Catena 263 (2026) 109728

Contents lists available at ScienceDirect

Catena

journal homepage: www.elsevier.com/locate/catena

Prescribed burning of scrubland only affects very sensitive topsoil properties: A decadal-scale study in encroached grasslands of the southern Pyrenees

Andoni Alfaro-Leranoz [1], David Badía-Villas [*], Clara Martí-Dalmau

GEOFOREST, Departamento de Ciencias Agrarias y del Medio Natural, Escuela Politécnica Superior de Huesca, Universidad de Zaragoza, Ctra. Cuarte s/n, 22071 Huesca, Spain

ARTICLE INFO

Dataset link: https://zaguan.unizar.es/?ln=es (Original data)

Keywords:
Prescribed fire
Soil organic carbon
Soil microbial activity
Ash
Shrub encroachment
Soil cm-scale

ABSTRACT

The decrease in grazing activity during the last century is causing an encroachment of the subclimatic grasslands in most European mountains, accelerated by climate change. To control this expansion, prescribed burnings (PBs) are used. However, their use can affect sensitive soil properties, particularly soil microbiology. Although many studies have already reported the immediate effects of shrub PBs on soils, very few of them have studied the long-term post-fire recovery. Therefore, the purpose of this study is to evaluate the long-term effects of the PBs on the topsoil biochemical properties over time, at a cm-scale (0–1, 1–2, 2–3 cm). The soil sampling was carried out in Asín de Broto (Central Pyrenees, Spain) and four treatments, depending on the time-since-fire, were selected: unburned (UB), immediately post-fire (B0), 5 (B5) and 9 (B9) years after burning. The results showed an immediate increase (B0) in chemical properties because of the ash incorporation, highlighting values 3.35, 1.64 and 1.55 times higher than in the UB samples for the electrical conductivity, the dissolved organic C and the total N, respectively. However, these increases were no longer noticeable after the mid-term (B5, B9). Regarding the biological properties, β-D-glucosidase (GLU) activity suffered a significant decrease over time (67 and 72 % less in B5 and B9, respectively), without affecting either the microbial biomass C or its activity. This study has proven that GLU is a useful indicator that can be used to assess the effects of low-severity fires.

1. Introduction

Throughout the human history, forest have been cut to provision human settlements with wood and burned to extend their cultivable lands and their grasslands for livestock production (García-Ruiz et al., 2020; Gómez, 2008). However, during the last half of the 20th Century, approximately 20 % of croplands and grasslands were abandoned and colonized by shrub communities (Gartzia et al., 2014), because of the loss of large mammal herbivores, that have a very important role in the evolution of vegetation on temperate areas (Amsten et al., 2024). In the Central Pyrenees, secondary grass communities can be found below the timberline, in the subalpine and montane stages (Alfaro-Leranoz et al., 2023). They are important reservoirs of C and N, even more than forests and scrublands (Nadal-Romero et al., 2018), and they are rich in species, in contrast to shrublands (Vélez, 2012). However, during the past decades, around 480,000 ha of grasslands have been encroached by woody species in the Southern Pyrenees, with a range of 6000 ha/year in the last decades (Gelabert et al., 2021).

Currently, there is a concern about the increase of woody fuel accumulation in the montane and subalpine stages, which is raising fire risk in ecosystems that, until now, have not been considered fire-prone (Pausas and Keeley, 2009; Turco et al., 2016). That situation, caused by the abandonment of agricultural and livestock use of the land, is expected to aggravate with the prognosticated increase of temperatures and seasonality of precipitations in the Pyrenees (OPCC-CTP, 2018). Moreover, the extremely suppressive fire-fighting policies applied by the public administrations for several years have also led to a rise of woody fuel accumulation and continuity (Turco et al., 2016). In the Central Pyrenees, one of the main colonizing shrub species is *Echinospartum horridum* (Vahl) Rothm., a leguminous spiny scrub that creates

---

[*] Corresponding author.
E-mail address: badia@unizar.es (D. Badía-Villas).
[1] Current address: Centro de Innovación en Bioeconomía Rural de Teruel, Centro de Investigación y Tecnología Agroalimentaria de Aragón (CITA), Calle Corinto 3, 44195 Teruel, Spain.

https://doi.org/10.1016/j.catena.2025.109728
Received 17 September 2025; Received in revised form 3 December 2025; Accepted 6 December 2025

practically monospecific, dense and highly flammable communities (Komac et al., 2013; Martín-Ramos et al., 2020). Fire is known to have a key role in the regulation of several ecosystems (Yang et al., 2020), particularly in the Mediterranean basin, where it has been leading the vegetation evolution and adaptation (Aponte et al., 2022). Nevertheless, fire has also induced land degradation around the world, typically in temperate areas with few annual precipitations (Mataix-Solera et al., 2011), which are quite vulnerable to climate change (Fletcher and Zielhofer, 2013).

Prescribed burnings (PBs) are low-intensity fires that are performed to achieve specific management purposes (Agbeshie et al., 2022), just as to reduce fuel quantity and its continuity, in order to minimize the fire risk in fire-prone ecosystems (Fernández-Guisuraga and Fernandes, 2024; Fontúrbel et al., 2021; Rabin et al., 2022). Additionally, they are employed to regenerate certain species or communities, like grasslands, and to train the wildfire extinction crews. These PBs are performed under specific environmental conditions, trying to ensure their low severity: less than 10 km h^{-1} of windspeed, atmospheric temperature below 15 °C, air relative humidity of 35–70 % and high soil moisture (Girona-García et al., 2019). Despite being performed in optimal conditions to minimize their effects on the edaphic environment, they can still affect some of the most temperature-sensitive soil properties, like the organic matter fractions and the biological properties (Alcañiz et al., 2018; Emran et al., 2025; Santín and Doerr, 2016). Also, PBs can have an impact on the ecological processes of the ecosystem and they can influence animal and plant biodiversity (Agbeshie et al., 2022). However, the impact of small PBs carried out in winter, with wet soils, tends to be much smaller than that of large summer wildfires and, in return, can generate a mosaic landscape that reduces their spread (Fontúrbel et al., 2024). The direct effect of PBs on soil properties is mainly driven by the heat transfer into the soil (Agbeshie et al., 2022), which is greatly influenced by soil moisture (Badía et al., 2017). Nevertheless, the post fire response can be highly variable and difficult to predict, due to the specific fire-related factors, like ash quantity and vegetation mortality, and the complex interaction between the post-fire physic-chemical and biological processes (Soria et al., 2023). In any case, field studies can provide data that is much closer to reality than controlled burns in the laboratory (Doerr et al., 2025).

Previous studies showed that soil responses to PBs are variable. However, the effects were generally negligible or positive, such as increases in ions and soluble nutrients, pH and black carbon coming from ash, which significantly alters soil chemistry (Agbeshie et al., 2022; Girona-García et al., 2018c). However, sometimes the effects can be negative as well. Changes in soil organic matter quality and quantity have been widely reported because of its importance to the soil quality (Agbeshie et al., 2022; Alcañiz et al., 2018). Soil organic carbon (SOC) may suffer changes after PBs, mainly depending on the fire intensity and its duration, the available biomass and its moisture content (Reyes et al., 2015). Generally, low-intensity fires, such as prescribed burns, tend to increase the SOC content or leave it unchanged, whereas the high-intensity ones tend to reduce it (Caon et al., 2014; Mora et al., 2021). Commonly, the loss of SOC happens because of its combustion at 200–500 °C (Badía-Villas et al., 2014; Santín and Doerr, 2016), but it can also happen because of the increase of microbial degradation, volatilization and labilization (Rodríguez-Cardona et al., 2020). On the other hand, the increase of SOC in PBs was often related with an increase in black C, due to the partial combustion of organic matter and the ash incorporation (Agbeshie et al., 2022; Caon et al., 2014; Gao et al., 2018). Nitrogen (TN) has a lower tolerance to temperatures than SOC, which makes it more prone to volatilization under temperatures above 200 °C (Caon et al., 2014; Mora et al., 2021; Santín and Doerr, 2016). Moreover, it can also be lost by erosion and leaching (Agbeshie et al., 2022). However, the ash produced in low-intensity fires can entail an important input of TN, increasing its content (Alcañiz et al., 2018; Wan et al., 2021). The ash is also rich in soluble ions and salts, which commonly leads to increases in pH and electrical conductivity (EC) (Agbeshie et al., 2022; Alcañiz et al., 2018; Muñoz-Rojas et al., 2016).

The changes in SOM quality and quantity, along with changes in N availability and post-fire plant cover evolution, can indirectly lead to changes in the soil biological properties (Agbeshie et al., 2022; Picariello et al., 2021). These properties are highly sensitive to environmental perturbations and to heat transfer into the soil, starting at 50 °C (Alfaro-Leranoz et al., 2024; Pereira et al., 2023; Santín and Doerr, 2016). Their sensitivity to perturbations makes them useful bioindicators of soil health (Fontúrbel et al., 2024; Vega et al., 2013). Among these, β-D-glucosidase activity (GLU) is particularly responsive (Alfaro-Leranoz et al., 2023; Alfaro-Leranoz et al., 2024; Lombao et al., 2021; Pereira et al., 2023). It can decrease directly because of enzyme denaturation by heat (Alfaro-Leranoz et al., 2024; Knicker, 2007), or indirectly through changes in the labile fraction of SOM (Alfaro-Leranoz et al., 2023; Girona-García et al., 2019; Wang et al., 2013), or changes on the soil cover (López-Poma and Bautista, 2014). The activity of the microbial community is often measured as the CO_2 efflux, also known as basal soil respiration (bSR). This parameter was often related with the quality and quantity of SOM (Alfaro-Leranoz et al., 2023; Castro-Díez et al., 2012) and with the microbial biomass carbon (MBC). This parameter can suffer changes depending on the intensity of the fire and the water content (Girona-García et al., 2019), and also on the availability of nutrients and ash incorporation (Agbeshie et al., 2022).

Fire also tends to make changes in soil water repellency (SWR). The effects on this property is very concerning management-wise because it directly affects water infiltration, runoff and, therefore, soil erosion (Weninger et al., 2019). SWR can be created or destroyed by fire, depending on the pre-fire SWR, the fire intensity and the organic matter characteristics (Agbeshie et al., 2022; Girona-García et al., 2018b). Normally, low-to-moderate-intensity fires tend to increase it, by the formation of hydrophobic compounds during the combustion of organic matter at low temperatures (Mataix-Solera et al., 2011), that can either stay at the soil surface together with ash (Jordán et al., 2016), or can migrate downwards deeper and condensate on cooler layers (Agbeshie et al., 2022; Alfaro-Leranoz et al., 2023; De Bano, 1981; Girona-García et al., 2018b). With the time, fire-induced SWR tends to disappear (Granged et al., 2011).

Given the variability of results shown by the aforementioned studies, it is necessary to investigate the consequences of this landscape management technique under every particular environmental conditions. Furthermore, most of the previous studies performed in the Pyrenean environment have studied the immediate to short-term effects of PBs on soil biochemical properties (Armas-Herrera et al., 2016; Girona-García et al., 2019; Girona-García et al., 2018a; San Emeterio et al., 2016; San Emeterio et al., 2013). Only Armas-Herrera et al. (2018) and Alfaro-Leranoz et al. (2023) extended the observation period, assessing the PBs effects up to 5 and 10 years, respectively, at the subalpine stage. Even though, these longer-term studies reported comparable results, likely reflecting the consistent environmental conditions under which they were conducted. Therefore, it is noticeable that there is a need for more research that studies the long-term post-fire soil recovery after PBs performed in different wet mountainous areas, where the fire is not naturally the responsible for ecological changes, unlike in the mediterranean ecosystems.

In the present study, we extended this line of research to an area situated at the transition between the montane and the subalpine stages, and we measured the temporal evolution of the effects of PBs using a chronosequence approach. The purpose of this study is, therefore, to evaluate the immediately post-fire, the medium- (5 years post-fire) and the long-term (9 years post-fire) effects of the PBs on the topsoil, in order to determine the long-term post-fire response under particular environmental conditions. Specifically, the changes at a centimetric scale (0–3 cm) were evaluated in sensitive soil biochemical properties, that can serve as indicators of the PB effects, such as soil reaction (pH), soil electrical conductivity (EC), total soil organic carbon (SOC) and nitrogen (TN), dissolved organic carbon (DOC), basal soil respiration (bSR), microbial biomass carbon (MBC) and ß-D-glucosidase activity (GLU), along with other related soil properties. Based on previous research, prescribed burning is expected to exert minimal immediate impacts on the studied soil properties. Nevertheless, alterations in the soil organic matter quality and quantity over short- to long-term may induce secondary negative effects, highlighting the importance of monitoring soil biochemical responses across temporal scales.

2. Material and methods

2.1. Study area

A study area located within the municipality of Asín de Broto, Central Southern Pyrenees, Huesca (Aragón), NE Spain (Fig. 1) was chosen for this study. The area, located around 1650 m above the sea level, is placed around the border between the montane and the subalpine bioclimatic stages, with 1120 mm of mean annual precipitation and 8.8 °C mean annual temperature (Girona-García et al., 2019). The predominant vegetation is *Echinospartum horridum* (Vahl) Rothm, Family Fabaceae, a plant community with very few species (n = 6 ± 3) and, therefore, very low diversity (Shannon index of 0.858 ± 0.467) and low pastoral value (Mora et al., 2022). Soils are classified as Endocalcaric Cambisol (Loamic, Humic) by WRB taxonomy (IUSS Working Group WRB, 2022), characterized by high organic matter content, high aggregate stability and neutral soil reaction, with silty loam texture. Its parent material is composed by Eocene turbidites (Girona-García et al., 2019). The average slope of the sampling plots ranges from 10 to 20 % and all of them are West facing.

2.2. Fire characteristics

Prescribed burns were performed 0, 5 and 9 years before the sampling time (Table 1) by firefighters of the EPRIF (Wildfire Prevention Team) of Aragón and the BRIF (Reinforcement Brigades against Wildfires) of Daroca units. According to available records, these plots have not been burned at any other time during the last 50 years. In all cases, the prescribed burnings were performed with backing fires, when the environmental conditions met the prescribed parameters for *E. horridum*: no heavy rainfall occurred prior to burning, the temperature range was between 5 and 15 °C, the air relative humidity was of 35–70 % and the wind speed of less than 10 km h^{-1} (Girona-García et al., 2019). The fire severity in the PBs was considered to be medium-low, as many partially carbonized remains from the litter layer were found on the surface, together with some ash deposits from the vegetation combustion (Fig. 2), although the duff layer was practically intact and there were no signs that the fire reached the mineral soil (Fernández et al., 2022). The *E. horridum* leaves were totally consumed and the stems were charred, but still in place. Furthermore, there was a slight fire-induced water repellency and no major changes in soil moisture content were observed at 1–3 cm depth.

Table 1
Dates of prescribed burning and plant biomass of each plot at the sampling time.

Treatment	Prescribed burning time	Date	Aboveground shrub dry biomass (t ha^{-1})
UB	Control (unburned)	–	34.1 ± 15.0
B0	Immediate (0 years)	March 2021	<1[a]
B5	Mid-term (5 years)	November 2016	3.0 ± 1.1
B9	Long-term (9 years)	January 2012	8.5 ± 1.1

[a] Charred residues.

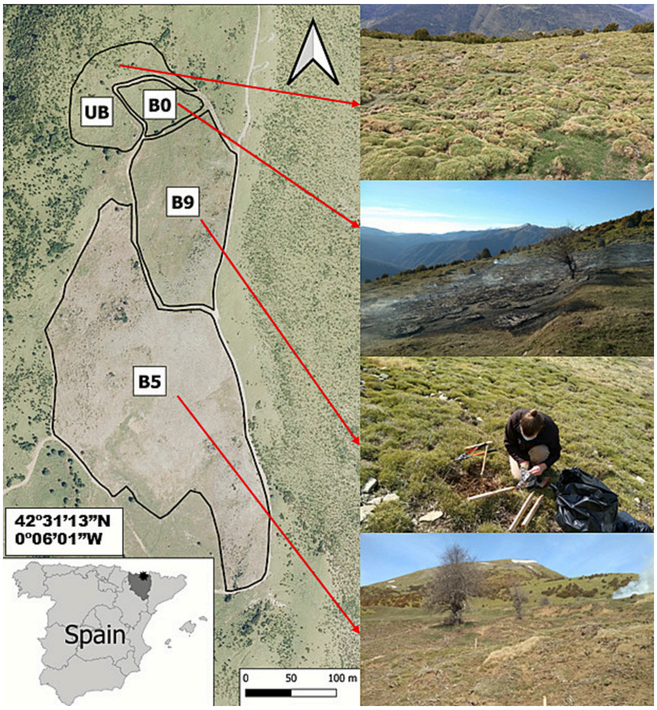

Fig. 1. Location of the area of study (Asín de Broto) in the Central Pyrenees (NE-Spain) and experimental plots: Unburned one (UB), burned and sampled on the day (B0), at 5 (B5) and 9 (B9) years after burning. In each plot, 4 soil replicates (n) for 3 soil depths (0–1, 1–2 and 2–3 cm) have been sampled.

2.3. Sampling

Soil sampling was conducted simultaneously across all the plots. Soils were sampled at three shallow depths (0–1, 1–2 and 2–3 cm) from four representative sampling points (sampling area 0.5 × 0.5 m) for each unburned and burned plots (UB, B0, B5, B9) to assess the immediate, mid-term, and long-term fire effects, respectively, on soil properties (n = 48). Previous studies show that there are no changes in soil properties at greater depths in prescribed fires performed at similar locations (Girona-García et al., 2019). In this regard, Doerr et al. (2025), when reviewing the maximum temperatures and duration of heating in mineral soils during forest fires and prescribed burns, also demonstrate that heat only penetrates the soil very superficially and for short periods of time. In each plot, the points were randomly selected with a five-meter separation from each other, similar to (Alfaro-Leranoz et al., 2023).

Fig. 2. Detailed view of a burned *E. horridum* individual in the B0 plot immediately after the prescribed fire.

Subsamples for biological determinations were separated and stored at 4 °C, and the rest of the soil was air-dried and sieved at 2 mm. Vegetation samples were also taken within each sampling point, to determine its dry biomass.

2.4. Laboratory analysis

The moisture content of the soil samples was determined gravimetrically after oven drying at 105 °C until a constant weight was reached, and all results were calculated on a 105 °C dry soil basis. Soil pH and electrical conductivity at 25 °C (EC) were measured in 1:2.5 *w/v* and 1:5 w/v soil aqueous suspensions, respectively. The total C (TC) and N (TN) concentrations were determined using a CN Vario Max elemental analyzer (Elementar, Hanau, Germany). The TC corresponds to the total soil organic carbon (SOC), since soils showed no $CaCO_3$ content. The dissolved organic carbon (DOC) was calculated as the K_2SO_4 extractable carbon and then measured by chromic oxidation (Vance et al., 1987). Soil water repellency (SWR) was calculated with water drop penetration test (WDPT), proposed by Bisdom et al. (1993), applying droplets of distilled water on non-sieved and air-dried soil samples. The results were categorized using the classes of Doerr (1998).

Microbial biomass carbon (MBC), expressed as g MBC per kg soil, was determined by the fumigation-extraction method (Vance et al., 1987), using an extraction factor of $K_{ec} = 0.38$ (Girona-García et al., 2019). Soil respiration or CO_2 efflux was measured in an incubation assay of soil samples under optimal temperature (25 °C) and moisture (50 % water-holding capacity) conditions. The released CO_2 was captured with NaOH traps (Anderson, 1982) and determined at intervals of 7 days during an incubation period of 28 days. Soil microbial respiration processes were distinguished as i) basal soil respiration (bSR) expressed as mg C-CO_2 per kg soil per day, ii) mineralization coefficient or normalized soil respiration (nSR) expressed as mg C-CO_2 per g SOC per day, and iii) metabolic quotient (qCO_2) expressed as mg C-CO_2 per g microbial biomass carbon (MBC) per day (Alfaro-Leranoz et al., 2023). The enzymatic β-D-glucosidase activity (GLU) was determined in soil incubated with 4-nitro-phenyl-β-D-gluconopyranoside enzyme substrate (Eivazi and Tabatabai, 1988) and expressed as mg ρ-nitrophenol (PNP) per g soil per h.

2.5. Data analysis

Statistical analysis was performed using the RStudio open-source software (R Core Team, 2023). The values reported in the text are expressed as the mean ± standard deviation unless otherwise noted.

2.5.1. Soil organic carbon mineralization modeling

The C-CO_2 emissions during the incubation period were fitted to a double exponential decay model, which divides the SOC into two pools, labile and recalcitrant, with different decomposition rates. The following equation describes the kinetics of both:

$$C_t = C_0 \times \left(1 - e^{-k_1 t}\right) + (SOC - C_0) \times \left(1 - e^{-k_2 t}\right)$$

where C_t is the accumulated C-CO_2 in mg C kg $soil^{-1}$ emitted at time t; C_0 is the labile C content in in mg C kg $soil^{-1}$ with a k_1 mineralization rate in $days^{-1}$, representing the daily flux rate; and SOC – C_0 is the recalcitrant C content with a k_2 mineralization rate in $days^{-1}$. The mean residence times of labile (MRT1) and recalcitrant (MRT2) organic carbon were calculated as $1/k_1$ and $1/k_2$ respectively.

2.5.2. Univariate analyses

A two factorial ANOVA was run to analyze the significant variance of soil response to the prescribed burning time and soil depths, and the interaction among the two factors (treatment x depth). The pairwise comparison of Tukey's HSD test was also used to evaluate the statistical significance of the differences in the response variables ($p < 0.05$). In order to satisfy the assumptions of the statistical tests (homogeneity of variance and normal distribution), the data were subjected to normality (Shapiro-Wilk, Lilliefors) and homoscedasticity (Levene, Bartlett) tests and were transformed whenever necessary ($p < 0.05$), using the boxcox function ("MASS" library). To analyze the significant variance of MRT to the prescribed burning time and soil depths, and the interaction among the two factors (treatment x depth), a non-parametric Scheirer-Ray-Hare test was performed. Subsequently, the Tukey's HSD test was conducted.

2.5.3. Multivariate analyses

An ANOVA-simultaneous component analysis (ASCA), using the "limpca" package (Thiel et al., 2023), was also conducted to obtain a synthetic view of the distribution of the samples depending on the different factors (treatment, depth) and to quantify the contribution of each measured variable to the total variation. In addition, a Pearson correlation was performed too to further identify relationships between the studied soil properties.

2.5.4. Resistance, resilience and recovery indexes

To assess the degree of resistance, resilience and recovery over time of the measured soil properties, the expressions used by López-Poma and Bautista (2014) were employed:

$$\text{Resistance (RS)} = -100\,[(UB - B0)/UB]$$

$$\text{Recovery (RC)} = -100\,[(UB - Bx)/UB]$$

$$\text{Resilience (RL)} = -(RS - RC)$$

where UB represents the value of the variable in unburned conditions, B0 corresponds to the value immediately post-fire, and Bx is the value at B5 (5 years post-fire) and B9 (9 years post-fire). RS and RC values of zero imply maximum resistance and full recovery at a specific time post-fire, respectively, while negative values are indicators of lower resistance and recovery; positive values indicate that the parameter increased above UB values at a given time post-fire. When RL has a value of zero, it means no resilience, positive values indicate resilience, and negative ones imply further decrease in the values of the response variable compared with the initial impact of fire.

3. Results

3.1. Soil parameters under the different treatments

ANOVA test showed no significant interaction between the two experimental factors (treatment, depth) in most studied parameters

Table 2

F and *p* values from the two-way ANOVA for the electrical conductivity (EC), pH, dissolved organic carbon (DOC), soil organic carbon (SOC), C/N relationship, β-D-glucosidase activity (GLU), microbial biomass carbon (MBC), basal soil respiration (bSR) at 7, 14, 21 and 28 incubation days, normalized soil respiration (nSR) and metabolic quotient (qCO_2).

	Treatment		Depth		Interaction	
	F - value	p - value	F - value	p - value	F - value	p - value
EC	74.998	**<0.001**	7.777	**0.002**	1.121	0.370
pH	72.739	**<0.001**	7.584	**0.002**	2.062	0.082
DOC	36.565	**<0.001**	31.194	**<0.001**	9.789	**<0.001**
SOC	9.777	**<0.001**	6.892	**0.003**	1.065	0.401
C/N	13.096	**<0.001**	6.992	**0.003**	0.986	0.449
GLU	10.436	**<0.001**	7.027	**0.003**	2.157	0.070
MBC	17.559	**<0.001**	4.216	**0.023**	0.702	0.650
bSR7	6.465	**0.001**	33.797	**<0.001**	2.178	0.068
bSR14	2.242	0.100	29.408	**<0.001**	1.168	0.345
bSR21	1.134	0.348	25.641	**<0.001**	0.784	0.588
bSR28	0.860	0.471	23.520	**<0.001**	0.745	0.618
nSR	7.168	**<0.001**	16.060	**<0.001**	0.524	0.786
qCO_2	1.978	0.135	1.788	0.182	0.622	0.711

(Table 2), suggesting that any potential differences in fire effects among depths were not statistically detectable.

Immediately after the PB event, the electrical conductivity (EC) significantly increased in all the studied depths (Fig. 3a). The effect of the fire on this parameter disappeared five years later, meeting the UB values. Soil pH was significantly correlated with EC (Table 3) and also significantly increased post-fire (Fig. 3b). Five and nine years after the fire, the pH values were still significantly higher than in the UB plots, but

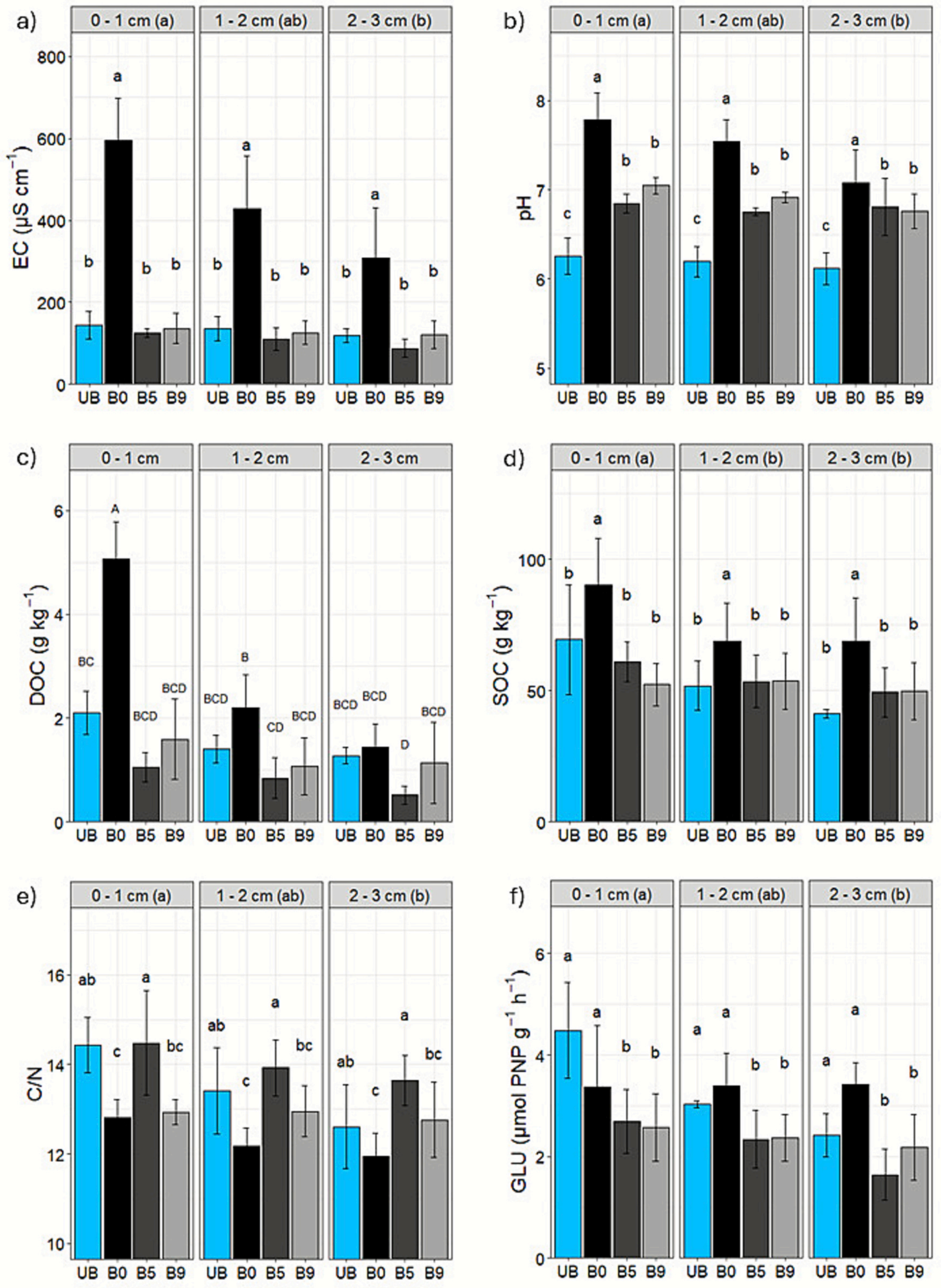

Fig. 3. Electrical conductivity (EC), pH, labile carbon (DOC), soil organic carbon (SOC), C/N relationship, and β-D-glucosidase activity (GLU) for the unburned (UB), immediately burned (B0), 5 years (B5) and 9 years (B9) post-fire samples, within the sampled soil depths (0–1, 1–2 and 2–3 cm). Lowercase letters on top of the bars indicate significant differences between treatments and those in brackets between depths ($p < 0.05$). Uppercase letters on top of the bars indicate significant differences within all samples when the interaction between treatment and depth was significant. In each bar, the mean ($n = 4$) and the standard deviation are represented.

Table 3

Pearson correlation coefficients, showing the degree of linear association between. Electrical conductivity (EC), pH, soil organic carbon (SOC), dissolved organic carbon (DOC), total nitrogen (TN), soil water repellency (SWR), microbial biomass carbon (MBC), basal soil respiration (bSR) at 7, 14, 21 and 28 incubation days, and β-D-glucosidase activity (GLU), for all treatments and all sampled depths ($n = 48$).

Variable	pH	EC	SOC	DOC	TN	SWR	MBC	bSR7	bSR14	bSR21	bSR28
pH											
EC	0.77**										
SOC	0.53**	0.68**									
DOC	0.52**	0.81**	0.67**								
TN	0.61**	0.77**	0.96**	0.70**							
SWR	0.59**	0.78**	0.53**	0.80**	0.57**						
MBC	0.29	0.46**	0.59**	0.46**	0.55**	0.40**					
bSR7	0.51**	0.61**	0.73**	0.82**	0.67**	0.55**	0.58**				
bSR14	0.41**	0.45**	0.65**	0.69**	0.56**	0.40**	0.55**	0.97**			
bSR21	0.34*	0.33*	0.59**	0.59**	0.48**	0.29*	0.52**	0.93**	0.99**		
bSR28	0.29*	0.26	0.55**	0.53**	0.43**	0.22	0.50**	0.89**	0.97**	1.00**	
GLU	0.03	0.39**	0.64**	0.52**	0.60**	0.14	0.46**	0.58**	0.56**	0.54**	0.53**

* $p < 0.05$; ** $p < 0.01$.

significantly lower than in B0. Therefore, both parameters showed positive resistance (RS) values (Table 4). However, they differed in the recovery (RC). While EC returned to its initial values in B9, resulting in RC values close to 0, while pH values remained higher than the UB ones.

Dissolved organic carbon (DOC) values exhibited a significant Treatment x Depth interaction (Table 2). Immediately post-fire DOC increased significantly in the upper layer (0–1 cm), whereas no changes were observed at 1–3 cm depth (Fig. 3c). Consequently, the RS values were much higher at the soil surface (Table 4). Five years post-fire, the DOC content decreased, similar to the UB values, but in B5 it was slightly lower than in B0 (non-significant), showing lower RC values than in B9. The soil organic carbon (SOC) showed a significant increase in all the studied depths (0–3 cm) immediately post-fire, but its value decreased in the mid- and long-term until returning to the UB levels (Fig. 3d). Therefore, the RS values were positive and the RC ones were either positive or close to 0. The C/N relationship decreased significantly immediately after the fire (B0), across all the studied depths, (Fig. 3e), resulting in negative RS values. In B5, this parameter showed significantly higher values than in B0, although the increase was not significant relative to the UB values, and RC values were slightly positive 0. However, nine years post-fire, this increase disappeared and the RC values were slightly negative. The resilience (RL) of this parameter was near to its maximum, with values quite close to 0.

The β-D-glucosidase activity (GLU) was not immediately affected by the fire, although it suffered a decrease over-time, in the mid- and long-term (Fig. 3f). However, a negative RS value was observed at the surface in B0 (Table 4). Both RL and RC values were negative for this parameter, indicating low recovery and no resilience.

The basal soil respiration (bSR) after a 28-day incubation period did not show any significant changes along the studied post-fire period (Table 2). Therefore, the RS, RC and RL values were close to their optimum values (Table 4). However, the bSR was only significantly higher immediately post-fire within the first 7 days of the incubation period and recovered its initial values after 5 years (Table 2, Fig. 4). These results corresponded with a high RS and RC. The mean residence time of the labile C (MRT1) did not suffer any significant effect with the fire, while, on the other hand, the mean residence time of the recalcitrant C (MRT2) suffered a significant increase immediately post-fire, followed by a significant decrease 5 and 9 years later, below pre-fire values (Table 5), resulting in negative RC and RL (Table 4).

The microbial biomass C (MBC) and the normalized soil respiration (nSR) showed significant differences in the ANOVA test for both treatment and depth factors, while the metabolic quotient (qCO_2) did not. However, the post hoc test revealed no significant changes immediately post-fire (B0) compared to the unburned soil (UB) for both parameters (Table A.1). Moreover, throughout the mid- and long-term, none of those parameters differed significantly from the UB values. Despite this, the nSR was significantly higher at 5- and 9-years post-fire compared with B0, resulting in positive RC and RL values. In addition, MBC was significantly lower in B9 than in B0, producing negative RC and RL values (Table 4).

Prior to the fire, the soil was completely hydrophilic (Fig. 5). However, right after the PB, half of the replicates showed a slight hydrophobia from 0 to 1 cm, while only one quarter showed it from 1 to 3 cm. The perturbation on this parameter due to the fire had completely vanished five years post-fire and, therefore, the RC values were optimum (Table 4).

3.2. *Interaction between parameters: ANOVA-simultaneous component analysis (ASCA)*

The score and loading plots in Fig. 6 show the results from the multivariate analysis. ASCA showed both significant differences in treatment (UB, B0, B5 and B9; $p < 0.01$) and sampled soil depth (0–1, 1–2 and 2–3 cm, $p < 0.01$). However, it did not show a significant interaction between the two factors ($p = 0.19$). The treatment explained 30.7 % of the total variance, while the depth explained the 22.31 %.

3.2.1. *Treatment*

For the treatment, the first two principal components accounted for 95.2 % of the variance (Fig. 6a, b). Principal Component 1 (PC1), which explained 80.9 % of the variance, was related to the direct and immediate effects of the prescribed fire. Principal Component 2 (PC2) explained the remaining 14.3 % of the variance and it was related with the time.

PC1 distributes the samples that were burned prior to the sampling (B0) in the right side of the axis, with high positive scores, whereas the unburned and the mid- and long-term burned samples are located on the left side, with lower scores (Fig. 6a). The variables with the highest positive (EC, TN, MRT2, pH, SWR, DOC, SOC) and negative (C/N) loadings are the ones which explain most of the variability (Fig. 6b). The positive ones are related with the B0 samples and experienced an immediate increase after the fire, whereas C/N, with a high negative loading, decreased post-fire. The closer the variables are to the center of the axis, the lower the variation they experienced with the fire.

PC2 distributes the samples by time. In the upper part, with the highest scores, the UB samples; in the center of the axes, the B0, and right under it B5; in the lowest part, the B9 samples can be found. The variable that showed the highest values before burning (GLU) is found in the upper part, with the highest positive loading, while the variable that was higher in B5 and B9 than in UB (pH) shows the highest negative loading.

Table 4
Resistance (RS), recovery (RC), and resilience (RL) after the prescribed burning for the electrical conductivity (EC), pH, dissolved organic carbon (DOC), soil organic carbon (SOC), C/N relationship, β-D-glucosidase activity (GLU), microbial biomass carbon (MBC), basal soil respiration (bSR) at 7, 14, 21 and 28 incubation days, normalized soil respiration (nSR) and metabolic quotient (qCO_2).

Variable	Depth	RS (%)	RC B5 (%)	RC B9 (%)	RL (%)
EC					
	0–1 cm	314.00	−12.87	−5.37	−319.37
	1–2 cm	214.47	−19.16	−8.03	−222.50
	2–3 cm	160.75	−25.51	2.43	−158.31
	Average	229.74	−19.18	−3.65	−233.39
pH					
	0–1 cm	24.46	9.43	12.63	−11.83
	1–2 cm	21.76	9.00	11.63	−10.13
	2–3 cm	15.78	11.32	10.47	−5.31
DOC					
	0–1 cm	140.78	−49.64	−24.24	−165.03
	1–2 cm	56.78	−39.97	−23.79	−80.57
	2–3 cm	12.63	−59.62	−10.62	−23.25
	Average	70.07	−49.74	−19.55	−89.62
SOC					
	0–1 cm	29.77	−12.17	−24.59	−54.36
	1–2 cm	32.90	3.23	3.47	−29.43
	2–3 cm	67.66	19.94	21.03	−46.63
	Average	43.44	3.67	−0.03	−43.47
C/N					
	0–1 cm	−11.17	0.33	−10.37	0.80
	1–2 cm	−9.19	3.85	−3.36	5.83
	2–3 cm	−5.17	8.28	1.26	6.42
	Average	−8.51	4.15	−4.16	4.35
GLU					
	0–1 cm	−24.68	−39.91	−42.60	−17.92
	1–2 cm	11.57	−22.80	−21.90	−33.47
	2–3 cm	41.73	−31.96	−9.72	−51.45
	Average	9.54	−31.55	−24.74	−34.28
MBC					
	0–1 cm	3.46	−8.42	−15.54	−18.99
	1–2 cm	56.08	17.14	−10.55	−66.63
	2–3 cm	26.33	−22.70	−20.84	−47.17
	Average	28.62	−4.66	−15.64	−44.26
MRT1					
	0–1 cm	−3.78	1.44	4.43	8.21
	1–2 cm	42.44	3.83	14.05	−28.38
	2–3 cm	−17.62	−25.52	−38.16	−20.54
	Average	7.01	−6.75	−6.56	−13.57
MRT2					
	0–1 cm	22,574.64	−36.14	−17.12	−22,591.77
	1–2 cm	5809.87	−56.25	−51.76	−5861.63
	2–3 cm	98.35	−98.24	−98.30	−196.65
	Average	9494.29	−63.54	−55.73	−9550.01
SWR					
	0–1 cm	484.38	0.00	0.00	−484.38
	1–2 cm	175.00	0.00	0.00	−175.00
	2–3 cm	156.25	0.00	0.00	−156.25
	Average	271.88	0.00	0.00	−271.88
bSR7					
	0–1 cm	52.17	−3.40	−8.83	−61.01
	1–2 cm	63.58	21.61	25.24	−38.34
	2–3 cm	32.97	−1.51	61.16	28.19
	Average	49.57	5.57	25.85	−23.72
bSR14					
	0–1 cm	24.75	−0.28	−8.23	−32.97
	1–2 cm	47.01	27.72	31.54	−15.47
	2–3 cm	28.80	6.87	68.77	39.96
	Average	33.52	11.44	30.70	−2.83
bSR21					
	0–1 cm	9.43	0.29	−8.19	−17.61
	1–2 cm	38.45	30.61	35.72	−2.72
	2–3 cm	28.29	14.07	68.36	40.08
bSR28		25.39	14.99	31.97	6.58
	0–1 cm	−0.52	−1.46	−8.96	−8.44
	1–2 cm	34.28	30.16	37.52	3.24
	2–3 cm	28.72	16.91	71.35	42.63
	Average	20.83	15.20	33.30	12.47
nSR					
	0–1 cm	−27.87	4.30	14.21	42.08
	1–2 cm	−2.27	21.78	30.26	32.54
	2–3 cm	−23.59	−2.35	35.28	58.87
	Average	−17.91	7.91	26.59	44.50
qCO_2					
	0–1 cm	4.79	3.31	6.51	1.72
	1–2 cm	−13.97	23.92	30.10	44.07
	2–3 cm	7.45	59.79	29.43	21.97
	Average	−0.58	29.01	22.01	22.59

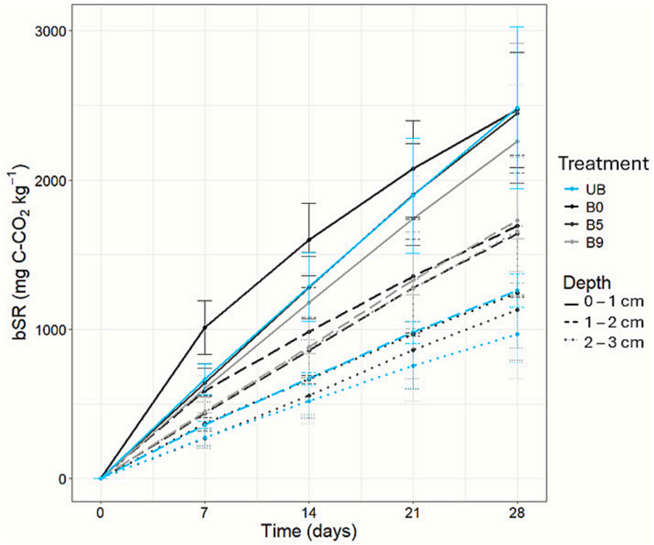

Fig. 4. Basal soil respiration (bSR) in the unburned (UB), immediately burned (B0), 5 years (B5) and 9 years (B9) post-fire samples, by soil depth (0–1, 1–2 and 2–3 cm); the error bars indicate standard deviation of the means (n = 4).

Table 5
Mean residence times of the labile (MRT1) and recalcitrant (MRT2) carbon fractions for the different treatments and depths. Lowercase letters indicate significant differences between treatments ($p < 0.05$).

Treatment	Depth	MRT1 (days)		MRT2 (years)	
	0–1 cm	20.33 ± 7.69		4.41 ± 1.80	
UB	1–2 cm	19.10 ± 12.89	a	8.52 ± 3.87	b
	2–3 cm	33.23 ± 9.91		254.14 ± 497.24	
	0–1 cm	19.57 ± 1.57		1000 ± 0.00	
B0	1–2 cm	27.20 ± 6.1.00	a	503.63 ± 573.16	a
	2–3 cm	27.38 ± 24.52		504.09 ± 572.63	
	0–1 cm	20.63 ± 0.73		2.82 ± 0.15	
B5	1–2 cm	19.83 ± 2.60	a	3.73 ± 0.26	c
	2–3 cm	24.75 ± 7.79		4.48 ± 0.30	
	0–1 cm	21.24 ± 5.04		3.65 ± 1.11	
B9	1–2 cm	21.78 ± 4.98	a	4.11 ± 1.16	c
	2–3 cm	20.55 ± 8.36		4.33 ± 0.89	

UB: unburned; B0: immediately post-fire; B5: 5 years post-fire; B9: 9 years post-fire.

3.2.2. *Soil depth*

For the depth, the first two principal components explain all the variance (Fig. 6c, d). PC1 explains 98.7 % of the variance and distributes the samples by depth, having the 0–1 cm samples the highest negative scores and the 2–3 cm samples the highest positive scores (Fig. 6c). The variables with the greatest differences between depths have the highest negative loadings, while the only variables that showed no significant differences between depths (MRT1, qCO_2) have a low positive loading (Fig. 6d).

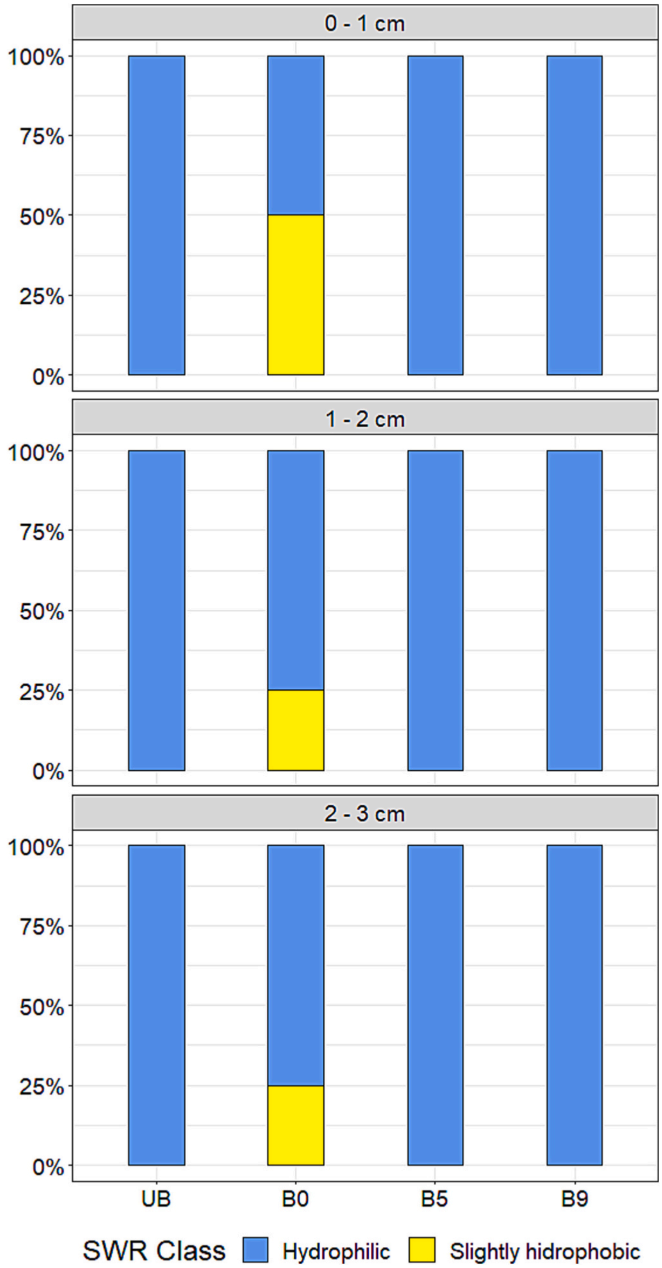

Fig. 5. Occurrence (%) of soil water repellency (SWR) according to the Water Drop Penetration Test (WDPT) for the for the unburned (UB), immediately burned (B0), 5 years (B5) and 9 years (B9) post-fire samples, within the different soil depths (0–1, 1–2 and 2–3 cm). SWR classes defined by Doerr (1998).

4. Discussion

4.1. Impact of the PB on soil organic matter

The fire impact was mainly driven by the incorporation of ash and partially charred materials. This effect can be explained by the immediate increase of EC, pH, DOC, SOC and MRT2, and the decrease in the C/N relationship. The ASCA analysis supports this hypothesis, as those parameters are the ones with the highest loadings and more related with the B0 treatment (Fig. 6a, b).

Ash, which is rich in soluble salts and ions, can produce short-term increases in EC when it is incorporated into the soil and confirms the results obtained in previous studies (Agbeshie et al., 2022; Alcañiz et al., 2018). Soil pH was significantly correlated with EC (Table 3) and its immediate significant post-fire increase was also related with ash incorporation (Mataix-Solera et al., 2009); as acid topsoil, unlike the alkaline ones, are not able to buffer the alkalinization effect of the ash (Certini, 2005). In Francos et al. (2019), low-severity PBs performed in a dry Mediterranean area also produced an immediate topsoil EC increase. Nevertheless in their research, EC values remained higher than in the UB plot even after 13 years, unlike in the present study, where higher annual precipitations, may have contributed for the mid-term decrease of EC due to leaching (Alfaro-Leranoz et al., 2023). However, they did not report immediate changes in pH due to the alkalinity of their soil, which was also reported by Soria et al. (2023). In similar wet areas from the Central Pyrenees, Girona-García et al. (2018c) reported immediate increases in soil pH in one of their sites, which they mainly attributed to the elimination of soil organic matter (SOM) acidic groups. Moreover, they also found a different behavior of pH and EC with the time, relating the EC decreases with the high annual precipitations that produced erosion and leaching.

The superficial increase in DOC (0–1 cm) may result from the partial combustion of the SOM (Alfaro-Leranoz et al., 2024; Dou et al., 2023), which generates pyrogenic organic carbon (Alexis et al., 2007; Bodí et al., 2014; Escuer-Arregui et al., 2025), as well as from the addition of ash, rich in soluble organic substances (Mora et al., 2021). This is consistent with the high correlation between DOC and EC (Table 3), and the proximity of MRT2 (recalcitrant C) and DOC in the ASCA analysis (Fig. 6b). Between 200 and 450 °C, the combustion of SOM is incomplete and, therefore, the formation of pyrogenic organic carbon occurs (Alcañiz et al., 2016; Bodí et al., 2014; Santín and Doerr, 2016). The effect of the fire on this parameter can be easily missed if the studied depth interval is too high, as happened to San Emeterio et al. (2016), who studied the PB effects on soils (0–10 cm) in Western Pyrenees up to 1.5 years. In contrast, other similar studies performed in the Central Pyrenees show no immediate changes in this parameter (Alfaro-Leranoz et al., 2023; Armas-Herrera et al., 2016; Girona-García et al., 2018a). However, all of them report that no ash was incorporated into the soil during the sampling process, unlike in the present study, where the windy conditions made it difficult to avoid ash incorporation into the samples.

The immediate increase in SOC was also related to the incorporation of ash and partially charred remains from the burned shrubs, mainly in the form of DOC. Evidence supporting this observation is provided by the high correlation observed between SOC, DOC and EC (Table 3). Generally, low-intensity PBs tend to produce either no changes or increases of SOC, as widely reported in the literature (Agbeshie et al., 2022; Alcañiz et al., 2018). However, some studies performed in similar wet mountainous areas have described immediate decreases in SOC, associated with high burning intensities, like Armas-Herrera et al. (2016) and Girona-García et al. (2018a). In contrast, other studies in comparable locations, have reported no changes in SOC under lower intensity PBs (Alfaro-Leranoz et al., 2023; Fontúrbel et al., 2024; Girona-García et al., 2019). Similarly to the present study, Cadenas et al. (2024) and Shakesby et al. (2015) found immediate increases of SOM following low severity fires in wet areas in NW Spain and Central Portugal. Furthermore, several studies performed in Mediterranean dry environments also align with our results, documenting immediate post-fire SOM increases after low intensity fires (Alcañiz et al., 2016; Lucas-Borja et al., 2022; Rodríguez et al., 2018; Úbeda et al., 2005).

The immediate decrease in the C/N relationship, produced by an increase of TN, greater than the one of SOC, was also observed in other studies. Although the TN is quite sensitive to the effect of heating, being volatilized at 200 °C (Francos et al., 2019; Mataix-Solera and Guerrero, 2007), its loss is sometimes compensated by the incorporation of ash and charred residues, as reported by San Emeterio et al. (2013), who found significant TN increases (up to 10 cm-deep) after a low-severity shrub PB in the Western Pyrenees. The strong correlation observed between DOC, EC and TN further supports this hypothesis (Table 3), suggesting that most of the organic N inputs were originated from ash incorporation, as

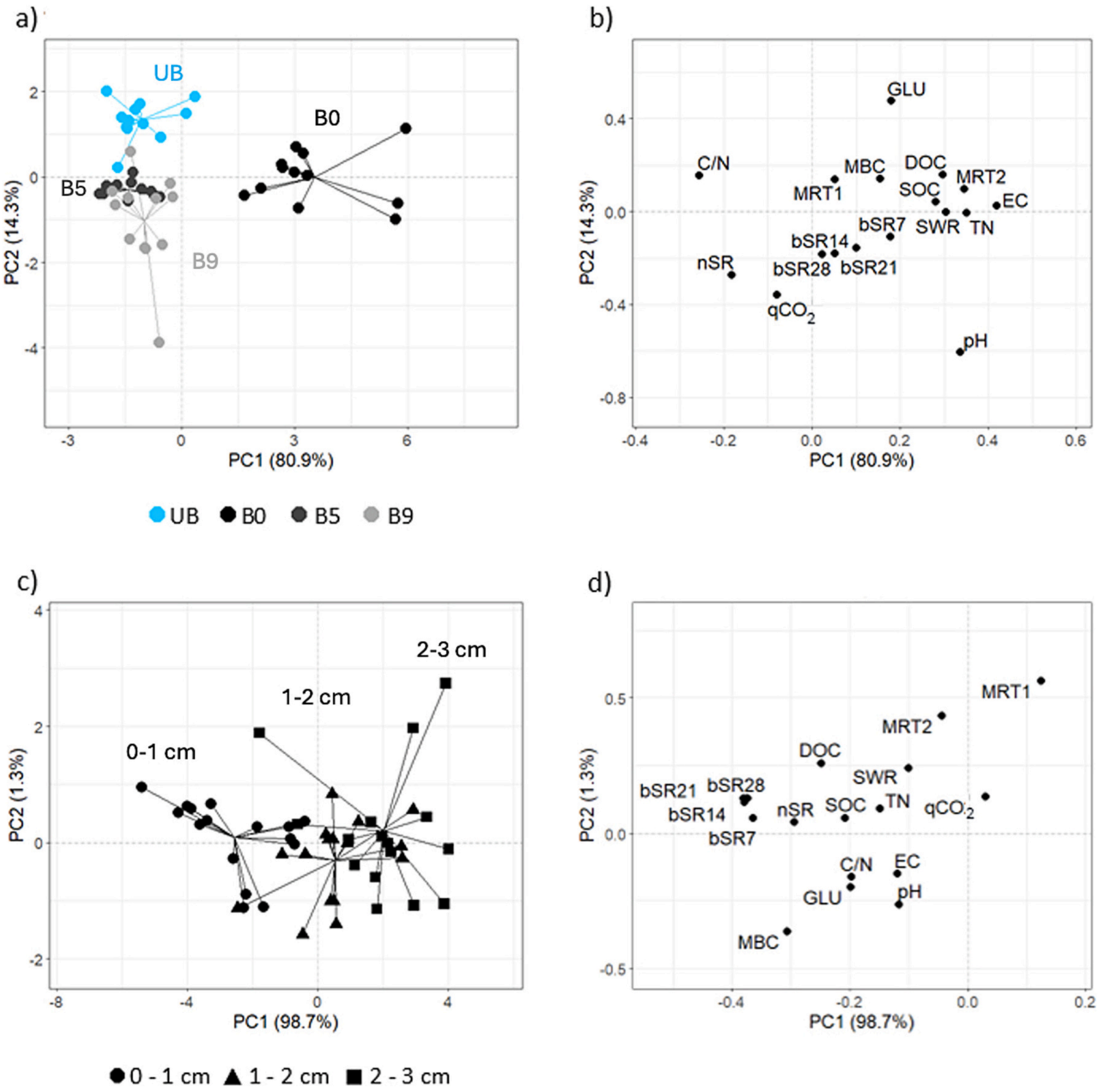

Fig. 6. Score and loading plots of the ANOVA simultaneous component analysis (ASCA): score plot for treatment (a), loading plot for treatment (b), score plot for depth (c), loading plot for depth (d). Abbreviations refer to unburned (UB), immediately burned (B0), 5 years (B5) and 9 years (B9) post-fire, electrical conductivity (EC), pH, soil water repellency (SWR), labile carbon (DOC), soil organic carbon (SOC), total nitrogen (TN), C/N relationship, microbial biomass carbon (MBC), basal soil respiration (bSR), normalized soil respiration (nSR), metabolic quotient (qCO$_2$) and β-D-glucosidase activity (GLU).

reported in previous studies (Alcañiz et al., 2016; Cadenas et al., 2024; Dou et al., 2023; Lucas-Borja et al., 2022). However, mineral N forms, like NH_4^+ and NO_3^-, were not measured in the present study, which limits the scope of this affirmation. In contrast to the present work, other shrub PB researches performed in the Central Pyrenees have reported either no immediate effects in the topsoil (0–3 cm) TN (Alfaro-Leranoz et al., 2023; Girona-García et al., 2019), or decreases (Armas-Herrera et al., 2016; Girona-García et al., 2018a; Mora et al., 2021). Notably, these studies did not report ash incorporation in the sampling, which may explain the observed differences. Thence, taking samples at a centimeter scale of soils recently affected by wildfires, in which the ash layer and other burnt residues were excluded, has shown that severe fires exerted a mineralization effect of the SOM by thermal oxidation in the most superficial part of the Ah horizon, with the consequent increase in available nutrients (Badía et al., 2014).

The immediate fire-induced effects on EC, DOC, SOC and TN typically disappear within the first years after the fire, as the present work shows. The short-to-mid-term stabilization to pre-fire values of SOM-related parameters have been widely reported and attributed mainly to microbial degradation, soil erosion and leaching. In this sense, Cadenas et al. (2024) and Shakesby et al. (2015) reported that the SOM and TN returned to pre-fire values, taking up to three years, due to microbial degradation and soil erosion, both due to soil exposure after the elimination of the shrub cover, after shrub low severity fires in wet areas in NW Spain and Central Portugal. Additionally, Alfaro-Leranoz et al. (2023), in PBs performed in Pyrenean shrublands, also reported mid-term losses of SOC, DOC and TN due to microbial degradation and/or soil erosion resulting from post-fire soil exposure. Post-fire soil erosion is likely to occur because of vegetation cover loss and fire-induced water repellency (Alados et al., 2019), or by wind (Girona-García et al., 2018c). Additionally, Alcañiz et al. (2016) observed, as in the present work, a gradual reduction of SOC over time, reaching the UB values 13 years post-fire. In addition, Dou et al. (2023), who studied the PB effects in a temperate pine forest in China, also found a significant immediate increase in TN, produced by the organic N oxidation by fire, creating ammonium and nitrate, which rapidly disappeared 6 months later. Ammonium and nitrate can be either leached by rain or consumed by plants during the post-fire vegetation recovery (Alcañiz et al., 2016;

Cadenas et al., 2024; Francos et al., 2019), especially in areas with high annual precipitations. If the shrub community, with its recalcitrant tissues, is replaced by pastures, the quality of decomposition of the leaf litter is modified, which has an idirect effect on the soil pH, Nutiretn availability and the quantity and quality of soil organic matter (Laorden-Camacho et al., 2025).

Soil water repellency (SWR) is closely related to the previously discussed soil biochemical properties (Alfaro-Leranoz et al., 2024), specially to SOC (Girona-García et al., 2018b; González-Pelayo et al., 2015). The development of hydrophobia immediately after a medium-low severity fire have been reported by many studies. Alfaro-leranoz et al. (2024), in an experimental burning of soil monoliths from the same location (Asín de Broto) at different intensities, found that SWR increased from 0 to 1 cm with the low and medium fire intensities, but decreased with the highest one. On the other hand, from 1 to 3 cm, the SWR increased even at the highest intensity. The development of hydrophobia in burned soils is commonly produced by the incorporation of aliphatic hydrocarbons and amphiphilic compounds (Mataix-Solera et al., 2009). This incorporation is certainly more significant in the upper layer of the soil, as most of the hydrophobic compounds come from the combustion of the vegetation and the duff and litter layers. However, those compounds can migrate at lower depths and condense, producing hydrophobia at deeper soil layers (De Bano, 1981). This immediate effect totally disappeared in all the studied depths in the mid- and long-term after the -fire. The strong correlation between DOC, SOC, EC and SWR (Table 3) evidence that the hydrophobic organic compounds were washed away and lixiviated with the time or consumed by soil microorganisms.

4.2. Effects on soil biological properties

The lack of immediate effects on the β-D-glucosidase activity (GLU) highlights the low severity of the PB, as GLU is one of the most heat-sensitive soil properties (Alfaro-Leranoz et al., 2023; Pereira et al., 2023). In low-intensity fires, the immediate effect on this enzyme is typically superficial. This was observed by Alfaro-Leranoz et al. (2024), who found significant decreases in GLU in low-to-medium intensity experimental fires performed on soil monoliths from a nearby site. However, as in the present study, they did not find significant effects on this parameter in the lowest intensity fire. The thermal-related effects on this parameter are normally produced by the enzyme denaturation (Alfaro-Leranoz et al., 2023). During higher intensity fires, the immediate effects on GLU increase, producing reductions deeper into the soil (Armas-Herrera et al., 2016; Pereira et al., 2023).

The mid- and long-term decreases on this enzyme activity were related with changes on the soil cover and on the quality of SOM. López-Poma and Bautista (2014) found that GLU was higher under shrubs than under bare soil or grass. This suggests that the plant community plays a crucial role in maintaining or enhancing GLU levels. In the present study, the woody vegetation load in B5 and B9 was still far below that of UB (Table 1), which may explain the observed differences. Additionally, other authors have reported a decline in GLU over the time due to a decrease in DOC, which is coherent with the significant correlation observed between these parameters in the present study (Table 3). Although the decrease in DOC at B5 and B9 relative to UB was not significant (Fig. 3c). Similar patterns have been observed in other studies performed in the Central Pyrenees, as Alfaro-Leranoz et al. (2023), who described a decrease in GLU even 10 years post-fire, or Girona-García et al. (2019), that reported reductions just 1 year post-fire.

The higher basal soil respiration values during the first week of the incubation period (bSR7) for the B0 samples were related to the microbial consumption of labile C compounds, which are believed to be the main drivers for changes in this parameter (Castro-Díez et al., 2012). Once these labile pools were depleted, the bSR was stabilized, allowing more recalcitrant fractions of soil organic matter to dominate the decomposition dynamics. These results are consistent with the high correlation between bSR7 and DOC, SOC and TN (Table 3). This shift is also reflected by the higher MRT2 in B0, calculated using a two-pool decay model, which reflects the slower turnover of pyrogenic and structural complex C compounds incorporated during the aerial vegetation consumption (Alexis et al., 2007; Bodí et al., 2014).The lower nSR observed in B0 further supports that these recalcitrant compounds were not readily metabolized by microorganisms. In contrast to our current findings, in a similar study, Alfaro-Leranoz et al. (2023) found a decrease in bSR 6 and 10 years after the fire, related with a decrease of DOC, SOC and TN content, which was influenced by the quality and quantity of SOM. Although they did not detect any significant changes in MBC, they observed a temporal increase in nSR and a long-term decrease in qCO_2, which was associated with increased recalcitrance of SOM.

The remaining biological parameters studied (MBC, nSR, qCO_2) showed no changes compared to the UB soil, providing further evidence of the low severity of the PB, as neither microbial biomass nor activity were affected. These parameters are generally sensitive to the direct effect of PBs, as reported in previous studies (Armas-Herrera et al., 2018, 2016; Girona-García et al., 2019, 2018a). Consistent with our results, other studies have also reported no changes in the microbial load and activity after low severity PBs in similar environments (Girona-García et al., 2019; San Emeterio et al., 2016). The absence of effects on qCO_2 further confirms that the soil microbial community did not experience any major stress conditions (Akburak et al., 2018). None of these parameters showed significant changes in the mid- and long-term, compared with the UB values, supporting no indirect effects of the fire on them.

5. Conclusions

The prescribed burning of scrub carried out in Asín de Broto did not directly generate any direct negative effect on the top centimeters of the soil (0–3 cm), confirming our hypothesis and evidencing the low fire severity. The fire produced an immediate significant increase in most of the chemical parameters studied (EC, pH, DOC, SOC, TN). Those changes were mostly related to the incorporation of ash and partially charred organic matter into the soil due to the combustion of the *E. horridum* shrubland and the partial combustion of the litter layer. This incorporation also induced a slight water repellency, especially on the topsoil centimeter (0–1 cm). In the medium term (B5), most of those changes had already disappeared, which was attributed to various processes such as leaching, erosion and microbial degradation, in an area with high annual precipitations.

Regarding the biological properties studied, none showed any immediate significant changes, which seemed related to the fact that the temperatures reached in the soil by the PB were not high. Only GLU suffered a significant decrease with time, confirming our initial hypothesis. The low input of leaf litter into the soil during the following years after the PB, due to the early stage of the shrub community, could be behind this decline. In fact, 9 years after the PB, the shrub biomass was far from reaching its original values. Therefore, this study has proven that GLU is a sensitive biological indicator that can be used to assess the effects of low-severity fires.

Moreover, most of the over-time fire-induced changes on soil properties are directly related to the fire effects on soil organic matter quality and quantity, as mentioned in previous studies. Consequently, soil organic matter plays a central role in these processes and it can be considered a key integrative indicator of the soil response to prescribed burning. However, the PB impact on this soil over time is minimal compared to previous research, which may be related to thermal heterogeneity among the different prescribed brush burns studied. However, we have concluded that, given the absence of significant direct negative effects on soil properties and the minimum impact of the PB over-time (GLU), PBs can be applied in this particular area without compromising soil health, supporting their integration into conservation-oriented land management. Nonetheless, further studies linking environmental variables such as temperature, precipitation, slope and aspect are needed to provide more precise recommendations regarding management.

CRediT authorship contribution statement

Andoni Alfaro-Leranoz: Writing – review & editing, Writing – original draft, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **David Badía-Villas:** Writing – review & editing, Writing – original draft, Visualization, Validation, Resources, Project administration, Funding acquisition, Conceptualization. **Clara Martí-Dalmau:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

A. Alfaro-Leranoz was funded by a DGA predoctoral research grant (BOA20200713012), financed by the Government of Aragón. We appreciate Fran Allué help driving us through difficult unpaved roads to the sampling area. We thank Rafael de Partearroyo and Raúl Vicente (EPRIF, Aragón) and Francho Aso (DGA) for allowing us to participate in the prescribed fire. We acknowledge José Antonio González Pérez (IRNAS-CSIC) for letting us use their equipment. We also thank Rosa Luis, Marta Escuer, Ana Paula Conte, Belén Aguado, José Antonio Manso and Esther López for their assistance in the field and laboratory during the study.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2025.109728.

Data availability

https://zaguan.unizar.es/?ln=es (Original data) (https://zaguan.unizar.es/?ln=es)

References

Agbeshie, A.A., Abugre, S., Atta-Darkwa, T., Awuah, R., 2022. A review of the effects of forest fire on soil properties. J. For. Res. 33, 1419–1441. https://doi.org/10.1007/s11676-022-01475-4.

Akburak, S., Son, Y., Makineci, E., Çakir, M., 2018. Impacts of low-intensity prescribed fire on microbial and chemical soil properties in a *Quercus frainetto* forest. J. For. Res. 29, 687–696. https://doi.org/10.1007/s11676-017-0486-4.

Alados, C.L., Saiz, H., Nuche, P., Gartzia, M., Komac, B., Frutos, D.E., Pueyo, Y., 2019. Clearing vs. burning for restoring Pyrenean grasslands after shrub encroachment. Geogr. Res. Lett. 45, 441–468. https://doi.org/10.18172/cig.3589.

Alcañiz, M., Outeiro, L., Francos, M., Farguell, J., Úbeda, X., 2016. Long-term dynamics of soil chemical properties after a prescribed fire in a Mediterranean forest (Montgrí massif, Catalonia, Spain). Sci. Total Environ. 572, 1329–1335. https://doi.org/10.1016/j.scitotenv.2016.01.115.

Alcañiz, M., Outeiro, L., Francos, M., Úbeda, X., 2018. Effects of prescribed fires on soil properties: a review. Sci. Total Environ. 613–614, 944–957. https://doi.org/10.1016/j.scitotenv.2017.09.144.

Alexis, M.A., Rasse, D.P., Rumpel, C., Bardoux, G., Péchot, N., Schmalzer, P., Drake, B., Mariotti, A., 2007. Fire impact on C and N losses and charcoal production in a scrub oak ecosystem 82, 201–216. https://doi.org/10.1007/s10533-006-9063-1.

Alfaro-Leranoz, A., Badia-Villas, D., Marti-Dalmau, C., Emran, M., Conte-Dominguez, A. P., Ortiz-Perpiña, O., 2023. Long-term evolution of shrub prescribed burning effects on topsoil organic matter and biological activity in the Central Pyrenees (NE-Spain). Sci. Total Environ. 888, 163994. https://doi.org/10.1016/j.scitotenv.2023.163994.

Alfaro-Leranoz, A., Badía-Villas, D., Martí-Dalmau, C., Escuer-Arregui, M., Quintana-Esteras, S., 2024. The effects of fire intensity on the biochemical properties of a soil under scrub in the Pyrenean subalpine stage. Fire 7, 452. https://doi.org/10.3390/fire7120452.

Amsten, K., Cromsigt, J.P.G.M., Kuijper, D.P.J., Loberg, J.M., Jung, J., Strömgren, M., Niklasson, M., Churski, M., 2024. Pyric herbivory in a temperate European wood-pasture system. J. Appl. Ecol. 1–14. https://doi.org/10.1111/1365-2664.14618.

Anderson, J.P.E., 1982. Soil respiration, in: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of soil analysis. Part 2. Chemical and microbiological properties. Agronomy monograph N^{o}9. Madison, pp. 831–871. https://doi.org/10.2134/agronmonogr9.2.2ed.c41.

Aponte, H., Galindo-Castañeda, T., Yáñez, C., Hartmann, M., Rojas, C., 2022. Microbial community-level physiological profiles and genetic prokaryotic structure of burned soils under Mediterranean Sclerophyll forests in Central Chile. Front. Microbiol. 13, 1–16. https://doi.org/10.3389/fmicb.2022.824813.

Armas-Herrera, C.M., Martí, C., Badía, D., Ortiz-Perpiñá, O., Girona-García, A., Porta, J., 2016. Immediate effects of prescribed burning in the Central Pyrenees on the amount and stability of topsoil organic matter. Catena 147, 238–244. https://doi.org/10.1016/j.catena.2016.07.016.

Armas-Herrera, C.M., Martí, C., Badía, D., Ortiz-Perpiñá, O., Girona-García, A., Mora, J. L., 2018. Short-term and midterm evolution of topsoil organic matter and biological properties after prescribed burning for pasture recovery (Tella, Central Pyrenees, Spain). L. Degrad. Dev. 29, 1545–1554. https://doi.org/10.1002/ldr.2937.

Badía, D., Martí, C., Aguirre, A.J., Aznar, J.M., González-Pérez, J.A., De la Rosa, J.M., León, J., Ibarra, P., Echeverría, T., 2014. Wildfire effects on nutrients and organic carbon of a Rendzic Phaeozem in NE Spain: changes at cm-scale topsoil. Catena 113, 267–275. https://doi.org/10.1016/j.catena.2013.08.002.

Badía, D., López-García, S., Martí, C., Ortíz-Perpiñá, O., Girona-García, A., Casanova-Gascón, J., 2017. Burn effects on soil properties associated to heat transfer under contrasting moisture content. Sci. Total Environ. 601–602, 1119–1128. https://doi.org/10.1016/j.scitotenv.2017.05.254.

Badía-Villas, D., González-Pérez, J.A., Aznar, J.M., Arjona-Gracia, B., Martí-Dalmau, C., 2014. Changes in water repellency, aggregation and organic matter of a mollic horizon burned in laboratory: soil depth affected by fire. Geoderma 213, 400–407. https://doi.org/10.1016/j.geoderma.2013.08.038.

Bisdom, E.B.A., Dekker, L.W., Schoute, J.F.T., 1993. Water repellency of sieve fractions from sandy soils and relationships with organic material and soil structure. Geoderma 56, 105–118. https://doi.org/10.1016/0016-7061(93)90103-R.

Bodí, M.B., Martin, D.A., Balfour, V.N., Santín, C., Doerr, S.H., Pereira, P., Cerdà, A., Mataix-solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 130, 103–127. https://doi.org/10.1016/j.earscirev.2013.12.007.

Cadenas, R.M., Castedo-Dorado, F., Valbuena, L., 2024. Medium-term comparative effects of prescribed burning and mechanical shredding on soil characteristics in heathland and Shrubland habitats: insights from a protected natural area. Fire 7. https://doi.org/10.3390/fire7050160.

Caon, L., Vallejo, V.R., Coen, R.J., Geissen, V., 2014. Effects of wildfire on soil nutrients in Mediterranean ecosystems. Earth-Science Rev. 139, 47–58. https://doi.org/10.1016/j.earscirev.2014.09.001.

Castro-Díez, P., Fierro-Brunnenmeister, N., González-Muñoz, N., Gallardo, A., 2012. Effects of exotic and native tree leaf litter on soil properties of two contrasting sites in the Iberian Peninsula. Plant and Soil 350, 179–191. https://doi.org/10.1007/s11104-011-0893-9.

Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143, 1–10. https://doi.org/10.1007/s00442-004-1788-8.

De Bano, L.F., 1981. Water Repellent Soils: A State-of-the-Art. US Dep. Agric. For. Serv. Gen. Tech, Rep.

Doerr, S.H., 1998. On standardizing the “water drop penetration time” and the “molarity of an ethanol droplet” techniques to classify soil hydrophobicity: a case study using medium textured soils. Earth Surf. Process. Landforms 23, 663–668. https://doi.org/10.1002/(SICI)1096-9837(199807)23:7<663::AID-ESP909>3.0.CO;2-6.

Doerr, S.H., Girona-García, A., Sánchez-García, Badía-Villas, Bryant, D., Dickinson, R., Mataix-Solera, M., Miesel, J., Robichaud, J., Stoof, P.R., Santín, C.R., 2025. Soil heating during wildfires and prescribed burns: a global evaluation. Int. J. Wildland Fire 34, WF25103. https://doi.org/10.1072/WF25103.

Dou, X., Köster, K., Sun, A., Li, G., Yue, Y., Sun, L., Ding, Y., 2023. Temporal dynamics of soil dissolved organic carbon in temperate forest managed by prescribed burning in Northeast China. Environ. Res. 237. https://doi.org/10.1016/j.envres.2023.117065.

Eivazi, F., Tabatabai, M.A., 1988. Glucosidases and galactosidases in soils. Soil Biol. Biochem. 20, 601–606. https://doi.org/10.1016/0038-0717(88)90141-1.

Emran, M., Alfaro-Leranoz, A., Martí-Dalmau, C., Badía-Villas, D., 2025. Evolution of Glomalin-Related Soil Proteins over ten years of prescribed scrub burning in the Southern Pyrenees. Geoderma Reg. e01039. https://doi.org/10.1016/j.geodrs.2025.e01039.

Escuer-Arregui, M., Alfaro-Leranoz, A., Badía-Villas, D., Conte-Domínguez, A.P., 2025. A burning intensity gradient modifies sensitive soil properties depending on sampled soil depth and the time since fire. Fire 8, 351. https://doi.org/10.3390/fire8090351.

Fernández, C., Fernández-Alonso, J., Keizer, J.J., Girona-García, A., 2022. Guía para la evaluación de campo de la severidad del fuego en el suelo. Ed. Xunta de Galicia, Spain 40 pp. Available online: https://lourizan.xunta.gal/es/actualidad/transferencia/guia-para-la-evaluacion-de-campo-de-la-severidad-del%20fuego-en-el-suelo.

Fernández-Guisuraga, J.M., Fernandes, P.M., 2024. Prescribed burning mitigates the severity of subsequent wildfires in Mediterranean shrublands. Fire Ecol. 20. https://doi.org/10.1186/s42408-023-00233-z.

Fletcher, W.J., Zielhofer, C., 2013. Fragility of Western Mediterranean landscapes during Holocene rapid climate changes. Catena 103, 16–29. https://doi.org/10.1016/j.catena.2011.05.001.

Fontúrbel, T., Carrera, N., Vega, J.A., Fernández, C., 2021. The effect of repeated prescribed burning on soil properties: a review. Forests 12. https://doi.org/10.3390/f12060767.

Fontúrbel, M.T., Jiménez, E., Merino, A., Vega, J.A., 2024. Contrasting immediate impact of prescribed fires and experimental summer fires on soil organic matter quality and microbial properties in the forest floor and mineral soil in Mediterranean black pine forest. Sci. Total Environ. 907. https://doi.org/10.1016/j.scitotenv.2023.167669.

Francos, M., Stefanuto, E.B., Úbeda, X., Pereira, P., 2019. Long-term impact of prescribed fire on soil chemical properties in a wildland-urban interface. Northeastern Iberian Peninsula. Sci. Total Environ. 689, 305–311. https://doi.org/10.1016/j.scitotenv.2019.06.434.

Gao, C., He, J., Cong, J., Zhang, S., Wang, G., 2018. Impact of forest fires generated black carbon deposition fluxes in great Hinggan Mountains (China). L. Degrad. Dev. 29, 2073–2081. https://doi.org/10.1002/ldr.2837.

García-Ruiz, J.M., Tomás-Faci, G., Diarte-Blasco, P., Montes, L., Domingo, R., Sebastián, M., Lasanta, T., González-Sampériz, P., López-Moreno, J.I., Arnáez, J., Beguería, S., 2020. Transhumance and long-term deforestation in the subalpine belt of the central Spanish Pyrenees: an interdisciplinary approach. Catena 195, 104744. https://doi.org/10.1016/j.catena.2020.104744.

Gartzia, M., Alados, C.L., Pérez-Cabello, F., 2014. Assessment of the effects of biophysical and anthropogenic factors on woody plant encroachment in dense and sparse mountain grasslands based on remote sensing data. Prog. Phys. Geogr. 38, 201–217. https://doi.org/10.1177/0309133314524429.

Gelabert, P.J., Rodrigues, M., de la Riva, J., Ameztegui, A., Sebastià, M.T., Vega-Garcia, C., 2021. LandTrendr smoothed spectral profiles enhance woody encroachment monitoring. Remote Sens. Environ. 262. https://doi.org/10.1016/j.rse.2021.112521.

Girona-García, A., Badía-Villas, D., Martí-Dalmau, C., Ortiz-Perpiñá, O., Mora, J.L., Armas-Herrera, C.M., 2018a. Effects of prescribed fire for pasture management on soil organic matter and biological properties: a 1-year study case in the Central Pyrenees. Sci. Total Environ. 618, 1079–1087. https://doi.org/10.1016/j.scitotenv.2017.09.127.

Girona-García, A., Ortiz-Perpiñá, O., Badía-Villas, D., Martí-Dalmau, C., 2018b. Effects of prescribed burning on soil organic C, aggregate stability and water repellency in a subalpine shrubland: variations among sieve fractions and depths. Catena 166, 68–77. https://doi.org/10.1016/j.catena.2018.03.018.

Girona-García, A., Zufiaurre Galarza, R., Mora, J.L., Armas-Herrera, C.M., Martí, C., Ortiz-Perpiñá, O., Badía-Villas, D., 2018c. Effects of prescribed burning for pasture reclamation on soil chemical properties in subalpine shrublands of the Central Pyrenees (NE-Spain). Sci. Total Environ. 644, 583–593. https://doi.org/10.1016/j.scitotenv.2018.06.363.

Girona-García, A., Ortiz-Perpiñá, O., Badía-Villas, D., 2019. Dynamics of topsoil carbon stocks after prescribed burning for pasture restoration in shrublands of the Central Pyrenees (NE-Spain). J. Environ. Manage. 233, 695–705. https://doi.org/10.1016/j.jenvman.2018.12.057.

Gómez, D., 2008. Chapter 7. Aspectos ecológicos de los pastos. In: Fillat, F., García-González, R., Gómez, D., Reiné, R. (Eds.), Pastos del Pirineo. Consejo Superior de Investigaciones Científicas, CSIC, Spain, pp. 61–73.

González-Pelayo, O., Gimeno-García, E., Ferreira, C.S.S., Ferreira, A.J.D., Keizer, J.J., Andreu, V., Rubio, J.L., 2015. Water repellency of air-dried and sieved samples from limestone soils in Central Portugal collected before and after prescribed fire. Plant Soil 394, 199–214.

Granged, A.J.P., Zavala, L.M., Jordán, A., Bárcenas-Moreno, G., 2011. Post-fire evolution of soil properties and vegetation cover in a Mediterranean heathland after experimental burning: a 3-year study. Geoderma 164, 85–94. https://doi.org/10.1016/j.geoderma.2011.05.017.

IUSS Working Group WRB, 2022. World Reference Base for soil resources. International soil classification system for naming soils and creating legends for soil maps. In: International Union of Soil Sciences (IUSS), Vienna, Austria, 4th ed.

Jordán, A., Zavala, L.M., Granged, A.J.P., Gordillo-Rivero, Á.J., García-Moreno, J., Pereira, P., Bárcenas-Moreno, G., de Celis, R., Jiménez-Compán, E., Alanís, N., 2016. Wettability of ash conditions splash erosion and runoff rates in the post-fire. Sci. Total Environ. 572, 1261–1268. https://doi.org/10.1016/j.scitotenv.2015.09.140.

Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 85, 91–118. https://doi.org/10.1007/s10533-007-9104-4.

Komac, B., Kefi, S., Nuche, P., Escós, J., Alados, C.L., 2013. Modeling shrub encroachment in subalpine grasslands under different environmental and management scenarios. J. Environ. Manage. 121, 160–169. https://doi.org/10.1016/j.jenvman.2013.01.038.

Laorden-Camacho, L., Tello-García, E., Tappeiner, U., Leitinger, G., Griguiz, K., Lyonnard, B., Colace, M.P., Gallet, C., Lavorel, S., 2025. Shrub encroachment modifies soil properties through plant resources economics tratis. Plant and Soil 514, 2083–2104. https://doi.org/10.1007/s11104-025-07506-3.

Lombao, A., Barreiro, A., Fontúrbel, M.T., Martín, A., Carballas, T., Díaz-Raviña, M., 2021. Effect of repeated soil heating at different temperatures on microbial activity in two burned soils. Sci. Total Environ. 799, 149440. https://doi.org/10.1016/j.scitotenv.2021.149440.

López-Poma, R., Bautista, S., 2014. Plant regeneration functional groups modulate the response to fire of soil enzyme activities in a Mediterranean shrubland. Soil Biol. Biochem. 79, 5–13. https://doi.org/10.1016/j.soilbio.2014.08.016.

Lucas-Borja, M.E., J., de las Heras, Moya Navarro, D., González-Romero, J., Peña-Molina, E., Navidi, M., Fajardo-Cantos, Á., Miralles Mellado, I., Plaza-Alvarez, P.A., Gianmarco Carrà, B., Wagenbrenner, J.W., Zema, D.A., 2022. Short-term effects of prescribed fires with different severity on rainsplash erosion and physico-chemical properties of surface soil in Mediterranean forests. J. Environ. Manage. 322. https://doi.org/10.1016/j.jenvman.2022.116143.

Martín-Ramos, P., Martín-Gil, J., Gómez-García, D., Cuchí-Oterino, J.A., 2020. On the physicochemical characteristics and applications of an “undesirable” Pyrenean thorny cushion dwarf: *Echinospartum horridum* (Vahl) Roth. Plants 9, 1180. https://doi.org/10.3390/plants9091180.

Mataix-Solera, J., Guerrero, C., 2007. Efectos de los incendios forestales en las propiedades edáficas. In: Mataix-Solera, J. (Ed.), Incendios Forestales, Suelos y Erosión Hídrica. Caja Mediterráneo CEMACAM, Alcoi, pp. 5–40.

Mataix-Solera, J., Guerrero, C., Arcenegui, V., Bárcenas, G., Zornoza, R., Pérez Bejarano, A., Bodí, M., Mataix-Beneyto, J., Góomez, I., García-Orenes, F., Navarro-Pedreño, J., Jordán, M.M., Cerdà, A., Doerr, S.H., Úbeda, X., Outeiro, L., Pereira, P., Jordán, A., Zavala, L.M., 2009. Los incendios forestales y el suelo: un resumen de la investigación realizada por el Grupo de Edafología Ambiental de la UMH en colaboración con otros grupos, in: Cerdà, A., Mataix-Solera, J. (Eds.), Efectos de Los Incendios Forestales Sobre Los Suelos de España. pp. 185–218.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., Zavala, L.M., 2011. Fire effects on soil aggregation: a review. Earth-Science Rev. 109, 44–60. https://doi.org/10.1016/j.earscirev.2011.08.002.

Mora, J.L., Girona-García, A., Martí-Dalmau, C., Ortiz-Perpiñá, J.O., Armas-Herrera, C. M., Badía-Villas, D., 2021. Changes in pools of organic matter and major elements in the soil following prescribed pastoral burning in the Central Pyrenees. Geoderma 402. https://doi.org/10.1016/j.geoderma.2021.115169.

Mora, J.L., Badía-Villas, D., Gómez, D., 2022. Fire does not transform shrublands of *Echinospartum horridum* (Vahl) Rothm into grasslands in the Pyrenees. J. Environ. Manage. 315, 115125. https://doi.org/10.1016/j.jenvman.2022.115125.

Muñoz-Rojas, M., Erickson, T.E., Martini, D., Dixon, K.W., Merritt, D.J., 2016. Soil physicochemical and microbiological indicators of short, medium and long term post-fire recovery in semi-arid ecosystems. Ecol. Indic. 63, 14–22. https://doi.org/10.1016/j.ecolind.2015.11.038.

Nadal-Romero, E., Otal-Laín, I., Lasanta, T., Sánchez-Navarrete, P., Errea, P., Cammeraat, E., 2018. Woody encroachment and soil carbon stocks in subalpine areas in the central Spanish Pyrenees. Sci. Total Environ. 636, 727–736. https://doi.org/10.1016/j.scitotenv.2018.04.324.

OPCC-CTP, 2018. El cambio climático en los Pirineos: impactos, vulnerabilidades y adaptación. Observatorio Pirenaico del Cambio Climático-Comunidad de Trabajo de los Pirineos Spain 149-pp. https://opcc-ctp.org/.

Pausas, J.G., Keeley, J.E., 2009. A burning story: the role of fire in the history of life. Bioscience 59, 593–601. https://doi.org/10.1525/bio.2009.59.7.10.

Pereira, J.S., Badía, D., Martí, C., Mora, J.L., Donzeli, V.P., 2023. Fire effects on biochemical properties of a semiarid pine forest topsoil at cm-scale. Pedobiologia - J. Soil Ecol. 96, 150860. https://doi.org/10.1016/j.pedobi.2022.150860.

Picariello, E., Baldantoni, D., Muniategui-Lorenzo, S., Concha-Graña, E., De Nicola, F., 2021. A synthetic quality index to evaluate the functional stability of soil microbial communities after perturbations. Ecol. Indic. 128. https://doi.org/10.1016/j.ecolind.2021.107844.

R Core Team, 2023. R: A Language and Environment for Statistical Computing.

Rabin, S.S., Gérard, F.N., Arneth, A., 2022. The influence of thinning and prescribed burning on future forest fires in fire-prone regions of Europe. Environ. Res. Lett. 17. https://doi.org/10.1088/1748-9326/ac6312.

Reyes, O., García-Duro, J., Salgado, J., 2015. Fire affects soil organic matter and the emergence of Pinus radiata seedlings. Ann. For. Sci. 72, 267–275. https://doi.org/10.1007/s13595-014-0427-8.

Rodríguez, J., González-Pérez, J.A., Turmero, A., Hernández, M., Ball, A.S., González-Vila, F.J., Arias, M.E., 2018. Physico-chemical and microbial perturbations of Andalusian pine forest soils following a wildfire. Sci. Total Environ. 634, 650–660. https://doi.org/10.1016/j.scitotenv.2018.04.028.

Rodríguez-Cardona, B.M., Coble, A.A., Wymore, A.S., Kolosov, R., Podgorski, D.C., Zito, P., Spencer, R.G.M., Prokushkin, A.S., McDowell, W.H., 2020. Wildfires lead to decreased carbon and increased nitrogen concentrations in upland arctic streams. Sci. Rep. 10, 1–9. https://doi.org/10.1038/s41598-020-65520-0.

San Emeterio, L., Múgica, L., Gutiérrez, R., Juaristi, A., Pedro, J., Canals, R., 2013. Cambios en el nitrógeno edáfico tras la realización de quemas controladas para mejora de pastos pirenaicos. Pastos 43, 44–53.

San Emeterio, L., Múgica, L., Ugarte, M.D., Goicoa, T., Canals, R.M., 2016. Sustainability of traditional pastoral fires in highlands under global change: effects on soil function and nutrient cycling. Agric. Ecosyst. Environ. 235, 155–163. https://doi.org/10.1016/j.agee.2016.10.009.

Santín, C., Doerr, S.H., 2016. Fire effects on soils: the human dimension. Philos. Trans. R. Soc. B Biol. Sci. 371, 28–34. https://doi.org/10.1098/rstb.2015.0171.

Shakesby, R.A., Bento, C.P.M., Ferreira, C.S.S., Ferreira, A.J.D., Stoof, C.R., Urbanek, E., Walsh, R.P.D., 2015. Impacts of prescribed fire on soil loss and soil quality: an assessment based on an experimentally-burned catchment in Central Portugal. Catena 128, 278–293. https://doi.org/10.1016/j.catena.2013.03.012.

Soria, R., Tortosa, A., Rodríguez-Berbel, N., Lucas-Borja, M.E., Ortega, R., Miralles, I., 2023. Short-term response of soil bacterial communities after prescribed fires in semi-arid Mediterranean forests. Fire 6. https://doi.org/10.3390/fire6040145.

Thiel, M., Benaiche, N., Martin, M., Franceschini, S., Van Oirbeek, R., Govaerts, B., 2023. Limpca: an R package for the linear modeling of high-dimensional designed data based on ASCA/APCA family of methods. J. Chemometr. 37, 1–16. https://doi.org/10.1002/cem.3482.

Turco, M., Bedia, J., Di Liberto, F., Fiorucci, P., Von Hardenberg, J., Koutsias, N., Llasat, M.C., Xystrakis, F., Provenzale, A., 2016. Decreasing fires in mediterranean Europe. PloS One 11. https://doi.org/10.1371/journal.pone.0150663.

Úbeda, X., Lorca, M., Outeiro, L.R., Bernia, S., Castellnou, M., 2005. Effects of prescribed fire on soil quality in Mediterranean grassland (Prades Mountains, north-East Spain). Int. J. Wildl. Fire 14, 379–384. https://doi.org/10.1071/WF05040.

Vance, E.D., Brookes, P.C., Jenkinson, D.S., 1987. An extraction method for measuring soil microbial biomass C. Soil Biol. Biochem. 19, 703–707. https://doi.org/10.1016/0038-0717(87)90052-6.

Vega, J.A., Fontúrbel, T., Merino, A., Fernández, C., Ferreiro, A., Jiménez, E., 2013. Testing the ability of visual indicators of soil burn severity to reflect changes in soil chemical and microbial properties in pine forests and shrubland. Plant and Soil 369, 73–91. https://doi.org/10.1007/s11104-012-1532-9.

Vélez, R., 2012. Introduction. in: Quemas prescritas realizadas por los EPRIF: método y aplicación, pp. 10–19. Chapter 1. Ed. Ministerio de Agricultura, Alimentación y Medio Ambiente. Madrid, Spain.

Wan, X., Li, C., Parikh, S.J., 2021. Chemical composition of soil-associated ash from the southern California Thomas fire and its potential inhalation risks to farmworkers. J. Environ. Manage. 278, 111570. https://doi.org/10.1016/j.jenvman.2020.111570.

Wang, Q., Zhou, F., Ding, X., Zhou, Z., Wang, C., Zhang, W., Li, L.K.Y., Lee, S.T., 2013. Microstructure and water-lubricated friction and wear properties of CrN(C) coatings with different carbon contents. Appl. Surf. Sci. 268, 579–587. https://doi.org/10.1016/j.apsusc.2013.01.027.

Weninger, T., Filipović, V., Mešić, M., Clothier, B., Filipović, L., 2019. Estimating the extent of fire induced soil water repellency in Mediterranean environment. Geoderma 338, 187–196. https://doi.org/10.1016/j.geoderma.2018.12.008.

Yang, S., Zheng, Q., Yang, Y., Yuan, M., Ma, X., Chiariello, N.R., Docherty, K.M., Field, C.B., Gutknecht, J.L.M., Hungate, B.A., Niboyet, A., Le Roux, X., Zhou, J., 2020. Fire affects the taxonomic and functional composition of soil microbial communities, with cascading effects on grassland ecosystem functioning. Glob. Chang. Biol. 26, 431–442. https://doi.org/10.1111/gcb.14852.