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Wildfire drives lasting changes in the topsoil chemistry of Alpine forests

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Abstract

Background Wildfires are natural disturbances that impact the soil altering the biogeochemical cycling of many elements. The European Alps are being increasingly affected by extreme wildfires, but little is known about the persistence of fire-induced changes in these soils, and the relationship with fire severity in this environment. We sampled an Alpine Scots pine (*Pinus sylvestris* L.) forest affected by a mixed-severity wildfire (tree mortality range 15–100%) and assessed the recovery of several soil properties three years after the event in surface and sub-surface horizons of unburnt and burnt plots belonging to different severity classes (mapped with a combination of remotely sensed and field data).

Results The burnt plots differed from unburnt plots by having greater bare soil exposure, and the dark-colored surface horizon was characterized by significantly higher pH and pyrogenic carbon content compared to the unburnt plots. The sub-surface horizons had significantly lower organic carbon content than the unburnt plots, higher pH, and less sand-sized aggregates. There was no proportionality between fire-induced changes in soil properties and fire severity class. The soils with the greatest water repellency (water drop contact angle > 110°) were mostly located in low severity plots.

Conclusions Most soil properties, including those that usually recover quickly (e.g., pH), were still different in burnt plots three years after the wildfire. Severity maps produced in these environments, although detailed, can fail in identifying the sites where the soil was most severely burnt, and recovery of soil properties is not proportional to the severity degree obtained with this approach.

Keywords Fire severity, Mid-term study, Mountain soils, *Pinus sylvestris* L., Pyrogenic carbon, Water repellency

Resumen

Antecedentes Los incendios de vegetación son disturbios naturales que impactan sobre el suelo, alterando el ciclo bio-geoquímico de muchos elementos. Los Alpes Europeos están siendo afectados por incendios forestales extremos, aunque poco se sabe sobre la persistencia de los cambios inducidos por el fuego en esos suelos, y su relación con la severidad de esos incendios en este ambiente. Muestreamos un bosque de pino silvestre (*Pinus sylvestris* L.) afectado por un incendio de severidad mixta (donde el rango de mortalidad fue entre el 15 y el 100%), y determinamos la recuperación de diferentes propiedades del suelo tres años después del evento de fuego, tanto en la superficie como en horizontes sub-superficiales de áreas no quemadas, y de quemadas a diferentes clases de severidad (mapeadas mediante una combinación de datos registrados por satélites y de campo).

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Resultados Las parcelas quemadas difirieron de las no quemadas en que tenían una mayor exposición de la superficie desnuda del suelo, y porque el horizonte superficial de color oscuro se caracterizaba por un pH significativamente mayor y también mayor contenido de carbono pirogénico comparado con las parcelas no quemadas. Los horizontes sub-superficiales mostraron, en parcelas quemadas, un más bajo contenido de carbono orgánico que las parcelas no quemadas, un pH más alto, y menos agregados del tamaño de granos de arena. No hubo una proporcionalidad entre los cambios en las propiedades de los suelos inducidos por el fuego y las clases de severidad. Los suelos con la mayor repelencia al agua (con un ángulo de contacto de la gota $> 110^\circ$) fueron los ubicados en las parcelas que tuvieron una baja severidad.

Conclusiones La mayoría de las propiedades de los suelos, incluyendo aquellas que se recuperan rápidamente (i. e. el pH), fueron también diferentes en parcelas quemadas luego de tres años del incendio. Los mapas de severidad producidos en esos ambientes, aunque detallados, pueden fallar en identificar los sitios donde el suelo fue más severamente quemado, y la recuperación de las propiedades de los suelos no es proporcional con la severidad que se obtiene mediante este abordaje metodológico.

Background

Wildland fire affects a yearly average of 400 million ha of land worldwide (Artés et al. 2019), imposing significant changes on ecosystems structure and composition (Ryan 2002). In Europe, about 281 million ha burnt between 2001 and 2018 (FAO 2020), and in the near future wild-fire frequency and extent are likely to increase due to climate and land use changes (Dupuy et al. 2020; Mantero et al. 2020). Southern Europe is being particularly exposed to wildfire occurrence as fuel continuity and the widespread adoption of fire suppression policies have led to the accumulation of ready-to-burn fuels (Pezzatti et al. 2013). An average of 1300 fires occur in the European Alps each year (Wastl et al. 2012), where *Pinus sylvestris* L. (Scots pine) is one of the dominant species, especially on south-facing slopes in dry inner-Alpine valleys exposed to a high fire risk (Vacchiano et al. 2015).

Fire behavior can be described in terms of intensity, i.e., the rate of energy release per unit time and area (reaction intensity) or fireline length (fireline intensity), and severity, which includes the change or loss in ecosystem biomass (Keeley 2009). Fire severity is commonly assessed after a disturbance either by remote sensing techniques (Fernández-García et al. 2018; Miller and Thode 2007) or by field-based measurements of fire effects (Key 2006), or by a combination of these two methods (Key and Benson 2006; Morresi et al. 2022). Maps are often created to classify fire severity based on a severity index, e.g., the composite burn index (CBI) (Key and Benson 2006), or some attributes describing fire effects (like tree mortality, crown scorch, depth of burn). In the absence of more specific information (like fire residence time), these maps may prove useful in identifying areas where fire-induced effects on soil properties are expected to be more significant. However, they may not be sufficiently accurate, or have insufficient resolution, to provide reliable information about soil conditions.

At the soil level, fire usually induces significant modifications only in the first few centimeters below the ground (Badía et al. 2014; Clement et al. 2011). For example, even with surface temperatures of 680–820 °C, temperatures might not exceed 25 °C below 5 cm (Bento-Gonçalves et al. 2012). For this reason, research is usually focused on the topsoil, considering either the first 10 cm (Andreu et al. 2001), 5 cm (Adkins et al. 2019; Varela et al. 2010), 3 cm (Fernández-García et al. 2019), or 2.5 cm (Arcenegui et al. 2008). To the degree that sub-surface mineral soil is heated, there can be major changes in biogeochemical cycles (Certini 2014) and, depending on fire type, soil type, ecosystem, and soil property, changes can be ephemeral, like ash deposition, or long-lasting (Certini 2005). These changes can persist for months or years following the fire event (Shakesby and Doerr 2006). For example, the biogeochemical cycling of phosphorus (P) can be significantly affected for years, with a conversion of organic P into plant-available inorganic forms (Schaller et al. 2015). Differences in P forms between burnt and unburnt sites can also persist years after the fire (Fernández-García et al. 2019; Vega et al. 2013), and the effects are usually more persistent in areas of greater organic matter consumption (Francos et al. 2020). Other changes in soil properties can be even more long-lasting, as in the case of carbon (C) biogeochemical cycles (Jones et al. 2019; Neary et al. 2005). The incomplete combustion of above- and below-ground biomass originates a heterogeneous mixture of thermally altered residues known as pyrogenic organic matter (PyOM) (Bird et al. 2015; Knicker 2011). PyOM is considered a long-term pool of C sequestered by the soil (Knicker 2007), since pyrogenic C compounds (PyC) are far more resistant to microbial decomposition than their unburnt precursors (Singh et al. 2012; Knicker et al., 2008). The higher the charring temperature, the more highly condensed

compounds are formed (Wiedemeier et al. 2015), with residence times that can span up to millennia (Czimczik et al. 2005; Hammes et al. 2008).

Another soil property that undergoes changes in the years after a fire is soil hydrophobicity, or water repellency. It limits water infiltration (DeBano 1981) and can be measured in the field (Ebel 2020; Tinebra et al. 2019) or under standardized laboratory conditions (Diehl et al. 2010; Negri et al. 2024; Papierowska et al. 2018). Soil water repellency determines the hydrological behavior of a soil, and when rainfall rates surpass infiltration rates, overland flow occurs (Moody and Ebel 2014), with episodes of post-fire surface runoff and erosion (DeBano 2000; Doerr et al. 2000). An increase in soil water repellency has been observed in many burnt sites (Larsen et al. 2009; Zavala et al. 2009) as, with rising temperatures, soil organic matter (OM) tends to coat mineral particles making them hydrophobic (Shakesby and Doerr 2006). Above a certain temperature, water repellency disappears and the soil becomes wettable; depending on heating duration, this may occur above 200 °C (Negri et al. 2024, 2021), 250 °C (Badía-Villas et al. 2014), or 300 °C (Atanassova and Doerr 2011), and it occurs even for soils that were originally extremely water repellent (Badía-Villas et al. 2014). Soil water repellency is known to be subjected to seasonal fluctuations (Plaza-Álvarez et al. 2018), but it is widely monitored in fire-affected landscapes as it allows easy identification of the areas where the soil has been directly affected by the fire (Mataix-Solera et al. 2013; Tinebra et al. 2019).

The post-fire recovery of soil properties depends heavily on litter inputs from trees. In Scots pine stands where fire induces a high tree mortality, the recolonization of the species is slow when seed trees are not nearby (Mantenero et al. 2023; Vilà-Cabrera et al. 2012). Therefore, we hypothesized that in Scots pine forests of the Alpine range the heat-induced organic matter consumption from significant soil heating and lack of organic matter replacement would cause multi-year changes in several soil properties, including those that usually recover more rapidly due to the soil buffering capacity.

We thus wanted to understand: (i) whether soil properties that recover slowly (like OM and PyOM contents) and others that usually recover more rapidly (like the abundance of available P, soil water repellency, and pH) would be different in unburnt and burnt sites, and (ii) whether a landscape-scale assessment of fire severity (obtained from a combination of medium resolution multispectral satellite data and field measurements) would be accurate enough to establish a direct proportionality between severity classes and fire-induced changes in soil properties.

Methods

Study area

We conducted the study in the European Alps, in Northern Italy, in the municipality of Mompantero (45°9′59.31″N, 7°3′7.60″E) (Fig. 1, top-left panel), on a south-facing slope with elevation ranging from 1400 to 1800 m a.s.l. The climate is warm-summer humid continental climate (Dfb in Koeppen's classification; Kottek et al. 2006). The mean annual precipitation is 840 mm and the mean temperature is 5 °C (<https://climate.mapresso.com/index.html?lat=45.164&lon=7.044>).

We focused on an almost pure Scots pine stand mostly resulting from secondary succession after land abandonment, which is among the dominant forest categories in the area. These stands are among the most affected by wildfires due to their high flammability and density (average tree density exceeding 1200 plants/ha), leading to potential crown fire behavior. In the whole Alpine region, the historical fire regime is typically characterized by winter-early spring low severity surface fires with low tree mortality (Reinhard et al. 2005), with some infrequent high severity crown fires with high tree mortality affecting dry conifer forests on the southern slopes.

The year 2017 was an extraordinary fire year in Europe (Rita et al. 2020), and it created the opportunity to study the impact of a high-severity wildfire on the soils of the Alps. On 22 October 2017, the Susa fire burnt a total of 4000 ha (2500 ha of forests, not only Scots pine stands) including our study area, producing a complex mosaic of severity patches (Morresi et al. 2022). The severity of the Susa fire was assessed in a previous work by combining multispectral satellite data from Sentinel-2 images and field data (Morresi et al. 2022), expressing fire severity as Relative differenced Normalized Burn Ratio (RdNBR) (Miller and Thode 2007) calibrated with CBI. We focused on a smaller area (<2 km²) with monospecific forest cover (Scots pine) and a fairly homogeneous lithology (i.e., serpentinite). The soils of the area are classified as Regosols and Cambisols according to the World Reference Base (WRB) for soil resources (WRB 2015). We provide the description of a representative soil profile in Supplementary information (Fig. S1). Data from local authorities (https://protciv.igr.piemonte.it/it/map/protciv_incendi/) indicate that no other fires have occurred in the area since the establishment of the fire monitoring service in the region (1997). In this area, ca. 70% of the Scots pine range in diameter between 20 and 40 cm. Using the map produced by Morresi et al. (2022), we identified areas classified as unburnt, and burnt with low, medium, and high severity (Fig. 1). The area is composed by 15% low severity patches, 20% medium severity, and 40% high severity patches. The field measurements of fire severity of Morresi et al. (2022) revealed that tree mortality

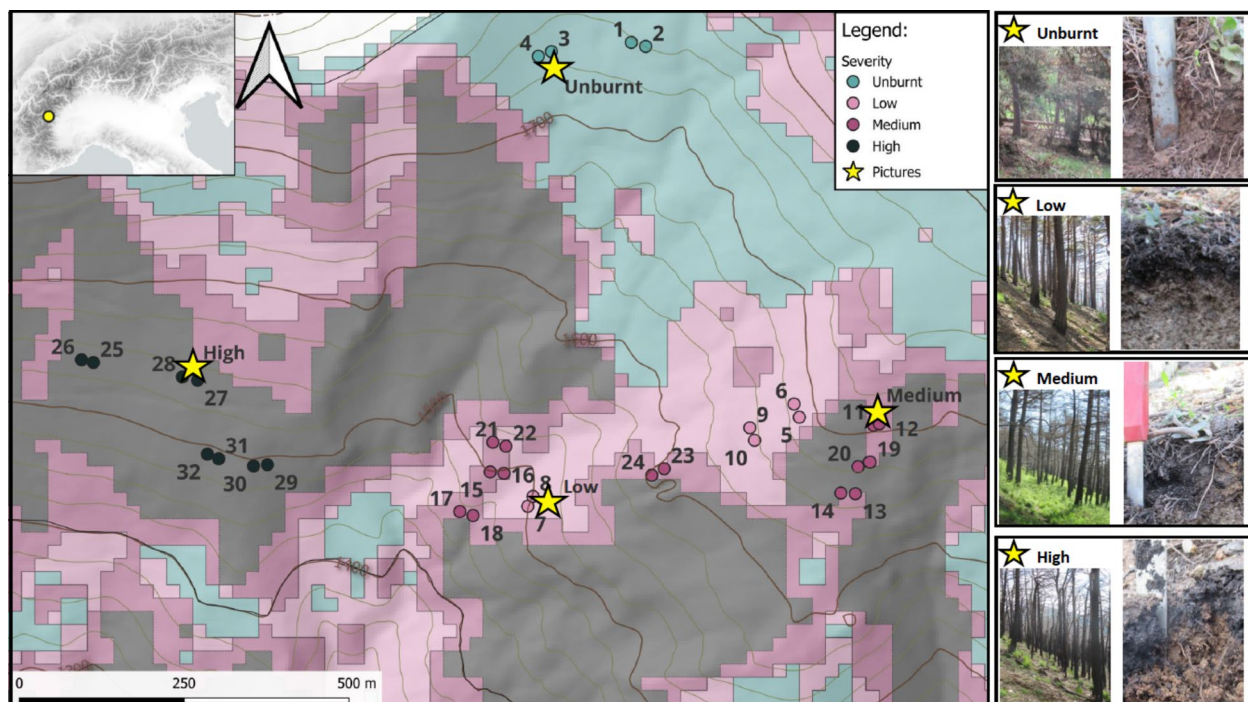


Fig. 1 Location of the study area and the sampling plots across the severity classes (unburnt, low severity, medium severity, high severity). Points (yellow stars) are where pictures of the stand and topsoil were taken

was ca. 15% in low severity, 40% in medium severity, and 100% in high severity patches, and the litter layer had consumption of ca. 50% in low severity, 65% in medium severity, and 90% in high severity patches.

A previous investigation carried out in a larger area (from 500 to 2500 m a.s.l.), with an experimental design aimed at inferring the short-term dynamics of post-fire Scots pine regeneration (Mantero et al. 2023), revealed no significant differences in topsoil features according to severity. This possibly occurred also because of the confounding effect of mixed lithologies, which we avoided in the current study by restricting our sampling to a smaller, homogeneous area. Three years after the fire, in 2020, Scots pine regeneration was abundant (>25,000 seedlings/ha) only within ca. 60 m from the surviving seed trees (Mantero et al. 2023).

Sampling design

We stratified plots among burn severity classes to broadly represent fire effects in the study site, with a proportional allocation of plots in the four severity classes (four plots in unburnt, six in low severity, 14 in medium, eight in high), as similarly performed in previous studies (Dzwonko et al. 2015; Fernández-García et al. 2019), for a total of 32 plots (Fig. 1). We

established the plots (circular, 2 m radius) in May 2020, three years after the Susa fire. The control plots, i.e., unburnt condition, presented no evidence of combustion or charring.

Mantero et al. (2023) assessed the abundance of Gramineae, forbs, shrubs, bare soil, coarse woody debris (CWD), and rock fragments in each of these plots (values expressed as a percentage of ground cover). Our plots presented an average slope of $24 \pm 6^\circ$. The unburnt plots were located upslope (elevation >1730 vs. <1705) (Fig. S2) and, according to the severity class, the plots differed for cover of Gramineae (average of 37% in unburnt vs. <8% in burnt plots, Fig. S2) and shrubs (average of 38% in high severity vs. <10% in the other plots). Differences in the exposure of bare soil among severity classes were at the limit of statistical significance ($P=0.053$), and it was highest in low severity areas (average of 82%), as seen in Fig. 1. An exemplificative picture of ground cover over a small surface area ($10 \times 10 \text{ cm}^2$ grid) is visible in Supplementary information (Fig. S3). It shows the paucity of litter cover in all the burnt plots.

In each plot, we sampled the top centimeters of soil, separating the superficial blackish layer (surface horizon), and the following 5 cm (sub-surface horizon). We opted for sampling by pedogenetic horizons to avoid diluting the effects induced by the passage of fire.

Soil analyses

Moisture and soil color

We evaluated moisture after oven-drying (105 °C until reaching constant weight) and expressed it on a moist mass basis. We air-dried the soil samples, sieved them (2 mm), and ground (0.5 mm) a portion of each.

We expected soil color to reflect the degree of organic matter combustion (Sazawa et al. 2013); thus, we measured the color of the surface horizon in the lab following existing protocols (Fu et al. 2020; Heil et al. 2020), as described below. We evenly distributed the soil over a flat surface and took a picture with a digital camera from a standard distance of ca. 20 cm. We used the air-dried (Fu et al. 2020) 0.5 mm ground material (Heil et al. 2020) to allow the highest level of homogenization and avoid any possible moisture-related bias. We took photos on the same day (in 1 h) with indoor illumination and placed a X-Rite Color Checker behind the samples to obtain a homogeneous exposure. We processed each image with MATLAB R2024b; we selected a region of interest (ROI) of ca. 700×700 pixels, avoiding the margins (Fig. S4). We converted the color of each pixel in the CIE L*a*b* color system, where the L* parameter expresses the lightness (Fu et al. 2020), ranging from black to white, with lower values for a darker matrix. Moreover, the a* and b* parameters are known to be proportional to the degree of organic matter condensation (relation with H/C and O/C atomic ratios) (Sazawa et al. 2013).

Water repellency

We assessed soil water repellency both in the field and in the lab. In the field, after having exposed the mineral soil surface, we applied the water drop penetration time (WDPT) test, evaluating the time required for the soil surface to incorporate water drops (values expressed in seconds, s) (Letey 1969). We placed five drops of distilled water (volume ca. 50 µL) over a 10×10 cm² grid in each plot; this number of drops is in line with numbers reported in previous studies (3–16) (Doerr et al. 2006; Mataix-Solera et al. 2013; Plaza-Álvarez et al. 2018). We recorded WDPT values up to 3600 s (Tinebra et al. 2019). Data were processed in two ways: (i) averaged, obtaining a single mean value for each plot, and (ii) expressed as a relative frequency of different water repellency classes, which are wettable (0–5 s), slightly hydrophobic (5–60 s), moderately hydrophobic (60–600 s), strongly and very strongly hydrophobic (600–3600 s), and extremely hydrophobic (≥3600 s), following the existing classification by Papierowska et al. (2018).

In the lab, we assessed soil water repellency by measuring the contact angle at the liquid-solid interface (values expressed in °). The contact angle refers to the internal angle of a water drop placed on the soil surface

and reflects the physical interface between both the liquid (in this case water) and the soil itself. Values <90° are assumed to represent a non-hydrophobic surface (Doerr et al. 2009; Letey 1969). We determined the contact angle with a Theta Lite (Biolin Scientific, method: sessile drop) equipped with a digital camera to monitor the drop behavior over time. We delivered deionized water on the sample with a tip size of 200 µL, drop size of 4 µL, and acquired images at 13 frames per second. We performed a preliminary analysis to select the sample fraction to be inspected: 2 mm vs. 0.5 mm. As visible in the Supplementary information (Fig. S5), the 2 mm samples presented an excessive roughness, making the assessment of contact angle not reliable. We thus used finely ground dry 0.5 mm samples placed over a glass slide, pressed the slide with a 100 g weight for several seconds (10–30 s) to flatten the surface, and measured contact angle on 5 drops per sample. We followed the evolution of the drops (and thereby of contact angle) for the first 10–15 s (Kholodov et al. 2018; Papierowska et al. 2018) and observed three distinct behaviors: (i) some soil samples maintained the initial (high) contact angle value, (ii) some samples had a decrease in contact angle at ca. 3 s, and (iii) some samples became totally wettable at ca. 3 s (i.e., the drops were completely absorbed, contact angle reaching 0°) (Fig. S6). We thus adopted the threshold of 3 s, determining contact angle on the 36th frame of each sample. We imported the images in ImageJ.exe software (<https://imagej.net/ij/>) and employed the Contact Angle Tool, calculating contact angle with the ellipse-derived drop fitting method, as in Diehl and Schaumann (2007). As for WDPT data, contact angle (CA) data were both: (i) averaged, obtaining a single mean value for each plot, and (ii) expressed in frequency classes (classification by Papierowska et al. 2018), for the following classes: wettable (<40°), slightly hydrophobic (40–90°), moderately hydrophobic (90–110°), strongly and very strongly hydrophobic (110–130°), extremely hydrophobic (≥130°). We carried out the water repellency analyses both on surface and sub-surface horizons.

Chemical properties

We determined soil pH and electrical conductivity (EC) in a soil:deionized water suspension (1:2.5, w:v) (van Reeuwijk 2002). We measured total soil C and N by dry combustion (Unicube CHNS Analyzer, Elementar, Langenselbold, Hesse, Germany) and determined organic C (OC) after inorganic C removal by HCl (Harris et al. 2001). These analyses were carried out both on surface and sub-surface horizons.

We adopted the Kurth-MacKenzie-DeLuca (KMD) method to estimate PyC concentration in the surface soil samples (Kurth et al. 2006). Briefly, 1 g of 0.5 mm ground

sample was subjected to digestion (20 mL 30% H_2O_2 and 10 mL 1 M HNO_3) in glass flasks placed on a hot plate. After swirling and letting them rest (30 min), we slowly brought the flasks to a temperature of 100 °C and left at 100 °C for 16 h. After cooling and filtering (Whatman #4), we collected the residual and oven-dried it (105 °C), weighed it, ground and homogenized it. We determined the C content by dry combustion taking into consideration the mass loss (Abney et al. 2019; Buma et al. 2014). We determined plant-available P by the bicarbonate-extraction method (Olsen 1954). We carried out these two analyses on the surface horizons.

We determined particle size distribution by sieving and sedimentation after sample dispersion with Na-hexametaphosphate (Gee and Bauder 1986), adopting the following size-classes: coarse sand (2000–200 μm), fine sand (200–50 μm), coarse silt (50–20 μm), fine silt (20–2 μm), and clay (<2 μm). This analysis was carried out on sub-surface horizons.

Statistical analyses

We considered the surface and sub-surface horizons separately and compared soil properties (response variables) using one-way ANOVA (effect=severity, P value threshold set at 0.05). For the properties analyzed both in the surface and sub-surface horizons, we inspected differences by the two-way ANOVA (effects=severity and horizon type). Tukey's HSD was used for pairwise comparisons. We calculated correlation coefficients between continuous variables by Pearson correlation analysis. We employed principal component analysis (PCA) on scaled and centered data to identify the main soil properties explaining dataset variance, and used biplots to depict the distribution of plots according to the severity gradient. We also calculated contributions (%) of the variables to the principal components. We used the permutational analysis of variance (PERMANOVA) of the Bray-Curtis distance matrix to test the differences in soil properties among burn severity classes (threshold set at 0.05), setting the number of permutations to 999.

As water repellency is a sensitive property often evaluated in fire-affected landscapes, we further investigated which plots presented the greatest hydrophobicity. To interpret the water repellency of the study area, we focused on the contact angle measure, considering it a more reliable indication than WDPT (since the contact angle analysis is performed in the lab, at standardized soil moisture, on a more homogeneous size fraction). According to the classification in use (Papierowska et al. 2018), a soil can be considered at least strongly hydrophobic if it has contact angle values >110°. Therefore, we quantified the relationship between a high surface water repellency (response variable: plots with average CA >110°)

and the following explanatory variables: soil properties of the surface horizon (thickness, moisture, pH, color, OC, N, C/N ratio, PyC/OC, available P), ground cover types (i.e., Gramineae, forbs, shrubs, bare soil, CWD, rock fragments), and fire severity classes, employing a random forest (RF) regression model. Prior to RF application, we performed predictors selection using PCA and correlation analysis. We split the dataset in train and test data (70:30 ratio) and tuned the number of trees to reduce the out-of-bag estimate of error rate. We carried out ten-fold cross validation to evaluate model performance. We assessed the accuracy by coefficient of determination (R^2) and root means square error (RMSE). We visualized relationships between the expression of a high water repellency and the top explanatory variables with partial dependence plots (Ehrlinger 2016). We performed all data analyses with R studio (R version 4.2.3.).

Results

The surface organo-mineral horizons presented variable thickness and moisture content (Fig. 2a and b). The pH was significantly higher in the plots that underwent burning (Fig. 2c), while the electrical conductivity did not vary significantly (Fig. 2d). Soil lightness (L^* color parameter) did not differ across the severity levels, due to the large variability in medium and high severity plots (Fig. 2e). The OC (Fig. 2f) and N (Fig. 2g) displayed decreasing trends with increasing severity, but they were not significant. Soil L^* was negatively related to OM content (OC vs. L^* : $R=-0.40$ and $P<0.05$; N vs. L^* : $R=-0.55$ and $P<0.01$). The C/N ratio (Fig. 2h) showed no significant variations according to fire severity. The total amount of PyC did not present a clear distinction between burnt vs. unburnt plots (Fig. 2i), while its proportion with respect to OC (PyC/OC) demonstrated an enrichment in the burnt plots (Fig. 2j). The available P was highly variable across all plots (Fig. 2k). The average infiltration time measured by WDPT was far higher in the unburnt plots (Fig. 2l) but, due to the high data dispersion, differences in mean values were not statistically significant. The contact angle was highly variable as well (Fig. 2m), especially in low severity plots. The a^* and b^* parameters of the CIE $L^* a^* b^*$ color system revealed that unburnt topsoils were clustered apart from the burnt ones (Fig. 3, $R=-0.40$ and $P<0.001$) and were associated with higher a^* and b^* values.

The sub-surface horizons were moister in unburnt plots than in low and medium severity plots (Fig. 4a). The pH was higher in medium severity (Fig. 4b), while the lowest electrical conductivity values were found in high severity plots (Fig. 4c). The OC and N were lower in the sites that underwent burning (Fig. 4d and e), irrespective of the severity degree. The C/N ratio

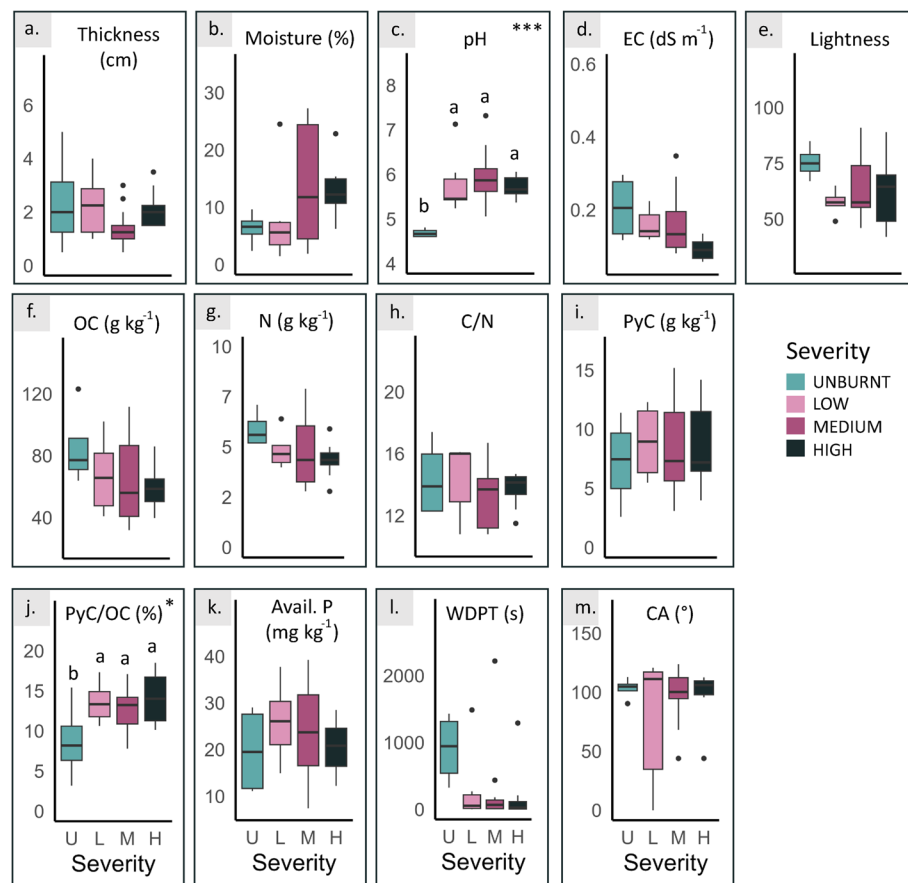


Fig. 2 Characterization of surface horizons according to severity class (unburnt, low severity, medium severity, high severity). **a** Thickness (cm), **b** moisture (%), **c** pH, **d** electrical conductivity (EC, dS m^{-1}), **e** soil color L^* parameter, **f** organic carbon (OC, g kg^{-1}), **g** nitrogen (N, g kg^{-1}), **h** carbon to nitrogen ratio (C/N), **i** pyrogenic carbon (PyC, g kg^{-1}), **j** amount of pyrogenic carbon to organic carbon (PyC/OC, %)*, **k** available phosphorus (P, mg kg^{-1}), **l** water drop penetration time (WDPT, s), **m** contact angle (CA, $^{\circ}$). Letters indicate statistically significant mean values (* for $P < 0.05$, ** for $P < 0.01$, *** for $P < 0.001$)

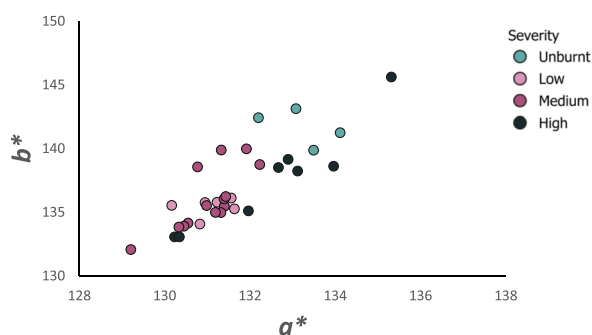


Fig. 3 Relationship between the a^* and b^* parameters of the CIE $L^*a^*b^*$ color system. Color according to severity class (unburnt, low severity, medium severity, high severity)

was lower in the controls and displayed an increasing but not significant positive trend with fire severity (Fig. 4f). The WDPT was on average higher in

unburnt plots (Fig. 4g) and still presented a huge data dispersion. The contact angle was highly variable in medium severity plots and far less in low severity plots (Fig. 4h). The soils had a sandy-loam texture (average sand = 65%, average silt = 27%), with some differences according to severity (Fig. 4i): the unburnt plots presented a greater content of fine sand, while the low severity plots presented more fine silt, and the medium severity plots presented more clay.

Comparing the surface and sub-surface horizons, the surface horizons had less moisture (average of $<10\%$ vs. $>16\%$, $P < 0.05$), were richer in OC (average >69 vs. <34 g kg^{-1} , $P < 0.001$) and N (average >4.9 vs. <2.7 g kg^{-1} , $P < 0.001$), and had an average double contact angle value (94° vs. 40° , $P < 0.001$). Values of pH, electrical conductivity, C/N, and WDPT did not statistically differ between horizons. Considering the division of water repellency values in classes (Papierowska et al. 2018), the surface horizons were more hydrophobic than the sub-surface

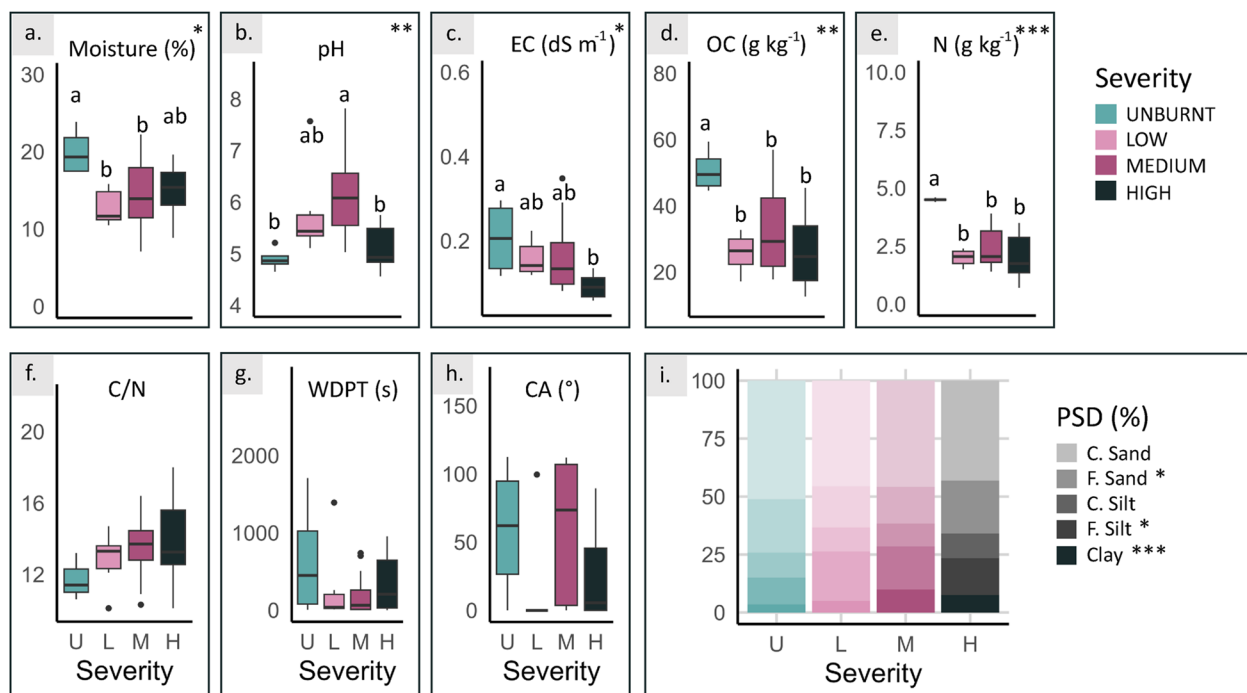


Fig. 4 Characterization of sub-surface horizons according to severity class (unburnt, low severity, medium severity, high severity). Abbreviations for variables are as in Fig. 2. **a** Moisture (%), **b** pH, **c** EC (dS m^{-1}), **d** OC (g kg^{-1}), **e** N (g kg^{-1}), **f** C/N, **g** WDPT (s), **h** CA ($^{\circ}$), **i** particle size distribution (%). Letters indicate statistically significant mean values (* for $P < 0.05$, ** for $P < 0.01$, *** for $P < 0.001$)

ones, both according to the WDPT test (Fig. 5a vs. b) and the CA method (Fig. 5c vs. d). The WDPT and CA values were not significantly related (for all the data together: $R = 0.12$ and $P > 0.05$; within the surface horizon: $R = -0.02$ and $P > 0.05$; within the sub-surface horizon: $R = 0.26$ and $P > 0.05$).

The PCA (Fig. 6) showed that most of the variance of the dataset was explained by pH of the surface and sub-surface horizon, followed by WDPT of the surface horizon, N content of the sub-surface horizon, and available P, content of pyrogenic C, and soil color of the surface horizon (PCA in Fig. 6a and b). According to these properties, the burnt plots were significantly different from the unburnt ones and oriented towards a darker surface soil color, lower WDPT and N contents, and higher values of PyC/OC, pH, and available P.

The most water repellent surface horizons, with an average contact angle value above 110° —corresponding to at least strongly hydrophobic soils—were found in low severity plots (Fig. S7). After having selected the predictors explaining this behavior (by PCA, in Fig. S8), the RF model showed that these soils were associated with high C/N values (Fig. 7a and b), more abundant organic C (Fig. 7c), intermediate values of PyC/OC (Fig. 7d), and a high percentage of bare soil exposed (Fig. 7e). The plots with the highest water repellency of the surface horizon

presented a patchy spatial distribution, both in the case of observed values and RF-estimated ones, with good agreement between the observed and the estimated values (Fig. S9, $R^2 = 0.75$, $\text{RMSE} = 0.25$).

Discussion

Are soil properties different in burnt plots with respect to unburnt plots?

Three years after the wildfire, the studied Alpine Scots pine forest soils still presented clear signs of fire effects, consistent with our hypothesis. As in other studies carried out in fire-affected coniferous forests (Dymov et al. 2021), we found an organo-mineral pyrogenic horizon of ca. 2 cm. The a^* and b^* color parameters of this horizon, which are known to be proportional to the degree of organic matter condensation (relation with H/C and O/C atomic ratios) (Sazawa et al. 2013), revealed that burnt topsoils were clustered apart from the unburnt ones and were associated to more condensed organic matter forms (Sazawa et al. 2013). The color lightness seems instead more related to OC content than to the disturbance, as already observed in the literature (Heil et al. 2020).

Our results contrasted the common observation that pH, a soil property that recovers rapidly after a fire, frequently returns to its initial value within 3 years (Capra et al. 2018; Certini 2005). The persistence of the pH

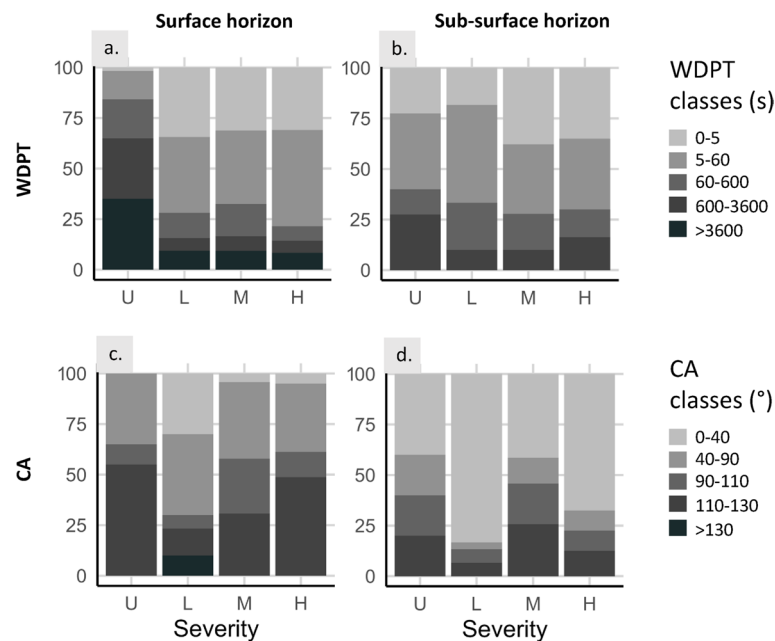


Fig. 5 Frequency of water repellency classes as assessed in the surface and sub-surface horizons by **a, b** WDPT and **c, d** CA

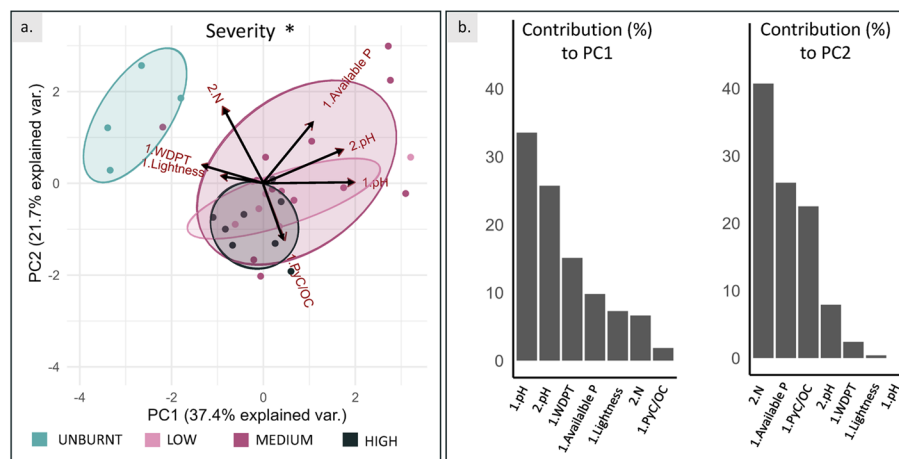


Fig. 6 **a** Biplot of soil properties of surface (1.) and sub-surface (2.) horizons according to severity class (unburnt, low severity, medium severity, high severity), and **b** contributions of variables to PC1 and PC2

increase in our study site can be related to the absence of new organic matter inputs in the years following the fire, which would have acidified the topsoil after the fire-induced base cations release. A paucity of litter cover was observed in almost all burnt plots (as can be inferred from the photographs in Fig. 1) and was not exclusive of the areas where tree mortality was high.

Soil water repellency, another dynamic soil property, was greater in surface horizons than in sub-surface horizons both according to WDPT and CA, as found also by other authors, attributed to the higher content

of OC in the topsoil (Badía-Villas et al. 2014; Doerr et al. 2009; Yang et al. 2022). We did not expect a direct relationship between WDPT and CA, since WDPT is known to be highly variable (Mataix-Solera et al. 2013; Tinebra et al. 2019). Also, samples with the same contact angle can have very different WDPT values (up to 1 order of magnitude of difference) (Negri et al. 2021). However, comparisons between lab-determined WDPT and CA can give moderate correspondence once both measures are grouped in water repellency classes (Papierowska et al. 2018). The absence of a relationship

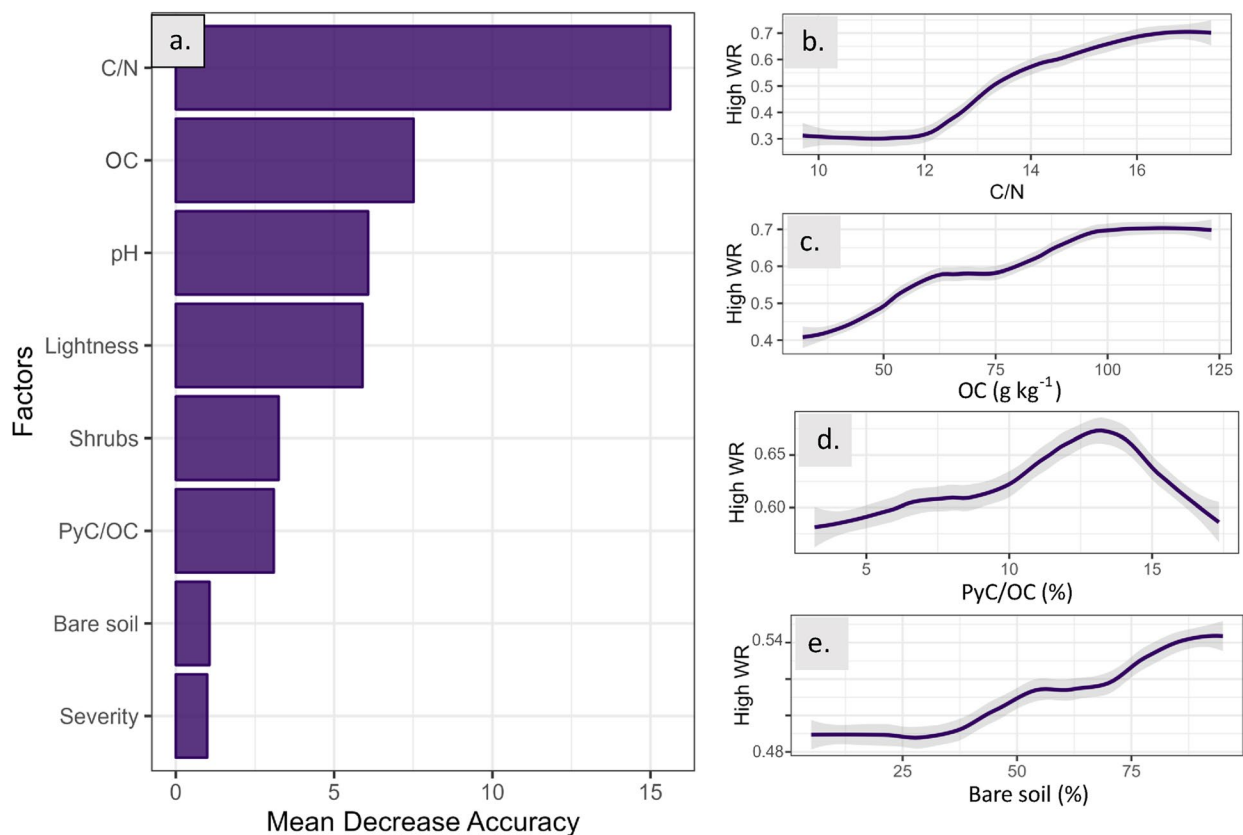


Fig. 7 **a** Variable importance plots and **b, c, d, e** partial dependence plots (on the y axis: 0–1 probability of occurrence of a high surface water repellency, WR) as a function of the top explanatory variables (on the x axis): carbon to nitrogen (C/N) ratio, organic carbon (OC, g kg⁻¹), amount of pyrogenic carbon to organic carbon (PyC/OC, %), bare soil exposure (%)

between field water repellency and lab contact angle was already documented in the literature (Doerr et al. 2009) and testifies that, since we carried out WDPT in field conditions and performed contact angle in the lab on air-dry finely ground soil, the difference in size fraction and moisture likely affected the measurement (Doerr et al. 2009; Plaza-Álvarez et al. 2018; Vogelmann et al. 2017; Yang et al. 2022). Considering the field measurement of water repellency (by WDPT), we conclude there was no increase in hydrophobicity as a consequence of fire either in the surface horizon or in the sub-surface horizon; we rather observed a decrease in water repellency in burnt plots. We partly expected this: Scots pine forests are known to be characterized by a naturally high soil water repellency (Šurda et al. 2021), especially if soils have a coarse texture (Doerr et al. 2000). In unburnt plots, in fact, soil water repellency was as high as in comparable nearby unburnt Scots pine forest soils (Negri et al. 2021). The heat of the fire likely altered organic matter and lead to the fragmentation of long-chain aliphatic compounds, decreasing the naturally high soil water repellency (Badía et al.

2017). Alternatively, if any sub-surface water repellent coating was generated by the fire, they were disrupted in three years (Shakesby and Doerr 2006).

Regarding the amount of available P, whose fire-induced changes should persist longer than the changes caused to pH or water repellency, we expected an increase in the available orthophosphate content as a consequence of the mineralization of labile organic P forms (García-Oliva et al. 2018). However, available P was not significantly different in the burnt plots with respect to the unburnt plots in our study area, while in the same period after fire (three years) a significant increase in available P in the topsoil was recorded in other pine forests, both in Mediterranean environments (Moya et al. 2019) and in Poland (Dzwonko et al. 2015). This difference suggests that part of the newly formed orthophosphate-P underwent runoff, which is plausible considering the steep slopes of the area. Indeed, it has been documented that water bodies become enriched with P following forest fires (Stein et al. 2012).

The OC content was expected to recover more slowly than the previous properties, as several authors observed a persistent decrease in soil OC proportional to the fire

severity (Certini 2005; Vega et al. 2013). Others found no difference between unburnt and burnt sites (Almendros et al. 1990), and others noted that the fluctuations in soil organic matter content are influenced by deposition, incorporation, and mineralization of organic compounds occurring at different rates in locations that, despite being spatially close, underwent different burn severities (Cui et al. 2014). In our surface soil horizon, the incorporation of partly combusted organic matter into the topsoil can be deduced from the absence of significant OC losses. The sub-surface burnt plots, in contrast, were characterized by lower OC contents, in line with the sensitivity of temperate coniferous forest soils to heat-induced organic matter loss (Li et al. 2021). This sensitivity is related to the high contents of poorly stabilized particulate organic matter in these environments (Pellegrini et al. 2022).

The lower N contents in burnt plots with respect to the unburnt plots can be explained by two mechanisms. Either the burnt soils of all severity classes underwent temperatures so high that they decreased the N pool (Tecimen and Sevgi 2011) instead of leading to an enrichment in thermostable N-containing organic compounds (Franco et al. 2018), or organic N was converted into inorganic mobile forms (i.e., nitrate) (McGranahan et al. 2022). Following the second mechanism, these inorganic forms were either absorbed by vegetation (Knicker 2007) or leached (Bodí et al. 2014). The C/N ratio did not undergo any significant changes in the burnt plots due to a concomitant decrease in both OC and N, implying that the temperatures were likely high enough to cause N losses (Knicker 2007). In a similar wildfire-affected Scots pine forest, Vacchiano et al. (2014) found a decrease in both soil OC and N contents only in the patches of high severity fire (Vacchiano et al. 2014). The losses of N in all our burnt plots suggest a higher fire severity in our area, attributable to lower soil moisture content at the time of fire spread.

The quality of organic matter, another property that recovers very slowly, appears to be affected by the fire as well, with the burnt plots displaying a higher proportion of pyrogenic carbon. An average increase by ca. 40% of soil PyC/OC was observed for wildfires occurring in coniferous forest (Li et al. 2021), and we obtained an average increase of 53%.

Finally, the burnt plots presented a slightly finer apparent texture presumably because of soil heating. There is no agreement with this result in the literature, as some studies found no effects of the fire on particle size distribution (Capra et al. 2018), while others found evidence of cementation (Vacchiano et al. 2014) or disaggregation (Andreu et al. 2001). In our case, the breakdown of sand-sized aggregates likely occurred since organic matter is the main binding agent in these montane forest soils, and the heat generated during the fire affected it (Negri et al. 2024).

Is there a direct proportionality between the mapped fire severity classes and fire-induced changes in soil properties?

There was an overall lack of proportionality between the fire severity classes and the degree of modification in soil properties. The thickness of the surface pyrogenic horizon was uncorrelated with the assigned degree of the disturbance, and soil OC contents were not proportional to fire severity class. Significant differences among the severity classes existed in the sub-surface horizon, but the burnt plots presented lower OC contents regardless of the fire severity degree. The high severity plots did not even display a significantly higher abundance of pyrogenic carbon.

The C/N ratio was one of the few parameters that showed a clear (though non-significant) trend of increasing C/N values with increasing fire severity in the sub-surface horizons, which may hint at the translocation of PyC to lower horizons (Abdalla et al. 2021). PyC is also known to accumulate in very deep soil horizons (Velasco-Molina et al. 2016), and in our sandy mountain forest soils, part of the PyC possibly moved downward throughout the soil pore network (Hilscher and Knicker 2011).

The plots belonging to the high severity class presented abundant PyC/OC, but they share this feature with other plots belonging to lower severity classes. A higher PyC content is known to be associated with burning of organic matter at high temperatures (Maestrini and Miesel 2017), so it is likely that all these plots (not exclusively those of high severity) underwent an intense warming. The high contents of PyC of the plots not assigned to high severity may be related to the great variability in fire behavior at small scales. This variability is a common feature of fire-affected landscapes, and its effects were observed for various soil properties like pH (Capra et al. 2018), OC content (Cui et al. 2014), and soil respiration (Martínez-García et al. 2017).

We observed non-null PyC/OC also in unburnt plots, and this could be caused by the presence of aromatic compounds in the native organic matter of these soils. Uncharred Scots pine needles can indeed contain ca. 50 g kg⁻¹ of C that is recovered as PyC (Maestrini and Miesel 2017). Although we did not identify any sign of fire passage in the unburnt plots, we cannot exclude that a fire has occurred a long time ago (i.e., decades ago, before 1997, when the fire monitoring service in this region was first established), since PyC has residence times of millennia (Czimczik et al. 2005; Singh et al. 2012).

We expected the amount of the available orthophosphate form to be proportional to soil burn severity (Vega et al. 2013). In our case, available P was higher in low and medium severity plots with respect to the unburnt and high severity plots (ca. 27 vs. 20 mg kg⁻¹). This pattern

suggests that the highest temperatures were reached in small smoldering spots (with long residence times) and not in areas of crown fire (Bento-Gonçalves et al. 2012). Alternatively, the absence of proportionality between PyC/OC and available P may be explained by the precipitation of unextractable calcium phosphate in areas of intense soil warming (Certini et al. 2005; Pereira et al. 2012).

Finally, water repellency measured by contact angle did not show a clear trend with the assigned fire severity classes. The topsoils with the highest contact angle values ($CA > 110^\circ\text{C}$) were common in plots of low severity, with a rather patchy distribution; again, this testifies to a high variability at fine scales. In the low severity plots, a combination of high C/N values (>14), high organic carbon contents ($>75\text{ g kg}^{-1}$), high percentage of bare soil exposure ($>50\%$), and intermediate PyC/OC values (of ca. 12%) triggered the insurgence of a very high topsoil water repellency. The soil possibly underwent intermediate temperatures that were high enough to induce the formation of water repellent coatings, but not too high to lead to an excessive organic matter loss.

Limitations, implications, and future perspectives

A limitation of the study was that we used a space-for-time approach to infer the changes induced by fire in Alpine Scots pine forest soils. To minimize bias, we selected a fairly homogeneous area for all the factors of pedogenesis but the disturbance. Conducting a study after some years (three) allowed us to examine the delayed recovery of soil properties in the topsoil of this environment. Future research to determine the amount of time required for soil to reach pre-fire conditions will require a much longer period (maybe more than 10 years). In burnt plots, the litter layer was consumed and not replenished in the years following the disturbance. Respect to the unburnt controls, this altered the composition of the organic matter yearly incorporated in the soil. The quality of organic matter supplied to the soil indeed depends on the re-establishment of vegetation (Rovira et al. 2012), and the soil C and N pools may take ca. 10 years to recover in fire-affected landscapes (Li et al. 2021). In Poland, in a similar forest ecosystem, Scots pine regeneration began to establish eight years after fire, reaching a cover of 30% only 13 years after the fire event (Dzwonko et al. 2015). Meanwhile, shrubs dominated the severely burnt sites (Dzwonko et al. 2015). Other study sites showed similar results, where shrub density was highest in medium and high severity patches in the time-window of 1.5 to 10 years after the fire (Yang et al. 2022). In the Alps, the revegetation process is gradual and affected by site conditions, competition with understory vegetation, density, and spatial distribution of safe sites

for regeneration, requiring up to 16 years to have substantial Scots pine recolonization (Vacchiano et al. 2015). Recolonization may then be delayed or unsuccessful in case of large and severe fires (Marzano et al. 2013; Moser et al. 2010), in presence of a lithology as harsh as serpentine (Burgess et al. 2015), and where post-disturbance management interventions hinder recovery dynamics (Beghin et al. 2010).

We did not make inferences about erosion and runoff, since our measurements were presumably taken after the period of highest vulnerability to soil loss. Runoff usually peaks one or two years following the fire (Meyer et al. 2001) and then decreases, with a low probability of extreme runoff generation in the third year (Ebel 2020). However, in our environment, we observed a considerable portion of bare soil in burnt areas (ca. 80% of ground cover in low severity plots, as also notable in Fig. 1). With fire occurrence, the interception of precipitation by vegetation can drop by 40–80% (McGuire et al. 2016; Stoof et al. 2012), and the protective canopy layer, herbaceous and litter layer are usually destroyed (Alcañiz et al. 2018; Andreu et al. 1996). This can cause significant changes in soil structural stability (Moody and Ebel 2014; Pausas et al. 2008) and aggregation (Negri et al. 2024). It also makes the soil vulnerable to further degradation (Doerr et al. 2000; Moody and Ebel 2014; Stiefel et al. 2021).

As a final remark, we would like to highlight that water repellency may give contrasting results when measured in different environmental conditions and on different size fractions. Also, the surface burnt horizons are different from sub-surface ones. We emphasize the importance of sampling according to horizon, rather than uniform depth, in order to properly identify development of soil features and prioritize any possible interventions.

Conclusions

Fire-affected topsoils of Alpine Scots pine forests are different from the unburnt condition even three years after fire occurrence and display the presence of a pyrogenic surface horizon. The main differences involve not only properties that are known to recover slowly, but also properties that usually recover more rapidly. They are higher pH than unaffected areas, overall lower organic carbon and nitrogen contents, finer texture, and enrichment in pyrogenic carbon. The changes in these features do not correspond to the degree of remotely sensed fire severity, revealing that remote sensing of fire severity, even when calibrated with field measurements, can fail to detect post-fire soil conditions.

In these soils, surface hydrophobicity has a highly patchy distribution, and a high water repellency is not consistently associated with higher fire severities. Rather, the unburnt soils were the most water repellent

when water repellency is measured in the field. In contrast, lab evaluation of water repellency measured at standard soil moisture and on a fine soil size fraction, reveals that higher water repellency ($CA > 110^\circ$) emerges in areas that underwent low fire severity, primarily because of the presence of abundant uncharred organic matter deriving from Scots pine litter.

The full recovery of soil properties in these environments is expected to occur slowly as a result of natural successional dynamics. Any post-disturbance management impeding, altering, or slowing down natural recovery should be carefully evaluated, while appropriate restoration interventions should be applied, when needed, to promote the regeneration dynamics.

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s42408-026-00493-5>.

Supplementary Material 1.

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Authors' contributions

Conceptualization: EB, RM, MG, MED, SN; Methodology: SN, EP; Formal analysis: SN, MED; Investigation: SN, GM; Data curation: SN; Validation: SN; Writing – original draft: SN; Writing – review and editing: all the authors; Funding acquisition: EB, MB, PQ.

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Data availability

The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare no competing interests.

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