer was completely consumed. The soil is partially covered by charred vegetal material and gray ash. The uppermost mineral soil horizon (0–2 cm) is dark and partially bare

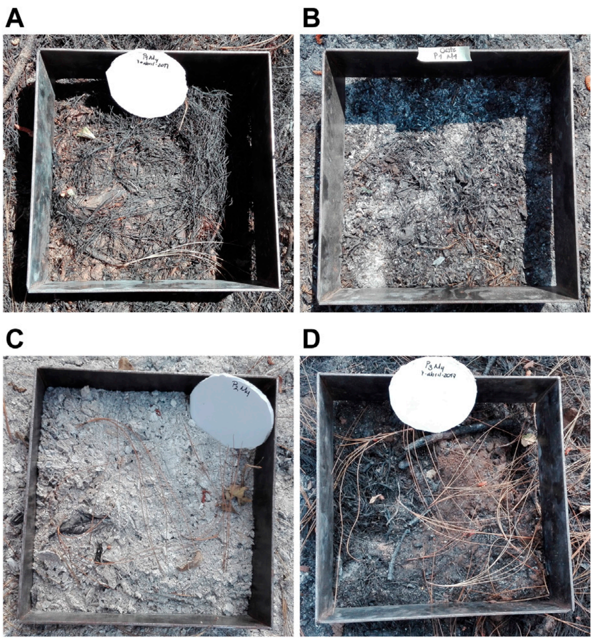

FIGURE 3 | Examples of the different levels of SBS after the prescribed burn in Ahuacapán Ejido: **(A)** low SBS (level 1), sample point P4M4; **(B)** low SBS (level 2), sample point P1M1; **(C)** high SBS (level 4), sample point P2M4; **(D)** high SBS (level 5), sample point P3M4. Photos by Faviola Castillo. The sampling metallic frames are 30 cm × 30 cm.

Changes in Physical and Chemical Soil Properties After the Prescribed Burn

Soil color measurements and mean moisture values for samples collected in the uppermost 5 cm showed changes after the prescribed burn (**Figure 4**). The mean values of the color parameters (L*, a*, and b*) decreased after soil burning, no matter the severity level. The reduction of luminosity (L*) mean values evidenced a soil darkening after the burn (**Figure 4A**). The decrease in the mean value of the a* and b* parameters showed fewer reddish and yellowish colors than the unburned condition (**Figures 4B, C**, respectively). Regarding soil moisture, this property tended to decrease slightly after the fire (**Figure 4D**). On the other hand, the soil bulk density increased 21.4% in the burned soils affected by low SBS and 22.6% by high SBS (**Figure 4E**).

Concerning the chemical soil properties, the pH values slightly increased after burning, especially in the high SBS (**Figure 5A**). The total organic carbon (TOC), no clear trend could be distinguished; it was reduced in soils affected by low SBS and increased in high SBS (**Figure 5B**). The TN concentrations did not show evident changes either (**Figure 5C**). Different trends in C/N ratios were evident because this ratio decreased in the low SBS. However, remarkable increases were found in the high SBS (**Figure 5D**). The TP values increased from unburned to burned condition 0.7 mg g^{-1} (**Figure 5E**).

Morphometric Analysis in Soils Affected by Different SBS

Images from thin sections with the organic layer and the upper 5 cm mineral soil from unburned and burned samples were morphometrically analyzed (**Table 2**; **Figure 6**). As logical, the lowest percentage of charred features was found in soils unaffected by fire (**Figure 6A**). All the scanned thin sections from the burn samples exhibited a darker layer on top (**Figure 6**). Samples with SBS level 1 showed the lowest number of unburned elements (e.g., aggregates, seeds, and plant tissues), coinciding with the highest porosity (46.3%, **Table 2**; **Figure 6B**) and followed by samples from SBS level 2 (37.5%, **Figure 6D**); the lowest porosity was found in the soil with SBS level 5 (14.5%). On the other hand, the morphometric analysis demonstrated that the highest number of elements with evidence of burning (e.g., charred and reddened aggregates, burned plant remains, and charcoal) was related to level 5 of the SBS index, reaching 50% of the considered area (**Figure 6E**; **Table 2**). Although level 4 showed fewer percentages of burning elements, only here mineral ash was detected (**Figure 6E**; **Table 2**).

In general, we identified a loss of structure and a reduction of the number of unburned elements and porosity with the increasing SBS. Additionally, ashes percentages and reddened aggregates were more significant in the highest SBS levels (4 and 5, respectively, **Table 2**). The morphometric data showed that the prescribed burn affected the soil organic layer and the first centimeter of the mineral soil for low

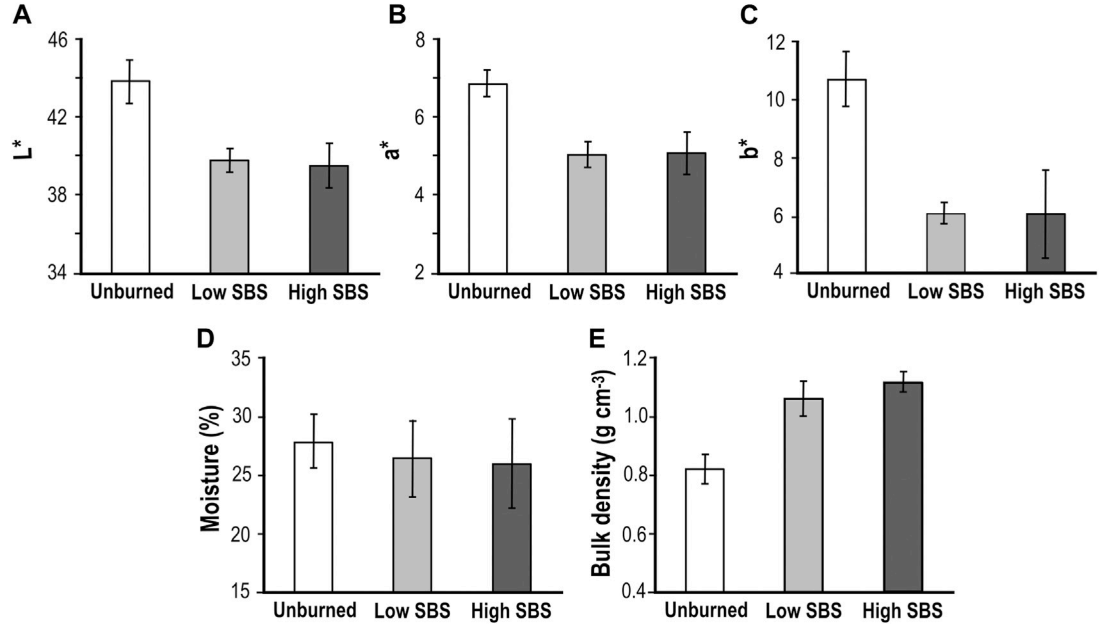

FIGURE 4 | Mean values and standard errors of physical properties of the soil samples collected from uppermost soil layer, before (unburned) and after the prescribed burn (low and high SBS). Low SBS corresponds to levels 1 and 2 (*n* = 6), and high SBS to levels 4 and 5 (*n* = 2). **(A)** L*; **(B)** a*; **(C)** b*; **(D)** Moisture; **(E)** Bulk density.

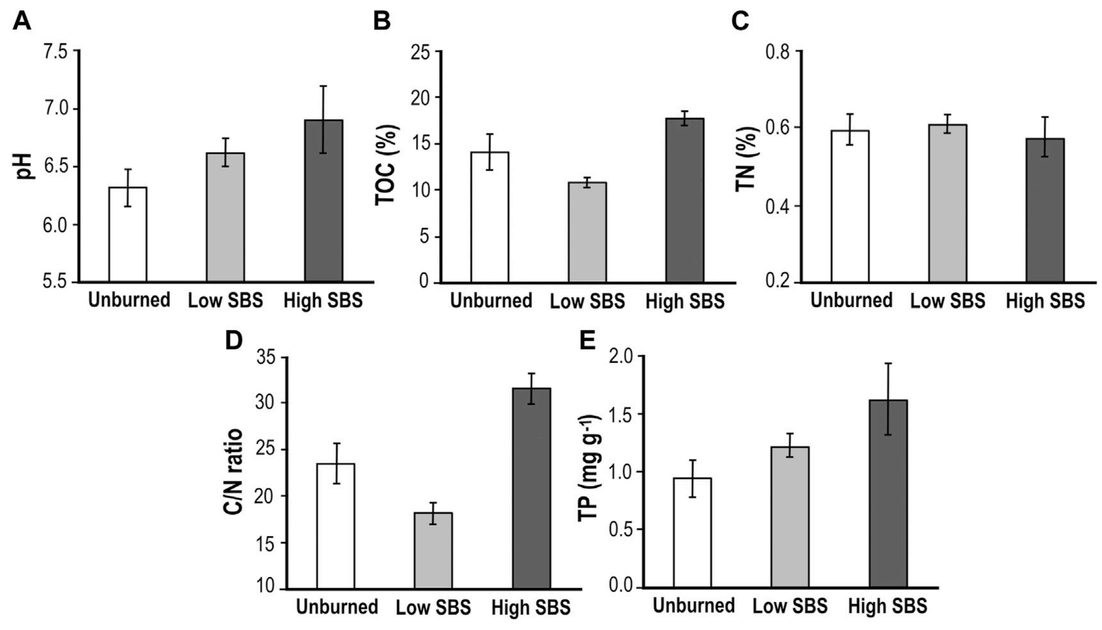

FIGURE 5 | Mean values and standard errors of chemical properties of the soil samples collected from uppermost soil layer, before (unburned) and after the prescribed burn (low and high SBS). Low SBS correspond to levels 1 and 2 (*n* = 6), and high SBS to the levels 4 and 5 (*n* = 2). **(A)** pH; **(B)** TOC; **(C)** TN; **(D)** C/N ratio; **(E)** TP.

SBS levels (**Figures 6B, C**) and the first 2 cm for high SBS levels (**Figures 6D, E**). The main proportion of burned remains corresponded to charred elements from the organic layer and the upper Ah horizon (**Figure 6**). It is important to mention that this examination might be including burning residues from previous fire events preserved in the soil. However, this previous evidence was discriminated by micromorphology analysis.

TABLE 2 | Soil morphometry results for the unburned and different SBS levels samples, showing the percentages related to the unburned and burned elements in a representative area (6 cm^2) of each thin section.

Classes	SBS level	Charred elements (%)	Reddened elements (%)	Ashes (%)	Total[a] (%)	Unburned elements (%)	Air and porous (%)
Unburned	0	4.2	0.03	—	4.2	78.2	17.6
Low SBS	1	19.7	7.8	—	27.6	26.0	46.3
	2	10.7	4.3	—	15.0	47.3	37.5
High SBS	4	10.3	3.4	15.0	28.7	48.7	22.5
	5	26.4	23.6	—	50.0	35.4	14.5

The data correspond to the soil organic layer and the upper 5 cm of the mineral soil.
[a]Elements with evidence of burning.

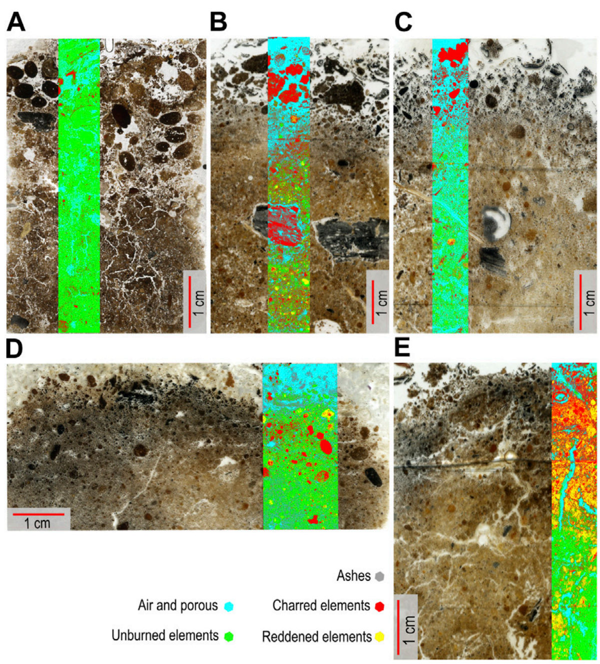

FIGURE 6 | Soil thin sections from the soil organic layer and the upper 5 cm of the mineral soil with the selected area used for the morphometry analysis: **(A)** Unburned topsoil with brown aggregates and fresh plant remains; **(B)** Sample at low SBS 1, burned aggregates at the surface soil; **(C)** Sample at low SBS 2, completely charred topsoil; **(D)** Sample at high SBS 4, ashes covering the burned soil. **(E)** Sample at high SBS 5, absent organic layer and the mineral soil with charred and reddened aggregates.

Soil Micromorphological Changes Due to the Prescribed Burn

The following descriptions will relate to the main micromorphological characteristics of unburned and burned samples from the organic layer, the top 5 cm of the mineral soil, and the burned features associated with each SBS level.

In the organic layer of unburned samples, fresh plant and animal residues, such as wood fragments (**Figure 7A**), seeds (**Figure 7B**), and shells (**Figure 7C**), were frequent and could be distinguished. The mineral topsoil showed an organic soil structure (sub-rounded aggregates pigmented by organic matter) (**Figure 7B**). Although these thin sections belong to unburned soils, a few rounded charcoal fragments were identified, possibly from previous fire events. A zoogenic structure (dark rounded peds) integrated by coprolites was also common (**Figure 7D**). Brown subangular aggregates were frequent from a depth of 3 cm and onward (**Figure 7C**).

At this microscopic scale, samples from low SBS level 1 areas showed an organic layer with partially charred plant tissues and some burned vegetal remains, such as charred wood, roots, and spikes. The mineral soil exhibited an organic structure, where some aggregates get darker after burning in all samples affected by this low SBS level (**Figures 8A–D**).

In the thin sections that showed a low SBS level 2, the prescribed burning heat induced a color change in the first centimeters of the soil, commonly covered by a black organic layer of 1 cm thick (**Figure 6C**). This layer comprised charcoal fragments from the charred organic layer remains, such as seeds, leaves, needles, stems, roots, and spikes in all samples (**Figures 8E–H**). These charred organic materials were frequently complete and preserved their cellular structure (**Figure 8F**). The surface soil aggregates for SBS level 2 was charred, fractured, and partially or totally disaggregated; a loss in the soil structure was also identified (**Figures 8G, H, 9B–D**).

The organic materials were reduced to ash in the samples showing SBS level 4, (gray material in **Figure 6D**, yellowish material associated with charcoal in **Figures 9B, C** and small shiny crystals in **Figure 9D**). Some ash particles fill the pores of the surface soil (**Figures 6E, 9C, D**). At this level, the combustion process of the organic layer was completed. In the mineral soil, the organic matter in aggregates, fine roots, and seeds were charred or turned to ash; rounded reddened aggregates appeared (**Figure 9A**); the porosity decreased; and the surface soil lost its structure (**Figure 6A**). The organic layer was absent in the SBS level 5, while the soil organic matter was charred in the first centimeter (black aggregates) (**Figure 6E**, **9E–H**).

Principal Components Analysis (PCA) of the Unburned and Burned Soils Data

In the PCA model, the first two principal components (PC) together explained 63% of the variance in the data (**Figure 10A**). The *L and pH are the variables that explain the most variance in PC1, while TOC and TN are in PC 2 (**Figure 10B**). On PC1, unburned samples are far from burned samples, mainly from those at SBS level 5 (**Figure 10A**). In contrast, on PC2 is possible to identify the overlaying by SBS level; in the center of the plot, the

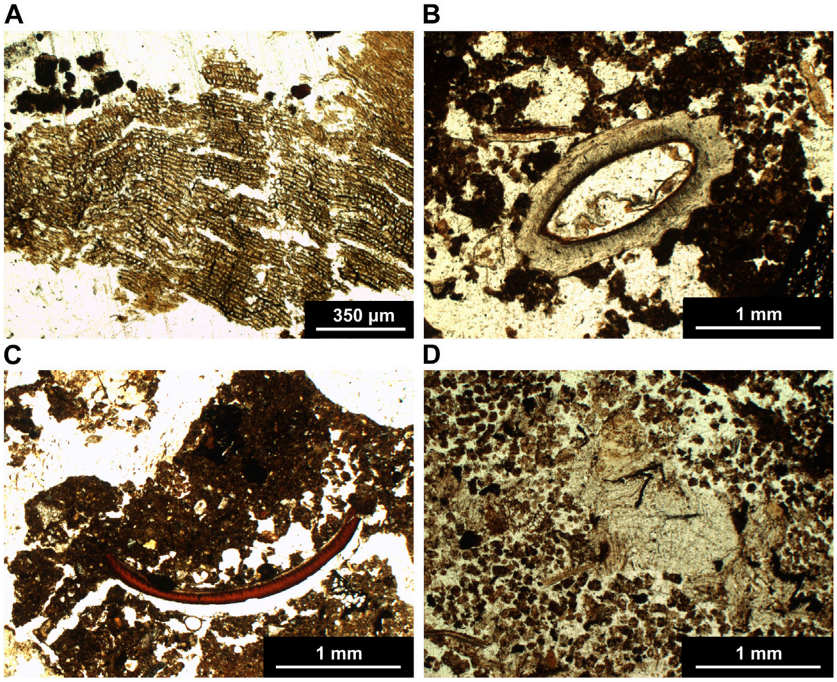

FIGURE 7 | Micromorphology characteristics from the unburned organic layer and unburned mineral soil: **(A)** fresh wood; **(B)** seed into the organic soil structure; **(C)** fresh shell with subangular soil aggregates; **(D)** rounded coprolites and vegetal remains.

unburned samples are together to SBS level 1. The SBS 2 samples are in lower values related to an increment of TN concentration and bulk density (**Figure 10**), while SBS levels 4 and 5 group are in higher values related to the increment of TOC concentration and reddened elements (**Figure 10**).

DISCUSSION

Changes in Soil Physical and Chemical Properties and Their Relationship With Micro-Scale Analyses

The effect of fire on soil physical-chemical properties has been described in several studies. Soil organic layer amount and composition can be affected (Nave et al., 2011; Merino et al., 2015; Santín et al., 2016). With respect to mineral soil, important changes in morphological, physical, and chemical properties can happen (Neary et al., 1999; Mataix-Solera et al., 2011), particularly in the uppermost 5 cm, where the highest temperature is reached (DeBano et al., 1979; Neary et al., 1999; Mataix-Solera et al., 2011; Badía et al., 2017). All these perturbations depend on the heating and are reflected as SBS levels. In the case of low-intensity prescribed burns, the heating caused minimal adverse changes in mineral soils (Debano, 2000; Mataix-Solera et al., 2011). The present study's PCA analysis separates the unburned from the burned samples by burn severity (**Figure 10A**), which agrees with the findings of Vega et al. (2013). The burned samples generally showed a darker color (L* value decrease), higher bulk density, higher pH values, and a higher percentage of charred and reddened elements than the unburned samples. Mainly, these properties explained 63% of the variance in the PCA analysis (**Figure 10A**). In addition, we observed modifications in other soil properties, such as moisture, TOC, TN, and TP.

Changes in the mean values of the chromatic parameters (L*, a*, and b*) indicate a darkening and a decrease in the redness and yellowness colors in the burned bulk samples (**Figure 4**). These results coincide with those obtained by Badía and Martí (2003a), where both value and chroma decrease as temperature increases, suggesting a darkening of the soil color of the charred samples. On the contrary, in the experiment conducted by Cancelo-González et al. (2014), the color parameters increased with the temperature rise in soils with different moisture contents (0%, 25%, and 50%). On the one hand, our unburned samples showed a mean value of soil moisture around 27%, and after the prescribed burn, a slight reduction of moisture was detected due to the water loss by vaporization in the uppermost 5 cm (Badía et al., 2017). On the other hand, Cancelo-González et al. (2014) experiment was done in soils derived from granitic rocks, whose mineralogy differs from that of our study (intermediate extrusive igneous rocks). Another difference was that we evaluated the color in sieved and homogenized samples, while Cancelo-González et al. (2014) measured the post-fire color on the undisturbed surface, which proved to be the best way for the evaluation. Nevertheless, the decrease in the L* parameter coincided with the morphometric and micromorphological

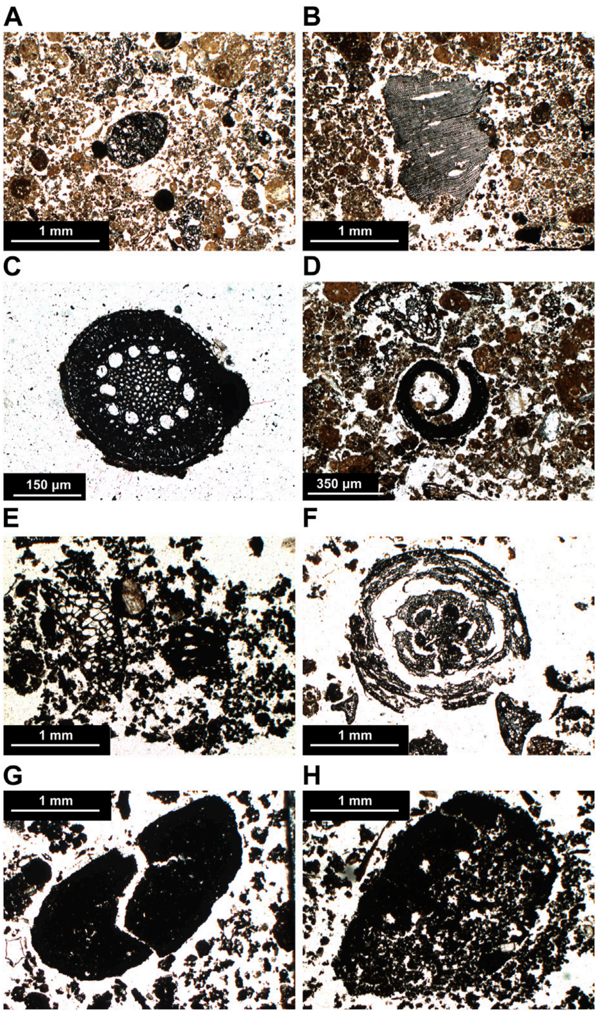

FIGURE 8 | Micromorphological combustion features of burned soils at low SBS levels 1 and 2: **(A)** charred aggregates and charcoal mixed with no burned aggregates, SBS 1; **(B)** charred wood, SBS 1; **(C)** charred root, SBS 1; **(D)** charred spike and needles in soil organic matter, SBS 1; **(E)** charred topsoil and charcoal, SBS 2; **(F)** charred pine stem, SBS 2; **(G)** charred and fractured aggregate, SBS 2; **(H)** charred partially disintegrated aggregate and soil structure disaggregation by fire, SBS 2.

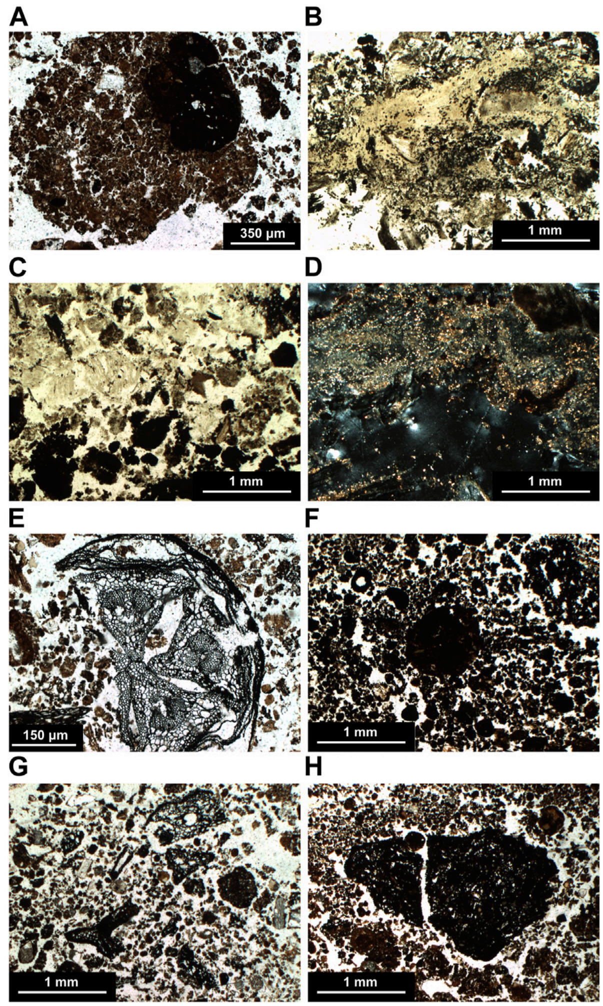

FIGURE 9 | Micromorphological combustion features of burned soils at high SBS levels 4 and 5; **(A)** reddened and rounded aggregate, SBS 4; **(B,C)** ashes in PPL, SBS 4; **(D)** ashes in XPL, SBS 4. PPL and XPL refer to plane and cross polarized light, respectively; **(E)** charred needles with its cellular structure well preserved, SBS 5; **(F,G)** charred aggregates and disaggregated soil structure, SBS 5; **(H)** charred and cracked aggregate, SBS 5.

analyses of the burn soils, where a dark black layer on the topsoils was identified. The darker color measured on burn samples with the colorimetric method corresponded to the abundant charred elements from the organic layer and the upper Ah horizon. Ketterings and Bigham. (2000) showed that soil became darker after fires at low temperatures (<250°C) of short duration because charred organic materials are dominant in the soil. Also, Cancelo-González et al. (2014) showed that organic matter content, temperature, and fire duration affected the color of burn soils. Concerning the reddened elements identified by micro-scale analyses of the samples with high SBS (**Figure 9A**), the colorimetric measurements did not detect them because this reddening was punctual, and the colorimeter performed an average of soil bulk samples color.

The observed increase of bulk density with heating (**Figure 5**) coincides with several studies where prescribed burn has been applied, independently of the kind of vegetation, brushes (Hubbert et al., 2006) or oak forest (Phillips et al., 2000), and for different soils heated under laboratory conditions (Badía and Martí, 2003a). Morphometric and micromorphological results showed the collapse of soil aggregates and the ash presence into the voids (**Figures 9G, H**), which can be other reasons to elevate the bulk density in the Ahuacapán Ejido. Likewise, Badía and Martí. (2003a) also found a moderate diminution of aggregate stability at 250°C, and Agbeshie et al. (2022) exhibited the breaking of organo-mineral aggregates and the clogging of porous by the ash-like causes of the bulk density increase. Moreover, this increase would indicate soil organic matter composition changes affecting the soil structure. Different

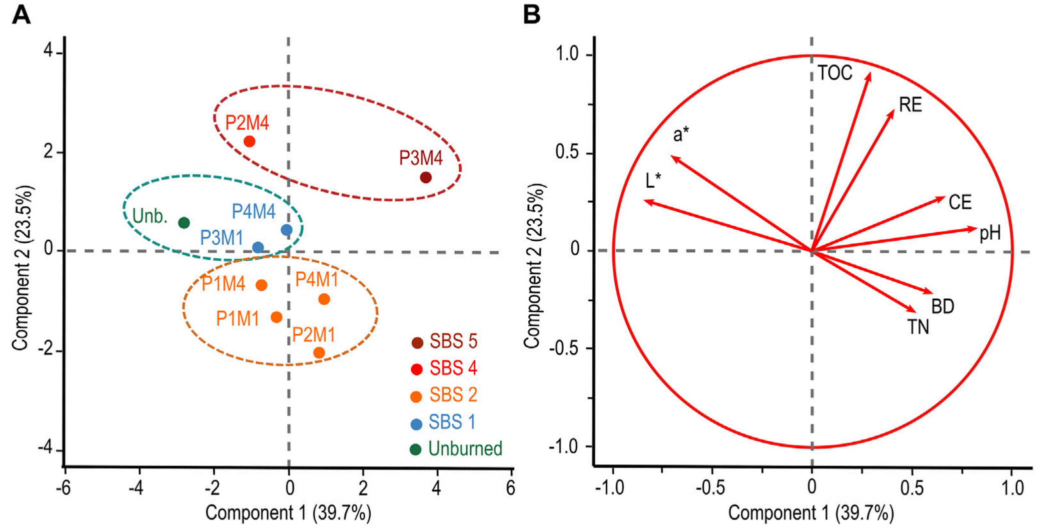

FIGURE 10 | (A) Principal components analysis for the unburned (average) and the burned samples before and after the prescribed burn, respectively; the soil burn severity level of the samples are represented in different colors. **(B)** The right plot showed the correlation between soil variables, considered for the analysis, and the principal components. TOC, total organic carbon; RE, reddened elements; CE, charred elements; BD, bulk density; TN, total nitrogen. L*: luminosity, a*: redness to greenness.

studies have revealed carbohydrate decreases with heating, implying lower aggregate stability (Gregorich et al., 1996; Kavdir et al., 2005; Merino et al., 2018). Another process involved in structure stability is the loss of binders, such as fine roots, hyphae, and microorganisms (Mataix-Solera et al., 2011). Additionally, decreases in biological properties in the soil surface after controlled burn have been reported both in the lab (Badía and Martí, 2003b) and in the field (Armas-Herrera et al., 2018; Girona-García et al., 2018; Alfaro-Leranoz et al., 2023).

We observed a decrease in TOC in the low SBS samples but an increase in the soils with a high-level SBS (**Figure 5**). The critical consumption of organic carbon starts between 200ºC and 250°C and is completed approximately at 460°C (Giovannini et al., 1988; Vega et al., 2013); however, when the temperature is higher than 500°C, a total loss of organic matter occurs (Terefe et al., 2008). Likewise, Badía and Martí. (2003a) found a coefficient of correlation of R = −0.84 (p < 0.01) between the SOM and heat, due to a significant decrease of the organic matter with temperature increase at 250°C and almost the completed consumption at 500°C. Therefore, no conclusive trend could be distinguished with the TOC, probably because of mixed effects: the unfinished combustion process of the organic matter and the integration of charred materials into the first centimeter of the mineral soil (Soto and Díaz-Fierros, 1993; Úbeda et al., 2005; Afif and Oliveira, 2006; Scharenbroch et al., 2012). Thus, it is possible that the light increases in TOC in the area showing high SBS were due to the presence of charcoal, as the high C/N ratios suggest it. This is supported by the important percentage of charred organic fragments integrated into the mineral topsoil detected in thin sections from the high SBS (**Figures 6D, E**).

Regarding another chemical property, the slight rise in pH registered after the prescribed burn is described by the whole oxidation of organic matter producing ash with high Ca, Mg, K, and carbonate content (Ulery et al., 1993; Arocena and Opio, 2003; Certini, 2005), as well as by the formation of Fe-hydroxides (Schwertmann and Fisher, 1973). Under the microscope, calcite crystals were detected in high-level SBS (**Figures 9F–H**). Higher pH values in burned samples are often reported (e.g., Alcañiz et al., 2018).

Little change in TN concentrations was noted after the prescribed burn (**Figure 5**). Similarly, Arocena and Opio. (2003) found no important modifications in the TN; the authors considered that TN loss was mainly restricted to the forest floor, or TN losses were compensated by atmospheric N deposition. However, this last possibility can be considered in the long term. Another explanation for this slight change in TN is that the maximum temperature on the soil surface of the prescribed burn was not enough to allow the organic nitrogen volatilization from the soil (Neary et al., 1999), and only a minor portion of the nitrogen was thermally mineralized and integrated into the soil from the forest floor as NH_4^+ (Dannenmann et al., 2011; Bird et al., 2015). On the other hand, Badía and Martí. (2003a) found a significant (R = −0.70; p < 0.01) reduction of TN at 500°C for two different soils under laboratory conditions. According to a review of several studies (Alcañiz et al., 2018), the behavior of TN after burning remains unclear. Due to the insignificant TN variations, the mean values of the C/N ratio showed the same trend as TOC concentration, with a reduction of the C/N ratio from the unburned to the burned samples at low SBS and an increment for the burned samples at high SBS.

The increase in the TP content after the prescribed burn (**Figure 5**) coincides with the results reported by other studies

(Wienhold and Klemmedson, 1992; Úbeda et al., 2005; Afif and Oliveira, 2006; Merino et al., 2018). Besides, the available P (Olsen extractable P) showed a significant increase by heating ($R = 0.48$; $p = 0.01$) in lab conditions (Badía and Martí, 2003a). This TP enrichment is mainly favored by the mineralization of organic matter (García-Oliva et al., 2018), together with the increase in pH, due to the liberation of basic cations during the organic matter combustion and the ash production (Ulery et al., 1993; Kennard and Gholz, 2001; Arocena and Opio, 2003; Certini, 2005).

Soil Burn Severity Index and Soil Micro-Scale Changes Due to Prescribed Burn

As mentioned before, one of the properties affected by fire is the soil structure. However, changes are not completely visible to the naked eye; consequently, the different SBS degrees are not identified. Micromorphological characterization is a complementary tool to evaluate the effect of the prescribed burns, as shown by Badía et al. (2020). The classification proposed by Vega et al. (2013) considers a macromorphological analysis, where the SBS 1 is associated with partial changes of the organic layer, while the mineral horizon remains unaltered. Our microscopic observations document some charred elements and burned features in the organic and mineral horizons (charred aggregates, partially charred plant tissues, burned roots, wood, and spikes). Although part of these charred materials can be formed during the present burn, it is important to notice that some features might also originated in previous fires (**Figure 6B**), as they were also identified in the unburned samples (**Figure 6A**). However, charcoals from prior fire events are easier to identify with micromorphological methods because they are more rounded than the fresh ones due to mechanical breaking during time (Ponomarenko, 1997; Ponomarenko and Anderson, 2001). Hobley et al. (2017) documented that charcoal fragments are quickly integrated into the soil profile after a fire event and can persist in the soil environment for a long time. In many archaeological sites, burnt features, identified at micro-scale, are well preserved, providing information about human impact on the ecosystem (Mallol et al., 2007; Huisman et al., 2019).

Similarly, in the SBS 2 Vega et al. (2013) index, the mineral soil is also undisturbed. However, the thin sections showed that the soil was covered by a 1 cm-thick black organic layer (**Figures 3B**, **6C**). Under the microscope, we observed a loss of the structure while charred aggregates are fractured and partially or completed disaggregated, in the first centimeter of the mineral soil (**Figures 8E–G**). This observation might explain the increase of the soil bulk density (**Figure 4E**). Greene et al. (1990) showed that frequent prescribed burnings can break down aggregates surface. These results have been also confirmed by Arocena and Opio. (2003), who demonstrated the presence of cracks on soil aggregates through SEM-EDS observations. They interpreted these cracks as a thermal shock related to the temperature growth. Other studies, however, have not found immediate effects on mineral topsoil aggregates after the low-intensity sand rapid prescribed burn in shrubland in Central Pyrenees, Spain (Badía et al., 2020). In agreement to our findings, DeBano et al. (1998) and Mataix-Solera et al. (2011) mentioned that even low-intensity fires can affect soil structure and aggregate stability due to the combustion and/or transformation of the organic matter compounds that binds soil aggregates; this generates a loss of soil structure, a porosity reduction, and a bulk density increase (DeBano, 1981). Moreover, Badía et al. (2017) revealed that controlled burning only affects the physical and chemical properties of the first centimeter of dry soils. Another explanation for the loss of aggregate stability after prescribed burnings, without significant loss of organic matter, was exposed by Albalasmeh et al. (2013) and Chief et al. (2012), who found that steam pressure can break down aggregates at temperatures of 150o̲C and 175°C in soils with high initial moisture. Nevertheless, it is important to mention when the fire temperature rises, an increment in the ped stability is noted because of the low contents of clay and Fe oxides in the soils (Thomaz, 2021). With these findings, we can complement the SBS Vega et al. (2013) index adding a slight perturbation of the first centimeter of the mineral soil at level 2.

For the SBS level 4, the micromorphological and morphometric descriptions agree with the general properties proposed by Vega et al. (2013), where the organic layer was completely consumed and transformed into a thick ash layer. The fire also affected the uppermost mineral soil structure and the surface fine roots and seeds (**Figures 3C**, **6D**, **9A–D**). The ash layer resulted from the oxidation of vegetal materials beyond charring during the fire, where the calcium oxalate was transformed into calcium carbonate by loss of TOC (Canti and Brochier, 2017). Calcite crystals were identified in the thin section of the SBS level 4 (**Figure 9D**). Reddened aggregates were also detected under the microscope. However, they were not recognized in the SBS index of Vega et al. (2013) because it is hard to identify them at a macroscopic scale. Our micro-scale study demonstrated that these mineralogical transformations expressed by reddened aggregates could start even before SBS level 5.

Regarding the Vega et al. (2013) index for the SBS 5, the micro-scale analyses showed that the soil organic layer was absent because it was completely consumed (**Figure 6E**). This characteristic can result from a higher temperature during the prescribed burn in this specific zone. The forest floor may have reached a temperature of around 460°C, in which the organic carbon consumption is completed (Giovannini et al., 1988), or the organic layer was absent in this area before de fire. In our study, the micromorphological and morphometric results showed that the aggregates and soil structure were slightly affected in the first 2 cm of the mineral soil (**Figure 6E**). Additionally, soil organic matter was charred and some aggregates were reddened in the surface soil (**Figure 6E**). The incipient soil reddening is a consequence of Fe oxides transformation due to heating; however, a continuous reddened layer covered by ash is expected only for severe fires and for SBS level 5 (Wells et al., 1979; Ulery and Graham, 1993; Ketterings and Bigham, 2000; Vega et al., 2013).

CONCLUSION

The physical-chemical characteristics of burned samples showed clear differences with unburned samples. After the prescribed burn, soils exhibited a darker color with a decrease in redness and yellowness, a slight reduction in soil moisture, an increase in bulk density, higher pH values, an inconclusive trend for TOC content, insignificant variation of TN, and an increase in TP content. Besides these changes in the physical and chemical properties, already well documented in other studies, micromorphological and morphometrical characteristics are useful techniques for revealing how deep the effect of prescribed burning reached into the soil and for complementing the diagnosis of post-fire soil physical alterations that SBS assessment cannot identify due to the evaluation scale.

Some of our results agree with to the SBS Vega et al. (2013) classification, particularly for the organic layer. However, the microscale analyses of mineral soil revealed that even from the SBS 2, the first centimeter of soil structure was affected by a prescribed burn. This might be attributed to changes in soil organic matter content and composition and to the thermal shock and steam pressure suffered by the aggregates due to temperature increase. Despite these features, some living seeds and fine roots could be distinguished, indicating a potential for plant and microorganism regeneration. Additionally, the Fe oxides transformations that reddened soil aggregates can start even before the soil gets the SBS 5. For high SBS levels, the micro-scale analyses revealed that the prescribed burn affected the first 2 cm of soil depth, disrupting soil structure, porous clogging with ash, and the reddening of soil aggregates. These effects resulted in higher bulk density and changes in surface soil porosity.

Land managers must consider these soil alterations at low and high SBS after applying prescribed burns to avoid post-fire erosion of the first-centimeters mineral soil. The actions to protect the soil must be applied following the prescribed burn and before the first rain, especially in the bare soil areas where aggregates disrupted by the fire can be susceptible to dispersion and erosion by the rain splash and runoff. Therefore, the SBS assessment can include an additional section about the microscopic characteristics of burn soils. Likewise, a micro-scale characterization of different SBS levels for wildfires could be interesting because it can reveal soil perturbation that we cannot see directly on the field.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

BC-V conceived the general idea and led the project. SQ-G and BC-V designed the sampling and took the samples. BC-V and MR-R performed the soil physicochemical analysis. MR-R prepare the thin sections. SM-R carried out the morphometric and micromorphological examination. SM-R and BC-V performed the statistical analysis. ES-R supervised the project. AM, BC-V, and ES-R contributed to the analysis of the results and the discussion section. SM-R wrote the manuscript with support of BC-V, ES-R, AM, and SQ-G. All authors contributed to the article and approved the submitted version.

CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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The Recovery of Mediterranean Soils After Post-Fire Management: The Role of Biocrusts and Soil Microbial Communities

Minerva García-Carmona, Fuensanta García-Orenes, Victoria Arcenegui and Jorge Mataix-Solera*

GETECMA- Soil Science and Environmental Technologies Group, Department of Agrochemistry and Environment, Miguel Hernández University, Elche, Spain

Although Mediterranean ecosystems are adapted to fire disturbances, soils are prone to degradation. Therefore, post-fire forest management is a critical step for ecosystem recovery: it can either reduce soil degradation or add a new disturbance. Post-fire management in Mediterranean burnt forests includes interventions with contrasting approaches, including the management of burnt trees, soil protection, or practices devoted to ecosystem restoration via the improvement of components or processes in the affected ecosystem. The consequences of forest management on soils are complex, thereby, in the context of the intensification of fire events and climate change, understanding the response of key soil components in managed ecosystems is critical for prioritizing soil conservation. One interesting component in the early post-fire stages is moss biocrust. The rapid colonization of biocrust-forming mosses in early successional stages post-disturbance stabilizes soils in their most vulnerable period. However, it is completely unknown further implications as active agents in the recovery and resilience of soils, in the transient stage before vascular vegetation regrowth. In combination with the biocrust, the response of soil microbial communities to forest management is crucial for evaluating the soil recovery progress, given their active role in fundamental ecosystem functions. The additive consequences of fires and forest management on biocrust emergence or microbial composition and functionality are usually neglected in the investigation of post-fire systems, although of major relevance to support strategies to preserve soils against functionality loss.

Keywords: biological soil crust, burnt wood, Mediterranean, microbial community, moss

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****Correspondence:***
Minerva García-Carmona
minerva.garciac@umh.es

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INTRODUCTION

Fire is an ecological and evolutionary force in most terrestrial ecosystems on Earth (Pausas and Bond, 2019). As a recurrent process, fire regimens have direct ecological effects on species traits, species interactions and community composition, carbon and nutrient cycling, and ecosystem functions (McLauchlan et al., 2020). The Mediterranean-type climate is an example of a fire-adapted ecosystem thanks to the climatic seasonality, precipitation in the mild winters that enable plant growth which became highly flammable during the dry and hot summers. Thus, several species have developed adaptive strategies to resist, promote, or recover from recurrent fires (Keeley et al., 2011). Land-use change, fire suppression policies, and climate predictions that point to intensification in drought frequency and warmer conditions, have the potential to magnify the wildfire impacts (Pausas, 2004; IPCC, 2018), threatening the resilience of ecosystems (Flannigan et al., 2009).

Intensification in frequency and severity of fire events is expected to result in detrimental effects on soils (Guénon et al., 2013; Pellegrini et al., 2018), through the magnification of the hydrological response, destruction of soil structure, modification of soil organic matter and soil biochemistry, and loss in soil biodiversity (Neary et al., 1999; DeBano, 2000; González-Pérez et al., 2004; Certini et al., 2021; Doerr et al., 2022). Fire effects on soils are coupled with changes aboveground. Rapid vegetation recovery is critical to guarantee soil protection against erosive forces, the main threat to Mediterranean soils after fires (Cerdà and Robichaud, 2009). Nonetheless, above-belowground interactions may suffer alterations under changing fire regimes, e.g., changing soil nutrient pools over time (Caon et al., 2014; Pellegrini et al., 2018; Dove et al., 2020). Understanding how vegetation regenerates is essential for mitigating the escalating fire effects in vulnerable ecosystems (Fernández-García et al., 2019).

Aboveground and belowground soil components are strongly linked; therefore, fire may modify the microbial communities by altering plant-induced changes in the soil environment (Hart et al., 2005; Knelman et al., 2015; Dove et al., 2021). Given the critical ecosystem processes soil microorganisms are involved in, including nutrient cycling, physical stability, carbon sequestration, or support for plant growth (Fultz et al., 2016), modifications in fire regimen could profoundly alter the microbial communities and lead to a great impact on soil functioning (Ferrenberg et al., 2013; Dove and Hart, 2017; Whitman et al., 2019; Sáenz De Miera et al., 2020). Considering the global change projections and new wildfire scenarios, additional work is necessary to better understand the resilience of fire-affected ecosystems exposed to additional disturbances such as human intervention through forest management, of major relevance to support strategies that preserve soils against functionality loss (Pereira et al., 2018; Tomao et al., 2020; Lucas-Borja et al., 2021; Averill et al., 2022).

Biocrust-Forming Mosses: Their Role in Soil Recovery

Biological soil crust, hereafter "biocrust," is a diverse community of photoautotrophic (e.g., cyanobacteria, algae, lichens, bryophytes) and heterotrophic (e.g., bacteria, fungi, archaea) organisms, living within the first centimeters of the soil surface. Soil particles are aggregated through their presence and activity, and the resultant living crust covers the surface of the ground as a coherent layer (Weber et al., 2022). Around 12% of Earth's terrestrial surface is covered by biocrust (Rodriguez-Caballero et al., 2018), dominating the plant interspace in many drylands thanks to specific adaptations to survive in unfavorable and often extreme environments (Belnap and Büdel, 2016). While mosses are typically found creating carpets in habitats where water is not a limiting factor (Weber et al., 2022), biocrust-forming mosses developed in drylands are adapted to cope with high insolation, low rainfall, and drought. In the semiarid Mediterranean region, biocrusts are dominated by lichens and bryophytes due to their physiological and morphological characteristics (Maestre et al., 2021; Ladrón De Guevara and Maestre, 2022).

Biocrust-forming mosses are ecosystem engineers: modulate soil properties, alter microbial communities, and intervene in key ecosystem processes such as water infiltration, nutrient cycling, or carbon sequestration (Ferrenberg et al., 2017; Ladrón De Guevara and Maestre, 2022). Above all, biocrusts are recognized as major soil stabilizers in drylands (Belnap and Büdel, 2016). The morphology of mosses (i.e., fine rhizoids and protonema mats) allows strong cohesion of soil particles providing high stability (Seppelt et al., 2016). This high resistance enables effective mitigation of soil erosion, directly, by creating a physical barrier and roughening the surface, and indirectly, by affecting soil properties mainly by increasing the organic matter content (Gao et al., 2020; Zhang et al., 2022). The biocrust effect on soil stability is subordinated to its development stage, which is influenced as well by the extent, intensity, and time since disturbances (Belnap and Büdel, 2016). Due to their implication in distributing surface flows, infiltration and runoff, and regulating soil moisture, biocrust have a major role in controlling local hydrological cycles in drylands (Eldridge et al., 2020). Biocrusts represent islands of fertility for plants and microorganisms through the concentration of essential elements in soils (Ferrenberg et al., 2018), promoting essential biochemical processes. Moss biocrust contributes directly to soil fertility by fixing carbon and nitrogen, increasing the organic matter in soils beneath the crust (Cheng et al., 2021), and contribute indirectly by acting as dust particle trappers (Reynolds et al., 2001). The nutrient status in the soil biocrust facilitates the development of microbial communities, playing fundamental roles in ecosystem multifunctionality and acting as hotspots of soil biodiversity (Delgado-Baquerizo et al., 2016; Maier et al., 2016; Zhang et al., 2022).

Natural recovery rates of biocrust after disturbances are known to be slow, especially after wildfire events, which can involve long-lasting consequences for biocrust community structure and diversity recovery (Johansen, 2001; Root et al., 2017). However, there is not a general consensus on how biocrusts respond to fire disturbances since it highly depends on the biocrust type, the ecosystem, and variables related to the fire, such as severity, frequency, and disturbance history (Zaady et al., 2016; Palmer et al., 2020). Under favorable climate conditions and soil stability, the initial cyanobacteria-dominated succession stages may be omitted to start with biocrust-forming mosses (Weber et al., 2016). This succession pattern is highly observed in fire-affected semiarid or temperate ecosystems (Bowker et al., 2004; Grover et al., 2020; Weber et al., 2022). Fire disturbances provide an opportunity for biocrust to develop, and demine temporarily, in areas that are commonly covered with vascular plants and plant litter. Eventually, biocrust will be diminished in abundance or replaced by vascular plant vegetation with natural recovery succession; however, persistent

FIGURE 1 | Patch of moss biocrust emerged after a wildfire stabilizing soils surrounded by bare soils exhibiting erosion symptoms.

stressful conditions for vascular plants, e.g., soil compaction provided by heavy machinery in post-fire management, might create conditions that support long-term persistence of biocrust in those environments (Gall et al., 2022a).

Bryophytes are recurrent elements in the post-fire vegetation succession in Mediterranean forests (During, 1979; De las Heras et al., 1994; Esposito et al., 1999; Castoldi et al., 2013; Stinca et al., 2020). After wildfires, ruderal mosses rapidly colonize bare soils in a transient succession stage before vascular plant colonization. This is especially documented after high-intensity fires, in which ecosystems are largely dominated by ruderal mosses during the first 2–3 years after the disturbance (De las Heras et al., 1994; Esposito et al., 1999), revealing the high resilience of mosses to the post-fire environment (Reed et al., 2016; Condon and Pyke, 2018). The reason for their quick response may be related to the wide dispersal of spores, the possible regeneration from dormant propagules in sub-surface soil banks, and rapid protonema and gametophyte growth facilitated by their ability to develop on unstable substrates like charred surfaces and ashes (Esposito et al., 1999; Smith et al., 2014). The colonization stage is characterized by the dominant presence of a few pioneers colonizing species such as *Funaria hygrometrica*, a specie that shows a very fast protonema development able to survive the desiccation that typically occurs in recently burned soils (During, 1979; De las Heras et al., 1994; Esposito et al., 1999).

Biocrust-forming mosses have received attention recently due to their efficiency in stabilizing the soil surface and controlling soil water erosion after wildfires (**Figure 1**) (Silva et al., 2019; Gall et al., 2022b), which makes them a promising technique to rehabilitate fire-affected soils (Grover et al., 2020; Muñoz-Rojas et al., 2021). Despite the growing body of knowledge demonstrating their role as ecosystem engineers, pioneer moss biocrusts are often neglected in studies assessing their effect on fire-affected ecosystems. The burgeoning biocrust is a valuable component in post-fire environments beyond soil stabilization. The early colonization of mosses mitigates the harsh conditions on the surface (e.g., desiccation, high temperature, and solar radiation), thereby facilitating microbial growth in biomass and diversity, thus accelerating key biochemical processes from nutrient cycling affected in the wildfire. Therefore, biocrust might play a critical role in the resilience of soil microbial communities affected by wildfires, an influence that persists and accentuates over time with biocrust development (García-Carmona et al., 2020; 2022). However, biocrust are highly vulnerable to physical disturbances and climate change (Rodriguez-Caballero et al., 2018; García-Carmona et al., 2020), thus more studies are needed to understand how biocrust-forming mosses will respond to the intensification of fire events in a scenario of climate change.

The Soil Microbial Response to Fire Disturbances

Soil microbial communities play an essential role in driving a wide variety of ecosystem processes and functions, including nutrient cycling, primary production, litter decomposition, climate regulation, and soil formation (Bardgett and Van Der Putten, 2014; Delgado-Baquerizo et al., 2020). Therefore, microbial diversity act as a precise indicator of soil process alterations and ecosystem recovery after wildfires (Muñoz-Rojas and Bárcenas-Moreno, 2019). Fire disturbances induce complex effects on ecosystem functioning, in which alterations can last months to years depending on the interactive plant and microbial communities' responses to fires (Kardol and Wardle, 2010; Pérez-Valera et al., 2019). In Mediterranean soils, microbial communities generally show high resilience to wildfires, and ecosystem functioning related to microbial performance recovers relatively quickly (Ferrenberg et al., 2013; Pérez-Valera et al., 2020). However, increasing disturbance pressure on soil microorganisms may hamper the recovery ability of the ecosystem (Villnäs et al., 2013; Mendes et al., 2015). Microbial functionality is linked to the soil post-fire status (Nelson et al., 2022) since the environment strongly filters the abundance and composition of microbial communities (e.g., pH, soil nutrients, climatic variables) (Bahram et al., 2018). The study of microbial communities (i.e., population abundance and taxonomic and functional diversity) and their relationship with soil properties (i.e., indicators of soil health, nutrient cycling, or soil carbon stock) becomes strategic to evaluate the recovery process after fires, monitor the soil biodiversity conservation, and to predict the ecosystem's resilience to further disturbances (Adkins et al., 2020; Dove et al., 2020; Guerra et al., 2021).

Fire profoundly alters the assembly of microbial communities, which are strongly affected by fire severity (Whitman et al., 2019), and with long-lasting consequences on the community complexity (Treseder et al., 2004; Holden et al., 2016; Cutler et al., 2017; Su et al., 2022). As warned in several meta-analyses, if microbial communities are not resilient to fire within a decade, the predicted increase in fire frequency can hinder the recovery of microbial communities and the important ecosystem processes they regulate (Dooley and Treseder, 2012; Pressler et al., 2019). After wildfires, the taxonomic structure is dominated by some groups as a response to their ecological strategy (Prendergast-Miller et al., 2017; Pérez-Valera et al., 2018). For instance, the

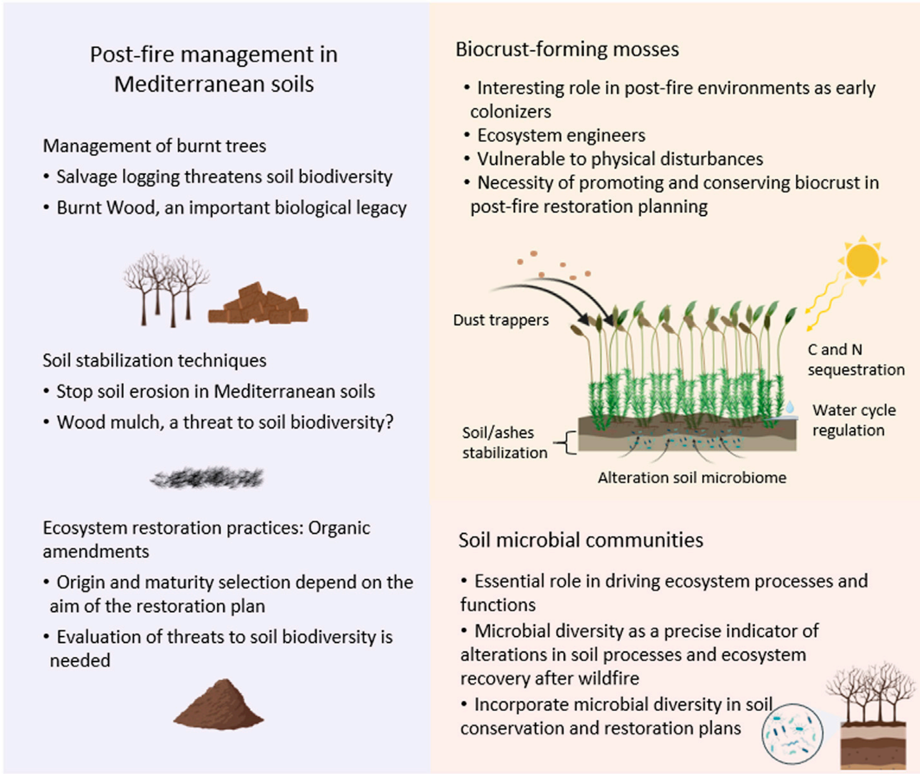

FIGURE 2 | Insights from the Discussion: An overview of post-fire management in Mediterranean soils and key roles of biocrust-forming mosses and soil microbial communities.

identification of responsive taxa to fire disturbances provides valuable information in order to predict the recovery of microbial functionality. In this sense, pyrophilous fungi are interesting indicator taxa after fires, which fruit abundantly due to heat stimulation, lack of competition, and tolerance to post-fire conditions (Reazin et al., 2016; Bruns et al., 2020; Raudabaugh et al., 2020; Fox et al., 2022). Pyrophilous fungi have been recently recognized for aggregating particles and increasing moisture in soils (Filialuna and Cripps, 2021), accelerating the ecosystem recovery process following a fire disturbance.

In drylands, biocrust promote soil microbial diversity (Delgado-Baquerizo et al., 2016; Zhang et al., 2022), being the macro component, either cyanobacteria, lichen, or bryophyte, the main driver of microbiome assembly (Maier et al., 2016; 2018). Moss biocrust is known to harbor a high diversity of bacteria and fungi beneath it, but communities are highly sensitive to disturbances (Xiao and Veste, 2017; Bao et al., 2019; Cheng et al., 2021). Whether new wildfire scenarios coupled with climatic projections may shift the structure of biocrust, switching to early-successional cyanobacteria, is relevant for microbial biodiversity conservation. Those shifts may strongly impact the functioning of recently fire-affected ecosystems through the profound alteration of soil microbial communities and biochemical processes (Maestre et al., 2015; Delgado-Baquerizo et al., 2018; Tucker et al., 2020; Tian et al., 2022). Those questions remain unanswered, but new approaches in the study of soil microbiome are expected to reveal valuable information in this regard.

Post-Fire Management in Mediterranean Forests: Restoring or Adding a New Disturbance

The management of fire-affected areas represents a crucial step for the fate of soils after fires. While Mediterranean ecosystems are resilient to fire events, soils are prone to degradation. Therefore, the management will determine the ecosystem's capacity to recover from the fire disturbance, combined with factors such as the fire history, ash properties, topography, post-fire weather, and vegetation recuperation (Pereira et al., 2018). Post-fire management planning in a Mediterranean burnt forest includes interventions with contrasting approaches, including the management of burnt trees, soil protection, or practices devoted to ecosystem restoration. The consequences of forest management in soils, especially in soil biology, are particularly complex and conditioned by multiple factors, often overlooked in the decision-making process (**Figure 2**).

Salvage logging is the most common post-fire management strategy in Mediterranean coniferous forests. Intensive salvage logging trigger soil degradation processes: soil compaction, delay of vegetation recovery (Wagenbrenner et al., 2016; García-Orenes et al., 2017), disturbance of nutrient cycling (Pereg et al., 2018), alteration in carbon fluxes (Serrano-Ortiz et al., 2011; Hartmann et al., 2014), and disruption in soil biodiversity directly or indirectly, e.g., disturbing the deadwood-dependent species (Thorn et al., 2020), reducing the cover of biocrust-forming mosses (García-Carmona et al., 2020), or altering the soil microbial communities (García-Carmona et al., 2021a). Soils can suffer persistent alterations, ultimately reducing forest productivity and ecosystem functionality (Hartmann et al., 2014; Chen et al., 2015). Nevertheless, the effects on soils are highly dependent on the context, the site characteristics, the soil erodibility, and the way to perform the management (Fernández and Vega, 2016; García-Orenes et al., 2017; Francos et al., 2018). On the other hand, burnt wood is a biological legacy of key relevance in burned forests (Thorn et al., 2018). The burnt wood act as a barrier for sediments against water erosion, constitutes a stock of nutrients that slowly fertilize soil through decomposition, and ameliorates the stress conditions by increasing soil moisture, enabling vegetation and microbial development and sustaining biodiversity and ecosystem services (Baldrian, 2017; Thorn et al., 2018; 2020; García-Carmona et al., 2021a; Juan-Ovejero et al., 2021). However, timber activities in Mediterranean forests are important from a social perspective, being non-interventionism is highly controversial (Castro, 2021). The creation of land diversification via patches of different wood extraction intensities could increase the forest's resilience to future disturbances (FAO et al., 2020).

Among the emergency stabilization techniques to face the risk of soil erosion, mulching is considered the most cost-effective intervention after wildfires (Robichaud et al., 2013; Girona-García et al., 2021). Straw mulches, the most commonly applied materials, are highly effective (Lucas-Borja et al., 2019), but their application presents some drawbacks like the introduction of non-native species and low wind resistance (Beyers, 2004; Kruse et al., 2004). In contrast, wood-based mulches exhibit great resistance to wind displacement and long longevity due to their decay resistance (Bautista et al., 2009; Jonas et al., 2019). However, vegetation regrowth can be hindered under a thick layer of mulch (Bautista et al., 2009), and thus endangered its crucial role in soil protection and recovery. While preventing soil loss in Mediterranean forests must be the main goal in post-fire planning, more research is needed regarding the potential threat to soil biodiversity conservation of wood-mulching if incorrectly performed. Multiple recommendations or guidelines exist in this aspect (Vallejo et al., 2012; Robichaud et al., 2013; Pereira et al., 2018; Castro, 2021): interventions should be limited to very specific situations, i.e., high risk of erosion, slow vascular plant recovery rate, risk downslope, *etc.* For instance, wood residues generated in the framework of logging operations are often applied where intensive logging operations may have created the necessity of the mulch application after triggering erosion processes (Castro, 2021). Wood-based mulch in soils is expected to produce positive effects in soils related to microclimatic improvement and nutrient supply, although the biological soil response and functionality recovery are still rather unexplored.

Restoration practices act on components or processes in the affected ecosystem in order to recover its functionality, for example, via the application of organic amendments (Hueso-González et al., 2018; Muñoz-Rojas, 2018). After the strong consumption of organic carbon in high-severity fires, the additional source of organic matter induces a cascade of effects in multiple components of the perturbed ecosystem (Heneghan et al., 2008; Costantini et al., 2016). The amendment selection, in relation to the decomposition rates of the organic materials, depends on the goals of the soil restoration plan in terms of the durability of effects on the soil response (Tejada et al., 2009; González-Ubierna et al., 2012; Larney and Angers, 2012; García-Carmona et al., 2021b). Studying the application effects on soil microbial diversity is highly necessary to identify possible threats to soil biodiversity, related to the introduction of new taxa, in order to correctly address biodiversity protection plans.

CONCLUSION

In order to support management practices that boost soil biodiversity and preserve ecosystem functionality, threats must be identified to formulate strategies that prioritize soil conservation (Guerra et al., 2021; Averill et al., 2022). In this sense, biocrust and soil microbial communities are highly vulnerable to post-fire physical disturbances, thus land diversification through different management intensities could be strategic for increasing the ecosystem's resilience. In the context of intensification of fire events and climate change scenarios, the investigation of two key components for soil recovery, i.e., microbial diversity and biocrust-forming mosses, might be key to guiding forest strategies toward accelerating recovery and resilience of semi-arid ecosystems prone to degradation.

AUTHOR CONTRIBUTIONS

MG-C conduct the study and wrote the manuscript, JM-S and FG-O lead the funding acquisition. All authors contributed to the article and approved the submitted version.

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CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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