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# Effects of wildfire, torrential rainfall and straw mulching on the physicochemical soil properties in a Mediterranean forest

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A R T I C L E I N F O

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Intense storm
Nutrients
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Post-fire management
Soil chemical properties

A B S T R A C T

With the effects of fire, weather and post-fire management on soil properties having been studied mostly individually, there is little understanding of the combined effects of wildfire, heavy storm and straw mulching. In this study, we evaluated the changes in soil properties following a high-severity fire, post-fire soil treatment using straw mulch, and a torrential storm in a forest stand in north-western Portugal. The main physicochemical properties of the soil were evaluated in different soil conditions: a) burnt and untreated, b) burnt and mulched areas, and c) unburnt sites on three survey dates 1) soon after wildfire, 2) after the rainstorm, and 3) one year after wildfire. We found that soil water repellency strongly increased ephemerally immediately after wildfire, decrease after the storm and disappeared after one year. Fire reduced the soil organic matter (by 50%, on average), total nitrogen (by 85–90%) and available phosphorous (by about 45%) at both the mulched and untreated sites, and this effect was not changed by the ensuing rainfall. In comparison to the unburnt sites, the pH increased in the burnt soils, and the electrical conductivity decreased. Finally, the dynamics of the major cations and minor elements were affected differently by the wildfire and rainfall under the different soil conditions. One year after the fire, the most notable changes compared to the unburnt soil were detected in magnesium, potassium and almost all the minor elements. Based on these findings, instructions were given to the land managers (aim to control soil hydrophobicity following wildfire, supply more soil organic matter to the soil to avoid a decline in soil fertility, and use alternative actions to straw mulching to control the soil chemistry) that were aimed at more effective post-fire management in severely burnt areas in Mediterranean forests.

## 1. Introduction

Mediterranean forests are prone to wildfires, which are a common and essential component of the ecological system (Bento-Gonçalves et al., 2012; Guz and Kulakowski, 2020). Wildfires, besides removing vegetation, heavily impact the physical, chemical and biological properties of soils, mainly due to soil heating and secondarily to plant burning (Agbeshie et al., 2022; Pereira et al., 2018). Soil chemistry can undergo noticeable changes after a wildfire due to ash formation, the combustion of vegetation and organic matter (SOM) and its incorporation into the soil (Pereira et al., 2014a). These impacts depend not only on the wildfire disturbance, but also on the pre-fire vegetation, time of sampling, and post-fire management and weather conditions (Pereira et al., 2018).

The impact of forest fires on soils has been extensively studied in recent decades (Alexakis et al., 2021; Andreu et al., 1996; Certini, 2005; Úbeda and Outeiro, 2009). The main results point to important changes in soil pH, the electrical conductivity (EC), SOM, available nutrients and ions after both low- and medium-severity fires (Murphy et al., 2006; Pereira et al., 2014b, 2019; Prieto-Fernández et al., 2004). After high-severity fires, the impacts on soil are variable and may last for several years before the pre-fire soil properties recover. The effects of wildfires

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generally lead to increases in surface runoff and soil erosion, especially after heavy rainstorms (Moody et al., 2013; Shakesby and Doerr, 2006; Zema et al., 2020). Some of the main consequences of these intense rainfalls are flash floods and non-tolerable soil loss.

Intense rainfall is very frequent in the Mediterranean ecosystem, especially in autumn (Serrano-Notivoli et al., 2017). Studies on climate change have forecast that these storms will become more frequent and intense (Tramblay and Somot, 2018). Following the disturbance of a wildfire, the soil is very sensitive, and heavy rainfall can wash it away and significantly degrade its quality. These effects may particularly increase with the forest fires typical of summer (McGuire et al., 2018; Rengers et al., 2020; Tang et al., 2019). In recent years, forest fires have been displaced in time in many Mediterranean countries due to their 'deseasonalisation' (Úbeda et al., 2021). Moreover, the 'window of disturbance' caused by fire is becoming longer and causing a time displacement of fires from summer to autumn (Prosser and Williams, 1998). In autumn, wildfires are more likely to coincide with storms. This coincidence can influence the post-fire recovery of soil properties and vegetation (Francos et al., 2016, 2019), which can lead to further increases in soil erosion and degradation (Wall et al., 2020). On the other hand, rainwater may leach the elements and compounds that are incorporated into soil due to ash release (Pereira et al., 2018), and this effect may smooth out the soil changes caused by fire.

The impacts of rainfall and the consequent degradation of burnt soils increase on steeper slopes, and therefore specific post-fire management actions are required in these areas in order to limit soil erosion and degradation and protect the forest ecosystem (Francos et al., 2021, 2022; Parente et al., 2022). Post-fire management affects soil properties (Francos et al., 2018a; Lucas-Borja, 2021; Pereira et al., 2018), with the degree of their impact being mainly determined by the type, intensity and spatial extension of the management actions and also by fire severity (Francos et al., 2018a; García-Orenes et al., 2017). It is essential for land managers to identify which management action is most appropriate after a forest fire, where necessary, especially given the aggressive climate conditions in Mediterranean areas (Lucas-Borja, 2021). Post-fire actions should be adopted before the first rains begin in order to head off the high fragility of burnt soils and their vulnerability to erosion, which depends on the rain intensity (Francos et al., 2016; Santos et al., 2020). Mulching has been the most common post-fire management action on the hillslope scale (Zema, 2021), since this technique is not cheap, it is effective at controlling soil erosion and degradation (Prosdocimi et al., 2016). Agricultural residues, such as straw, are commonly used as mulching materials, due to their wide availability and low cost in rural areas. Several studies have evaluated the effects of straw mulching on soil, reaching generally satisfactory conclusions (Carrà et al., 2021, 2022; Keizer et al., 2018; Lopes et al., 2020; Prats et al., 2012, 2016a, 2016b; Vieira et al., 2018). However, limit efficacy of mulching in some environmental contexts have also been reported in some studies (Fernández et al., 2012; Lucas-Borja et al., 2018), such as decreases in water infiltration and increases in the soil's hydrological response. It is therefore essential to clarify whether mulching is effective or not at preventing soil erosion and limiting the degradation of its physicochemical properties in recently burnt areas, especially under the effects of intense rainfall.

Several studies have analysed the impacts of intense rainfall on erosion in soils not affected by fire and treated with straw mulch (Jourgholami et al., 2020; Li et al., 2021a; Parhizkar et al., 2021). Other studies have analysed the post-fire impacts of intense rainfall on soil properties, but little attention has been paid to burnt soils treated by straw mulching (Francos et al., 2016). Moreover, the effects of straw mulching on wildfire-affected soils have not been analysed in areas where heavy rainfall has occurred (Lucas-Borja et al., 2018, 2019, 2020; Vieira et al., 2018). Consequently, only a few studies are available on the combined effects of straw mulching and intense rainfall on burnt soils (Khan et al., 2016; Rahma et al., 2017). This lack of relevant information is due to the difficulty in studying two concurring extreme and natural events, such as wildfire and heavy rainfall, in forest areas treated by mulching. This is an important gap in the literature, since the sequence of forest fire–torrential rainfall–mulching has a complex effect on soil and therefore on the soil's hydrological response. Therefore, there is a need to evaluate the soil processes and dynamics occurring following disturbances involving sequences of high-severity fire, heavy rainfall and straw mulching.

To fill this gap, in this study, we evaluated the main physicochemical properties of soils burnt by a wildfire and treated with straw mulching to protect a steep soil in a forest in north-western Portugal following a torrential rainfall event. The specific objectives of the study were to evaluate the changes in soil properties between a) burnt and untreated, b) burnt and mulched areas in comparison to c) unburnt sites: 1) immediately following a wildfire; 2) following a rainstorm; and 3) one year after the wildfire. Finally, based on the findings of this study, we provided information to land managers with the aim of promoting more effective post-fire management in severely burnt areas in Mediterranean forests.

## 2. Material and methods

### 2.1. Study area

The study site was located in Braga (41°32′14.69″N, 8°22′54.67″W)– a steep wildland–urban interface in north-western Portugal (Fig. 1). The wildfire started in Leitões (Guimarães municipality) on 12 October 2017, arriving in Braga municipality on 15 October. This fire affected 1007 ha (967 ha of forest and 40 ha of shrub) in an area predominantly covered by eucalyptus (*Eucalyptus globulus* L.). In this area, there was also a significant cover of oak (*Quercus robur* L.), cork (*Quercus suber* L.) and kermes oak (*Quercus coccifera* L.) (Bento-Gonçalves et al., 2019). The severity of the fire was classed as high because 100% of the tree crowns was burnt (Úbeda et al., 2006) and gray/white ash covered the topsoil (Moreno and Oechel, 1989; Pereira et al., 2012), as surveyed immediately after the wildfire was extinguished. The understory vegetation was mainly composed of *Ulex parvifolius* Pourr., *Genista scorpius* L., *Pistacea lentiscus* L. and *Erica arborea* L. The geological substrate mainly consisted of mineral soils from granites and similar rocks, formed by human influences (azonal soils) (Sousa et al., 2004). The soil was classed as an Anthrosol Cumúlico Dístrico (FAO, 1998). The mean annual temperature of the study site is 13.8 °C and mean annual rainfall ranges between 1500 and 1600 mm according to the Braga meteorological airport gauging station (41°55′12″N, 8°42′49″W) classified as *Csb* by Kottek et al. (2006). Following the wildfire, there was a heavy rainfall event (Storm Ana), the characteristics of which have been reported in Vieira et al. (2019) and Bento-Gonçalves et al. (2019).

### 2.2. Experimental design

At the beginning of December 2017, just before the Storm Ana, three sites were selected in the study area and three plots, one per site, were identified. The first site (41°32′12″N, 8°22′33″W) was selected as a reference (hereafter designated the 'unburnt control', [UC]) in the burnt area. The second site (41°32′15″N, 8°22′57″W) was in the wildfire-affected area but had not received any soil treatment ('burnt and untreated' [BU]). The third site (41°32′15″N, 8°22′55″W) was in another burnt area and the topsoil had been mulched ('burnt and mulched', [BM]) at a dose of 2.5 Mg/ha of straw. We made sure that the selected areas had similar environmental characteristics (slope, aspect, lithology, vegetation and soil type) in order to prevent biases due to external factors in the comparisons. Each plot was 30 m long by 10 m wide, covering 300 $m^2$. The soil profiles of the plots had mean slopes of 20% and north-western aspects.

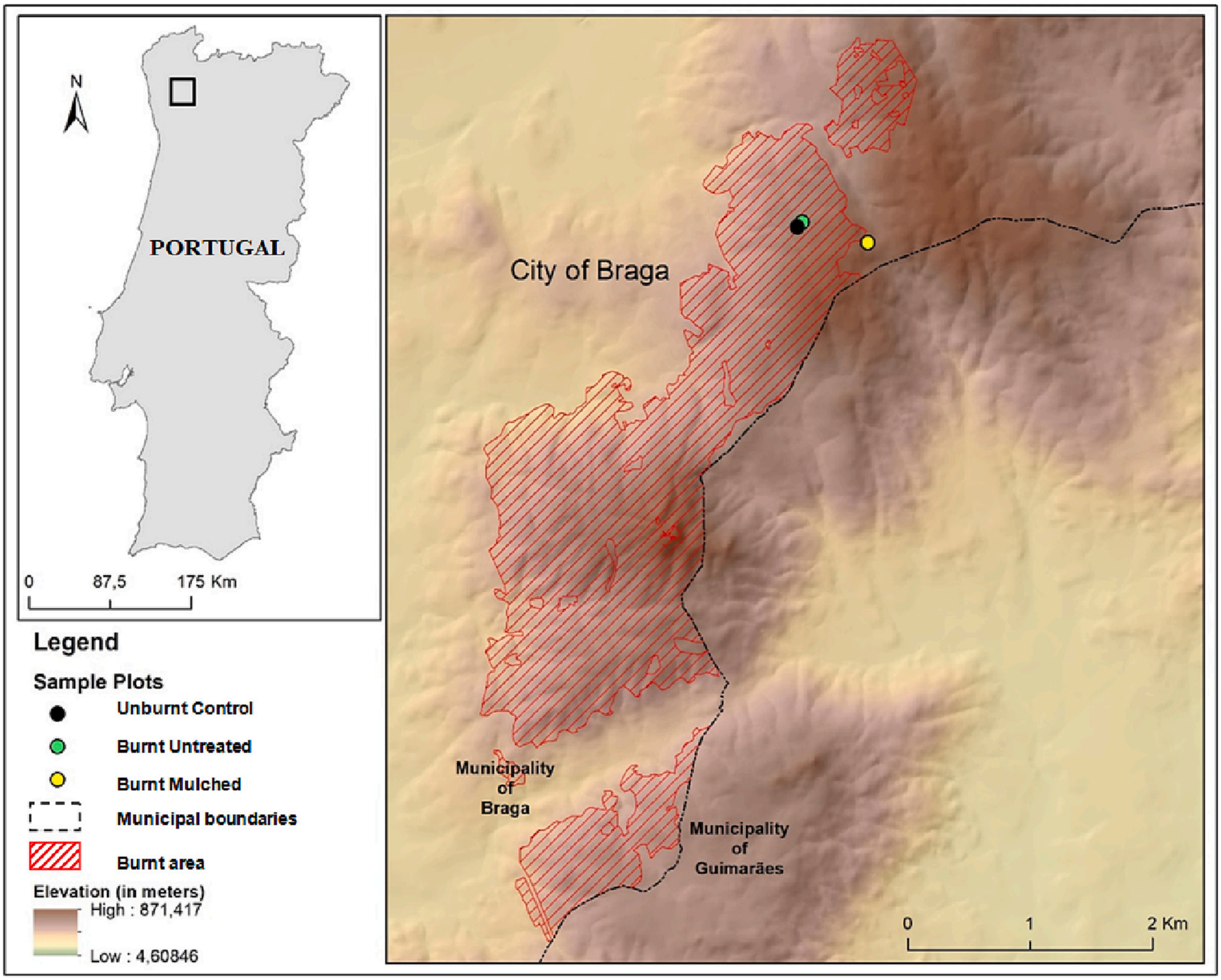


**Fig. 1.** Location of the study sites.

### 2.3. Soil sampling

The soil from each plot was sampled in a transect aligned perpendicularly to the slope at 2-m intervals. The soil was sampled on three dates: 1) December 2017, about two months after the wildfire ('AF', after fire) being the precipitation between the wildfire and first sampling characterized as light drizzle and a total of 63 mm in two months; 2) May 2018, five months after the torrential rainfall ('AR', after rainfall) and 3) October 2018, one year after the wildfire ('A1Y', after one year). On each sampling date, nine samples of soil were collected for laboratory analysis—this sample number is considered viable for representative statistical analysis. Each time, the soil was sampled to a depth of between 0 and 5 cm. The delay between the wildfire and the first sampling was due to the need to obtain authorization from the landowner (who was often difficult to identify).

### 2.4. Laboratory analysis

The soil samples were left for 7 days at room humidity and a constant temperature of 23 °C. The samples were sieved through 2-mm sieves and the fines fraction was analysed. The soil water repellency (SWR) was determined using the water drop penetration time method of Wessel (1988). In more detail, six drops of deionised water (total volume of 0.05 ml) were dripped on to the top of each sample, and the time required for each drop to completely penetrate the soil was recorded. Four categories of SWR level were distinguished by Doerr (1998) based on the time needed for the drops to penetrate soil—wettable or non-repellent soils (< 5 s), low repellency (6–60 s), strong repellency (61–600 s) and severe repellency (601–3600 s).

Concerning the chemical properties of soils, the total nitrogen (TN in advance) was determined using a Flash 112 Series analyser and the relevant data calculated using Eafer 300 software (Pereira et al., 2012). The soil organic matter (SOM) and inorganic carbon (IC) were measured using the loss-on-ignition method (Heiri et al., 2001). The soil pH [1:2.5] and EC were analysed with an extraction [1:2.5] of deionised water. The extractable element (potassium [K], sodium [Na], calcium [Ca] and magnesium [Mg]) contents were determined using an extraction [1:20] of ammonium acetate (Knudsen et al., 1982; Úbeda et al., 2009). The available phosphorus (P) was analysed following the Olsen–Gray method (Olsen et al., 1954). The soil extractable aluminium (Al), manganese (Mn), iron (Fe), zinc (Zn), boron (B), silicon (Si) and sulphur (S) were measured with an extraction [1:20] of ammonium acetate using a PerkinElmer ELAN-6000 spectrometer and a PerkinElmer Optima-3200 RL spectrometer. The ratio between the organic carbon (C) and TN (C:N ratio) was calculated.

### 2.5. Statistical analysis

Prior to statistical processing, the data normality and homogeneity were assessed using the Shapiro–Wilk and Levene's tests. When the data were normally distributed and their variances were homogenous, a two-way analysis of variance (ANOVA) test was applied, otherwise the

Kruskal–Wallis non-parametric test was used. In both cases, soil condition (UC, BU, BM) and sampling time were assumed as factors. The Tukey post-hoc test was applied to identify statistical differences at $p < 0.05$ among the sites and sampling dates. A redundancy analysis (RDA) were carried out to identify relationships between the soil properties (SWR, TN, IC, SOM, C:N, pH, EC, extractable Ca, Mg, Na, K, available P, extractable Al, Mn, Fe, Zn, B, Si and S). The significance of the RDA Monte Carlo analyses were carried out by means of a Monte Carlo Permutation test with 499 Permutations under the full model. The results obtained showed significant differences in AF ($p$-value = 0.0020; number of variables 19; F-ratio = 19.23), AR (p-value = 0.0020; number of variables 19; F-ratio = 10.85) and A1Y (p-value = 0.0020; number of variables 19; F-ratio = 20.62). All statistical analyses were implemented using SPSS 23.0 and CANOCO 4.5 for Windows.

## 3. Results

### 3.1. Soil water repellency

Differences in the SWR were significant for both factors (soil condition and sampling time) considered by ANOVA, and their interaction (soil condition x sampling time) (Table 1).

The information relating to the data shown in the different figures has been compiled in table format as supplementary material (see Table 1S). In the UC plots, there were no notable changes detected in SWR over time. By contrast, the SWR increased over time in the BU and BM soils (Fig. 2A). Regarding the differences among the different soil conditions, SWR was significantly lower in the UC plots compared to the BU and BM plots during AF (first soil sampling). Due to the decrease in SWR over time, this property was significantly lower in the burnt soils on the subsequent sampling dates. Following the rainfall (second sampling), the SWR was higher in the BM soils than the BU soils, with the reverse being detected A1Y (third sampling) (Fig. 2A). In the BU soils, about 10% of the samples had low SWR in the second sampling date (AR), with this frequency increasing to 35% A1Y period. On this sampling date, all the soil samples were wettable or slightly repellent at both the BU and BM sites, whereas the SWR remained from strong to severe in the UC soils (Fig. 2B).

### 3.2. Carbon, nitrogen and phosphorous

In terms of the different forms of C (IC and SOM), TN and the C:N ratio, significant differences were observed only in relation to the soil condition (Table 1). The SOM in the soil was always higher in the UC plots compared to the BU and BM plots during the first sampling period (AF). These properties were very similar in the BU and BM plots AR, although the BM soils had a slightly higher content compared to the BU plots. At A1Y, the maximum SOM content was again detected in the UC soils, whereas it was lower in the BU and BM soils (Fig. 3).

The IC in the UC sites was higher compared to the burnt sites immediately AF and A1Y, whereas the highest IC content AR was measured in the BM plots. The BU plots had the highest IC immediately AF and also during the third sampling (A1Y), when this property was comparable between the BU and BM soils. During second sampling period (AR), the maximum value of IC was measured in the BM soils and the minimum in the BU soils (Fig. 3).

The TN was significantly higher in the UC plots compared to the BU and BM plots AF. TN significantly differed in the UC compared to the BU soils AR, and was higher in the UC plots compared to the BU and BM plots A1Y (Fig. 3).

The C:N ratio was always higher in the UC soils compared to the BU and BM soils, and decreased in the BU and BM plots, the latter showing the lowest value A1Y (Fig. 3).

The content of available P showed a maximum in the UC plots on the second and third sampling dates, while these plots showed the minimum value on the first date. Comparing the burnt soils, P was the highest in the BM plots than in the BU soils AF, and the lowest in the BU plots AR (Fig. 3).

**Table 1**
Results of a two-way ANOVA of physicochemical soil properties under three soil conditions (UC, BU, BM) and on three sampling dates (AF, AR, A1Y) at a forest site (Braga, north-western Portugal).

| Soil properties | ANOVA factor | F-value | p-value |
|---|---|---|---|
| Soil Water Repellency | Sampling date | 47.45 | *** |
| | Soil condition | 10.64 | *** |
| | Sampling date x Soil condition | 14.41 | *** |
| Total Nitrogen | Sampling date | 0.95 | n.s. |
| | Soil condition | 28.12 | *** |
| | Sampling date x Soil condition | 0.56 | n.s. |
| Inorganic Carbon | Sampling date | 0.32 | n.s. |
| | Soil condition | 4.38 | * |
| | Sampling date x Soil condition | 2.01 | n.s. |
| Soil Organic Matter | Sampling date | 0.32 | n.s. |
| | Soil condition | 66.48 | *** |
| | Sampling date x Soil condition | 6.22 | n.s. |
| C/N | Sampling date | 0.04 | n.s. |
| | Soil condition | 4.83 | * |
| | Sampling date x Soil condition | 0.43 | n.s. |
| Available Phosphorous | Sampling date | 19.30 | *** |
| | Soil condition | 2.02 | n.s. |
| | Sampling date x Soil condition | 5.08 | *** |
| pH | Sampling date | 12.11 | *** |
| | Soil condition | 55.77 | *** |
| | Sampling date x Soil condition | 3.80 | ** |
| Electrical Conductivity | Sampling date | 9.86 | *** |
| | Soil condition | 46.65 | *** |
| | Sampling date x Soil condition | 3.83 | ** |
| Extractable Calcium | Sampling date | 4.81 | * |
| | Soil condition | 1.17 | n.s. |
| | Sampling date x Soil condition | 1.69 | n.s. |
| Extractable Magnesium | Sampling date | 169.07 | *** |
| | Soil condition | 116.51 | *** |
| | Sampling date x Soil condition | 47.25 | *** |
| Extractable Sodium | Sampling date | 5.12 | ** |
| | Soil condition | 4.96 | ** |
| | Sampling date x Soil condition | 0.73 | n.s. |
| Extractable Potassium | Sampling date | 121.34 | *** |
| | Soil condition | 110.98 | *** |
| | Sampling date x Soil condition | 33.92 | *** |
| Extractable Aluminium | Sampling date | 0.98 | n.s. |
| | Soil condition | 29.06 | *** |
| | Sampling date x Soil condition | 1.14 | n.s. |
| Extractable Manganese | Sampling date | 10.90 | *** |
| | Soil condition | 14.70 | *** |
| | Sampling date x Soil condition | 3.16 | * |
| Extractable Iron | Sampling date | 1.06 | n.s. |
| | Soil condition | 31.46 | *** |
| | Sampling date x Soil condition | 1.02 | n.s. |
| Extractable Zinc | Sampling date | 2.76 | n.s. |
| | Soil condition | 13.00 | *** |
| | Sampling date x Soil condition | 3.34 | * |
| Extractable Boron | Sampling date | 26.68 | *** |
| | Soil condition | 8.17 | *** |
| | Sampling date x Soil condition | 6.95 | *** |
| Extractable Silicon | Sampling date | 8.65 | *** |
| | Soil condition | 3.69 | * |
| | Sampling date x Soil condition | 4.40 | ** |
| Extractable Sulphur | Sampling date | 4.32 | * |
| | Soil condition | 118.04 | *** |
| | Sampling date x Soil condition | 4.18 | ** |

Note: * = $p < 0.05$; (**) $p < 0.01$; (***) $p < 0.001$; n.s. = not significant ($p < 0.05$).

### 3.3. pH, electrical conductivity and major ions

Significant differences were recorded in the soil pH, EC, Mg and K for the two ANOVA factors and their interaction. By contrast, sampling date was a significant factor in the differences in Ca, whereas the differences were significant in relation to sampling date and soil condition for Na, and sampling date and interaction sampling date x soil condition for P

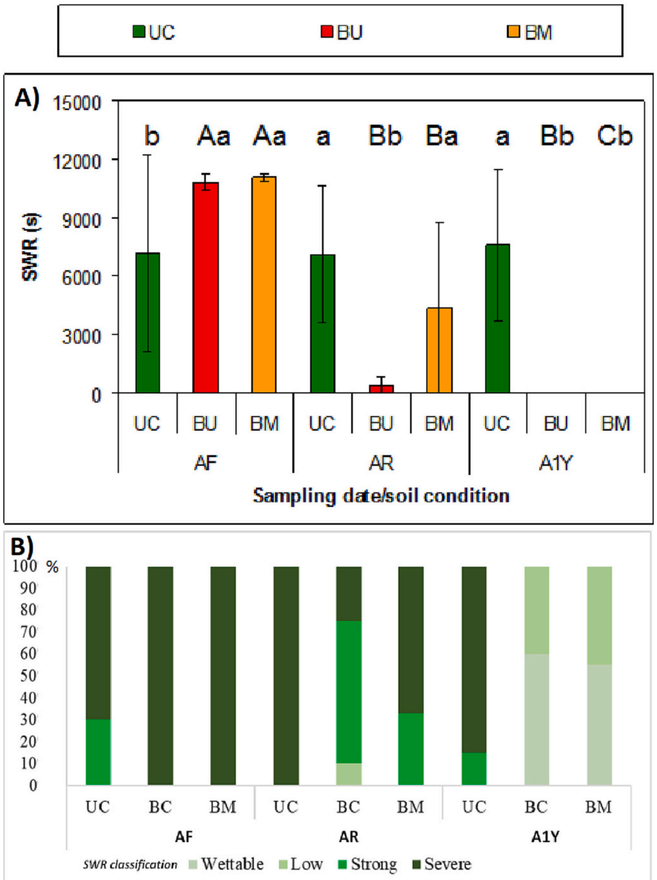


**Fig. 2.** A) Mean ± standard deviation ($n = 9$) of SWR and B) Relative frequency of SWR (in %) for SWR under three soil conditions (UC, BU, BM) and on three sampling dates (AF, AR, A1Y) at a forest site (Braga, north-western Portugal). Note: In A) different letters represent significant differences at $p < 0.05$ among the sampling dates (capital letters) and soil conditions (lowercase letters).

(Table 1).

The lowest pH and highest EC were measured in the UC soils on all the sampling dates. The pH was comparable between the BU and BM plots, in the range of 4.37 ± 0.18 for the BU plots A1Y to 5.30 ± 0.89 AF. At A1Y, the BM soils showed a higher pH than the BU soils. The EC was much lower and similar (except AR) in the BU and BM plots compared to the UC plots during AF, with this difference decreasing over time (Fig. 4).

The dynamics of the extractable cations differed both in time and among soil conditions. At AF, the Ca and Na contents were higher in the UC plots and similar between the BU and BM plots. Also AF, by contrast, Mg and K were the lowest in the UC soils and increase in the BU and BM soils. At AR, the UC plots showed the lowest Ca, Mg and K, and the highest Na. Still in AR, the BU plots evidenced the highest Mg and K, and the lowest Na, whereas, in the BM plots, Ca had the highest value. At A1Y, a gradient of UC < BU < BM was observed for Ca, Mg and K, while for Na, this gradient was reversed (UC > BU > BM) (Fig. 4).

### *3.4. Minor elements*

The soil Mn, B, Si and S were significantly different depending on the two ANOVA factors and their interaction. Soil condition determined significant differences in the Al and Fe contents, while differences in the soil Zn were significant in relation to soil condition factor, and the interaction soil condition x sampling date (Table 1).

At AF, the UC plots showed the highest contents of all the minor elements analysed for, except for Mn, which was the lowest among the soil conditions. Moreover, the contents of these elements were similar between the burnt soils. At AR, compared with the values measured in the UC plots, the contents of all elements decreased in the burnt plots, except for Mn. On the same date, the BM plots showed higher Al, Fe, B, S and Si, and lower Mn and Zn compared to the BU plots. At A1Y, the gradient UC < BU < BM was observed for Mn, while the gradient UC > BU > BM was observed for Al, Fe and S. At the same time, B and Si were the lowest in the UC soils and the highest in the BM soils, whereas Zn was the lowest in the BU soils and the highest in the UC soils (Fig. 5).

### *3.5. Multivariate analysis*

The RDA based on soil samples collected AF explained 61.3% of the variance in the original variables. Factors 1 and 2 of the RDA explained 49.8% and 11.5% of this variance, respectively (Fig. 6a). The SWR, pH, Mg, K and Mn values were grouped close to the scores of the burnt sites (BU and BM) on the chart reporting Factor 1 vs Factor 2. At these sites, the values of those soil properties were the highest among the three soil conditions. The other soil properties (TN, IC, SOM, C:N, EC, Ca, Na, P, Al, Fe, Zn, B, Si and S) were grouped close to the scores of the UC plots, characterized by the maximum values of those variables (Fig. 6a).

In the RDA related to the sampling AR (Fig. 6b), the first factor explained 48.7% of the total variance in the soil properties, while Factor 2 explained another 44.3%, for a total explained variance of 93%. A similar trend to that from the previous RDA was evident, although no clear groups of sites and soil properties could be discriminated. The variables associated with the burnt sites were pH, Ca, Mg, K, B and Si, while a higher influence of SWR, TN, IC, SOM, C:N, Na, Al, Fe, Zn and S on the UC plots could be observed (Fig. 6b).

Finally, the RDA associated with the samples collected A1Y (Fig. 6c) showed that Factors 1 and 2 explained 49.9% and 45.2% of the original variance, respectively (for a total of 95.1%). The same grouping of sites variables as in the RDA of the first sampling period was observed, with higher values of pH, Ca, Mg, K, P and Mn in the burnt areas (BU and BM) and higher SWR, TN, IC, SOM, C:N, EC, Na, Al, Fe, Zn, B, Si and S close to the scores of the unburnt plots (UC) (Fig. 6c).

## 4. Discussion

### *4.1. Soil water repellency*

Fires with different levels of severity exert notable influences on several soil properties, and these influences depend on several factors (Certini et al., 2021; Verma and Jayakumar, 2012). The most notable impact of fire is the increase in SWR immediately after burning, although this increase is generally ephemeral, since SWR quickly decreases over time (Mataix-Solera et al., 2013). The unburnt soil in this study showed a natural strong to severe repellency, which was increased by fire. However, this increase in SWR lasted until ash released by the fire was incorporated into the soil following the first rainfall (Pereira et al., 2014b). The areas affected by fire that were mulched showed the same SWR as the burnt and untreated soils prior to the strong rainfall. However, at the treated sites, the repellency levels were not reduced by the rainfall because, due to the presence of cover, the soil directly received rainwater without experiencing rapid changes in the variables, such as repellency (Pereira et al., 2018). By contrast, the burnt soil showed a sharp decrease in SWR, which may be due to the increase in soil water content that is well known to reduce hydrophobicity (DeBano, 1971, 2000). One year later, the SWR disappeared from the burnt areas, while remaining in the unburnt plots. These results suggest that fire does increase SWR in the short term, but that this effect is smoothed out by rainfall in untreated areas and is not limited by mulching treatments. These findings are in accordance with the findings of other studies, which have shown how increases in SWR due to fire are generally high, but ephemeral (e.g., Zavala et al., 2014; Alcañiz et al., 2018), and that mulching does not noticeably change SWR in the short term (Carrà et al.,

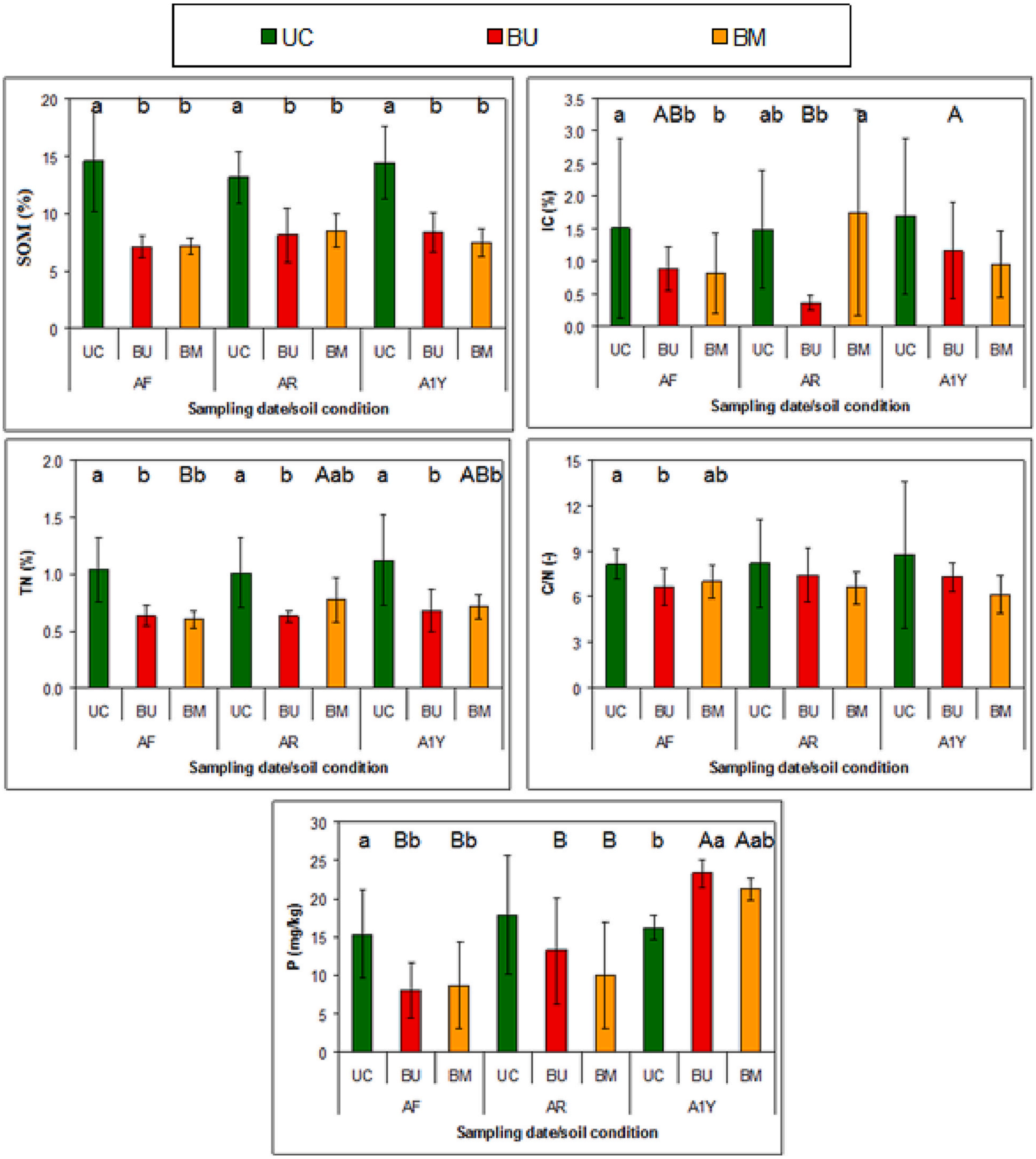


**Fig. 3.** Mean ± standard deviation ($n = 9$) for SOM, IC, TN, C:N ratio and available P under three soil conditions (UC, BU, BM) and on three sampling dates (AF, AR, A1Y) at a forest site (Braga, north-western Portugal). Note: Different letters represent significant differences at $p < 0.05$ among the sampling dates (capital letters) and soil conditions (lowercase letters).

2021, 2022; Prats et al., 2016a; Zavala et al., 2014).

### 4.2. Carbon, nitrogen and phosphorous

Compared with the unburnt soils, the SOM decreased after the fire in the burnt soils. Moreover, neither the rainfall nor mulching altered this decrease. Immediately after a wildfire, it is common to detect a reduction in soil SOM (Badía et al., 2014), and this reduction often take more than a year to recover after fire (Francos et al., 2016, 2019). The decrease in SOM detected in the burnt soils is due to the combustion and mineralisation, volatilisation and solubilisation of organic compounds (Agbeshie et al., 2022; Binkley and Fisher, 2019; Certini, 2005; Rodriguez-Cardona et al., 2020). These effects are due to the soil being heated to high temperatures, which reduces the amount and quality of SOM (Merino et al., 2018; Pereira et al., 2018). We also discovered that mulching did not affect the dynamics of the organic C after the fire. Lucas-Borja et al. (2021) reported a significant increase in soil SOM in mulched areas subjected to an intense storm immediately following a wildfire in comparison to untreated sites. Other authors have observed that, following forest fires, mulch preserves the SOM content, although in those studies, no notable reductions in SOM due to fire were detected (José et al., 2019). This absence of short-term differences between the

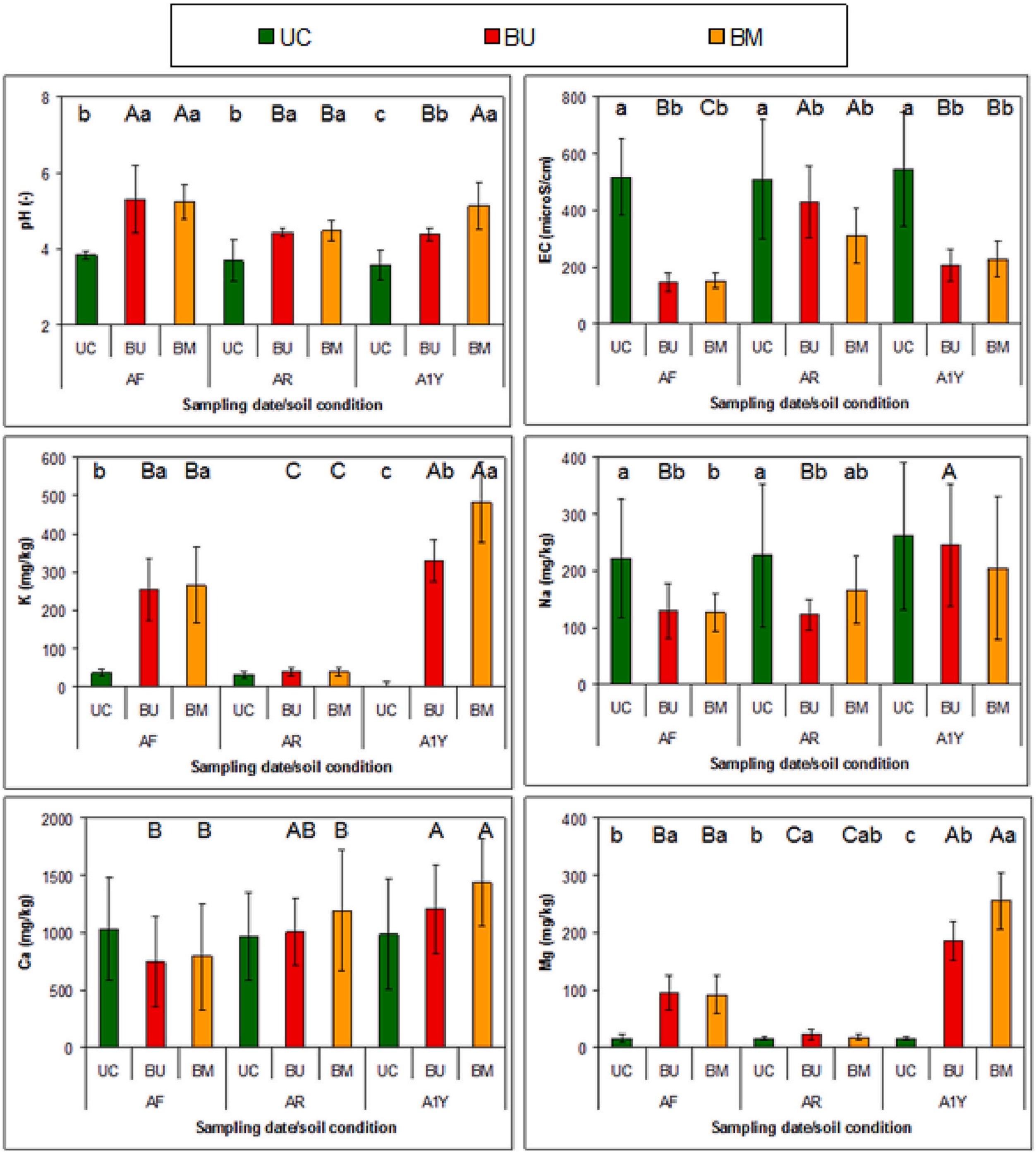


**Fig. 4.** Mean ± standard deviation ($n = 9$) for pH, EC and extractable cations (K, Na, Ca, Mg) under three soil conditions (UC, BU, BM) and on three sampling dates (AF, AR, A1Y) at a forest site (Braga, north-western Portugal). Note: Different letters represent significant differences at $p < 0.05$ among the sampling dates (capital letters) and soil conditions (lowercase letters).

burnt and untreated and burnt and mulched areas following fire leads us to deduce that the treatment does not negatively affect the soil stock of SOM. Some authors have reported increases in soil SOM in burnt and mulched areas, although these increases were observed several years after the fire and not in the short term (Lucas-Borja et al., 2022). Presumably, the short time that had elapsed between the wildfire and mulching treatment in our study was too little to allow a determination of the incorporation of the vegetal residues supplied through mulching and link this with the subsequent increase in SOM content in the soils (Bombino et al., 2019, 2021). However, according to Certini (2005), the recovery of SOM in the burnt areas, which started with the natural or artificial reintroduction of vegetation, was generally fast, and therefore we think that SOM depletion after fire is a reversible and ephemeral effect.

Concerning the dynamics of soil nutrients, wildfires disrupt their cycles by changing their forms, distributions and amounts (Alcañiz et al., 2018). In our study, as was observed for SOM, fire decreased the soil TN compared with the unburnt plots (Murphy et al., 2006), and the changes in this element were very similar to the SOM dynamics (i.e. no differences in TN between the BU and BM plots during the first post-fire period, either before or after the storm). The decrease in TN exhibited by the burnt sites (treated or not) may have been caused by the

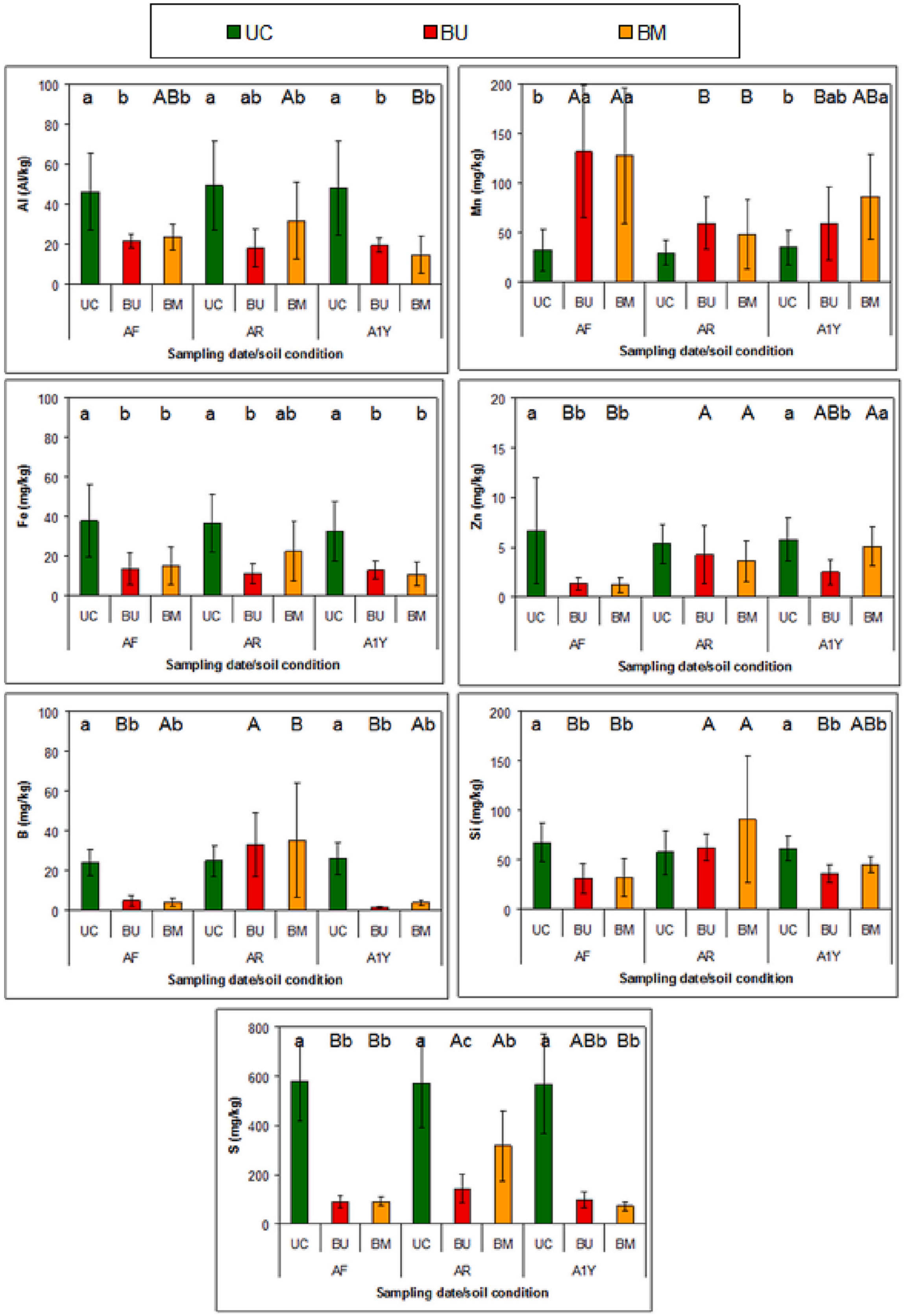


**Fig. 5.** Mean ± standard deviation (n = 9) for the minor extractable elements (Al, Mn, Fe, Zn, B, Si and S) under three soil conditions (UC, BU, BM) and on three sampling dates (AF, AR, A1Y) at a forest site (Braga, north-western Portugal). Note: Different letters represent significant differences at $p < 0.05$ among the sampling dates (capital letters) and soil conditions (lowercase letters).

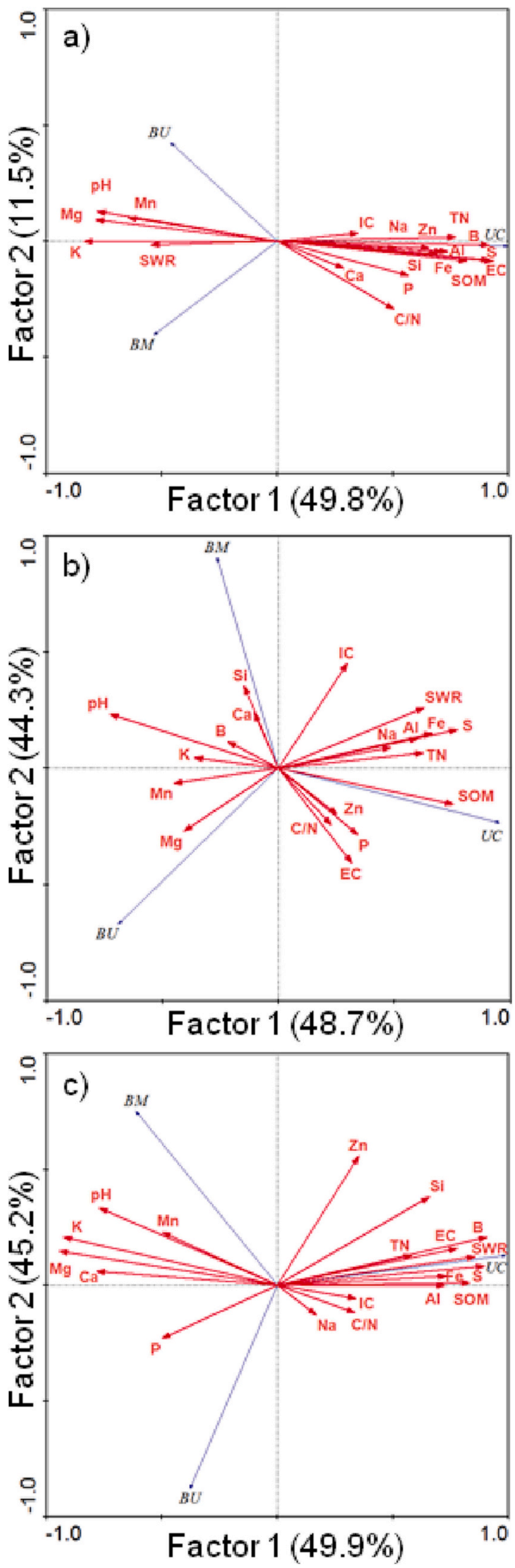


**Fig. 6.** Results of the RDA with the two relevant factors (Factors 1 and 2) of soil samples collected under three soil conditions (UC, BU, BM) and on three sampling dates (AF = a, AR = b, A1Y = c) at a forest site (Braga, northwestern Portugal).

volatilisation losses that occur during high-intensity fires, as explained by Binkley and Fisher (2019), Caldwell et al. (2002) and Certini (2005). These changes can last for up to one year after fire, with the pre-fire values recovering during the medium to long term, depending on the fire severity (Francos et al., 2018b, 2018c; Li et al., 2021b). The application of straw did not affect the amount of TN in the soil in the short term; this finding is in line with that of Thomas et al. (2000). Notable changes in soil TN are common when erosion due to heavy rainfall is significant (Gómez-Rey et al., 2013a, 2013b).

Due to the similar dynamics of C and N compounds, the C:N ratio was not notably altered between the burnt and unburnt soils. The fire ephemerally reduced the C:N ratio in the soil, and the pre-fire values were quickly restored a few months after the fire. Moreover, the differences in this ratio due to fire and post-fire management and over time (i.e., before and after the torrential rainfall and one year after the fire) were not significant. The slight differences detected between the unburnt areas and those affected by fire (with or without mulching) could be due to the different volatilisation and mineralisation of C and N (Baird et al., 1999), and to the immobilisation of N (which was not observed in the SOM) in recalcitrant heterocyclic structures (Francos et al., 2018c). Over time, the changes in the C and N contents were smoothed, presumably due to the quick growth of herbaceous vegetation (Úbeda et al., 2006).

Regarding the fate of P due to fire, rainfall and mulching, this element also experienced a decrease in the burnt areas soon after fire. However, while for C and N these changes lasted throughout the entire monitoring period, the soil P content increased over time at the burnt sites. The reductions in P measured soon after fire could be ascribed to volatilisation (Certini, 2005) due to the high temperatures, whereas the heavy rainfall did not have any effect. Fires result in an enrichment in available P (Certini, 2005; Serrasolsas and Khanna, 1995), although this increase is destined for a quick decline. The post-fire increases in soil P content after fire are mainly due to basic cations released by SOM, the formation of ash and its incorporation into the soil and the low pH that favours P retention (Alcañiz et al., 2018; Kennard and Gholz, 2001).

### 4.3. pH, electrical conductivity and major ions

The pH values were notably altered by the fire, increasing immediately afterwards. Such an increase is common in severely-burnt soils due to the denaturation of organic acids (Certini, 2005) and the increase in Na and K oxides, carbonates and hydroxides (Pereira et al., 2018; Ulery et al., 1993). The increase in soil pH by 1.5–2 points detected in this study is lower than what has been reported by Certini (2005) and Zavala et al. (2014), who found that, after a high-severity fire, the pH of the surface soil could increase by 3–5 points. Subsequent rainfall presumably diluted the pertinent compounds, although the pH did not recover to pre-fire values. However, this fire effect could be beneficial to soil health because it helps to decrease the acidity of the soil, which may be a limitation on forest plant growth. The pH levels were similar between the burnt plots, except for one year after the fire, when a small increase was detected in the mulched soils.

The EC significantly decreased in the burnt soils in comparison to the unburnt soils—an effect that was unexpected. As a matter of fact, several authors have reported increases in EC after fire (Alcañiz et al., 2020; Granged et al., 2011; Scharenbroch et al., 2012), due to the incorporation of ash (Fonseca et al., 2017; Scharenbroch et al., 2012; Úbeda and Outeiro, 2009) and the release of soluble ions during the combustion of SOM (Alcañiz et al., 2018; Certini, 2005). Our results are more in line with those of Hernández et al. (1997) and Naidu and Srivasuki (1994), who reported decrease in EC after fire. The reason for the immediate decrease in EC in the burnt plots in our study may have been the absence of the leaching effect of ions incorporated into ash, due to the lack of ash incorporation into the soil. In other words, the ash released by the fire did not infiltrate the soil due to be easily mobilized by wind (Francos et al., 2016). This result supports by Zavala et al. (2014), who explained that salts are quickly leached or transported by runoff and wind after burning. Moreover, after the subsequent rainfall throughout the year following the wildfire, the EC declined when the ash layer was depleted and the retained minerals were consumed; but no EC modifications were recorded as a result of the treatment, so decomposition of the EC has not yet taken place and there are little or no soil erosion marks after the fire (Díaz-Raviña et al., 2012).

Compared with the unburnt soils, the changes in soil extractable cations were significant for some dates and different between the BU and BM plots. Immediately following the fire, Mg and K increased, whereas Ca and Na decreased, regardless of the soil conditions (with or without treatment). The concentrations of some cations, such as Ca, Mg and K, can notably increase in soil solution immediately after burning (Certini, 2005; Khanna and Raison, 1986). The rainfall had an appreciable effect only on the Mg and K contents of the soil. By contrast, Ca and Na were not affected by leaching due to rainfall. One year after the fire, the burnt soils were enriched in all the major cations, and this may be because of the ash released during rainfall (Alcañiz et al., 2018; Cawson et al., 2012; Zavala et al., 2014). The dynamics of these ions in the soil depends on the chemical composition of the ash after wildfire; it commonly consists of Ca, Mg, K, Si and P, and occasionally Al, Mn, Fe and Zn (Etiegni and Campbell, 1991; Khanna et al., 1994; Zavala et al., 2014).

### *4.4. Minor elements*

As with the major cations, the dynamics of the minor elements differed among the soil conditions and was variable over time. In general, all elements, except for Mn, decreased immediately following the fire in the burnt soils, with no differences being found between the mulched and untreated plots. However, the burnt soils responded differently to each element in relation to the subsequent storm. In detail, we recorded increases in Zn, B, S and Si, small to no changes in S and Fe, and decreases in Mn and Al. The changes detected in the burnt and mulched soils were similar as those at the untreated sites for Zn and B, and different for the remaining elements. These contents remained substantially unchanged even one year after the fire, except for some decreases in Zn, B, Si and S found in the burnt untreated soils. In contrast, the mulched soils showed decreases in Al, Fe, B, Si and S, and increases in Mn and Zn. According to Parra et al. (1996), who investigated forest soils in pine stands of Central Spain, the total content of Mn can significantly increase after wildfires due to supply from ash, in line with our findings. Also the post-fire dynamics of Fe, Cu and Zn is presumably similar as Mn, since these elements move downwards very little (Certini, 2005). In general, fire plays an important action on availability of these elements, since heating modifies the composition of soil microbial communities (Perry et al., 1984). In this regard, ash plays a key role in the variety and amounts of micro-elements in the burnt soils. The chemical composition of ash deposited by wildfires commonly includes significant amounts of several micro-elements (Etiegni and Campbell, 1991; Khanna et al., 1994), and their actual proportion mainly depends mainly on the composition of fuel and burning temperatures during burning (Misra et al., 1993; Demeyer et al., 2001; Zavala et al., 2014). However, the fate of micro-elements, such as Fe, Mn, Cu, Zn, B and Mo, in the burnt soils is not well understood, because targeted studies on this aspect are lacking (Certini, 2005). This is an important literature gap, since noticeable increases or decreases in these micro-elements due to wildfire may lead to alteration of soil quality in delicate ecosystems, such as the Mediterranean forests. Leaching due to rainstorms may smooth these alterations, but, at the same time, can results in excessive percolation into the deeper soil layers and, therefore, in groundwater pollution. Therefore, further investigations should be suggested, in order to quantify the magnitude of these two effects (release of micro-elements due to wildfire and soil leaching after torrential precipitation).

### *4.5. Implications for post-fire management*

In this study, we have demonstrated how, the combination of wildfire, post-fire treatment using mulching and rainstorm results in notable changes in soil properties, with the effects differing among the studied soil conditions and evolving over time.

The main effect we found was the increase in SWR in burnt soils. This effect requires land managers to be cautious because, in hydrophobic soils, water infiltration is commonly reduced (Carrà et al., 2021; Zema, 2021; Zema et al., 2021a, 2021b). Decreased infiltration leads to higher surface runoff and soil erosion, with rates that can increase by one or two orders of magnitude (Cawson et al., 2012; Moody et al., 2013; Shakesby et al., 2000; Shakesby and Doerr, 2006). We demonstrated on one side that this effect is ephemeral, since the SWR decreased some weeks after the fire event, when the early rainfall began. On the other side, soil mulching was found not to be effective in reducing soil hydrophobicity. The absence of any monitoring of surface runoff and erosion, which could have verified the effectiveness of this treatment in controlling the soil's hydrological response, was a limitation of this study, although several studies have proven this beneficial effect under variable environmental conditions (Carrà et al., 2021, 2022; Lucas-Borja et al., 2018, 2019).

The decrease in SOM and nutrients (such as N and P) in the burnt areas may be an adverse effect of wildfire. A decline in organic C and N after fire, although not being irreversible, may reduce the soil functionality. This reduction consists of a decrease in soil fertility (which hampers post-fire vegetation regeneration) and structure (with lower aggregate stability and, therefore, decreased infiltration capacity for water and air in the root systems of plants) (DeBano, 2000; González-Pérez et al., 2004; Lucas-Borja et al., 2020, 2022). We also showed that straw mulching is not able to limit the decreases in SOM and N due to fire, which could be due to the long time required for the incorporation of vegetal residues in soil (Bombino et al., 2019, 2021). We suggest increasing the mulching dose (which may increase the amount of SOM supplied to the soil) using other vegetal residues with lower lignin contents (which might be more biodegradable compared with straw), and shredding the mulch material into finer pieces (to encourage faster mulch incorporation into the soil). These approaches need to be researched and tested in the field and with case studies to prove their effectiveness and balance pros and cons.

## 5. Conclusions

Immediately after a wildfire, straw mulching and torrential rain in a forest in north-western Portugal, SWR strongly ephemerally increased. Then, SWR sharply declined after the storm, and disappeared after one year. Mulching was not effective in limiting this increase. Fire reduced the SOM, N (which then remained stable over time) and P (increased throughout one year after the fire) at both the mulched and untreated sites (however, without notable differences), and this effect was not affected by the ensuing rainfall. In comparison with the unburnt sites, the pH increased and the EC decreased in the burnt soils, but without notable differences between the mulched and untreated areas. The rainfall did not change the pH values in the burnt soils, but notably increased the EC (particularly in the untreated soils). Wildfire and rainfall induced notable changes in Mg, K and almost all the minor elements compared to the unburnt soils, while the effects of mulching was much lower.

Overall, we can conclude that: 1) there is a need to control the increase in soil hydrophobicity after a wildfire, to avoid and probably a parallel increase in soil's hydrological response; 2) the notable decline in the contents of SOM and nutrients in the burnt soils may result in degradation of soil quality and fertility, which requires mitigating actions (e.g., addition of organic materials, to increase the C and N stocks in forest soils); and 3) soil mulching is not able to limit the water repellency and preserve fertility of soils, and therefore more effective post-fire management actions are needed.

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## CRediT authorship contribution statement

**Marcos Francos:** Conceptualization, Methodology, Data curation, Writing – original draft, Funding acquisition, Writing – review & editing. **António Vieira:** Conceptualization, Methodology, Supervision, Funding acquisition. **António Bento-Gonçalves:** Conceptualization, Methodology, Supervision, Funding acquisition. **Xavier Úbeda:** Conceptualization, Supervision, Writing – original draft, Formal analysis. **Demetrio Antonio Zema:** Methodology, Writing – original draft, Funding acquisition. **Manuel Esteban Lucas-Borja:** Methodology, Writing – original draft, Funding acquisition, Formal analysis.

## 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.

## Data availability

Data will be made available on request.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2023.106987.

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