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# Medium-term effects of wildfire severity on soil physical, chemical and biological properties in *Pinus halepensis* Mill. woodland (Southern Italy): an opportunity for invasive *Acacia saligna* colonization?

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ABSTRACT

Wildfire frequency and severity have greatly increased in Mediterranean areas in recent decades affecting ecosystems functioning due to alteration in the above- and below-ground process. This study aimed to investigate how wildfire severity, in the medium-term (2–5 years), impacts soil properties within a *Pinus halepensis* woodland located in the Special Area of Conservation (SAC) of the Natura 2000 network (IT9130006 - Pinewoods of the Ionian Arch). In 2021, four years after a large wildfire in 2017, the woodland still exhibited fire effects with evidence of low, medium or high burn severity in different sites (named LBS, MBS, HBS, respectively). In addition, we observed an area burnt at medium severity that was invaded by *Acacia saligna* (MBSA site), a fast-growing, highly invasive, drought-tolerant nitrogen-fixing plant, thus we also examined the combined effect of wildfire and *A. saligna* on the soil. We compared soil properties across burnt sites with a nearby unburnt site (control). Thickness, weight and organic carbon of litter (L) and fermentation (F) layers were measured, alongside physical, chemical and biological properties in the underlying mineral soil (0–10 cm). Our results show that wildfire destroyed the organic layers and these had not recovered four years after the wildfire (except for L-layer within LBS) with a consequent loss (~2 t C ha$^{-1}$) of this carbon pool. In mineral soil we identified fourfold increases in N mineralization and nitrification rates across all burnt sites, regardless of the burn severity and *A. saligna* presence, suggesting an alteration of N-cycle processes. On the contrary, total microbial biomass and soil respiration as well as most of the physical and chemical properties of the soil were comparable between the burnt and control soils. Principal Component Analysis (PCA) highlighted that burn severity affected soil variables with lower changes in LBS than in other burnt soils. Also, the HBS soil did not show greater negative impacts compared to MBS sites. This is probably due to the increased post-fire colonization by herbaceous plants in HBS, favoured by the complete destruction of trees. In this case, waiting for natural vegetation recovery can be a valid management option, but periodic monitoring of fire-soil-vegetation interaction mainly to avoid invasive species widespread is advocated.

## 1. Introduction

Wildfires are becoming more prevalent globally, through combined human- and climate-induced actions. Fires impact not only the above-ground biomass but also the belowground soil environment and its ability to maintain critical ecosystem functions and services (Adhikari and Hartemink, 2016; Baveye et al., 2016; Pereira et al., 2018a). However, the integrity of global soils is seriously threatened and is facing increasing challenges because of land-use change and climate change-induced events such as wildfires (IPCC, 2021). The latter are uncontrolled fires capable of burning for many days and covering many hectares (Tedim et al., 2018), but fires are also a natural ecological factor in ecosystems characterized by alternating wet and dry seasons (Keeley et al., 2011a; Turco et al., 2017). Mediterranean ecosystems

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have historically been fire-influenced, stimulating adaptive responses in plants (Carrión et al., 2003; Keeley, 2012; Keeley et al., 2011b). Despite this, the observed increase in wildfire frequency and severity seriously threatens the ecosystems' health. (Flannigan et al., 2000; Ganteaume et al., 2013; Mueller et al., 2020; Pausas et al., 2008).

Fire can affect the canopy, trunk, root systems and ecophysiology of plants (Ducrey et al., 1996; Niccoli et al., 2019, 2023) and at the same time it also alters a wide range of physical, chemical and biological soil characteristics depending on its severity (Certini, 2005). Fire severity refers to the impact of a fire on the environmental components, mainly vegetation, but also the soil (Ryan, 2002). Generally, fire severity is directly assessed by (i) observing in the field the degree of scorch on tree trunks, the amount of vegetation burned and the percentage of dead plants and/or indirectly (ii) evaluating the estimated effect of heat transfer from vegetation burning on the soil surface and the combustion of organic materials (Keeley, 2009). Typically, it is expressed according to a qualitative classification of low, medium and high in relation to the damage observed (Bento-Gonçalves et al., 2012; Neary et al., 2005). The degree of fire-impacted vegetation and soil change can also be assessed, along with field-specific survey methods (Composite Burn Index (CBI); Key and Benson, 2006), by means metrics derived from Earth Observation (Mallinis et al., 2018; Parks et al., 2014).

Several studies report the multiple effects of fire on soil physical and chemical properties such as consumption of organic matter (González-Pérez et al., 2004), increase in soil erosion (Gehring et al., 2019), alteration of nutrient cycling (Rodríguez et al., 2017), emission of $CO_2$ from soil (Xu et al., 2021) and potential loss of organic nitrogen through either volatilization or mineralization (Pellegrini et al., 2018). Fire can also negatively affect the soil microbial community (Rutigliano et al., 2013; Weber et al., 2014), influencing in turn ecosystem services as well as post-fire recovery (Dominati et al., 2010; Singh and Gupta, 2018). Microbes can be affected by fire, either as a direct result of heating or as an indirect effect of changes in the soil physical and chemical properties (Docherty et al., 2012; Fontúrbel et al., 2012). Microorganisms usually respond to environmental perturbations very quickly, therefore, they can be used as indicators of the effects of disturbance factors on the soil (Barreiro and Díaz-Raviña, 2021; Memoli et al., 2020). Ultimately, fire can cause changes in microbial biomass and activity determining alterations in microbial community structure and diversity (Lucas-Borja et al., 2019; Moya et al., 2022).

The reduction or complete elimination of vegetation cover and the alteration of the soil characteristics strongly influence post-fire recovery processes and secondary succession (Gimeno-García et al., 2007), and encourage the invasion of fast-growing alien species which may outcompete native species so reducing biodiversity. According to Kerns et al. (2006), the invasion of an environment can be triggered by fire (or other disturbances) as in the case of plants belonging to the genera *Acacia,* native to Australia including *Acacia saligna* (Labill.) H.L. Wendl. (=*Acacia cyanophylla* Lindl.) (Fabaceae). Historically, in the Mediterranean environments this species was introduced in coastal areas to stabilize dunes, for reforestation purposes and as windbreaks to protect coastal *Pinus* forests (Richardson et al., 2015). Then, the invasive plant spread without control across many Italian regions colonizing mainly the areas between the maquis and evergreen woodlands on fixed dunes (Del Vecchio et al., 2013), where it induces negative impacts on several Habitats of Community Interest *sensu* Habitat Directive 92/43/EEC (HD) (Lazzaro et al., 2020). This alien species is a highly fire-resistant plant with fast post-disturbance growth via sexual reproduction or acting as a resprouter and can also generate a persistent and rich seed bank in the soil (Gibson et al., 2011; Strydom et al., 2012). Although several data are reported in the literature on the negative effect of *A. saligna* on vegetation (Maitre et al., 2011; Tozzi et al., 2021), in the Mediterranean environment there are limited results on its effect on fire prone areas. This could be crucial for the management of burnt areas during post-fire succession, especially to conserve areas of naturalistic importance.

Even with the management efforts of the relevant authorities to prevent wildfires, the year 2017 is remembered as particularly critical for fire incidence. An extremely high number of fires were recorded worldwide, in Europe alone they burned about one million hectares of natural surfaces and it was the worst fire season in Italy since 2007, with a total burnt area of 161,987 ha (San-Miguel-Ayanz et al., 2017).

This study complements a larger research project monitoring the conservation status, *sensu* HD, of the Habitat of priority interest 2270* - Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*, within the Special Areas of Conservation (SAC) IT9130006 (Apulia, Southern Italy). Here, we analyzed relationships between vegetation and soil characteristics across a range of fire severity conditions. These conditions are considered likely to trigger multiple successional pathways, potentially leading to habitat change/degradation, including the invasion of alien species, as *A. saligna*. In the literature, there are numerous studies aimed at evaluating the short-term (<2 years post-fire) impacts on soil properties (Badía et al., 2014; Li et al., 2019), but studies on how fire severity affects soils in the medium (2–5 years post-fire) or in long (>5 years post-fire) term are scarce (Muñoz-Rojas et al., 2016).

The specific aims of this work were to assess: i) medium-term (4 years post-fire) effects of varying fire severity on soil physical, chemical and biological features in a *Pinus halepensis* Mill. woodland and ii) the effect of the alien species presence in burnt area as further ecological factor influencing soil properties. We hypothesize that: i) soil physical, chemical and biological properties have still not recovered to pre-fire conditions 4 years post-fire compared to the control site and ii) the *A. saligna* presence in burnt area, probably favoured by fire, may have an additive effect on the soil properties.

The multidisciplinary approach, i.e., soil analysis together with burn severity assessed by field and remote sensing techniques, is suggested as a methodology to be applied also in other contexts. Outputs from this research will improve our understanding of these complex interactions to identify timely management actions on this and other Natura 2000 sites.

## 2. Material and methods

### 2.1. Study area

The study was carried out in a *Pinus halepensis* Mill. subsp. *halepensis* (Pinaceae) woodland within the Special Area of Conservation (SAC) of the Natura 2000 network IT9130006 - Pinewoods of the Ionian Arch (Apulia region, Southern Italy) comprising both the most important remnants in Italy of spontaneous Aleppo pine forest and Mediterranean pines plantations (Francini, 1953). The study area (Fig. 1a) is located on the west bank of the Galaso river, it extends for approximately 36 ha at about 200 m from the coastline with an average slope of 5%.

Understory vegetation is comprised of a shrub-layer of evergreen species as *Cistus salviifolius* L., *Phillyrea angustifolia* L. and *Pistacia lentiscus* L. (Pazienza G., personal observations). Here, according to the SAC data form (https://natura2000.eea.europa.eu/natura2000/SDF.aspx?site=IT9130006; last accessed December 2022), the Priority Interest Habitat 2270* - Wooded dunes with *Pinus pinea* and/or *Pinus pinaster* is recorded. Moreover, for this Habitat, *A. saligna* was reported as an allochthonous species (http://vnr.unipg.it/habitat/cerca.do?formato=stampa&idSegnalazione=31), thus a possible threat.

The climate of the study area is typically Mediterranean (Fig. 2a; Table S1) with precipitation concentrated in autumn and winter (Table S1-B), with maximum values in October (72.2 mm) and November (88.2 mm) and minimum values in August (14.7 mm). The average annual temperature is 17.0 °C, the coldest month is January with an average temperature of 8.7 °C and the hottest month is August with an average temperature of 26.6 °C (Table S1-A). Thermo-pluviometric trend of the study area from wildfire event up to sampling time (2017–2021) is reported in Fig. 2b. According to the FAO Digital Soil World Map (DSMW; FAO, 2007), the soil of the study area is classified as *Gleyic Luvisol*.

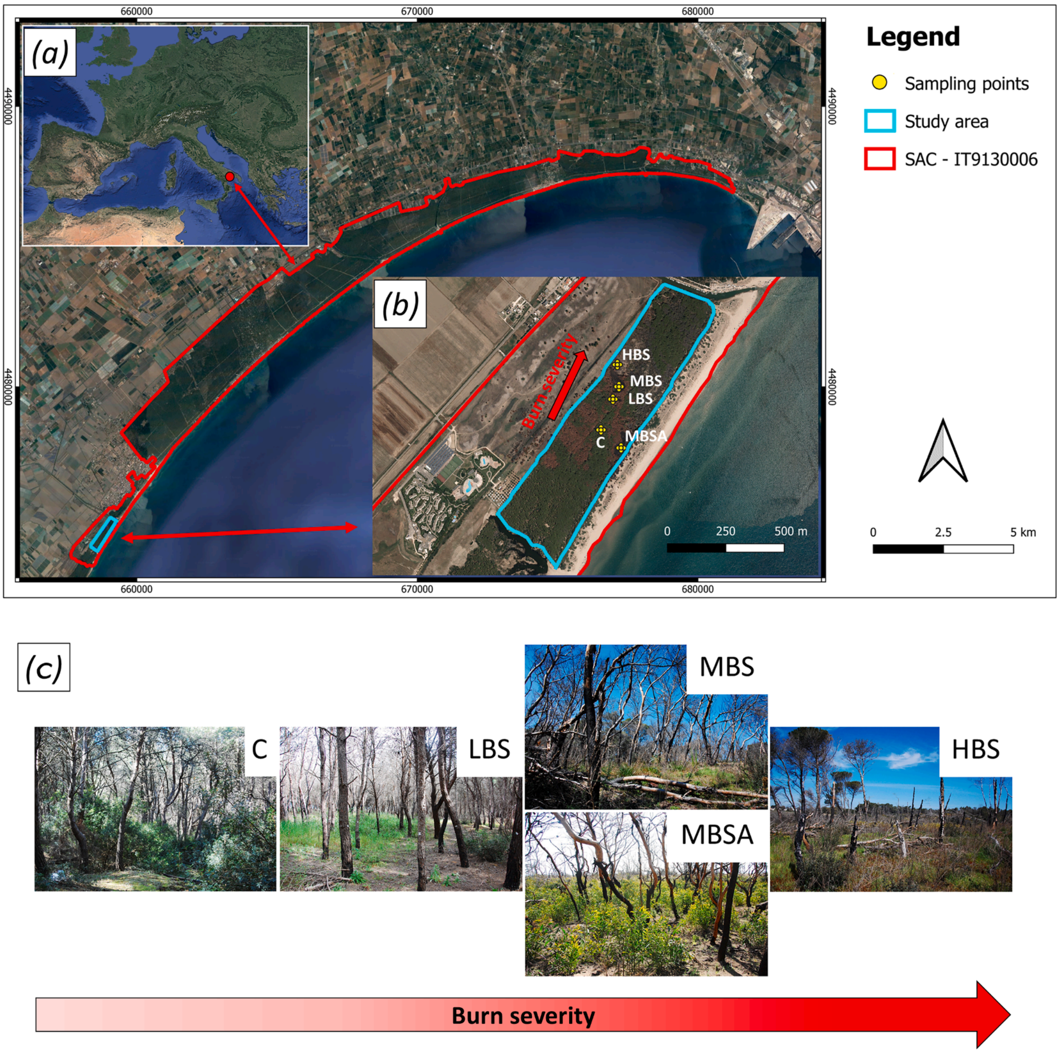



**Fig. 1.** a) Location map; b) map of the study area, the yellow symbols indicate the sampling sites along the burn severity gradient (C, control; LBS, low burn severity; MBS, medium burn severity; MBSA, medium burn severity with *A. saligna* invasion and HBS, high burn severity) and c) pictures of the sampling points (Photos Marfella L. and Pazienza G), the bottom red arrow indicates increasing burn severity.

The study area is affected by the recurrence of wildfires and by the colonization of the invasive alien species *A. saligna*, which widespread in many Italian regions (Galasso et al., 2018), but reported in this SAC for the first time in the present work.

We focus on a large wildfire event which happened on 2 July 2017, with firefighting operations concluded the day after, affecting a total of 13.8 ha and resulting in different burn severities across the study area (Fig. 1b). No salvage logging was performed in the burnt sites and deadwood (woody debris, downed logs and snags) was left in place.

### 2.2. Sampling design and burn severity assessment

A synchronic soil sampling was carried out in March 2021, four years after the 2017 wildfire, across five sites selected along three burn severity classes (low, medium, high) transect (Fig. 1b, c); 1. unburnt (control C; 22.2 ha), 2. low burn severity (LBS; 6.3 ha), 3. medium burn severity (MBS; 3.1 ha); 4. medium burn severity with the *A. saligna* invasion (MBSA; 3.2 ha) and 5. high burn severity (HBS; 1.2 ha).

The control site is 50–60 years overstocked pine plantation (830 living trees ha$^{-1}$, average height of 13.5 m). The medium burn severity site with *A. saligna* plant invasion was identified upon field observations; considering the wide distribution and abundance of young suckers and root suckers, the alien species was already present in this site before the fire even though at a lower density of individuals. On the contrary, no signs of *A. saligna* presence were found in *P. halepensis* stand identified as control site or in burnt sites other than MBSA. The definition of burn severity classes was based on field visual assessment of the state of the vegetation. Since this study was not initiated right after the disturbance, we were not able to use the CBI method; therefore, broad reference to this method was made by considering only the intermediate trees (5–20 m height) vegetation stratum (average vegetation height was around 20 m). Sites assigned to the low severity class were characterized by only light burn signs on the bark at the base of the trees, while those assigned to the medium severity class showed extensive burn signs on tree bark and crown desiccation, finally those assigned to the high severity class had dead trees either still standing or present as snags (Fig. 1c).

To support the assessment carried out in the field, burn severity was also assessed by means of one of the most used metrics of burn severity, the satellite remote sensing bitemporal spectral index delta (or differenced) Normalized Burn Ratio (dNBR = prefireNBR - postfireNBR) (Key and Benson, 2006, Mallinis et al., 2018). This is calibrated through dNBR values retrieved from the unburnt area surrounding the fire perimeter and in the same vegetation type (dNBR offset) to mitigate the inter-annual variations in plant phenology and moisture content (Parks et al., 2014, Miller and Thode, 2007; Meddens et al., 2016), according to the following equations:

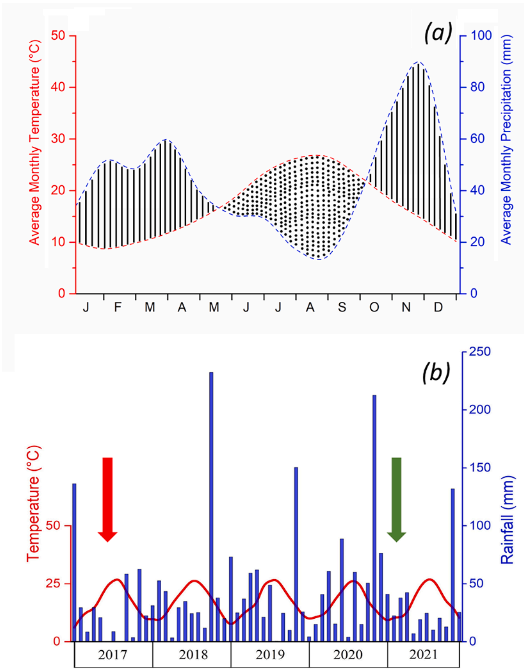



**Fig. 2.** a) Thermo-pluviometric diagram 2010–2021 for Ginosa Marina meteorological station (data from Civil Protection Section Apulia Region; last accessed September 2022) with the wet and dry periods indicated by vertical lines and punctuation, respectively; b) Values of average monthly temperatures (°C, red curve) and total monthly rainfall (mm, blue bars) of the study area during the 2017–2021, with the red and green arrow indicating wildfire and sampling time, respectively.

$$NBR = \frac{NIR - SWIR2}{NIR + SWIR2} \quad (1)$$

where NIR stands for near infrared and SWIR2 for short-wave infrared.

$$calibrated_{dNBR} = (prefireNBR - postfireNBR) - dNBRoffset \quad (2)$$

where (prefireNBR – postfireNBR) is the satellite remote sensing bitemporal spectral index delta normalized burn ratio (dNBR) and dNBRoffset is the normalized burn ratio retrieved from the unburnt area surrounding the fire perimeter and in the same vegetation type.

Sentinel-2 images (MultiSpectral Instrument (MSI) onboard Sentinel-2A (S2A) and Sentinel-2B (S2B) satellites) 2016 (Level 1C), 2017, 2018, 2019 and 2020 (Level 1A) images with similar acquisition timing (July) were used to compute the calibrated dNBR, i.e., dNBR_2016-2017 (difference between pre and immediately post disturbance), dNBR_2016-2018 (difference between pre and one vegetation season after disturbance), dNBR_2016-2019 (difference between pre and two vegetation seasons after disturbance) and dNBR_2016-2020 (difference between pre and three vegetation seasons after disturbance and prior to field sampling). Since a high NBR value indicates healthy vegetation while a low value indicates bare ground and recently burnt areas, although dNBR ranges can vary from case to case, it can be assumed that the calibrated dNBR takes higher values in reduced vegetation biomass/vitality conditions. The 2016 image was pre-processed to correct from Top-of-Atmosphere (TOA) to Bottom-of-Atmosphere (BOA) reflectance products obtained with the Sen2Cor 2.8.0 processor (Mueller-Wilm et al., 2019). For the computation of NBR, Sentinel-2 Band 8 and Band 12 were used (Mallinis et al., 2018). Image analysis was performed using the Sentinel Application Platform (SNAP) and maps were output via the open-source software QGIS Desktop 3.12.1.

### 2.3. Soil data collection

In each site, soil was collected in five field replicates taken at the center and at the four cardinal points in a circular sampling unit (13 m radius; Fig. S1a). Samples of the litter (L-layer), fermentation (F-layer) layers (where found) and of the 0–10 cm mineral soil (S-layer) underneath were collected, in squares of 40 × 40 cm size (Fig. S1b). In each square, L- and F- layers were entirely removed (after thickness measurement) and placed in paper bags, and five soil samples were collected from the mineral soil underneath with a cylindrical sampler (10 cm height and 6 cm diameter), placed into polythene bags and mixed to obtain a homogeneous sample (Soil Survey Staff, 2014a). Sampling was performed at 10 cm (after Mehdi et al., 2012; Memoli et al., 2021; Stinca et al., 2020) to consider the possible vertical transport of partially burnt organic matter that may occur post-fire (Knicker, 2011; Velasco-Molina et al., 2016) so stimulating microbial activity even at this depth. Thus, each field replica was in turn composed of five soil cores for a total of 125 cores. An intact soil core was also sampled near each 40x40 cm-square to determine soil water holding capacity, bulk density and porosity. All soils samples, stored in field refrigerator, were transported to the laboratory. L- and F- layers samples were air-dried to measure weight (LW and FW) and organic carbon content (L-$C_{org}$ and F-$C_{org}$). Soil samples were sieved (2 mm mesh size) to exclude coarse fragments and larger plant roots (Soil Survey Staff, 2014b). Two soil subsamples were taken: one air-dried for most chemical analyses and the other stored at 4 °C for biological analyses as well as for ammonium and nitrate content determination.

### 2.4. Soil analysis

Water content (WC) and water holding capacity (WHC) were measured using the gravimetric method (Allen, 1989). Bulk density (BD) was assessed on undisturbed soil cores of known volume dried for 48 h at 105 °C and porosity (Po) was calculated from bulk density (Soil Survey Staff, 2014a). Soil pH was determined on a soil/deionized (DI) water suspension (1:2.5 ratio) through the potentiometric method (HANNA Instruments, HI8424). Electrical conductivity (EC) was assessed by using a digital conductivity meter (COND 51$^{+}$) on aqueous soil extracts (1:2 soil/DI water ratio). Cation exchange capacity (CEC) was performed by treating the soil with barium chloride and triethanolamine solution at pH 8.2 and then measured by complexometric titration (Soil Survey Staff, 2014a). Organic carbon content ($C_{org}$) of dried L, F and S-layers samples was measured by wet oxidation with a 0.33 M $K_2Cr_2O_7$ followed by back titration with 0.2 M Fe(II)$SO_4$ solution (Springer and Klee, 1954). Total nitrogen ($N_{tot}$) was determined on dry soil, using a CHN elemental analyser (LECO TruSpec® Micro) and after the C/N ratio was calculated from $C_{org}$ and $N_{tot}$ contents. Mineral nitrogen content (as ammonium and nitrate) was evaluated by the potentiometric method using ion-selective electrodes specific for $NH_4^+$-N (ORION, Mod.9512BNWP) and $NO_3^-$-N (ORION, Mod.9707BNWP), after extraction with 0.5 M $K_2SO_4$ in 1:5 soil/extractant ratio (Castaldi et al., 2011). By means of $NH_4^+$-N and $NO_3^-$-N measured on soil samples incubated aerobically (60% of field capacity, at 25 °C, in the dark) at the beginning and end of a 14-day period, the nitrogen mineralization (Min) and nitrification (Nit) were calculated (Grilli et al., 2021). Microbial biomass carbon ($C_{mic}$) was estimated by the fumigation–extraction method; both chloroform-fumigated and unfumigated soil samples (5 g of soil each) were extracted with 0.5 M $K_2SO_4$ (1:4 ratio) and the $C_{org}$ content in each extract was determined by chemical digestion with a 0.33 N $K_2Cr_2O_4$ solution followed by back titration with 0.033 N Fe(II)$SO_4$ solution (Vance et al., 1987). From the $C_{org}$ content in fumigated and non-fumigated samples, the microbial carbon ($C_{mic}$) was calculated; the $C_{org}$ in non-fumigated soil samples corresponds to the extractable organic carbon ($C_{ext}$). According to ISO 16078 (ISO, 2002), total microbial activity was evaluated by measuring the soil potential respiration (R; mg $CO_2$-C $kg^{-1}$ d.w. $d^{-1}$) during an incubation of 13 days in standard conditions (25 °C, 55% of WHC, in the dark). The $CO_2$ evolved from soil samples, which bound to a 0.1 NaOH trap, was measured at 3, 7, 10, and 14 days titrating the NaOH excess with 0.05 M HCl. During two consecutive incubation periods, soil samples were re-incubating after new NaOH addition to ensure to bind all evolved $CO_2$. The respiration of the last incubation period corresponds to basal respiration. Mineralizable carbon ($C_{min}$) was also calculated fitting cumulative $CO_2$ evolved *vs.* incubation time, using a first-order pool kinetics model (Riffaldi et al., 1996), with the following equation:

$$C = C_0 \times \left(1 - e^{-k \times t}\right) \tag{3}$$

where "C" is the cumulated mineralized carbon; "t" is the incubation time in days; "$C_0$" is the asymptotic maximum quantity of $CO_2$ evolved from the samples (henceforth called $C_{min}$; g $CO_2$-C $kg^{-1}$ d.w.); and "k" is the mineralization rate constant in $days^{-1}$. Lastly, three microbial indexes from $C_{org}$, $C_{mic}$, basal respiration and $C_{min}$ data were calculated: (i) microbial percentage of total $C_{org}$ ($C_{mic}$%$C_{org}$; Anderson and Domsch, 1993), (ii) quotient of mineralization (qM; $CO_2$-C % $C_{org}$) derived from the asymptotic maximum quantity of $CO_2$-C ($C_0$ of the equation (3) or $C_{min}$) and $C_{org}$ (Dommergues, 1960) and (iii) metabolic quotient ($qCO_2$; mg $CO_2$-C % $C_{mic}$ $d^{-1}$), calculated from basal respiration and $C_{mic}$, after Anderson and Domsch (1993).

### 2.5. Statistical analysis

Descriptive statistics (means and standard deviations) for all variables were calculated for each layer (L, F, S-layers). The normality test (Shapiro-Wilk) and the equal variance test (Levene) were applied to test for normal distribution of the data before performing parametric tests; the data were transformed by $\log_{10}$ (except for pH) when not normally distributed (Sokal and Rohlf, 2011). For each variable, one-way ANOVA (followed, if required, by Student-Newman-Keuls post-hoc test) was applied in order to determine the significance of differences ($p < 0.05$) among the five study sites. Principal Component Analysis (PCA) was used for multivariate analysis by applying it on a matrix composed of 25 sites and 25 variables. Cluster Analysis (i.e., agglomerative hierarchical clustering, AHC) was also performed to check the similarity between sites, using Euclidean distance and Ward's method on the matrix of site's factor scores obtained from PCA. Pearson correlation coefficient ($r$) was used to assay the correlation among all considered variables i.e., the 25 soil variables, the PCA axes, the burn severity (BS) based on the field visual assessment expressed on a scale ranging from 0 to 3 (0: control; 1: LBS; 2: MBS and MBSA; 3: HBS), as well as the four differenced normalized burn ratio (dNBR) calculated indexes. PCA, Cluster Analysis and Pearson correlation were performed by XLSTAT, all other statistical analyses were carried out using the software program SigmaPlot 12.5 (Sigma Stat, Jandel Scientific).

## 3. Results

### 3.1. Burn severity assessment using remote sensing

Remote sensing analysis confirmed our field-based assessment of burn severity and provided evidence of lagged disturbance consequences on the vegetation cover across the sites, again supporting our characterization of LBS, MBS and HBS sites (Fig. 3a, b, c). Indeed, in 2017 immediately after the fire, vegetation damage was initially evident only in HBS site which showed higher values of dNBR_2016-2017 compared to all other sites ($p < 0.001$) and this trend remained similar one vegetation season after the fire (Fig. 3c). Subsequently, two and three vegetation seasons after the fire event, compared to pre-fire (dNBR_2016-2019 and dNBR_2016-2020; Fig. 3c), the effect of the disturbance on the vegetation had extended over a wider area also including MBS, MBSA, and at a lower extent the LBS site (Fig. 3b). Calibrated dNBR_2016-2020 was significantly ($p < 0.001$) higher in MBS, MBSA and HBS than in LBS and control site (Fig. 3c).

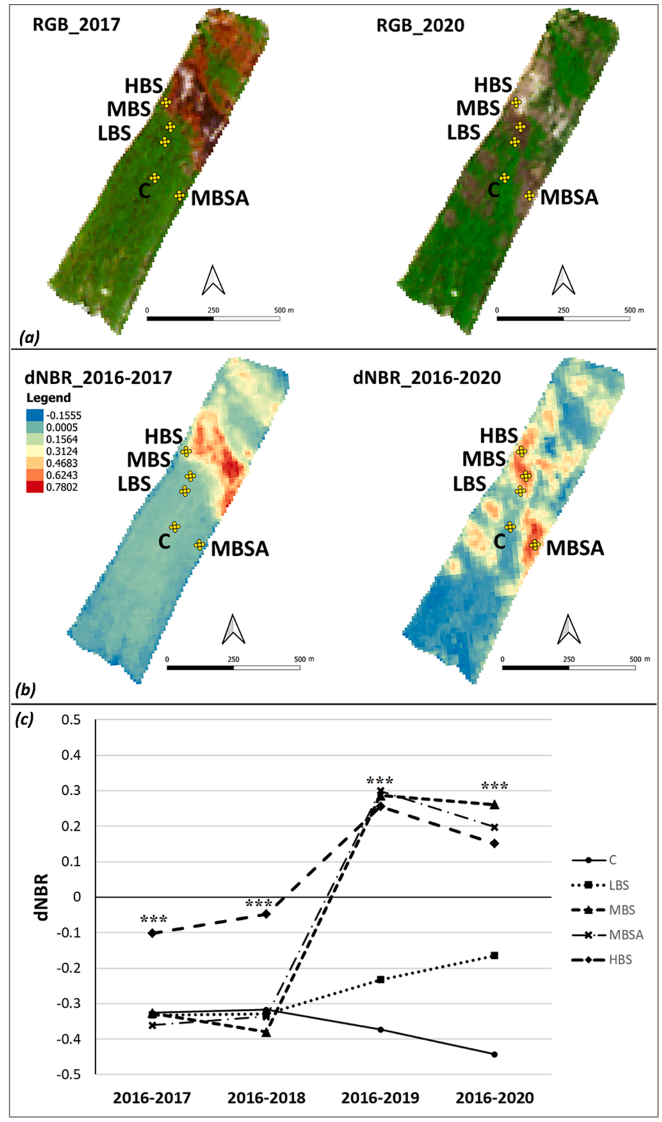



**Fig. 3.** a) Sentinel 2 natural color maps (RGB_2017 and RGB_2020) and b) calibrated dNBR maps 2016–2017 and 2016–2020, determined as described in the equation (2). In both a) and b) maps the yellow symbols indicate sampling points. c) Average dNBR values (y axis) at each sampling site for the 2016–2017, 2016–2018, 2016–2019 and 2016–2020 (x axis). The result of the one-way ANOVA is also reported (*** = $p < 0.001$) between the sites for each considered time.

### 3.2. Fire effects on organic (L and F) layers

Four years after the disturbance the different burn severity sites were still identifiable in terms of soil organic layer characteristics. The L- and F- layers (each about 0.5 cm thickness in control site) were completely absent in MBS, MBSA and HBS sites (Table 1A). In the LBS site, only the L-layer was recovered (thickness ~ 1 cm) with no significant change in weight compared to control, but with a significant decrease in the $C_{org}$ content expressed in g $kg^{-1}$ d.w. (Table 1A). On average, the MBS, MBSA and HBS sites lost $C_{org}$ from the organic layers to the extent of 2.62 t $C_{org}$ $ha^{-1}$ (Table 1B), assuming soil $C_{org}$ content in these sites, before wildfire, was equivalent to that found in the L- and F- layers of the undisturbed control. This amount corresponds to a loss of 19.6 t of $C_{org}$ for the combined MBS, MSBA and HBS burnt areas (total surface of 7.5 ha). On the contrary, within the L-layer at LBS site, $C_{org}$ content had recovered (~140 g $C_{org}$ $m^{-2}$) and was comparable to control site, but no F-layer had developed. Therefore, in LBS, we estimate a loss in $C_{org}$ of ~ 1.28 t

**Table 1**

*A)* Mean values (±standard deviations; n = 5) of the weight and organic carbon content ($C_{org}$) in L and F layers for control (C) and burnt sites (LBS, MBS, MBSA and HBS). Significant differences between the sites by the one-way ANOVA (*** = $p < 0.001$) are reported in the last column. Superscripts letters indicate statistically significant ($p \leq 0.05$) differences among sites assayed by Student-Newman-Keuls test. For plot labels see Fig. 1. *B)* C losses calculated for burnt sites.

| *A)* | | Control | LBS | MBS | MBSA | HBS | one-way ANOVA |
|---|---|---|---|---|---|---|---|
| L-layer | Weight (g $m^{-2}$) | 365 (±137)$^a$ | 423 (±134)$^a$ | $0^b$ | $0^b$ | $0^b$ | *** |
| | $C_{org}$ (g $kg^{-1}$d.w.) | 371 (±30.6)$^a$ | 336 (±29.5)$^b$ | $0^c$ | $0^c$ | $0^c$ | *** |
| | $C_{org}$ (g $m^{-2}$) | 134 (±45.6)$^a$ | 140 (±31.9)$^a$ | $0^b$ | $0^b$ | $0^b$ | *** |
| F-layer | Weight (g $m^{-2}$) | 433 (±139)$^a$ | $0^b$ | $0^b$ | $0^b$ | $0^b$ | *** |
| | $C_{org}$ (g $kg^{-1}$d.w.) | 290 (±37.6)$^a$ | $0^b$ | $0^b$ | $0^b$ | $0^b$ | *** |
| | $C_{org}$ (g $m^{-2}$) | 128 (±57.4)$^a$ | $0^b$ | $0^b$ | $0^b$ | $0^b$ | *** |
| *B)* | | | | | | | |
| **Calculated** | t $ha^{-1}$ | | 1.28 | 2.62 | 2.62 | 2.62 | |
| **C loss**[(1)] | t[(2)] | | 8.06 | 8.12 | 8.38 | 3.14 | |

[(1)] Calculated by assuming that burnt sites should show the same $C_{org}$ content of control in L and F layers if wildfire was not occurred;
[(2)] Calculated based on total surface of LBS (6.3 ha), MBS (3.1 ha), MBSA (3.2 ha) and HBS (1.2 ha).

$C_{org}$ $ha^{-1}$ corresponding to ~ 8.06 t $C_{org}$ across the 6.3 ha (Table 1B). For the total burnt area, including all sites, we estimate a loss of 27.7 t of $C_{org}$ based on current measurements compared to the control site.

### 3.3. Fire effects on mineral soil

Compared to control site, burnt sites did not show differences in the WC, WHC, BD and Po of topsoil (0–10 cm). However, the LBS site showed the lowest BD value and the highest WHC and Po values compared to other sites, with significant differences only when compared to the MBS site ($p < 0.05$; Table 2). The control soil was found to be slightly alkaline and had average values of EC, CEC and $C_{org}$ of 239 ± 37.4 µS $cm^{-1}$, 7.88 ± 3.45 cmol $kg^{-1}$ d.w., and 25.3 ± 6.80 g $kg^{-1}$ d.w., respectively. A decrease in pH from 7.3, in the control, to 7.0 ($p < 0.01$), in LBS, MBSA and HBS sites, was detected (Table 2). The lowest EC value was found at the MBS site (162 ± 19.7 µS $cm^{-1}$), with significant differences compared to control and LBS sites ($p < 0.001$), and the highest value (271 ± 17.3 µS $cm^{-1}$) was found at the LBS site. No effect ($p > 0.05$) of wildfire was observed for CEC, $C_{org}$ and $C_{min}$ and $N_{tot}$ content (Table 2). Conversely, a significant decrease in $C_{ext}$ was found between the control and burnt sites ($p < 0.05$), except for HBS. A significant decrease in $NH_4^+$-N content, compared to control (4.13 ± 1.48 mg $kg^{-1}$d.w.), was found in MBS (1.85 ± 0.31 mg $kg^{-1}$d.w.; $p < 0.05$), the latter site also showed the highest C/N ratio (17.2 ± 3.88). MBS and HBS showed significantly higher $NO_3^-$-N (173 ± 65.7 mg $kg^{-1}$d.w. and 142 ± 47.0 mg $kg^{-1}$d.w., respectively) content compared to other sites (Table 2). No fire effect was found for $C_{mic}$, soil respiration (R), $C_{mic}/C_{org}$ ratio and qM (Fig. 4a,b,e,g). On the contrary, an alteration in the nitrogen cycle can be observed with a significant increase in Min and Nit rates in burnt sites ($p < 0.01$), in both parameters four times greater compared to the control, irrespective of burn severity and *A. saligna* presence (Fig. 4c, d). Compared to control (8.63 ± 2.19 mg $CO_2$-C % $C_{mic}$ $d^{-1}$), $qCO_2$ showed higher values in HBS and MBSA (21.4 ± 12.0 mg $CO_2$-C % $C_{mic}$ $d^{-1}$ and 22.3 ± 9.75 mg $CO_2$-C % $C_{mic}$ $d^{-1}$, respectively) with significant differences only for MBSA ($p < 0.05$; Fig. 4f).

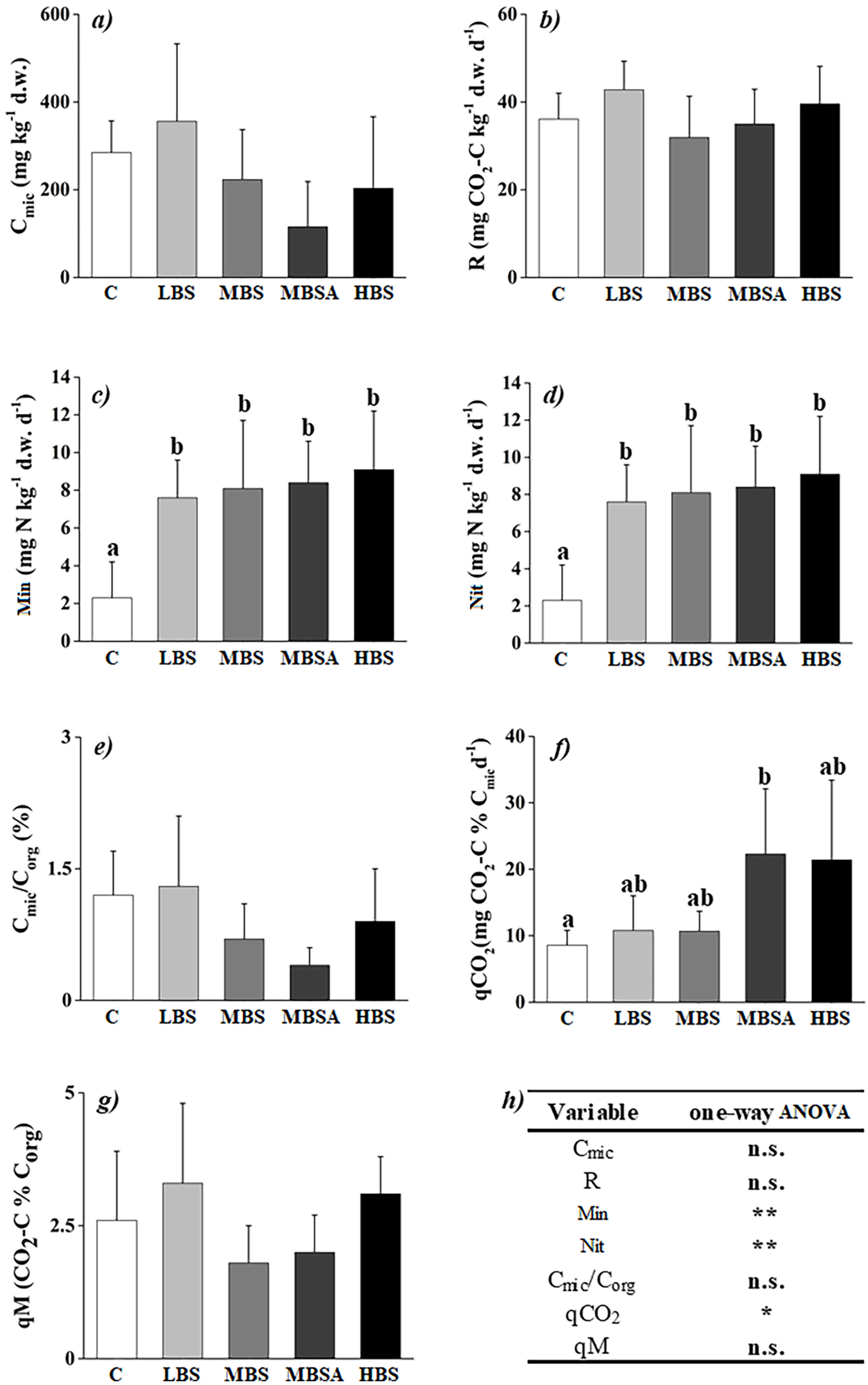
### 3.4. Overall relationships across sites and variables

The PCA, carried out by considering all variables together, allows a better understanding the role of wildfire in regulating soil variables (Fig. 5a) and clarifies which variables were the main responsible for cluster separation which emerged from the Cluster Analysis (Fig. 5b). The first two axes of the biplot deriving from the PCA explained 52.77 % of variance. Along axis 1 (explaining 27.99 % of variance) cluster I (including IA and IB), comprising MBS and most MBSA and HBS sites, was separated by cluster II, comprising control (C, cluster IIA) and LBS (cluster IIB) sites, in the latter also some MBSA ($MBSA_2$, $MBSA_3$) and HBS ($HBS_1$, $HBS_3$) sites were included. Along axis 2 (explaining 24.78 % of variance) the control sites (cluster IIA) clearly separated from all others. Both axes were correlated with field assessed burn severity (BS), the axis 2 positively (0.59, $p < 0.01$), the axis 1 negatively (-0.57, $p < 0.01$), on the contrary the dNBR_2016-2017 and dNBR_2016-2018 were not correlated with PCA axes but only with BS ($p < 0.01$; Table 3). The dNBR_2016-2019 and dNBR_2016-2020 were correlated with both axes ($p < 0.001$ axis 1 and $p < 0.05$ axis 2) and with BS ($p < 0.001$; Table 3). Furthermore, with regards to the soil variables, they were more correlated with dNBR_2016-2019 and dNBR_2016-2020 (LW, L-$C_{org}$, FW, F-$C_{org}$, pH, EC, $C_{ext}$, $N_{tot}$, C/N, $NH_4^+$-N, $NO_3^-$-N, $C_{mic}$, Min, Nit, $C_{mic}/C_{org}$ and $qCO_2$) than those correlated with

**Table 2**

Mean (±standard deviations; n = 5) values of soil water content (WC), water holding capacity (WHC), bulk density (BD), porosity (Po), pH, electrical conductivity (EC), cation exchange capacity (CEC), total organic C ($C_{org}$), extractable C ($C_{ext}$), mineralizable C ($C_{min}$), total N ($N_{tot}$), C/N ratio and mineral N ($NH_4^+$-N and $NO_3^-$-N) in soil of control and burnt sites (LBS, MBS, MBSA and HBS). Results of the one-way ANOVA (n.s. = not significant, * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$) among sites are reported in the last column. Superscript letters indicate statistically significant ($p \leq 0.05$) differences among sites assayed by Student-Newman-Keuls test. For plot labels see Fig. 1.

| Variable | Control | LBS | MBS | MBSA | HBS | one-way ANOVA |
|---|---|---|---|---|---|---|
| WC (%) | 9.78 (±2.48)$^{ab}$ | 11.9 (±2.25)$^b$ | 6.68 (±2.04)$^a$ | 9.39 (±3.07)$^{ab}$ | 11.1 (±1.60)$^b$ | * |
| WHC (%) | 29.2 (±3.46)$^{ab}$ | 38.4 (±11.3)$^a$ | 25.0 (±2.85)$^b$ | 29.5 (±6.53)$^{ab}$ | 35.0 (±6.25)$^{ab}$ | * |
| BD (g $cm^{-3}$) | 1.09 (±0.11)$^{ab}$ | 0.86 (±0.14)$^a$ | 1.16 (±0.12)$^b$ | 0.92 (±0.16)$^a$ | 0.93 (±0.13)$^a$ | ** |
| Po (%) | 59.0 (±4.07)$^{ab}$ | 67.6 (±5.19)$^a$ | 56.2 (±4.49)$^b$ | 65.3 (±6.14)$^a$ | 64.8 (±4.81)$^a$ | ** |
| pH | 7.30 (±0.19)$^a$ | 7.04 (±0.11)$^b$ | 7.19 (±0.02)$^{ab}$ | 7.00 (±0.12)$^b$ | 7.04 (±0.05)$^b$ | ** |
| EC (µS $cm^{-1}$) | 239 (±37.4)$^{ac}$ | 271 (±17.3)$^a$ | 162 (±19.7)$^b$ | 199 (±35.2)$^{bc}$ | 203 (±16.5)$^{bc}$ | *** |
| CEC (cmol $kg^{-1}$d.w.) | 7.88 (±3.45) | 9.53 (±1.45) | 6.77 (±2.55) | 9.67 (±1.92) | 8.95 (±1.54) | **n.s.** |
| $C_{org}$ (g $kg^{-1}$d.w.) | 25.3 (±6.80) | 26.8 (±6.63) | 32.8 (±5.36) | 28.3 (±7.09) | 22.2 (±3.80) | **n.s.** |
| $C_{ext}$ (g $kg^{-1}$d.w.) | 0.17 (±0.03)$^a$ | 0.06 (±0.05)$^b$ | 0.04 (±0.01)$^b$ | 0.07 (±0.04)$^b$ | 0.11 (±0.06)$^{ab}$ | * |
| $C_{min}$ (g $CO_2$-C $kg^{-1}$d.w.) | 0.58 (±0.12) | 0.82 (±0.23) | 0.58 (±0.22) | 0.54 (±0.13) | 0.68 (±0.20) | **n.s.** |
| $N_{tot}$ (g $kg^{-1}$d.w.) | 3.10 (±0.82) | 2.47 (±0.68) | 1.95 (±0.24) | 1.92 (±0.68) | 2.36 (±0.67) | **n.s.** |
| C/N | 8.59 (±2.84)$^a$ | 11.3 (±3.64)$^{ab}$ | 17.2 (±3.88)$^b$ | 16.4 (±6.81)$^{ab}$ | 10.2 (±3.98)$^{ab}$ | * |
| $NH_4^+$-N (mg $kg^{-1}$d.w.) | 4.13 (±1.48)$^a$ | 3.45 (±0.84)$^{ab}$ | 1.85 (±0.31)$^b$ | 2.34 (±1.21)$^{ab}$ | 3.45 (±1.12)$^{ab}$ | * |
| $NO_3^-$-N (mg $kg^{-1}$d.w.) | 83.0 (±18.6)$^a$ | 77.1 (±31.3)$^a$ | 173 (±65.7)$^b$ | 57.0 (±10.1)$^a$ | 142 (±47.0)$^b$ | *** |




| Variable | one-way ANOVA |
|---|---|
| $C_{mic}$ | n.s. |
| R | n.s. |
| Min | ** |
| Nit | ** |
| $C_{mic}/C_{org}$ | n.s. |
| $qCO_2$ | * |
| qM | n.s. |

**Fig. 4.** Mean (+standard deviation, n = 5) values of a) microbial biomass C ($C_{mic}$), b) respiration (R), c) nitrogen mineralization (Min), d) nitrification (Nit), e) $C_{mic}/C_{org}$ ratio, f) metabolic quotient ($qCO_2$), g) quotient of mineralization (qM) and h) results of one-way ANOVA (n.s. = not significant, * = $p < 0.05$, ** = $p < 0.01$) for the study sites (C, control; LBS, low burn severity; MBS, medium burn severity; MBSA, medium burn severity with *A. saligna* invasion and HBS, high burn severity). For each variable, significant ($p \leq 0.05$) differences among experimental conditions (assayed by Student-Newman-Keuls test) were reported as different letters on bars. Please note the different scales on the y axes.

dNBR_2016-2017 (none) and dNBR_2016-2018 (only $C_{org}$). Along axis 1, LBS and control sites occupied similar position, while along axis 2 control sites were at the bottom of the biplot and LBS at the top. In addition, axis 1 was positively correlated with WC, WHC, Po, EC, $C_{min}$, $N_{tot}$, $NH_4^+$-N, $C_{mic}$, R, $C_{mic}/C_{org}$ ratio, qM, LW, L-$C_{org}$ and FW. Axis 1 was also negatively correlated with BD, C/N and $NO_3^-$-N (Table 3). Thus, sites of cluster I, placed on the left side of the biplot, were generally associated with less favourable soil conditions for plants and soil organisms, compared to sites of cluster II. Moreover, HBS sites (included partially in IA and partially in IIB) showed higher values of variables indicating more favourable conditions for biotic community than most medium severity burnt sites (except for $MBSA_2$ and $MBSA_3$). Axis 2 of PCA was positively correlated with WHC, Po, CEC, $C_{min}$, R, Min, Nit and negatively correlated with BD, pH, $C_{ext}$, $N_{tot}$, FW and F-$C_{org}$. Further other correlations among the measured variables were found (Table 3).

## 4. Discussions

### 4.1. Burn severity

Fire effects on vegetation (an indicator of burn severity) were derived from field observation four years after the fire and from remote

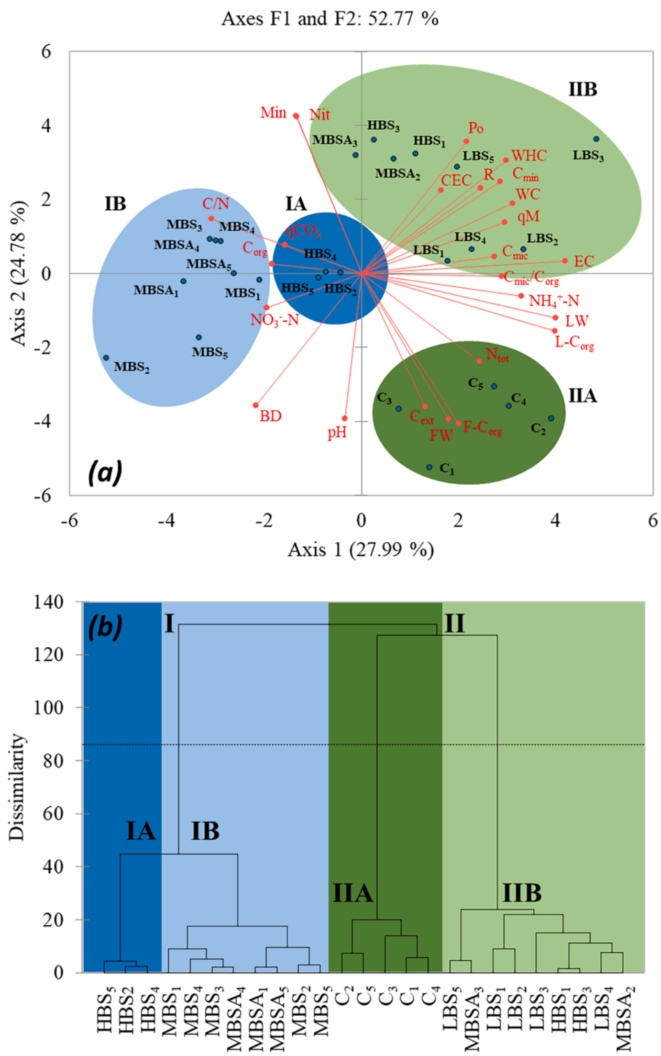



**Fig. 5.** Biplot deriving from the Principal Component Analysis, PCA (a) and dendrogram deriving from Cluster Analysis (b) referring to 25 sites, corresponding to 5 field replicates for each burn severity (C, control; LBS, low burn severity; MBS, medium burn severity; MBSA, medium burn severity with *A. saligna* invasion and HBS, high burn severity) and 25 variables. In the PCA biplot (a) the 25 sites are indicated in black and the following 25 variables are in red (with vector length reflecting the strength of each driving factor): weight of litter and fermentation layers (LW and FW), total organic C (g $m^{-2}$) in L (L-$C_{org}$) and F (F-$C_{org}$) layers, soil water content (WC), water holding capacity (WHC), bulk density (BD), porosity (Po), pH, electrical conductivity (EC), cation exchange capacity (CEC), soil organic C ($C_{org}$), extractable C ($C_{ext}$), mineralizable C ($C_{min}$), total nitrogen ($N_{tot}$), C/N ratio, mineral nitrogen ($NH_4^+$-N and $NO_3^-$-N), N mineralization (Min), nitrification (Nit), microbial C ($C_{mic}$), respiration (R), $C_{mic}/C_{org}$ ratio, metabolic quotient ($qCO_2$), quotient of mineralization (qM). The lines in the dendrogram (b) represent the automatic truncations, leading to four main clusters (IA, IB, IIA and IIB).

sensing analysis for the period 2016–2020. Other authors used this combined approach to assess the burn severity (Marcos et al., 2021). The fire behavior, burn severity and ecosystem post-fire responses are profoundly influenced by forest ecosystem conditions (e.g. microsite, composition, structure, etc.) that are not homogenous at the microscale and this is why within forest fires some areas remain unburned or burnt at low severity ("*fire refugia*"; Meigs and Krawchuk, 2018). The 2017 wildfire produced large and immediate effects on the vegetation only at the HBS site, completely destroying the canopy. In contrast, it did not produce evident direct and immediate effects on the canopy, as highlighted by the remote sensing analysis (dNBR_2016-2017), at the MBS, MBSA and LBS sites even though these appeared burnt based on field observations (trunk burn scars, crown desiccation, litter consumption, burnt residues on the ground). Data suggest that in these sites trees probably suffered from physiological alteration due to fire (Bär et al., 2019) that may have made them more sensitive to changes in weather or microclimatic conditions and pathogen attacks (Bär et al., 2019; Loehman et al., 2017; Marcos et al., 2021). Indeed, four years after fire, all pine trees were dead in MSB and MSBA as well as in HSB. In LSB the 25 % of tree pine was alive confirming the lower burn severity in this site (Mairota et al., unpublished data). In the following 4 years after fire, in HSB, according to the dNBR index, a recovery of herbaceous vegetation was found. In fact, HSB had on average twice the herbaceous cover (20%) of other burnt sites and 41% shrub cover (similar to both control and MSB) with approximately 1200 pine seedlings $ha^{-1}$, similar to the average of MSB and MSBA. Furthermore, in MSBA, *A. saligna* suckers' cover was around 60%. On the contrary, shrub, herbaceous and tree recruitment layers were absent at the LSB site (Mairota et al., unpublished data).

### 4.2. Fire effects on organic (L and F) layers

The direct effect of the 2017 wildfire was initially only observed in the vegetation of the HBS site where the canopy was completely lost as evident from the Sentinel-2 data. The lack of the organic layers (L- and F-layers) in MBS, MBSA and HBS sites, four years post-wildfire, suggests that the canopy fire developed only in HBS site, then the fire spread also to other sites but with the flame closer to the ground (surface/ground fire), thus destroying the organic layers which in MBS, MBSA and HBS have not yet been restored 4 years after disturbance. Moya et al. (2020) discovered very negative impacts on ecosystem functions following fires of significant severity mainly due to the loss of litter layer because they leave the soil exposed to erosion and run-off. Likewise, Moghli et al. (2021) found that litter removal due to wildfire disturbance leads to a decline in habitat complexity. It is clear from the available literature that the restoration time of post-fire litter may depend on many factors, in addition to the extent of disturbance, such as the type of biome and plant communities, the dominant and understorey species, post-fire interventions, and climate (Certini et al., 2011; Eugenio et al., 2006; Fioretto et al., 2003; López-Poma et al., 2014). In our study, the L-layer was found at the LBS site, where F-layer was not detected, demonstrating that wildfire destroyed also here both organic layers (L + F layers), but four years after the fire, the L-layer was almost recovered to pre-fire level while the F-layer has not yet had time to form. According to Trabaud et al. (1985) who studied the litter recovery from a burnt *P. halepensis* forest, litter retrieval started in the year after the fire as a result of pine needles being killed by the heat (secondary impact) but not directly burnt by the fire. Moreover, they conclude that litter progressively increases as trees reoccupy the burnt areas and mainly when pines overtop the understory communities and begin to shed needles (around 25–30 years). The heat effect could justify the amount of litter found in LBS site (similar to control), in addition to the normal litter production by pine trees, considering that only 25 % of the trees were alive. In other burnt sites (HBS, MBS, MBSA), where all pines were dead four years after fire, litter layer was not recovered.

Although the recovery of the litter layer in terms of weight was evident at the LBS site, the $C_{org}$ concentration (g $kg^{-1}$ d.w.) of the new litter showed a significant decrease compared to the control. This could be due to the premature fall of the leaves (heat effect) which did not give the pines time to carry out the eco-physiological retranslocation of nutrients (nitrogen, phosphorus, etc.) during leaf senescence, before fall, to the younger leaves (de las Heras et al., 2017; Fife et al., 2008), thus probably in LBS site litter was relatively richer in nutrient (N, P, etc.) and, as a consequence, poorer in C.

Overall, a net $C_{org}$ loss from soil of 27.7 t in the 13.8 ha of burnt area (on average about 2 t $ha^{-1}$) from organic layers of burnt sites was recorded four years after the fire event. The $C_{org}$ pool had not recovered

**Table 3**

**Table 3.** Pearson correlation coefficients (r) among the biplot axes data (n=25, see Fig. 5), the burn severity (BS), the normalized burn ratio indexes (dNBR_16-17, dNBR_16-18, dNBR_16-19 and dNBR_16-20) and the following 25 variables: weight of litter and fermentation layers (LW and FW), organic C (g $m^{-2}$) in L (L-$C_{org}$) and F (F-$C_{org}$) layers and, the following variables apply for topsoil samples: water content (WC), water holding capacity (WHC), bulk density (BD), porosity (Po), pH, electrical conductivity (EC), cation exchange capacity (CEC), organic C ($C_{org}$), extractable C ($C_{ext}$), mineralizable C ($C_{min}$), total N ($N_{tot}$), C/N ratio, mineral N ($NH_4^+$-N and $NO_3^-$N), microbial biomass carbon ($C_{mic}$), respiration (R), nitrogen mineralization (Min), nitrification (Nit), $C_{mic}/C_{org}$ ratio, quotient of C mineralization (qM) and metabolic quotient ($qCO_2$). Significant r values (* $p$ <0.05; ** $p$ <0.01; *** $p$ <0.001) are shown as bold text.

| Variables | Axis 1 | Axis 2 | LW | L-$C_{org}$ | FW | F-$C_{org}$ | WC | WHC | BD | Po | pH | EC | CEC | $C_{org}$ | $C_{ext}$ | $C_{min}$ | $N_{tot}$ | C/N | $NH_4^+$-N | $NO_3^-$-N | $C_{mic}$ | R | Min | Nit | $C_{mic}/C_{org}$ | qM | $qCO_2$ | BS | dNBR_16-17 | dNBR_16-18 | dNBR_16-19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| LW | 0.81*** | -0.23 | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| L-$C_{org}$ | 0.81*** | -0.30 | 0.99*** | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| FW | 0.41* | -0.78*** | 0.43* | 0.50* | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| F-$C_{org}$ | 0.37 | -0.76*** | 0.40* | 0.47* | 0.98*** | | | | | | | | | | | | | | | | | | | | | | | | | | |
| WC | 0.64*** | 0.36 | 0.31 | 0.29 | 0.00 | -0.05 | | | | | | | | | | | | | | | | | | | | | | | | | |
| WHC | 0.61** | 0.59** | 0.41* | 0.34 | -0.14 | -0.13 | 0.59*** | | | | | | | | | | | | | | | | | | | | | | | | |
| BD | -0.44** | -0.69*** | -0.21 | -0.17 | 0.27 | 0.23 | -0.51** | -0.75*** | | | | | | | | | | | | | | | | | | | | | | | |
| Po | 0.44** | 0.69*** | 0.21 | 0.17 | -0.27 | -0.23 | 0.51** | 0.75*** | -1.00*** | | | | | | | | | | | | | | | | | | | | | | |
| pH | -0.07 | -0.76*** | 0.18 | 0.24 | 0.53** | 0.51** | -0.44* | -0.57** | 0.67*** | -0.67*** | | | | | | | | | | | | | | | | | | | | | |
| EC | 0.86*** | 0.06 | 0.71*** | 0.70*** | 0.25 | 0.19 | 0.81*** | 0.47* | -0.39 | 0.39 | -0.19 | | | | | | | | | | | | | | | | | | | | |
| CEC | 0.33 | 0.43* | 0.02 | 0.00 | -0.17 | -0.25 | 0.77*** | 0.33 | -0.32 | 0.32 | -0.52** | 0.59** | | | | | | | | | | | | | | | | | | | |
| $C_{org}$ | -0.38 | 0.05 | -0.16 | -0.14 | -0.07 | -0.03 | -0.14 | -0.15 | 0.09 | -0.09 | -0.11 | -0.18 | 0.03 | | | | | | | | | | | | | | | | | | |
| $C_{ext}$ | 0.27 | -0.69*** | 0.22 | 0.27 | 0.68*** | 0.65*** | 0.13 | -0.26 | 0.30 | -0.30 | 0.43* | 0.19 | -0.06 | -0.27 | | | | | | | | | | | | | | | | | |
| $C_{min}$ | 0.50* | 0.44* | 0.29 | 0.25 | -0.21 | -0.22 | 0.27 | 0.50* | -0.31 | 0.31 | -0.19 | 0.29 | 0.27 | -0.22 | -0.14 | | | | | | | | | | | | | | | | |
| $N_{tot}$ | 0.50* | -0.46* | 0.34 | 0.38 | 0.58** | 0.54** | 0.46* | -0.02 | 0.10 | -0.10 | 0.12 | 0.56** | 0.30 | -0.06 | 0.52** | -0.08 | | | | | | | | | | | | | | | |
| C/N | -0.63*** | 0.29 | -0.38 | -0.39 | -0.39 | -0.35 | -0.46* | -0.18 | 0.04 | -0.04 | -0.11 | -0.53** | -0.20 | 0.61** | -0.44* | -0.17 | -0.77*** | | | | | | | | | | | | | | |
| $NH_4^+$-N | 0.67*** | -0.12 | 0.53* | 0.53** | 0.38 | 0.33 | 0.45* | 0.46* | -0.36 | 0.36 | -0.16 | 0.51** | 0.12 | -0.27 | 0.31 | 0.10 | 0.39 | -0.51** | | | | | | | | | | | | | |
| $NO_3^-$N | -0.40* | -0.18 | -0.36 | -0.37 | -0.23 | -0.24 | -0.28 | -0.26 | 0.46* | -0.46* | 0.17 | -0.40* | -0.20 | -0.07 | -0.04 | -0.06 | -0.02 | -0.11 | -0.14 | | | | | | | | | | | | |
| $C_{mic}$ | 0.56** | 0.09 | 0.54* | 0.56** | 0.13 | 0.12 | 0.05 | 0.28 | -0.16 | 0.16 | 0.14 | 0.36 | -0.04 | -0.05 | -0.22 | 0.47* | 0.07 | -0.14 | 0.20 | -0.18 | | | | | | | | | | | |
| R | 0.58** | 0.48* | 0.32 | 0.29 | -0.11 | -0.13 | 0.45* | 0.61** | -0.40* | 0.40* | -0.22 | 0.43* | 0.32 | -0.21 | -0.14 | 0.82*** | -0.05 | -0.16 | 0.21 | -0.31 | 0.43* | | | | | | | | | | |
| Min | -0.27 | 0.82*** | -0.45* | -0.49* | -0.68*** | -0.63*** | 0.15 | 0.27 | -0.38 | 0.38 | -0.44* | -0.21 | 0.21 | 0.15 | -0.66*** | 0.20 | -0.45*** | 0.36 | -0.38 | -0.12 | 0.09 | 0.21 | | | | | | | | | |
| Nit | -0.27 | 0.82*** | -0.45* | -0.49* | -0.67*** | -0.63*** | 0.15 | 0.27 | -0.38 | 0.38 | -0.44* | -0.21 | 0.20 | 0.15 | -0.66*** | 0.20 | -0.45*** | 0.36 | -0.36 | -0.12 | 0.10 | 0.21 | 0.99*** | | | | | | | | |
| $C_{mic}/C_{org}$ | 0.59** | -0.02 | 0.48* | 0.50* | 0.20 | 0.17 | 0.07 | 0.18 | -0.06 | 0.06 | 0.22 | 0.40 | -0.01 | -0.34 | -0.04 | 0.45* | 0.11 | -0.30 | 0.20 | -0.17 | 0.93*** | 0.42* | -0.01 | 0.00 | | | | | | | |
| qM | 0.60** | 0.27 | 0.35 | 0.30 | -0.06 | -0.09 | 0.32 | 0.47* | -0.25 | 0.25 | -0.06 | 0.37 | 0.20 | -0.69*** | 0.09 | 0.83*** | -0.02 | -0.45* | 0.23 | -0.05 | 0.35 | 0.74*** | 0.02 | 0.02 | 0.50* | | | | | | |
| $qCO_2$ | -0.32 | 0.15 | -0.46* | -0.48* | -0.33 | -0.33 | 0.15 | -0.11 | 0.07 | -0.07 | -0.32 | -0.18 | 0.24 | -0.13 | 0.18 | -0.08 | -0.06 | -0.03 | -0.06 | 0.14 | -0.74*** | 0.01 | -0.01 | -0.01 | -0.63*** | 0.01 | | | | | |
| BS | -0.57** | 0.59** | -0.79*** | -0.83*** | -0.75*** | -0.71*** | -0.05 | 0.03 | -0.14 | 0.14 | -0.48* | -0.52** | 0.06 | -0.03 | -0.39 | -0.02 | -0.43* | 0.27 | -0.34 | 0.39 | -0.36 | -0.03 | 0.64*** | 0.64*** | -0.34 | -0.05 | 0.50* | | | | |
| dNBR_16-17 | 0.02 | 0.21 | -0.30 | -0.31 | -0.18 | -0.17 | 0.28 | 0.25 | -0.13 | 0.13 | -0.19 | -0.07 | 0.07 | -0.31 | 0.17 | 0.10 | 0.14 | -0.30 | 0.12 | 0.34 | -0.08 | 0.15 | 0.25 | 0.26 | 0.02 | 0.19 | 0.28 | 0.58** | | | |
| dNBR_16-18 | 0.11 | 0.23 | -0.30 | -0.24 | -0.13 | -0.12 | 0.33 | 0.29 | -0.29 | 0.29 | -0.26 | 0.01 | 0.09 | -0.42* | 0.20 | 0.05 | 0.16 | -0.36 | 0.19 | 0.18 | -0.06 | 0.12 | 0.25 | 0.25 | 0.06 | 0.20 | 0.25 | 0.53** | 0.95*** | | |
| dNBR_16-19 | -0.76*** | 0.47* | -0.87*** | -0.91*** | -0.67*** | -0.64*** | -0.29 | -0.13 | 0.03 | -0.03 | -0.40* | -0.71** | -0.02 | 0.16 | -0.48* | -0.12 | -0.50* | 0.44* | -0.49* | 0.37 | -0.48* | -0.15 | 0.54** | 0.54** | -0.49* | -0.22 | 0.50* | 0.88*** | 0.29 | 0.22 | |
| dNBR_16-20 | -0.69*** | 0.57** | -0.77*** | -0.83*** | -0.75*** | -0.71*** | -0.21 | 0.005 | -0.03 | 0.03 | -0.48* | -0.62** | 0.06 | 0.22 | -0.55** | 0.04 | -0.52** | 0.44* | -0.49* | 0.40* | -0.44* | -0.05 | 0.58** | 0.57** | -0.51** | -0.12 | 0.46* | 0.84*** | 0.23 | 0.12 | 0.95*** |

in MBS, MBSA and HBS sites, whereas LBS had recovered about 51.2 % of $C_{org}$ lost. The loss of the organic layer and, consequently $C_{org}$ pool, from the whole burnt area causes a loss of ecosystem services related to it, such as C sequestration, nutrient cycling regulation, etc. (Moya et al., 2020). Giuditta et al. (2018) found a reduced evaporation flux from litter of burnt areas to the atmosphere, at least up to 18 months after prescribed fire, due to the thinner litter layer in burnt area, compared to the control, which caused lower amount of water to be stored in the litter.

### 4.3. Fire effects on mineral soil

The values of water content (WC) and water holding capacity (WHC) recorded four years post-fire in the mineral soil (0–10 cm) did not show significant changes in association with increasing burn severity. Similarly, Pourreza et al. (2014) and Li et al. (2019) did not observe water content changes between burned and unburnt sites one year and six months after the event, respectively. In contrast, Weber et al. (2014) reported a significant decrease in water content in relation to increasing fire severity in a *Pinus ponderosa* stand, but three months after the fire. Nevertheless, Debano et al. (1976) found high soil moisture contents in areas hit by high severity fires due to the destruction of the hydrophobic or litter layers and the decrease in the rain interception capacity of fire-damaged plants (Hamman et al., 2007). Bulk density (BD) and porosity (Po) also did not appear to have been directly affected by fire, regardless of the fire severity, contrary to the findings of Mehdi et al. (2012) who reported higher BD with increasing fire severity, one-year post-fire. In our study, we observed some variations in BD and Po, with LBS site showing the lowest BD and the highest Po values compared to other sites. According to heating experiment by Giovannini et al. (1988), the BD increased with increasing soil heating in sandy-loam soil, as a direct consequence of organo-mineral aggregates destruction and the filling of empty spaces with ash and dispersed clay minerals (Durgin and Vogelsang, 1984), while Po decreases. However, a seven-year study carried out by Xue et al. (2014) showed that after an initial increase, BD decreased after four years until it returns to a value comparable with the control after seven years.

Even though is widely known that soil pH increases after wildfire (Certini et al., 2011) or prescribed fire (Catalanotti et al., 2018), our data did not confirm this trend. Indeed, pH at all burned sites decreased slightly but significantly when compared to the control. The pH increase, reported in the first year after the disturbance is due to the higher content of oxides, hydroxides and carbonates of potassium and sodium in the ashes. However, this may be only a temporary effect because these compounds are very soluble and easily leached (Úbeda et al., 2009). In a controlled heating experiment, Marcos et al. (2007) found no significant differences in pH after subjecting soils to temperatures of 100 and 200 °C, whereas Badía and Martí (2003) documented a reduction in pH levels between 220 and 300 °C in the studied soils. On the contrary, increase in soil pH were reported with soil temperature reaching 400–500 °C (Certini, 2005). Thus, in our study, the pH decreases in burnt sites may be due to a low temperature over a short time period reached at ground level or most likely due to severe runoff of ash and soluble cations content.

Usually, like pH, the EC of soil tends to increase after a fire mainly due to the release of ions from the combustion of organic matter (Lasanta and Cerdà, 2005; Notario del Pino et al., 2008; Prieto-Fernández et al., 2004), but also due to the use of seawater to extinguish the fire (Bogunovic et al., 2015; Park et al., 2015). In the study area, no change in the salt content of the soil extracts were observed in burnt sites compared to the control (except for MBS site showing lower value), consistent with the results of other studies stating that a significant change in EC is only observed immediately after the fire (Bárcenas-Moreno et al., 2011; Granged et al., 2011; Pereira et al., 2018b). No changes due to fire were observed for the CEC as well as the organic carbon content, regardless of the burn severity. In general, fire tends to reduce soil CEC due to the combustion of soil organic matter and the decrease in clay content (Ekinci, 2006; Hatten et al., 2005). However, Mehdi et al. (2012) showed that after one year from the fire, at least in a low fire severity site, the CEC returned to control values. Liang et al. (2006) stated that charred carbon, a common by-product of wildfire, increases the CEC. In our burnt sites the unchanged total $C_{org}$ content, compared to control, explains the unchanged CEC. Similarly at the total $C_{org}$ content, the short-term mineralizable carbon content ($C_{min}$) was also comparable in the burnt and control soils. On the contrary, the fraction of more labile organic C ($C_{ext}$) decreased in the burnt soils compared to the control and was negatively correlated with dNBR_2016-2019 and dNBR_2016-2020, indicating an effect of burn severity. A decrease in soil $C_{org}$ as a function of increasing fire severity has been found in previous studies (Pourreza et al., 2014; Tulau et al., 2020). However, the available studies on this topic are inconclusive because increases, decreases or no change of $C_{org}$ content have all been reported in burnt soils, compared to controls (Martín Lorenzo et al., 2021; Moya et al., 2019; Prendergast-Miller et al., 2017; Stinca et al., 2020; Xu et al. 2022a), making it difficult to generalize the consequences of fire on soil $C_{org}$ (González-Pérez et al., 2004), therefore highlighting the need for more research across different soil types regarding fire impact on different C pools. Several explanations of soil mineral $C_{org}$'s failure to change in this study, four years post-fire, are possible: i) the soil temperature was probably not very high during the fire so $C_{org}$ content was not affected; ii) $C_{org}$ content decreased immediately after fire, but in the following four years it returned to that of the control site; iii) the loss of organic C during fire and possible loss due to increase in mineralization of labile C fraction in the following period, caused by microbial activity stimulation, could have been balanced by the accumulation of more recalcitrant forms of carbon (charred material, free lipids, colloidal fractions such as humic acids and fulvic acids, etc.) which are more resistant to biodegradation (González-Pérez et al., 2004). In our study, no clear effect of fire on $N_{tot}$ content was found, in contrast to the results of Martín Lorenzo et al. (2021) who reported a decrease in $N_{tot}$ in the burn area which has also been observed in other studies (Li et al., 2019; Pourreza et al., 2014). On the contrary, Fernández-García et al. (2019) detected no shifts in soil $N_{tot}$ immediately after wildfire in a *P. pinaster* wood, while Giuditta et al. (2019) found a $N_{tot}$ increase in *P. pinea* coastal plantation and no change in 600 m a.s.l. *P. pinaster* plantation in the first year after prescribed fires performed in Southern Italy. Total N may be lost during fire via volatilization, but this effect is generally temporary (Bárcenas-Moreno and Bååth, 2009). In MBS and MBSA sites, the slight decrease (not significant) in $N_{tot}$, associated to a slight increase (not significant) of $C_{org}$ content, compared to the control, was reflected in the increase in the C/N ratio. This is inconsistent with no changes in C/N ratio reported by Vega et al. (2013), who observed decrease in both $C_{org}$ and $N_{tot}$. Mineral nitrogen content, evaluated here as the individual contributions of $NH_4^+$-N and $NO_3^-$-N ions, showed a decrease in $NH_4^+$-N in MBS site compared to the control and a negative correlation with dNBR_2016-2019 and dNBR_2016-2020; whereas an increase in $NO_3^-$-N was observed in MBS and HBS sites. Generally, $NH_4^+$-N content increases immediately after the fire, due to the low temperature combustion of the forest floor and the incorporation of nitrogen-enriched ash (Alcañiz et al., 2018; Fultz et al., 2016), but within one year it decreased until pre-fire level (Wan et al., 2001). Nitrate also increased post-fire but at a lesser rate compared to ammonium content, and then declined some years after the fire (Turner et al., 2007; Xu et al., 2022b). This is in-line with other studies (Kong et al., 2019; Múgica et al., 2018) and can be explained as the next step after the immediate post-fire flash of ammonium and the activation of nitrifying bacteria. High availability of nitrate in burnt sites can lead to a rapid recovery of the undergrowth as it is a form of nitrogen that is highly absorbable by plants but at the same time is also prone to leaching.

Despite compared to the control, no significant changes were detected in $C_{mic}$, R, $C_{mic}/C_{org}$ ratio and qM, the $C_{mic}$ and $C_{mic}/C_{org}$ ratio were affected by burn severity as demonstrated by negative correlations with dNBR_2016-2019 and dNBR_2016-2020. In accordance with our findings, Moya et al. (2019) also found no significant changes related due to fire in soil respiration and qM after three and five years in a *P. halepensis* forests located in the SE of Iberian Peninsula. However, the probability of finding differences in soil respiration is usually higher in the first year after fire, monitoring this parameter for at least one year (with sampling for each season, after Plaza-Álvarez et al., 2021). Vega et al. (2013) observed a significant depletion in soil respiration at all eight studied sites (*P. pinaster* stands and shrublands) as a function of increasing fire severity, but 3–7 days after the wildfire event. In several studies microbial biomass showed an increase in the short-term (from the first weeks to < 2 years) post-fire followed by a decrease (Kara and Bolat, 2009; Rutigliano et al., 2007) which may be explained by the indirect impact of the fire on the plant cover and soil properties that affect the edaphic microflora. In fact, early after a wildfire, it is possible to find a stimulating effect on the microbial biomass that may be due to the greater availability of nutrients contained in the ash and of partially burnt organic matter that promotes the growth of microorganisms that have survived the disturbance (Snyman, 2004). Subsequently, in the medium to long term (from 2 to > 5 years), other environmental factors could strongly influence soil microorganisms such as the higher seasonal fluctuation of water availability in burnt areas still not completely covered by vegetation (Dooley and Treseder, 2012; Memoli et al., 2021). Wang et al. (2012), in a *meta*-analysis on the effects of wildfire on forest soils, described a significant reduction in microbial biomass for more than three years post-fire. Likewise, Jhariya and Singh (2021), in a tropical seasonally dry forest, found higher microbial biomass at the control site than those impacted by different fire severities, while Moya et al. (2019) showed decreases in microbial carbon and $C_{mic}/C_{org}$ ratio in burned areas after five years. In the present work lower values of $C_{mic}/C_{org}$ ratio (even if not significant) in burnt areas of medium severity (MBS and MBSA), compared to control and LBS sites, were found, which can be ascribed to a stronger fire impact on $C_{mic}$ than $C_{org}$ (Bastida et al., 2008). Several authors reported that $C_{mic}/C_{org}$ ratio values below 2% reflect a depletion of organic matter (Anderson, 2003; Paz-Ferreiro and Fu, 2016), therefore, our results suggest a large decrease in organic matter in the MBS and MBSA sites, where $C_{mic}/C_{org}$ ratio values were always below 1 %. The metabolic quotient ($qCO_2$), which indicates the efficiency of C resource use by microorganisms, showed an increase in MBSA and HBS sites, compared to control, significant only in MBSA. Thus, in these sites a reduced efficiency in the carbon use by microbes was observed compared to the control, LBS and MBS sites respectively. According to a *meta*-analysis on the fire effects on the soil microbial metabolic quotient, fire can increase the $qCO_2$ (Liu et al., 2023). Indeed, in the Mediterranean maquis environment, a significant increase in soil respiration and $qCO_2$ was reported by Rutigliano et al. (2002) 14–16 months after a wildfire. In another area of Mediterranean maquis, an increase in soil respiration and $qCO_2$ was found in the first three months after an experimental fire, but no difference thereafter (Fierro et al., 2007; Rutigliano et al., 2007). Finally, according to Hanan et al. (2016) findings, nitrogen mineralization and nitrification increased in all burnt sites, irrespective of fire severity and *A. saligna* presence, compared to the control. N mineralization and nitrification may be stimulated in burnt soil because of changes in soil conditions, such as pH, temperature, soil moisture, substrate availability, ash addiction as well as competition with plants (Knicker, 2007; Knoepp et al., 2004). The nitrification increase may be explained by the reduced competition for $NH_4$ between nitrifiers and plants. Unburnt mature stands could be more N limited than recently burnt stands, so that plants compete with nitrifying microorganisms slowing nitrification (Kong et al., 2019). In our study, we associate the nitrification increase with an increase in $NO_3^-$-N concentration observed in MBS and HBS sites, but not in LBS and MBSA sites which exhibited values akin to the control. This may be explained by higher nitrate uptake by plants in LBS (less impacted by the fire) and MBSA sites (whereby *A. saligna* recovery after wildfire might be more efficient at fixing and accumulating N within its tissues rather than in the soil), higher microbial immobilization and/or higher denitrification or leaching loss. Annual herbs may play a key role in retaining N during the early stages of post-fire recovery. For example, in burnt chaparral, the post-fire herbaceous plants absorbed available N so effectively that N accumulation in the plants exceeded the net mineralization flux measured in the soil (Hanan et al., 2016). An interaction of different factors probably occurred in our sites that may explain changes in N compounds and N process rates. In the LBS site, the recovery of the litter layer provided substrates for N-mineralizing microorganisms and nitrifiers, with no nitrate accumulation in the soil probably due to root uptake by pine trees. On the other hand, in sites most affected by fire (MBS, MBSA and HBS), the negative impact on the tree cover did not allow the litter layer to recover, but it encouraged the colonization of herbaceous plants. This allowed recovery of the $NH_4^+$-N content in MBSA and HBS, with an increase in $NO_3^-$-N content only in HBS. On the contrary, in MBS the $NH_4^+$-N content had not recovered and $N_{tot}$ was slightly (not significantly) lower than in control, in the meantime nitrate N content, as well as N mineralization and nitrification, increased suggesting that MBS soil was losing N. The N loss probably did not occur in MBSA due to intense post-fire colonization by *A. saligna*, which can naturally fix N, so reducing the loss by leaching and could N-enrich the soil thanks to its symbiotic N-fixing bacteria (Slingsby et al., 2017; Yelenik et al., 2007). However, it has to be underlined that this apparent positive effect on soil N conservation by the invasive *A. saligna* could favor the establishment of nitro-ruderal species potentially leading to negative effects on native species, as recently found in similar habitat by Calabrese et al. (2017) and Lazzaro et al. (2020). The fire passage and subsequent vegetation colonization can change the intensity of seasonal fluctuations in soil climate conditions which are the main factor influencing microbial activity related to the nitrogen cycle in the Mediterranean area (Rutigliano et al., 2009). Zhou et al. (2009), in a prescribed fire experiment applied in the Mongolian steppe, observed that inter-annual climatic variations may affect the microbial communities involved in the nitrogen cycle more than fire.

### 4.4. Overall effect of wildfire on burnt sites

By considering all evaluated variables together through multivariate analysis, it has been possible to generate overall information on fire-induced changes to soil health. PCA evidenced a clear effect of wildfire occurrence and severity on the soil system four years after the disturbance event. However, wildfire effects on soil did not increase proportionally with increasing wildfire severity. Indeed, the following gradient of fire impact on soil system in different sites may be recognized by PCA: MBS > MBSA > HBS > LBS. Surprisingly, compared to most medium severity burnt sites, HBS sites generally showed higher values of variables that contribute to soil health in the sense, according to Marzaioli et al. (2010a), of "*more is better*" (*i.e.*, the best soil functionality is associated with high values). The total tree destruction in HBS probably stimulated faster colonization by herbaceous vegetation that quickly enriched the soil of labile organic substrates (higher $C_{min}$) rich of nitrogen (higher $N_{tot}$ and $NH_4^+$-N), stimulating microbial community growth (higher $C_{mic}/C_{org}$ ratio). It has to be underlined that several soil variables were not significantly different among burnt and control sites, among these soil $C_{org}$, which is an important C pool that positively affects soil microbial biomass and activity (Marzaioli et al., 2010b).

The wildfire has a stimulating effect on diffusion of *A. saligna* only where the species was already present (MSBA); in other burnt sites we did not find resprouter or seedlings four years after fire. However, the presence of *A. saligna*, does not seem to have negatively affected the soil properties at the MBSA site compared to MBS site affected by the same burn severity. On the contrary, some soil properties (BD, Po) appeared even improved by *A. saligna* presence, compared to MBS. This plant is among the main invasive alien species of the Mediterranean coasts with a significant transformative capacity (Richardson et al., 2000) and, thus, capable of influencing ecosystem services (Cohen and Bar, 2017). In the literature, there is ample evidence that *A. saligna* induces several changes concurrently in plant and soil communities influencing litter development, soil moisture regimes and soil nutrient levels, linked to its symbiotic relationship with nitrogen fixing bacteria (Abd El-Gawad and El-Amier, 2015; Gaertner et al., 2011; Maitre et al., 2011; Tozzi et al., 2021). According to Yelenik et al. (2004), *A. saligna* appears to be able to alter N-cycle through long-term (about 30 year) invasions in nutrient-poor soils such as those associated with fynbos vegetation. Surprisingly, in our study no effect on $N_{tot}$ content was found in MBSA, compared to MBS, except for a significant decrease in $NO_3^-$-N. This decrease cannot be considered a negative effect because nitrate may be quickly lost by leaching (Padilla et al., 2018). However, since the effect of the invasive plant on soil may increase, many potential negative changes could occur over time.

### 4.5. Conclusions

This study contributes to understanding the medium-term effects of wildfire at different severity on the soil physical, chemical and biological properties of a *P. halepensis* woodland, where the presence of the alien species *A. saligna* has been recorded. Our hypotheses were partially confirmed. Indeed, although four years after the disturbance many physical and chemical properties of the mineral soils, as well as microbial biomass and soil respiration, were comparable between the burnt soils and the control, nevertheless, the following alterations were observed:

- A total lack of the soil organic layers (litter and fermentation layers) in high and medium severity (HSB, MSB and MSBA) sites, and the presence of litter alone in the low burn severity site (LBS). Since no increase in $C_{org}$ in mineral soil occurred, this led to an estimate of approximately 2 t $ha^{-1}$ of organic carbon lost from the edaphic compartment of the forest ecosystem and potentially transferred into the atmosphere, which could have been, at least in part, compensated by the C sequestration from plant regeneration.
- Changes in the N cycle processes were found. Nitrogen mineralization and nitrification at the burnt sites, regardless of burn severity or the presence of *A. saligna*, were four times greater than in the control. This may have caused N loss, if the release of mineral N was not compensated by plants or microbial uptake.
- No negative effects on the soil have been so far observed in presence of *A. saligna* which has been reported for the first time in this study in the SAC-IT9130006. However, the alien plant density in MSBA has increased and may increase further in the coming years.
- The effect of the wildfire did not increase proportionally with the increasing burn severity, with the high burn severity site showing better soil health than the medium burn severity sites.

Our data suggest that in areas where the fire has destroyed the forest, the option of waiting for the natural recovery of the vegetation may be considered but being careful in the spread of the invasive plant and/or fire recurrence. The periodic monitoring of fire-soil-vegetation interaction is advocated, particularly to control the expansion of invasive species. Mapping the distribution of *A. saligna* can provide a powerful tool for monitoring the invasion over time and for planning possible mitigation/eradication applications. Finally, the assessment of burn severity by combining field visual assessment and remote sensing can be a valid methodology for constructing the fire history when detailed information immediately after the disturbance is not available.

### CRediT authorship contribution statement

**Luigi Marfella:** Conceptualization, Methodology, Investigation, Software, Formal analysis, Visualization, Writing – original draft. **Rossana Marzaioli:** Conceptualization, Methodology, Investigation, Software, Formal analysis, Visualization, Writing – review & editing. **Gaetano Pazienza:** Conceptualization, Methodology, Investigation, Writing – review & editing. **Paola Mairota:** Conceptualization, Methodology, Investigation, Validation, Resources, Project administration, Writing – review & editing. **Helen C. Glanville:** Conceptualization, Validation, Resources, Writing – review & editing. **Flora A. Rutigliano:** Conceptualization, Methodology, Investigation, Validation, Resources, Project administration, Writing – review & editing.

## 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 material

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