



The effects of wildfire frequency on post-fire soil surface water dynamics

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Abstract

Increasing wildfire frequency in the Mediterranean Basin could affect future plant–soil–water–dynamics. The capacity of soils to retain water is a key parameter affecting plant post-fire regeneration. Yet, few research has looked at how different soil properties related to water retention is affected by increasing wildfire frequency. This study aimed at understanding the relationship between wildfire frequency, soil–water-related properties and the dynamics of surface water in soils. To this, after a 2012-summer wildfire in Portugal, three sets of three replicate maritime pine stands with contrasted wildfire frequency were selected (0 vs. 1 vs. 4 fires since 1975). At each of the nine study sites, three re-sprouter shrubs and neighbouring bare soil were chosen (54 microsites). There, soil cover, soil–water retention curves and surface (0–5 cm) soil–water-related properties (texture, bulk density, organic matter content, soil moisture, soil surface water repellency) were monitored for one year. Furthermore, records of post-fire soil moisture dynamics were analysed continuously using 72 probes installed at 2.5 and 7.5 cm depth. The hillslopes affected by 1 fire showed higher plant recovery than the 4 fires hillslopes. During the dry season, the threshold for water stress was reached 17 days sooner in the 4 fires hillslopes, and also 10 days sooner on bare microsites. Periods of plant water stress were longer and bare soil patches size bigger. The increase in wildfire frequency promoted high soil organic matter contents but less available water content, stressing the importance of soil organic matter quality characterization in water-related properties.

Keywords Wildfire frequency · Soil moisture content · Effective soil–water content · Soil–water properties · Soil cover

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Introduction

Mediterranean ecosystems are subjected to the combined effect of both climate and land-use changes, leading to an increase on wildfire frequency (Pausas and Fernandez-Muñoz 2012). Warmer and drier scenarios may transform the high-biomass forestry systems into low-biomass shrublands (Acacio et al. 2009; Pausas and Bond 2019). On the other hand, the increase in wildfire frequency in fire-prone ecosystems such as Portugal can lead to land degradation through long-term cumulative effects on vegetation and soil (Vallejo and Alloza 2015). Several studies had shown delays in plant re-generation (Mayor et al. 2007; Gimeno-García et al. 2007), declines in plant productivity (Eugenio and Lloret 2004), changes in plant composition and structure (Eugenio et al. 2006a; Santana et al. 2010, 2016) as well as changes on soil hydrology (González-Pelayo et al. 2010; Shakesby 2011), soil fertility (Campos et al. 2016; Hosseini et al. 2017) and soil carbon balances (Santana et al. 2016). However, studies evaluating the cumulative effects of wildfire frequency on soil–water dynamics are scarce (Mayor et al. 2016).

Several research have found that soil moisture content (SMC) used to be higher in burnt than in unburnt forests, and this was attributed to the loss of the vegetation capacity to intercept, retain and evaporate water from precipitation (Soto and Diaz-Fierros 1997; Cardenas and Kanarek 2014; Nolan et al. 2014). Vegetation type and distribution affects SMC due to changes in soil evaporation (Katra et al. 2007), plant transpiration, root activity and organic matter incorporation, which in turns affects the soil hydraulic conductivity (Famiglietti et al. 1998). On the other hand, the volume of water stored in the soil that is effectively available for plants (effective soil–water content (ESWC); Porporato et al. 2002) depends on climate (rainfall regime, temperature and evapotranspiration), vegetation (rooting depth and plant physiological characteristics) and soil properties (soil texture and SOM, which mainly define the pF-derived parameters, such as permanent wilting point (PWP) and field capacity (FC)) (Ankenbauer and Loheide 2017). The soil available water content (AWC; the difference between FC and PWP) has been calculated to assess the plant water stress (van Genuchten et al. 1991; Stolte 1997). Plant water stress is considered maxima if the soil–water is below PWP, and minima at FC (Ebel 2012; Ankenbauer and Loheide 2017). By affecting SOM and soil texture, wildfires can have an effect in soil hydrological properties and the volume of soil–water available for plants, but the effects of wildfire frequency on soil–water availability are not totally understood (González-Pelayo et al. 2006). Furthermore, contradictory conclusions in literature are found as regards the

soil–water availability in burnt areas; Silva et al. (2006) found SMC decreases for the upper layers, presumably due to the higher evaporation and heating of soil after fire, while Soto and Diaz-Fierros (1997) and Nolan et al. (2014), describes higher SMC due to plant water uptake following fire. According to Shakesby (2011), a deep study in this regard is justified.

The extends at which the increase in wildfire frequency affects the dry/wet switching mechanisms of SMC redistribution are not very well understood. The moisture content on the first 5–10 cm of the topsoil determines the hydrological processes, especially through different states of hydrophilic–hydrophobic topsoil layers (González-Pelayo et al. 2015; Malvar et al. 2016) which are also related to the presence of biotic and abiotic soil cover components (Ruiz-Sinoga et al. 2010a). An increase in wildfire frequency drastically reduces the biotic cover, such as vegetation and litter (Gimeno-García et al. 2007) and enhances the remaining abiotic components, such as bare soil and stones (Ruiz-Sinoga et al. 2010a; Gabarrón-Galeote et al. 2013). As a consequence, it depletes the topsoil organic layers (Eugenio et al. 2006b), increases the soil nutrient losses and finally changes the spatial patterns of bare-vegetation microsites (Mayor et al. 2016; Santana et al. 2016). In addition, the effect of post-fire salvage logging operations also affects the SMC redistribution, by increasing soil compaction and bare soil cover, for sandy granitic (Wagenbrenner et al. 2015; Fernandez and Vega 2016; Prats et al. 2021) and silty schist soil types (Santana et al. 2016; Malvar et al. 2017). All these effects strongly affect the SMC redistribution in frequently burnt areas and compromises the ecosystem resilience. Emergency post-fire mitigation treatments such as mulching (i.e. the application of an organic cover over the soil surface) are envisaged to reduce post-fire soil erosion, but they have also shown positive effects in increasing the soil–water storage capacity (Fernandez and Vega et al. 2016) mainly during dry periods (Prats et al. 2016a, b, 2022). However, to the best of our knowledge, no studies have assessed the extent to which the increase in the organic cover of the soil (by means of litter or plants) will impact the soil–water balance in frequently burnt areas.

The main goal of this research is to assess the effects of wildfire frequency on plant recovery, soil properties, soil moisture content (SMC) and effective soil–water content (ESWC) available for plants during the first post-fire year in fire-prone areas of North-Central Portugal with three different wildfire frequencies: unburnt mature pine forest (no wildfires in the period 1975 to 2012), burnt only once (affected by the 2012 wildfire) and burnt 4 times (affected by 1978, 1985, 2005 and 2012 wildfires). The specific objectives are:

1. To assess the effects of three wildfire frequencies (unburnt, 1 fire vs 4 fires) on vegetation recovery (plant height, stem diameter and stem length), soil cover and soil properties (texture, bulk density, pH, SOM, pF-curves, soil–water repellency) at the end of the first year after the 2012 wildfire,
2. To assess the effects of three wildfire frequencies (unburnt, 1 fire vs. 4 fires) on the seasonal variations of SOM, SMC and ESWC during the first year after the 2012 wildfire through soil sampling campaigns,
3. To compare the effects of two wildfire frequencies (1 fire vs. 4 fires), two microenvironments (plant vs. bare soil) and two soil depths (2.5 and 7.5 cm) on the daily readings SMC and ESWC during the first year after the 2012 wildfire.

Material and methods

Study area and sites

This study was conducted in Várzea, a hamlet near of Viseu (40°46′059″N, 7°51′726″W), North-central Portugal, after a wildfire on 9th September 2012 that burnt through a *Pinus*

pinaster Ait. Forest area. Climate is humid Mediterranean, classified as Csb (FAO 2006), with a dry period from June to August and peak rainfalls between October and March. Mean annual rainfall and temperature are 1200 mm and 14 °C, respectively (SNIRH 2015). The earliest available wildfire records of this fire-prone area are from 1975 to present, and no prescribed fires have been recorded from this time interval (ICNF 2013). Within this area, we selected three 50 × 50 m hillslopes for each of the following wildfire frequencies (Fig. 1): (1) unburnt from 1975 to 2012; (2) burnt 1 time by one wildfire on Sept 2012 (1 fire); and (3) burnt 4 times namely by four wildfires on 1978, 1985, 2005 and 2012 (4 fires). The average wildfire return period corresponded, respectively, to > 39 years, 37 years and 9 years. Soil burn severity for the 2012 wildfire was estimated as moderate at the two burnt areas (using soil visual indicators described in Vega et al. 2013) with all the litter being transformed to a uniform black layer of charcoal, although in the hillslopes burnt 4 times the ash colour was grey in some spots, denoting a higher fire severity in some spots (Table 1). Pre-fire vegetation cover was similar and comparable between sites. The unburnt and the areas burnt once (1 fire) were covered by mature *Pinus pinaster* woodland, with *Pterospartum tridentatum* (L.) Willk., *Calluna vulgaris* (L.)

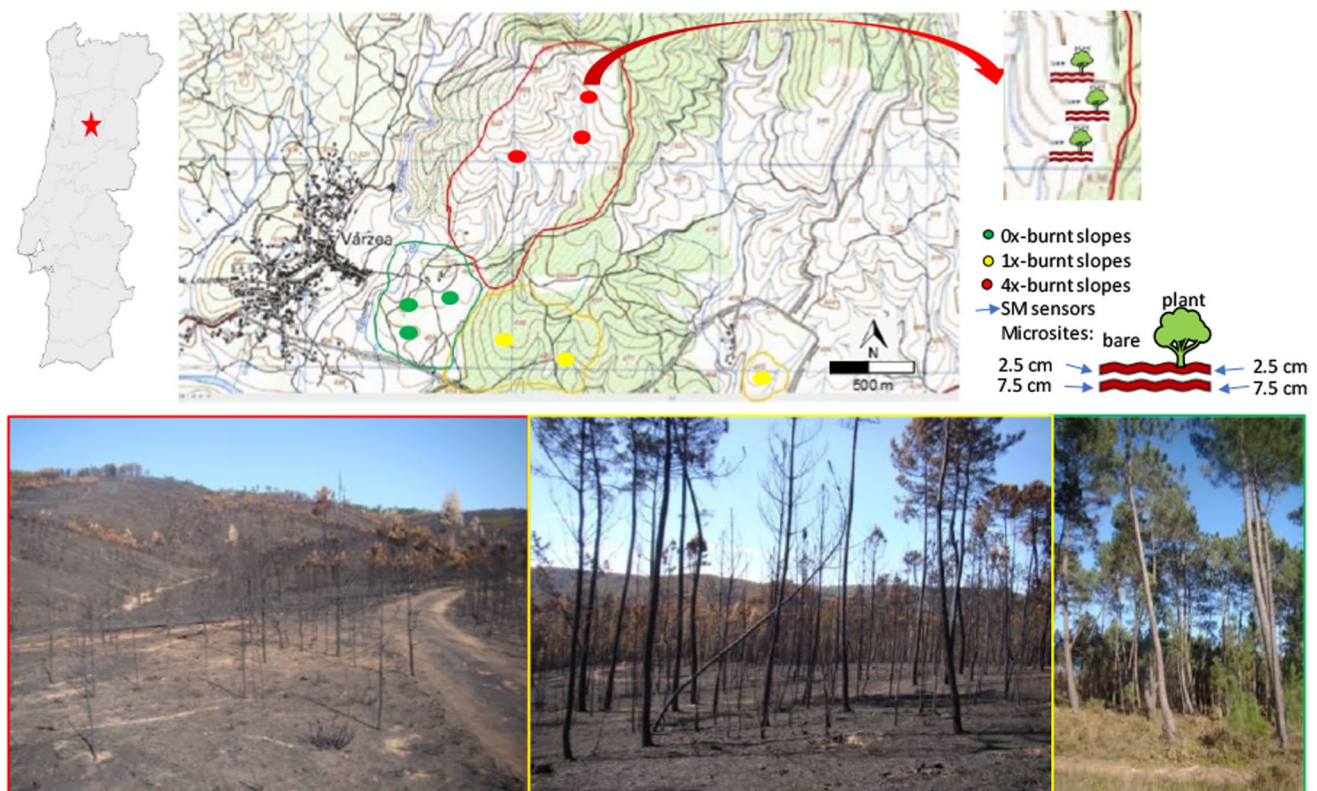


Fig. 1 Experimental set-up showing the unburnt (green circles), 1 fire (yellow circles) and 4 fires hillslopes (red circles) as well as a scheme of the microsites (plant and bare) and soil depths (2.5 and 7.5 cm)

monitored in this study. Bottom pictures (from left to right) correspond to 4 fires, 1 fire and unburnt hillslope

Table 1 General site characteristics, soil type and general soil properties characterization in the Ah topsoil layer (0–5 cm depth) of each site (unburnt, 1 or 4 fires)

| Site | Unburnt | 1 fire | 4 fires |
|-------------------------------------|-------------------------|-----------------|--------------------|
| <i>General characteristics</i> | | | |
| Slope aspect (°) | 210 | 210 | 200 |
| Slope angle (°) | 7–14 | 7–15 | 7–17 |
| Pre-fire vegetation | <i>Woodland</i> | <i>Woodland</i> | <i>Shrubland</i> |
| Fire severity | Unburnt | Moderate | Moderate |
| <i>Soil type and properties</i> | | | |
| Soil type (IUSS, 2022) | Eutric Cambisol (Humic) | Umbric leptosol | Epileptic Umbrisol |
| Soil depth (cm) | 30–35–40 | 15–20–25 | 30–35–40 |
| Texture class | sandy loam | loam | loam |
| Sand/silt/clay (%) | 55a/29b/15a | 46b/38a/16a | 48b/35a/16a |
| Bulk density (g cm ⁻³) | 1.0(0.2)a | 0.9(0.2)b | 1.0(0.1)ab |
| SOM (%) | 13(3) b | 17(1) a | 19(3) a |
| pH | 4.9(0.1)a | 4.6(0.1)a | 4.7(0.1)a |
| <i>Soil hydrological properties</i> | | | |
| pF0(%) | 52(13)a | 56(19)a | 48(10)a |
| pF1(%) | 52(13)a | 55(18)a | 46(10)a |
| pF2(FC; %) | 38(10)a | 40(14)a | 34(7)a |
| pF2.5(%) | 23(4)a | 25(7)a | 25(4)a |
| pF3.2(%) | 12(1)c | 16(3)b | 18(3)a |
| pF4.2(PWP; %) | 5(2)c | 9(3)b | 11(3)a |
| AWC (FC-PWP; %) | 33(11)a | 31(16)a | 23(9)b |
| WHC (mm/profile) | 0.38 | 0.40 | 0.34 |

Mean (and standard deviation) for the granulometric composition (%), $n=54$), bulk density (g cm⁻³, $n=54$), soil organic matter content (SOM in %, $n=54$), pH ($n=54$), pF-curves (pF1, pF2, pF2.5, pF3.5, and pF4.2, in %, $n=54$), available water content (AWC in %, $n=54$) and water holding capacity (WHC, mm/profile) on each area followed by a different letter were statistically different at $p < 0.05$. Comparisons were analysed per mean values as well as per plant and bare microenvironments

Hull and *Agrostis* spp., as the dominant understory vegetation. The areas burnt 4 times (4 fires) were characterized by sparse 7 years old young pines regenerated from the previous wildfire in 2005, and dominated by *Pterospartum tridentatum* shrubs and the co-occurring species *Agrostis* spp., *Calluna vulgaris*, *Cistus* spp. and *Halimium* spp. (Fig. 1).

All selected sites had similar elevation (450–550 m.a.s.l.), slope aspect (200–210° azimuths) and steepness (7–8° on top to 14–17° on bottom slope positions), and the soils were developed from pre-Ordovician schist of the Hesperic Massif. According to IUSS (2015), the soils were classified at the unburnt hillslopes as an association of Eutric cambisol (humic) and Epileptic umbrisol. In the 1 fire hillslopes corresponded to Umbric leptosol and in the 4 fires hillslopes to Epileptic umbrisol. These soils were all acidic sandy loam in the unburnt areas and loam in the 1 and 4 fires areas. Soils were somewhat deeper in the unburnt and the 4 fires hillslopes (30–40 cm) as compared to 1 fire hillslopes (less than 25 cm). Topsoil Ah layers showed similar bulk density and pH, although the unburnt soils showed a somewhat coarser texture and lower SOM than the two burnt soils (Table 1).

Analytical methods

Rainfall

The rainfall data were recorded using 3 rainfall automatic gauges (ECRN-100 from Decagon devices) as well as 3 storage gauges in each of the three areas. Daily rainfall volumes and maximum rainfall intensities in 30 min (I_{30}) were calculated from each automatic gauge.

Field measurements

During early November 2012, three alive re-sprouts of *Pterospartum tridentatum* shrubs were randomly selected along each hillslope (Fig. 1). This target shrub species was selected because it was the most representative species present in all wildfire-recurrence levels (hereafter, plant microsite). In April, July and September 2013, the maximum plant height as well as the length and diameter of five stems (total n plants: $3 \times 9 = 27$ and n stems: $5 \times 3 \times 9 = 135$) were measured. Additionally, a bare soil microsite near to each plant was selected to account for a balanced number of plant/

bare experimental microsites (total of 54 microsites), and to monitor the direct effects of wildfire on plant recovery and soil cover, as well as the indirect effects of wildfire on soil properties.

Soil cover was measured on the six microsites of each hillslope in December 2012 and November 2013. Soil cover was assessed by lying a 0.5 × 0.5 m plot frame with 100-grid points over the microsite, and recording one of the following five categories in each intersection point: litter and vegetation (biotic components), ash, bare soil and stones (abiotic components) (Ruiz-Sinoga and Romero-Díaz 2010; Prats et al. 2019). Soil–water repellency (SWR) was measured in five bare microsites along the hillslope, in only one hillslope per area (unburnt, 1 and 4 fires), at monthly intervals from December 2012 to November 2013 ($n = 5 \times 3 \times 12 = 180$) using the molarity ethanol droplet (MED) method (Doerr et al. 1998). Each measurement involved applying three droplets of increasing ethanol concentration to fresh parts of the soil at the surface Ah horizon, after removal of the surface ash or stones, until infiltration of at least two of three droplets of the same concentration within 5 s. Following Malvar et al. (2016), the nine volumetric ethanol percentage concentrations and, in between brackets, corresponding SWR classes were used: 0% (very wettable), 1%, 3%, 5% (wetable/slightly water-repellent), 8.5%, 13% (moderate/strongly water-repellent); 18%, 24% (very strongly water-repellent), 36% and > 36% (extremely water-repellent).

Soil sampling and laboratory characterization

Study sites characteristics and general soil properties were identified, immediately following the last fire, at each of the nine hillslopes by a proper soil profile description (IUSS 2015) and laboratorial analyses (Table 1). The superficial 0–5 cm of the topsoil was sampled with a 5 cm-height steel core sampler after removing the above ash and/or litter layer. In each hillslope, 6 samples were gathered with 250 cm³ steel cores (3 from shrubs and 3 from bare soil microsites, for a total n of 54 samples). These samples were used to assess soil texture (Guitian & Carballas 1976), bulk density (Malvar et al. 2016), pH (ISO 10390 2005) and soil organic matter (SOM; %) after sieving at < 2 mm and incineration by loss-on-ignition at 550 °C 4 h (ISO 11465 1993). A second set of 54 samples (using 100 cm³ steel cores) was used for the assessment of soil–water retention curves using a sandbox apparatus (Van der Harst and Stakman 1961) and a pressure plate device (Stolte 1997). The amount of water retained in the soil was calculated at five soil–water potentials: –1, –10, –33, –124 and –1550 kPa; corresponding, respectively, to: pF1 (saturation water), pF2 (field capacity, FC), pF2.5, pF3.2 and pF4.2 (permanent wilting point, PWP). Available water content (AWC; %) was calculated as

FC—PWP, once water retained at PWP is not accessible for plant uptake (González-Pelayo et al. 2006; Chen et al. 2007).

Additionally, soil samplings were carried out in other 5 periods (April, June, July, September and November 2013) in order to determine the changes of soil organic matter (SOM) content and the gravimetric soil moisture content in wet and dry periods. Some of the sampling times were not complete, due to the rainy conditions that interrupted the work during some field sampling campaigns (Table 2).

Table 2 *Pterospartum tridentatum* recovery (mean plant height, stem diameter and length) and soil response (soil characterization by mean gravimetric soil moisture content-SMC and soil organic matter content-SOM at 0–5 cm depth) during the drying period of April 2013 to September 2013

| Month | Unburnt | 1 fire | 4 fires |
|---|------------|------------|------------|
| Plant response | | | |
| <i>Height (mm)</i> | | | |
| Apr 2013 | 664(271)b | 68(25)e | 90(38)e |
| June 2013 | 752(284)a | 286(77)d | 308(79)d |
| Sept 2013 | 695(230)ab | 430(146)c | 397(104)c |
| <i>Stem diameter (mm)</i> | | | |
| Apr 2013 | 4.4(1.9)b | 1.6(0.3)f | 1.8(0.4)ef |
| June 2013 | 4.8(2)a | 2.0(0.5)de | 2.4(0.6)cd |
| Sept 2013 | 5(2)a | 3.0(0.9)c | 2.7(0.8)c |
| <i>Stem length (mm)</i> | | | |
| Apr 2013 | 678(315)b | 57(18)f | 76(33)f |
| June 2013 | 722(324)a | 267(88)e | 315(89)de |
| Sept 2013 | 722(331)a | 398(182)cd | 385(105)c |
| Soil response | | | |
| <i>Soil organic matter (SOM; %)</i> | | | |
| Apr 2013 | 11(4)d | 18(5)b | 17(3)b |
| June 2013 | 13(3)cd | 20(5)a | 19(3)ab |
| July 2013 | 14(7)c | 21(5)a | 18(4)ab |
| Sept 2013 | 12(5)cd | 21(6)a | 17(3)b |
| Nov 2013 | – | 18(3)b | 18(2)ab |
| <i>Gravimetric soil moisture content (SMC; %)</i> | | | |
| Apr 2013 | – | – | – |
| June 2013 | 12(3)bc | 11(3)bc | 13(3)bc |
| July 2013 | 9(4)d | 11(5)cd | 5(3)e |
| Sept 2013 | 4(2)f | 3(1)fg | 2(1)g |
| Nov. 2013 | – | 13(4)b | 17(4)a |
| <i>Effective soil–water content (ESWC; %)</i> | | | |
| Apr 2013 | – | – | – |
| June 2013 | 41(30)ab | 27(10)bc | 38(13)ab |
| July 2013 | 34(36)bc | 27(13)c | 16(10)d |
| Sept 2013 | 16(24)d | 7(3)e | 7(3)e |
| Nov. 2013 | – | 26(8)c | 43(11)a |

Mean (standard deviation) values followed by different letters indicate significant differences between unburnt, 1 fire or 4 fires areas at $p < 0.05$

Soil moisture probes

A total of 72 soil moisture sensors (Decagon Inc.) were installed in November 2012 in order to carry out a detailed assessment of the daily variations in volumetric SMC only between the 1 fire and 4 fires hillslopes. These burnt hillslopes were both affected by the 2012 wildfire and differed only in the wildfire frequency (9 years vs 37 years). Therefore, the sensors were installed in the six microsites (plant and bare), at two soil depths (at 2.5 and 7.5 cm depth) in each of the six burnt hillslopes, totalling 72 sensors ($n = 6 \times 2 \times 6 = 72$). The EC5 and GS3 dielectric capacitance soil moisture sensors were connected to Em5b dataloggers and determine volumetric soil moisture content (SMC; v/v) by measuring the dielectric constant of the media using capacitance/frequency domain differences. A capacitance sensor uses the soil as a capacitor element and use the soil charge storing capacity to calibrate to water content. They have a measuring range of 0.0–1.0 ($\text{m}^3 \text{m}^{-3}$) and an accuracy of $\pm 0.03 \text{ m}^3 \text{m}^{-3}$ typical in mineral soil solutions that have an $\text{EC} < 8 \text{ dS/m}$. Besides volumetric water content, GS3 sensors can also measure bulk electrical conductivity in a range of 0–25 dS/m . To avoid influence of the hillslope aspect, all probes were oriented to 200–210° azimuths. Volumetric SMC was measured at 5-min time intervals, which were averaged to daily SMC ($n = 350$, from 21st Nov. 2012 to 6th Nov. 2013). All the EC5 and GS3 sensors were calibrated in the laboratory before installation and the offset differences between sensors were minimized by making use of individual 4-point calibration in fluids with known apparent dielectric permittivity (following user's guide available at www.metergroup.com). In this way, the mutual differences between sensors, which is especially important in the dry range, could be minimized and a high relative measuring accuracy could be obtained (van den Elsen et al. 2014).

Data analyses

The daily average volumetric SMC (%) and the results obtained from pF-curves (FC, PWP) were used to calculate the ESWC (%). It was assumed that the availability of soil–water to plants is at best at FC, and it declines with decreasing SMC (Chen et al. 2007). At PWP, it is generally accepted that the soil–water is no longer available for mesophilic plants. The effective soil–water content (% of the total available water) was calculated using the following formula (Porporato et al. 2002):

$$\text{ESWC} = (\text{SMC}_{\text{actual}} - \text{PWP}) / (\text{FC} - \text{PWP})$$

where $\text{SMC}_{\text{actual}}$ is the average daily volumetric SMC measured with the soil moisture sensors (% volume). FC (field

capacity, % volume) and PWP (permanent wilting point, % volume) were calculated from the pF-curves assessed from the soil samples collected (Van Genuchten et al. 1991).

Linear mixed-effects statistical models (Littell et al. 2006) with repeated measures on plants were used to assess the differences in the plant variables (plant height, stem diameter, stem length) with the wildfire frequency (three levels: unburnt, 1 fire, 4 fires) and the month of measurement (three levels: April, June, September 2013) as the fixed factors, while the plant location was considered as the random factor. In order to assess the differences in SOM, gravitational SMC and ESWC measured through the soil sampling campaigns, linear mixed-models without repetition were constructed. While wildfire frequency, month of measurement (April, June, July, September, November 2013) and microsite (plant, bare) were the fixed factors, the sample location was considered as the random factor. Similarly, to assess the differences in the daily volumetric SMC and ESWC, linear mixed-models with repeated measures on the soil moisture sensors were constructed. Four fixed factors were considered: wildfire frequency (two levels: 1 fire, 4 fires), microsite (plant and bare), soil depth (2.5, 7.5 cm) and season (five levels: autumn 2012, winter 2012–2013, spring 2013, summer 2013, autumn 2013). The individual soil moisture sensors were included as the random factor. The covariance structure of the repeated measures was modelled using a compound symmetry function or autoregressive heterogeneous variance, as it gave the lowest -2 Res Log likelihood model-fitting values (Littell et al. 2006). In order to assure a normal distribution of the model residuals, plant variables, SOM and gravimetric SMC were fourth-root transformed, while daily volumetric SMC and ESWC were log-transformed. Explanatory continuous variables were tested as covariates in a forward selection procedure, including daily rainfall, maximum 30-min rainfall intensity (I_{30}) and accumulated rainfall amount from 1 and until 14 days and were included in the volumetric SMC or ESWC models if they were significant. As the multiple rainfall characteristics are related, they were tested independently and only the variable with highest F-value was included in the model, following the principle of forward selection. Linear regressions and coefficient of determinations (R^2) between the explanatory variables and the daily volumetric SMC and ESWC were calculated.

Differences in soil cover, bulk density, pH, pF-values and on SWR, were tested using mixed-effects models with or without repeated measures similar to the models described earlier, except no covariates were used. If the assumptions for equal variance and normality were not met, as it was the case of SWR, relative frequencies of each class were calculated as the percentage of any ethanol concentration, over the total measurement in each site and time. The nonparametric

Wilcoxon test was used to assess the differences between each wildfire frequency and soil depths.

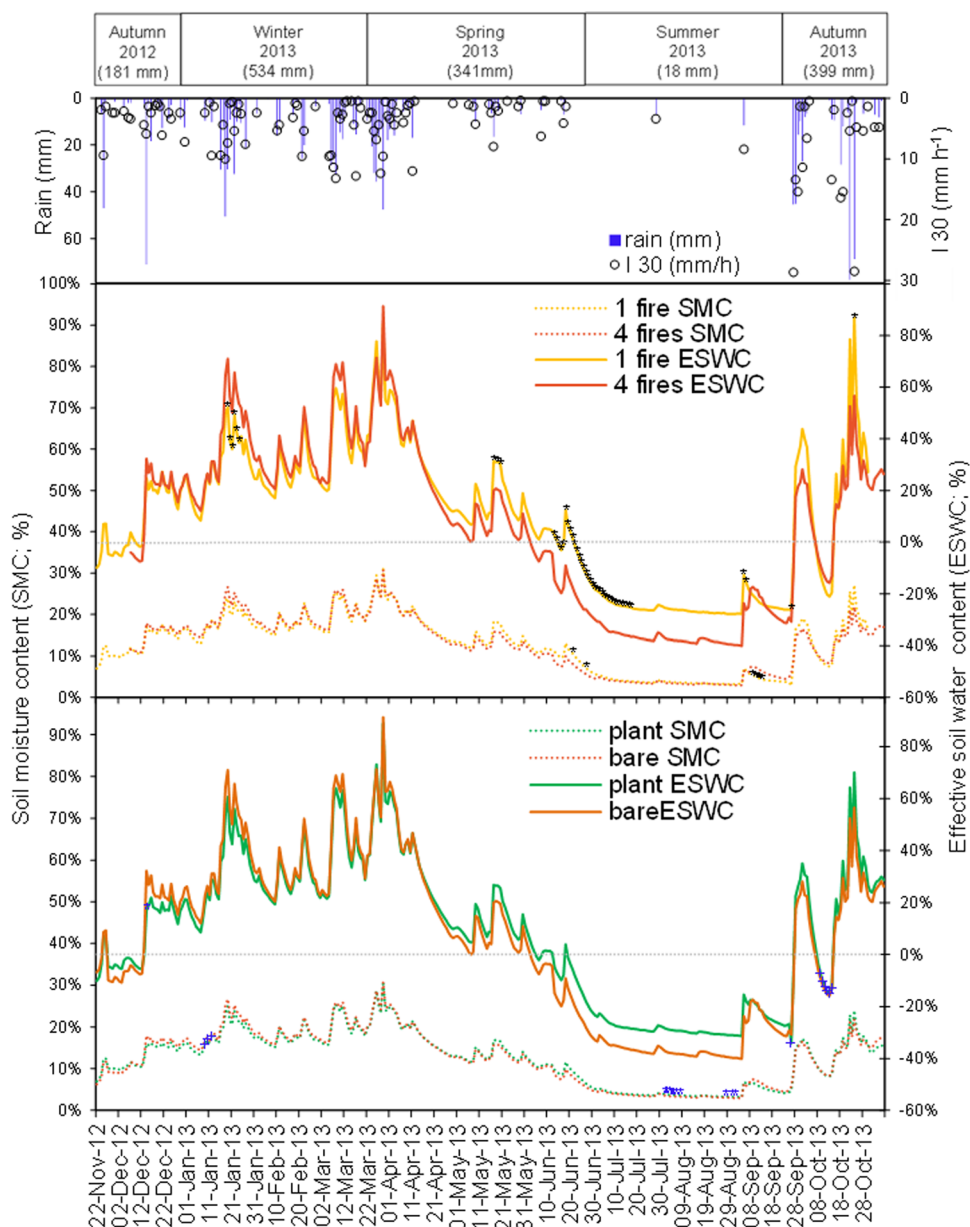
Comparisons among the fixed effects as well as differences between the levels of the factors were tested by least-squares means and adjusted by the Tukey–Kramer method (Kramer 1956). All statistical data analyses were carried out using the SAS 9.4 software package (SAS Institute, Inc. 2008), and all statistical tests used $\alpha=0.05$.

Results

Rainfall

From the analysed 350 days, 138 were rainy days, with a total rainfall of 1473 mm. Total rainfall per season was calculated as follows: Autumn 2012 (21/11/2012–21/12/2012, 181 mm), Winter 2013 (22/12/2012–21/03/2013, 534 mm), Spring 2013 (22/03/2013–21-06-2013, 341 mm), Summer 2013 (22-06-2013–21-09-2013, 18 mm) and Autumn 2013 (22-09-2013–06/11/2013, 399 mm). Maximum rainfall intensities in 30 min were measured in March and September 2013 (14 and 29 mm h⁻¹). Peak daily rainfall happened

Fig. 2 Daily, seasonal rainfall amount (mm) and maximum intensity in 30 min (I30, mm h⁻¹). Average daily volumetric soil moisture content (SMC, %) and effective soil–water content (ESWC, %) measured with the soil moisture sensors at the 1 fire (n = 36) and 4 fires hillslopes (n = 36) as well as for plant and bare soil microsites. Black asterisks indicate significant daily differences between 1 and 4 fires. Blue symbols indicate significant differences between plant and bare microsite for the 1 fire hillslopes (blue crosses) or the 4 fires hillslopes (blue asterisks) at the significance level of $p < 0.05$



in autumn 2013 (Fig. 2). Out of the summer drought (July, August and September 2013), the average and the median number of days between rain events were 4 and 3, respectively, while maximum days without rain amounted to 26.

Plant recovery, soil cover and related soil–water properties

The plant response at the two burnt areas was very similar, although at the end of the last dry period of September 2013, the 4 fires hillslopes showed a slightly lower plant height, stem diameter and stem length as compared to the 1 fire hillslopes (Table 2), although without significant

differences. Plant height was very similar in the unburnt hillslopes through time, and significantly higher than the 1 and 4 fires hillslopes.

Soil cover differed between unburnt and burnt areas (Fig. 3). Immediately after fire, abiotic soil surface components such as ashes, bare soil and stones amounted to 81% in the 1 fire and 99% in the 4 fires hillslopes. One year later, the unburnt hillslopes remained completely covered by biotic litter and vegetation, while in the 1 fire and 4 fires hillslopes amounted to only 49% and 16% of the biotic soil cover, respectively (Fig. 3).

Soil–water repellency (SWR) did not display statistically significant differences between 1 and 4 fires area in the overall data pool (Fig. 4). However, in the unburnt area, moderately/strongly SWR levels were more frequent as compared to the burnt sites. Overall, annual SWR frequency at 2–3 cm depth was non-repellent (MED class 0) in 44 and 43% of the measurements on the 1 and 4 fire areas, respectively, and only 17% on the unburnt areas. The 4 fires areas showed very strong and extremely repellent categories in 21% of the measurements, while the 1 fire and the unburnt areas were close to 40%.

The pF-curves revealed no differences at the gravitational soil–water content (< pF2) and field capacity (FC, pF2) due to wildfire frequency (Table 1). However, the soil–water retained at higher pF-values (pF3.2 and pF4.2 or PWP) increased as wildfire frequency (Fig. 5). Consequently, AWC (AWC = pF2–pF4.2) was significantly lower on the 4 fires hillslopes as compared to both the 1 fire and the unburnt hillslopes (Table 1).

The unburnt, 1 and 4 fires hillslopes reflected differences in SOM, in the gravimetric SMC and in the ESWC through the seasons of the first post-fire year (Table 2). SOM did not vary between the five soil sampling campaigns, but the unburnt hillslopes showed significantly lower SOM contents than the burnt hillslopes. Gravimetric SMC decreased consistently during the dry periods, and markedly so in

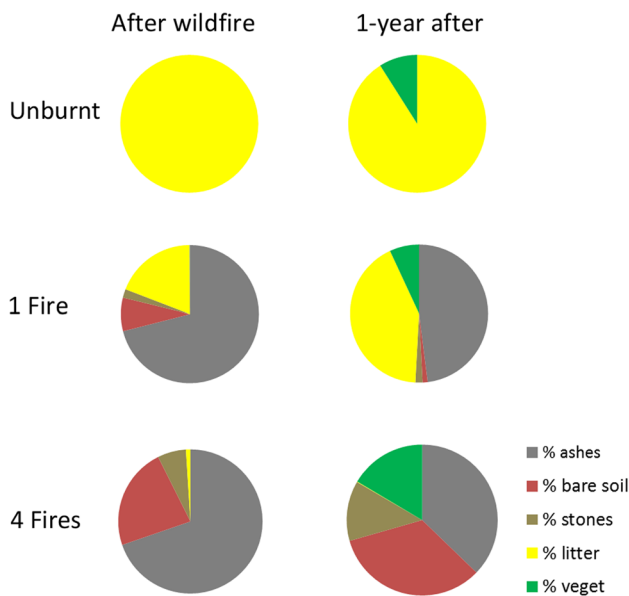
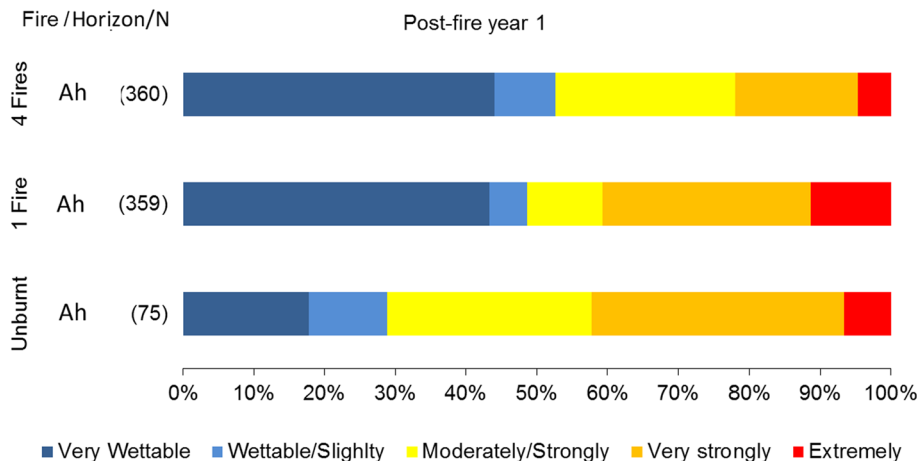


Fig. 3 Soil cover (%). Biotic (litter and vegetation) and abiotic (ash, bare soil, stones) soil surface components on the unburnt, 1 fire and 4 fires areas after the 2012 wildfire (December 2012) and one year later (November 2013)

Fig. 4 Relative frequency (x-axis) of soil–water repellency (SWR) determined by the molarity ethanol drop test (MED) for the soil Ah surface layer at the unburnt, 1 fire and 4 fires areas. SWR classes were grouped as follows: 0% ethanol, very wettable; 1%, 3% and 5% ethanol, wettable; 8.5% and 13%, moderately/strongly; 18% and 24% very strongly; 36% and > 36%, extremely repellent



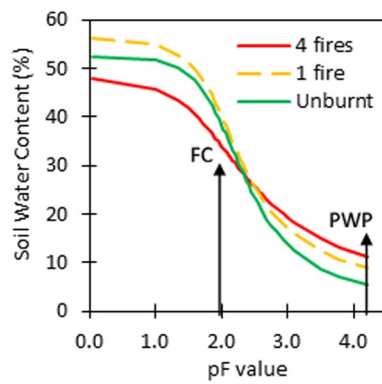


Fig. 5 Representation of the soil–water retention curves (% in volume) for the unburnt, 1 fire and 4 fires hillslopes at a range of selected tensions (pF-values from 0 to 4.2; n=54). The curves were fitted to all data points following Van Genuchten et al. (1991)

the 4 fires hillslopes by July 2013, which were statistically drier than the 1 fire and the unburnt hillslope (Table 2). In contrast, during the wet sampling of November 2013, the 4 fires hillslopes showed a significantly higher gravimetric SMC than in the 1 fire hillslopes (17% vs. 13%). ESWC varied in the wake of gravimetric SMC. ESWC for the 4 fires hillslopes was significantly lower during July 2013 (16%),

and significantly higher during November 2013 (43%), as compared to the 1 fire hillslopes.

Factors affecting soil moisture sensors readings

The post-fire overall difference in volumetric SMC between the 1 and 4 fires hillslopes was not statistically different (Table 3) and reached, respectively, 12.7% and 13.0% (S1). There was not a significant microsite effect (S1) but there were strong depth and season effects, and both were statistically significant (Table 3). The presence of significant interactions indicated that the effect of wildfire frequency was not straightforward for all the levels of the factors. For example, wildfire frequency and microsite showed complex relations: the plant microsite was wetter than the bare soil microsite on the 1 fire hillslopes, but the opposite was found on the 4 fires hillslopes (Table 4). These patterns were consistent throughout all the seasons, but any of the within-season differences were significant.

On the other hand, the difference in ESWC between the 1 and 4 fires hillslopes was almost significant (Table 3), with average values of 9.0% and 7.3% (see S1). Plant microsites registered higher overall ESWC values than bare soil microsites (9.9% vs. 6.4%), but only for the 1 fire hillslopes, and the differences were not significant. The effect of soil depth

Table 3 Summary of the F- and p-values obtained from the linear mixed-effects soil moisture content (SMC) and effective soil–water content (ESWC) models for the 1 and 4 fires areas

| | SMC | | ESWC | |
|-----------------------------|---------------|-------------------|---------------|-------------------|
| | F-value | p-value | F-value | p-value |
| <i>Fixed effects</i> | | | | |
| Fire | 2.1 | 0.1505 | 3.1 | 0.0791 |
| Microsite | 0.4 | 0.5381 | 0.4 | 0.53 |
| Depth | 99.5 | <0.0001 | 63.5 | <0.0001 |
| Season | 103.1 | <0.0001 | 60.9 | <0.0001 |
| <i>Interactions</i> | | | | |
| Fire*season | 1.5 | 0.2016 | 1.5 | 0.2045 |
| Fire*microsite | 15.1 | 0.0001 | 2.6 | 0.1099 |
| Season*microsite | 0.7 | 0.5954 | 0.4 | 0.7879 |
| Fire*season*microsite | 1.2 | 0.3304 | 0.8 | 0.5117 |
| Fire*depth | 3.8 | 0.0523 | 1.8 | 0.1718 |
| Season*depth | 2.1 | 0.0844 | 1.8 | 0.1273 |
| Fire*season*depth | 0.6 | 0.6764 | 0.6 | 0.6636 |
| Microsite*depth | 0.1 | 0.7974 | 0.4 | 0.5499 |
| Fire*microsite*depth | 2.7 | 0.1037 | 1.6 | 0.2055 |
| Season*microsite*depth | 0.1 | 0.9841 | 0.1 | 0.9815 |
| Fire*season*microsite*depth | 0.5 | 0.7659 | 0.3 | 0.8594 |
| <i>Covariates</i> | | | | |
| Ant_rain7_ | 2173.4 | <0.0001 | 1955.3 | <0.0001 |
| Ant_rain7_*depth | 285.5 | <0.0001 | 19.1 | <0.0001 |
| Ant_rain7_*season | 360.0 | <0.0001 | 287.9 | <0.0001 |

F- and p-values highlighted in bold for the fixed effects, covariates or their interactions are statistically significant at $p < 0.05$

Table 4 Seasonal mean values of volumetric soil moisture content (SMC, %) and effective soil–water content (ESWC, %) based on fire recurrence (1 vs. 4 fires), microsite (plant vs. bare soil) and season

| Fire | Microsite | SMC (%) | | | | | ESWC (%) | | | | |
|---------|-----------|-----------|--------|----------|---------|-----------|----------|--------|---------|----------|----------|
| | | Autumn | Winter | Spring | Summer | Autumn | Autumn | Winter | Spring | Summer | Autumn |
| | | 2012 | 2013 | 2013 | 2013 | 2013 | 2012 | 2013 | 2013 | 2013 | 2013 |
| 1 Fire | plant | 12.6bcdef | 17.4ab | 16.9abc | 10.2def | 12.6bcdef | 0.6abcd | 18.6ab | 19.1ab | −5.5bcd | 2.4abcd |
| | bare | 10.7def | 17.3ab | 15.3bcde | 9f | 9.3ef | −9.9bcd | 20.5ab | 13.4abc | −12.2 cd | −7.9bcd |
| 4 Fires | plant | 11.8cdef | 17.8ab | 14.6bcde | 9.7ef | 9.8def | −1.4bcd | 34a | 17.4ab | −9.5bcd | −1.7abcd |
| | bare | 16.4abcd | 19.8a | 16.1abcd | 9.6ef | 12.9bcdef | 15.8abc | 35.2a | 16.7ab | −18.6d | 4.2abcd |
| Overall | 1 Fire | 11.7ed | 17.4ab | 16.1abc | 9.6e | 11ed | −4.6ed | 19.6ab | 16.3abc | −8.8e | −2.8ed |
| | 4 Fires | 14.1 cd | 18.8a | 15.3bc | 9.7e | 11.3ed | 7.2 cd | 34.6a | 17bc | −14.1e | 1.3ed |

SMC or ESWC values followed by different letters indicate significant differences at $p < 0.05$

was significant and the surface soil layers exhibited much lower ESWC than the deeper soil layers (−4.1% vs. 20.4%; S1).

The mean seasonal volumetric SMC readings were significantly lower for summer 2013, due to the drought period, and highest for winter 2013 (Table 4). ESWC was significantly lower for the same dry season, when the ESWC was below 0%, or in other words, below PWP and thus, plants were under water stress. The daily variations of volumetric SMC and ESWC showed that the hillslopes burned 1 time were more stable (i.e. less oscillations) than the hillslopes burned 4 times (Fig. 2). The threshold of 0% in ESWC (no water available for plants) was reached 17 days sooner in the 4 fires hillslopes than on the 1 fire hillslopes, and also 10 days sooner on the bare than plant microsites (respectively, 15 and 3 days, for the 1 fire and 4 fires hillslopes). Although the differences were small, both volumetric SMC and ESWC were significantly higher on the 1 fire hillslopes than the 4 fires hillslopes during some of the driest days of summer 2013, and significantly lower during some of the wettest days of winter 2013 (Fig. 2). The plant microsites were slightly wetter than the bare soil microsites, although the daily differences were marginal (Fig. 2). The seasonal variations were highly correlated with rainfall amount. However, the most important covariate for both volumetric SMC and ESWC was the cumulated 7-days rainfall amount. The seasonal analysis revealed very low correlation coefficients during the dry summer seasons of autumn 2012 and summer 2013 ($r^2 = 0.54$ and 0.33), when the soils were much drier and hydrophobic (S2).

Discussion

Effects of wildfire frequency on plant recovery and soil properties

A previous study in the area showed that the increase in wildfire frequency had important effects on vegetation composition and structure (Mayor et al. 2016). Vegetation changed from pine woodland on the unburnt and 1 fire hillslopes to open shrubland on the 4 fires hillslopes. The occurrence of repeated wildfires killed the pine trees before they reached reproductive age, which left a footprint that constrains vegetation recovery and compromise forest productivity (Mayor et al. 2016; Kowaljow et al. 2018). Understorey plant recovery (*Pterospartum tridentatum*) was slightly faster on the 1 fire than the 4 fires hillslopes, which can be attributed to the higher litter and vegetation soil cover on the 1 fire than the 4 fires hillslopes (Fig. 3) or reduced regrowth capacity due to partial depletion of carbohydrate reserves or root damage. These cover categories protect the soil–water from evaporation at high soil temperatures and reduce the albedo (Sankey et al. 2012). However, exceptions were found between the higher water demand of plants as compared to bare agriculture lands during water stress periods on deep soil layers (Bellot et al. 1999) or the lower SMC measured on the vegetated areas as compared to burnt gorse shrubland (Soto and Díaz-Fierros 1997).

The higher plant recovery on the 1 fire hillslopes may also be a consequence of the higher soil–water availability for plants, which is a function of soil texture, bulk density and SOM (Vereecken et al. 1989; Ankenbauer and Loheide 2017). Wildfires can decrease the clay fraction (via soil erosion) and also increase the bulk density, due to the loss of SOM (Boix-Fayos 1997; Certini 2005). However, we observed that wildfire frequency increased the SOM and maybe, the silt fraction. We cannot be assertive since we are lacking the pre-fire sampling, so the difference in the silt fraction could be due to previous differences between sites.

However, the fining of the particle size distribution after burning can possibly be explained by physical weathering of the sand-sized particles into silt and clay-sized particles (Stoof et al. 2010; Rhoades et al. 2004), which are more easily removed by differential water erosion (Hosseini et al. 2016). On the other hand, SOM decreased after intense wildfires due to complete combustion (González-Pérez et al. 2004; Certini 2005), but it can also increase after moderate wildfires (Boix-Fayos 1997; De la Rosa et al. 2012) due to incomplete combustion and, as happened in our study, to its cumulative effect by repeated wildfires. Furthermore, the increase in SOM in our burnt hillslopes did not lead to higher soil–water retention, as it will be expected from other soils with high SOM (Minasny and McBratney 2017). Further possible improvements of the current study are the post-fire SOM quality characterization in water-related properties (Merino et al. 2018), which could explain odd results.

Effects of wildfire frequency on soil–water retention

Researchers have found difficulties in stating the subtle differences in soil–water retention due to soil heating, in laboratory (Badía and Martí 2003; Stoof et al. 2010) as well as field prescribed burning (Silva et al. 2006), single and repeated experimental fires (Soto and Diaz-Fierros 1997; González-Pelayo et al. 2006; 2010) or forest wildfires (Boix-Fayos 1997; Ebel 2012; Sankey et al. 2012). In general, soil–water retention decreases with fire temperature, and Badía and Martí (2003) found a decrease in the FC significantly correlated with the silt fraction increase and stated a significant reduction in the soil–water content, at both pF2 and pF4.2, with the progressive soil heating. Boix-Fayos (1997) related the decrease in the AWC with the changes in the proportion of small aggregates, silt and SOM, while González-Pelayo et al. (2006) found decreases in the AWC in burned vs. non-burned Mediterranean shrubland soils, concluding that the fire effect could favour high water holding at low pF values, anyway this relation is not straight forward. Stoof et al. (2010) stated that: (a) heating below 200 °C induces mild decreases in the amount of soil gravitational water at low pF tensions, (b) heating at 300 °C strongly decreased both gravitational water and soil capillary water held at high pF tensions, and (c) heating at 500 °C attained an intermediate position, as the strong reduction of SOM is offset by the increase in the finer silt and clay fractions. In fact, the fate of the organic matter can play a role in water storage capacity (Minasny and McBratney 2017; De la Rosa et al. 2012; González-Pérez et al. 2004). Although we reduced the sources of variation on our hillslopes (same aspect, steepness, hillslope length, parent material) and maximized the number of measurements per wildfire frequency (plant and bare microsities replicated 3 times in each of the 3 hillslopes; $n = 54$), the differences were small. The 1 fire hillslopes

showed the highest gravitational and FC soil–water, followed by the unburnt and by the 4 fires hillslopes, while the 4 fires showed the highest water retention at PWP (Fig. 5). On the one hand, the higher gravitational water on the 1 fire soils can be attributed to the incorporation of charcoal to the soil, which enhanced the water retention at low pF suction (Ebel 2012). On the other hand, the 4 fire soils had comparatively higher fire temperatures than the 1 fire, and fuels were likely transformed into less charcoal due to a more complete combustion. These temperatures could also destroy the repellent substances of the soil (Jimenez-Morillo et al. 2017), together with the increase in the less hydrophobic pyrogenic carbon fractions (Jones et al. 2019). Consequently, the incorporation of a small fraction of char to the 4 fires soils can explain the lower gravitational water as compared to the 1 fire soils (Stoof et al. 2010). Subsequently, soil–water repellency (SWR) appeared to decrease at increasing wildfire frequency (Fig. 4) and was higher in the unburnt than the 1 or the 4 fires hillslopes. Soil–water infiltration and storage can increase with the removal of the SWR, so Badía-Villas et al. (2020) showed as SWR decrease coincides with a sixfold increase in the unsaturated hydraulic conductivity on a prescribed burned soil; therefore, the lack of vegetation cover alters soil surface microstructure that, in the long run, could modify SMC. SWR can be also more persistent on coarse-textured soils and under certain vegetation types (González-Pelayo et al. 2015), as it was the case of the sandy-loam and vegetated unburnt hillslopes as compared to the loam and bare burnt 1 and 4 fires hillslopes.

Effects of microsite, topsoil depth and season on post-fire SMC and ESWC

Microsities exerted an effect on SMC and ESWC, but this effect differed between areas. Plant microsities retained more soil–water than bare microsities in the 1 fire hillslopes, but the opposite was true for the 4 fires hillslopes (Table 4). In his review, Sankey et al. (2012) found that plant microsities accumulated more SOM, nutrients and soil–water than bare microsities, the effect being stronger on the wildfire-affected steppes than in the unburnt areas, as it happened in our 1 fire hillslopes. The existence of soil moisture radial gradients between plant to bare microsities have been well described for Mediterranean arid (Sarah 2002; Katra et al. 2007) and semi-arid conditions (Gimeno-García et al. 2011), and less pronounced for humid Mediterranean environments with rainfall volumes of more than 700 mm of rain per year (Lavee et al. 1998; Ruiz-Sinoga et al. 2010b; Gabarrón-Galeote et al. 2012; Martínez-Murillo et al. 2013). The humid Mediterranean climate, with rainy winters and dry summers result in rapid fuel built-up, and a higher wildfire frequency. The more frequent wildfires destroy

the microtopographic shrub mounds and lower elevation bare interspaces. During the window of disturbance, the plants lose their capacity to buffer extreme temperatures and accommodate surface water relative to bare microsites (Sankey et al. 2012), which was likely the case of the 4 fires hillslopes. In fact, the 4 fires hillslopes produced eight times higher sediment losses than in the 1 fire hillslopes (Hosseini et al. 2016). Additionally, both microsites are subjected to wind erosion, but plants can be deflated at a greater rate than bare interspaces (Sankey et al. 2011).

Our results showed that both volumetric SMC and ESWC significantly increased with soil depth, which agreed with literature about post-fire soil moisture patterns (Soto and Díaz-Fierros 1997; Silva et al. 2006; Nolan et al. 2014; Cardenas and Kanarek 2014) and on other unburnt Mediterranean forest/shrubland soils (Sarah 2002). The lower volumetric SMC and ESWC in the upper soil layer are attributed to the higher evaporation in post-fire soil surfaces, due to the different albedo as a result of the lack of litter and the presence of black ashes (Bellot et al. 1999; Soto and Díaz-Fierros 1997).

The gravimetric SMC and ESWC data of the seasonal soil sampling campaigns fitted in well with the volumetric soil moisture from sensor data series. The wet season soil sampling campaign of November 2013 (Table 2) showed that increasing wildfire frequency led to greater quantities of water. On the other hand, the dry soil sampling of July 2013 showed that the more frequently-burnt soils had difficulties to retain soil–water. Sankey et al. (2012) described that plants afforded less time to recover between fires in high fire frequency sites, and soils were subjected to decreases in their water retention capacity. The same conclusions arose from the soil moisture sensors (Table 4 and Fig. 2): An increase in wildfire frequency led to maximum volumetric SMC values during winter periods, and minimum during dry periods, with longer periods of soil–water stress for plant growing. In overall for our study, the increase in wildfire frequency increased the water stress period in 17 days, during the first post-fire year, and the bare microsites increased the water stress in 10 days as compared to the plant microsites. This contrasting effect will depend on plant recovery to pre-fire levels, which depends on both wildfire frequency, and plant physiology to recover after disturbances (Sankey et al. 2012). Plant water uptake plays a complex effect in the soil–water retention in bare/plant microsites during wet/dry seasons in burnt areas. Silva et al. (2006) did not find any significant effect, but others (Soto and Díaz-Fierros 1997; Nolan et al. 2014; Prats et al. 2016b) found an effect of plant water uptake that reduced SMC during the dry seasons. The two first studies measured higher SMC in burnt than unburnt/vegetated soils, while the third study attributed the higher SMC in burnt than hydromulched soils to the higher plant cover during the dry end of the first post-fire year.

Management implications

Post-fire managers should follow strategies to preserve and restore pre-fire vegetation structures, due to their important role in water conservation and redistribution and prevent catastrophic tipping points towards desertification (Lavee et al. 1998; Ludwig et al. 2005) even in humid Mediterranean forestlands. Forest managers have applied post-fire emergency rehabilitation treatments after large fires (e.g. mulching, grass seeding, erosion barrier installation) in order to reduce soil degradation (soil and fertility losses, flooding, reservoir siltation) as well as mid-term silvicultural treatments (e.g. thinning, shrub removal, acorn seeding, seedlings planting) to minimize plant competition and restore vegetation (Vallejo and Alloza 2015). There is, however, an ongoing scientific debate about whether, how and when to implement which post-fire restoration actions, to ensure adequate tree recruitment and fast ecosystem natural regeneration (Taboada et al. 2017). Additionally, the impacts and the timing of the soil mechanical disturbance by the post-fire salvage logging should be taken into account (Fernandez and Vega 2016; Malvar et al. 2017). Within the research community, there is a growing consensus that after a wildfire, the focus should be paid in mitigating soil erosion on severely burned areas where protective soil cover is lacking (Shakesby 2011). A widely used technique because their cost-effectiveness is the application of a protective cover of different mulching types thus mimicking natural litter cover (Fernandez and Vega 2016; Lucas-Borja et al. 2019; Prats et al. 2022). The effectiveness of mulch in the first post-fire year can be extremely large (Prats et al. 2022), in comparison with seedling treatments (Badía and Martí, 2000), and its use in burned environments strongly reduces soil–water losses (Lucas-Borja et al. 2019, 2020; Fernandez and Vega 2016), enhancing plant production in semiarid areas (Badía and Martí, 2000), providing soil protection and accelerating seedling establishment (Santana et al. 2014; Prats et al. 2022), although its effectiveness varies markedly depending on topography, rainfall regimes, burn severity, and time of application (Girona-García et al. 2021). Mayor et al (2016) suggest that increased wildfire frequency changes the vegetation from forest to shrubland and increases the bare (abiotic) soil cover. The longer window of water stress for plants during the dry season can compromise its restoration. We recommend the application of local and readily available mulch materials (Prats et al. 2022) on the bare interspaces, not only of high severity burnt areas but also of high wildfire frequency burnt at moderate fire severity. These areas deserve special attention, first and foremost because its restoration is compromised, and both soil erosion mitigation treatments as well as re-introduction of new seedlings will be needed. The management of these burnt areas is rather complex, but some experiments have shown that the two

targets (i.e. reduce soil losses and restore the natural vegetation) can be achieved by means of mulch. Community-based initiatives have shown that mulching can serve both objectives (Prats et al. 2022). Anyway, for a further advance on post-fire restoration treatments, researchers are developing more robust post-fire soil erosion models (Lopes et al. 2021).

Conclusions

The conclusions about the effect of wildfire frequency on plant recovery, soil properties and soil–water dynamics throughout the first year after the 2012 wildfire in unburnt hillslopes, hillslopes affected by 1 fire and 4 fires, are as follows:

- (i) The increase in wildfire frequency significantly decreased the litter and vegetation (biotic) soil cover and might have increased the SOM content by char incorporation, while it slightly decreased soil–water repellency (SWR).
- (ii) Increasing wildfire frequency did not affect the overall annual/seasonal soil moisture content (SMC), although soil–water retention was significantly lower at field capacity (FC) and significantly higher at the permanent wilting point (PWP) on the 4 fires as compared to the 1 fire soils, resulting, in overall, in a lower available water content (AWC) as increasing wildfire frequency.
- (iii) However, the threshold for plant water stress (ESWC < 0%) was reached sooner (17 days more) in the 4 fires hillslopes than on the 1 fire hillslopes, and also in the bare than plant microsites (10 days). As a consequence, the effective soil–water content (ESWC) was lower on the 4 fires than in the 1 fire area, and plants were under water stress for longer.
- (iv) Both volumetric SMC and ESWC were significantly higher at deeper soil layers than on the superficial ones and were also significantly affected by seasonal influences. The microsite was not a significant factor, although overall mean values revealed higher ESWC under plant, as compared to bare microsites. Antecedent rainfall in 7 days was a significant covariate for both volumetric SMC and ESWC.
- (v) Finally, the hillslopes burnt at high wildfire frequency suffer longer water stress periods during the dry seasons.

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Author contributions The study conception belong to Coen Ritsema, Susana Bautista and Jan Jacob Keizer. Ther study design belong to Jan Jacob Keizer and Oscar González-Pelayo. Material preparation, data collection and analysis were performed by Oscar González-Pelayo, Sergio Prats, Maruxa Malvar end Erik van den Elsen. The first draft of the manuscript was written by Oscar González-Pelayo, and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Data availability Not applicable.

Declarations

Conflict of interest The present study does not present conflicts of interest/competing interests.

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