J. For. Res. (2020) 31(5):1647–1661
https://doi.org/10.1007/s11676-019-00936-7

ORIGINAL PAPER

Effect of pre- and post-wildfire management practices on plant recovery after a wildfire in Northeast Iberian Peninsula

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Received: 21 November 2018 / Accepted: 19 December 2018 / Published online: 4 April 2019

Abstract Fire and pre- or post-fire management practices shape the distribution and richness of plant species. Here, the effects of pre- and post-fire management on vegetation recovery were studied at different times, up to 18 months after a wildfire. Two months after a 2015 wildfire, 18 study plots were established (three 4-m^2 plots for each treatment), vegetation regrowth was monitored and vegetal species richness (S), evenness (I_T), density (D), diversity (H') and maximum diversity (HMax) after 2, 10 and 18 months. The treatments were (1) control, unaffected by 2015 wildfire; (2) no treatment (NT), burned in 2015 wildfire and not managed; (3) managed in 2005 and burned in 2015 (M05B); (4) managed in 2015, 2 months before wildfire (M15B); (5) cut and manual removal after the 2015 wildfire (CR); (6) cut and no trunk removal randomly deposited on topsoil after the 2015 wildfire (CL). All the treatments were carried out in a *Pinus halepensis* Miller forest. At 10 and 18 months after the wildfire, vegetation recovery was greater in NT, CR and CL plots than in M05B and M15B the plots. By 18 months after the wildfire, *Brachypodium retusum* (Pers.) P. Beauv. and *Rosmarinus officinalis* L. were still dominant, especially in M15B, corroborating the belief that pre-fire treatment reduced ecosystem resilience and vegetal recovery compared to the NT and post-fire managed plots. Richness was significantly lower 10 months after wildfire in control plots, and I_T was significantly higher in that inventory than previously in M15B. Eighteen months after the wildfire, H' was significantly lower in M15B. Ten months post-wildfire, HMax was significantly lower in the control plots. Eighteen months after the wildfire, HMax, was significantly higher in CR, CL and M05B than in the control and M15B plots. Overall, pre-fire management was detrimental to post-fire vegetation recovery, while manual post-fire management proved beneficial.

Keywords Forest management · Vegetation recovery · Diversity · Richness · Evenness

Project funding: This study was supported by POSTFIRE_CARE Project (CGL2016-75178-C2-2-R) sponsored by the Spanish Ministry of Economy and Competitiveness and AEI/FEDER, UE. Support was also received from the FPU Program (FPU 014/00037) sponsored by the Ministry of Education, Culture and Sports, the EST16/00183 to a short stay in Mykolas Romeris University (Vilnius, Lithuania) to the Ministry of Education, Culture and Sports and project 2017SGR1344 of the Generalitat de Catalunya.

The online version is available at http://www.springerlink.com

Corresponding editor: Chai Ruihai.

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s11676-019-00936-7) contains supplementary material, which is available to authorized users.

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Introduction

Wildfires, a global phenomenon and natural element of the ecosystems (Bowman et al. 2009) are increasing in severity as a consequence of climate change, land use change, land abandonment, monospecific plantations, the expansion of urban areas into wildlands and effective fire suppression policies among other factors (Oliveira et al. 2014; Pereira et al. 2018). In parallel with this, various studies have examined how the vegetation recovers after wildfires in Europe (e.g., Francos et al. 2016a; Viana-Souto et al. 2017), North America (e.g., Hart et al. 2005; Hart and Chen 2004; Coppoletta et al. 2015), Africa (e.g., Dwomoh and Wimberly 2017; Chergui et al. 2018a, b), Asia (e.g., Goto et al. 1996; Han et al. 2015) and Oceania (e.g. Caccamo et al. 2015; Nicholson et al. 2017).

In the Mediterranean, fire is a natural disturbance and a cultural element (Chergui et al. 2018a, b; Santana et al. 2018). Mediterranean vegetation is resilient to fires, and several species actually resprout (e.g., *Quercus suber* and *Quercus cocciferae*) or spread their seeds (e.g., *Pinus pinaster* and *Pinus halepensis*) (García-Jiménez et al. 2017; Pausas and Keeley 2017) as an adaptation to fire disturbance. The response of vegetation to fire disturbance depends on the recurrence of fire, the topography of the burned area, the severity of the episode, post-fire weather conditions, pre- and post-fire management and other factors (Pereira et al. 2018). Moreover, measures to suppress fires appear to have increased the amount of biomass available in wildland environments and, hence, the risk of high severity wildfires (Lavaux et al. 2016). To reduce the biomass, several management options have been adopted, including prescribed fires (Úbeda et al. 2018), grazing (Mena et al. 2016) and mechanical vegetation removal (Hevia et al. 2018). In areas affected by wildfire, the most common post-fire management practice is the logging of burnt wood and site preparation (Peterson et al. 2015; Pereira et al. 2018). Pre- and post-wildfire management and prescribed fire modify the vegetation structure (Broncano et al. 2005), the degree of change depending on the type of management. For example, thinning and prescribed fire reduce the understorey biomass and temporarily increase the number of light-demanding species and species richness (Abella and Springer 2015; Trentini et al. 2017; Willms et al. 2017). Post-wildfire management, in contrast, especially when done immediately after a fire and with heavy machinery (e.g., salvage logging), induces a marked soil disturbance (Pereira et al. 2018), influencing vegetation regeneration capacity (Vallejo and Alloza 2012). However, results are far from conclusive. For example, some studies have found that salvage logging had no effect on vegetation recovery (Fernandez and Vega 2016), while others reported a negative impact (Griffin et al. 2013; Garcia-Orenes et al. 2017; Francos et al. 2019). Vegetation recovery apparently can be seriously affected by soil compaction and reduced water availability as a consequence of heavy-machinery traffic (Page-Dumroese et al. 2006; Marañón-Jiménez et al. 2013).

Several studies have examined the impact of wildfire (e.g., Broncano et al. 2005; Carnicer et al. 2013), post-wildfire management (e.g., Knapp and Richie 2016; Moya et al. 2015) and pre-wildfire management (Tempel et al. 2015; Tucker and Kashian 2018) on vegetation recovery. However, little information is available about pre- and post-management impacts in areas affected by a single wildfire. Such knowledge is vital for understanding the responses of vegetation of similar characteristics to pre- and post-fire management practices and whether they are detrimental for ecosystems. The aim of this study was to identify the impacts of pre- and post-wildfire management on (1) vegetation cover, (2) species distribution and (3) richness (S), evenness (I_T), density (D), diversity (H') and maximum diversity (HMax).

Materials and methods

Study area

The study area is located in Ódena (41°38′42″N–1°44′21″E) in the province of Barcelona (NE Spain). A fire broke out in the municipality of El Bruc on 26 July 2015, burning 1274 ha. The burned area had been previously affected by fire in 1986. Pre-fire vegetation was mainly composed of *Pinus halepensis* Miller, *Pinus nigra* Arnold and *Quercus ilex* L., with an understorey layer composed of *Genista scopius* L. and *Pistacia lentiscus* L. The geological substrate of the area is sedimentary (Panareda-Clopes and Nuet-Badia 1993), and the soils are classified as Fluventic Haploxerept (Soil Survey Staff 2014). The average temperature of the study area is 14.2 °C, and the mean annual precipitation is between 500 and 600 mm according to the El Bruc automatic meteorological station (41°34′44″N–1°46′57″E) classified as *Csa* by Köppen (1990).

Experimental design and vegetation inventories

Two months after the wildfire, 18 plots with similar slopes (< 10%) and a northeast exposure were established in the burned area in zones subject to distinct pre- and post-fire treatment. In each zone, three 4-m^2 subplots were set up to monitor vegetation recovery after the wildfire. The following treatments were considered: (1) control, unaffected by the 2015 wildfire; (2) no treatment (NT), affected by the 2015 wildfire, but not subject to any management; (3) managed in 2005 and affected by the 2015 wildfire (M05B); (4) managed in 2015, 2 months before the 2015 wildfire (M15B); (5) cut-and-remove management after the 2015 wildfire (CR), with trunks cut and removed manually after the wildfire; (6) cut-and-leave management (CL), with trunks cut after the 2015 wildfire and left randomly on the soil. The pre-wildfire management was a clear-cut operation, with 1000 trees left per hectare and the cut branches no longer than 1 m left on the soil. The branches were cut to a length of 1 m, leaving fine and medium wood on the soil as done by Francos et al. (2018a). Both M05B and M15B treatment plots were managed before the wildfire to attain 1000 trees/ha from an initial density > 2000 trees/ha. In plots managed post-wildfire, tree trunks were cut after the episode and left randomly on the soil (Table 1). At 2, 10 and 18 months after the fire, we visually recorded the individuals of each species in each plot, then transformed the values into percentage species abundance (Úbeda et al. 2006) and the following indices: richness (*S*), evenness (I_T), density (*D*), diversity (H') and maximum diversity (HMax). Richness is the number of different plant species recorded in each plot, while evenness is a measure of the relative abundance of the different species making up the richness of the plot. The Shannon–Weaver diversity index (H'), which combines *S* and I_T was calculated by applying the formula $H' = -\sum p_i \ln p_i$, where p_i is the proportion of individuals of each species. Maximum diversity is the log *S* and defines the maximum diversity observable in each area at any moment (Garcia-Orenes et al. 2017; Weaver and Shannon 1949). The fire was classified as high severity because tree crowns had completely combusted (Úbeda et al. 2006).

Statistical analyses

Shapiro–Wilk and Levene tests were used to verify data normality and homogeneity at a value of $p > 0.05$. Two-way ANOVA was considered to identify differences between time and treatment. When significant differences were found at $p < 0.05$, we applied Tukey's significant difference (HSD) post hoc test. All statistical analyses were conducted using SPSS 23.0 (IBM, Armonk, NY, USA).

Results

Post-wildfire rainfall

Post-wildfire rainfall is shown in Fig. 1. Total precipitation in the 18 months following the fire was 641.6 mm, giving a monthly mean of 35.64 mm with a coefficient of variation of 66.08%.

Vegetation cover and distribution

In the control area, vegetation remained unchanged 2, 10 and 18 months after the fire and covered 100% of the plot. The most abundant species were *Rubia peregrina* L. (47.62%) and *Pinus halepensis* Miller (41.27%). Two months after the fire, vegetation cover in all the other plots was greatly reduced: just 5% in NT, CR and CL and 3% in M05B and M15B. With the exception of the CR plots, *Brachypodium retusum* Pers. was the most common species in all the fire-affected plots, while *Euphorbia serrata* L. was the most abundant in CR. The least common species in the different plots were as follows: NT. *Pistacia lentiscus* L., *Arbutus unedo* L., *Asparagus acutifolius* L., *Pistacia lentiscus* L., and *Quercus ilex* L. (2.70%); M05B, *Euphorbia serrata* L. and *Psoralea bituminosa* L. (1.61%); M15B, *P. halepensis* and *Quercus coccifera* L. (3.57%); CR, *Lonicera implexa* Aiton (2.38%); and CL, *Bupleurum* sp. and *P. lenticus* (4%) (Table S1).

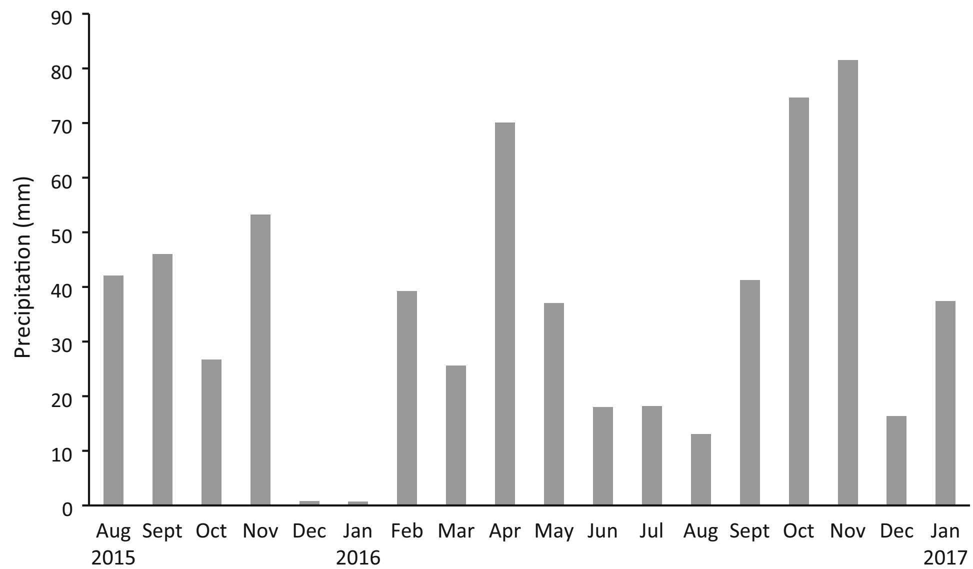
Ten months after the wildfire, both the vegetation cover and its overall structure had changed. The highest percentage cover was observed in NT and CR (16%) and the lowest in M15B (9%). *B. retusum*. continued to be the most abundant species in M05B (43.66%) and CL (21.05%), while in NT and CR the most abundant species was *R. peregrina* (27.16 and 40.20%, respectively) and *Rosmarinus officinalis* L. (27.45% of both sites). In M15B, *R. officinalis* covered the largest area (27.45%). The least abundant species by plot were as follows: NT, *Filago germanica* L., *P. halepensis*, *P. lenticus*, and *Q. coccifera* (1.23%); M05B, as *Erica multiflora* L. (1.41%); M15B and CR, *Coris monspeliensis* L. (1.96 and 0.98%, respectively); and, CL, *P. lenticus* and *Rubus ulmifolius* Schott (3.51%) (Table S2).

Eighteen months after the wildfire, the percentage cover was greatest in CR (57%) and lowest in M15B (25%). *B. retusum* was the most common species in M05B (34.38%), M15B (44.29%) and CR (14.58%), whereas in NT the most common were *Dorycnium pentaphyllum* Scop. (23.33%) and *Smilax aspera* L. (22.33%). In CR the most abundant

Table 1 Study area, dates of fire, and date and type of management in each area

Study area	Years of fire(s)	Date of management	Type of management
CUn	1986	Not managed	Not managed
NT	1986 and 2015	Not managed	Not managed
M05B	1986 and 2015	2005	Cut and leave on topsoil
M15B	1986 and 2015	2015 (2 months before fire)	Cut and leave on topsoil
CR	1986 and 2015	2015 (2 months after fire)	Cut and remove manually
CL	1986 and 2015	2015 (2 months after fire)	Cut and leave on topsoil

Fig. 1 Post-wildfire rainfall at the meteorological station of Ódena (41°36′19″N–1°38′18″E 423 m a.s.l.) closest to the study area



species was *D. pentaphyllum* (20.91%). The least common species in NT were *Arbutus unedo* L., *Bupleurum* sp., *P. lentiscus*, *R. officinalis* and *Thymus vulgaris* L. (1.67%). In M05B, the least abundant species was *Sedum acre* L. (1.56%), while in M15B it was *Globularia alypum* L (1.43%). In CR, the least common species were *Bupleurum* sp., *P. halepensis*, *P. bituminosa*, *R. officinalis*, *Sedum acre* L. and *Viburnum tinus* L. (0.91%), while in CL they were *A. unedo*, *E. serrata*, *P. lenticus,* and *Q. ilex* (2.08%) (Table S3).

Two months after the wildfire, M05B and CR had been colonized by herbaceous species, while in NT and CL shrub species were the most common. M15B presented a similar number of both types. In all the treatment plots, herbs occupied the largest area (Fig. 2a, d). Ten months after the fire, shrub species were more common in CL and MB05, while in CR and M15B herbs were more dominant. Numbers of herbs and shrubs were similar in NT. The area occupied by herbs was greater in NT, M05B and M15B. Lianas covered a greater area in CR, while herbs and shrubs presented a similar cover in CL (Fig. 2b). Eighteen months after the wildfire, in all plots but M15B, the number of shrub species was always higher than that of herbs. The appearance of scrub species in all the plots also acquired a degree of importance, while mosses were important in NT, CR and CL. Shrub coverage was highest in all the treatments (Fig. 2c, f).

Two months after the wildfire, with the exception of M15B, resprouters were always more numerous than seeders. In fact, in all cases, the area covered by resprouters was higher than for other life forms (Fig. 3a, d). Ten months after the wildfire, with the exception of CR, the number of seeders was higher than that of resprouters. However, in common with the observations recorded 2 months after the wildfire, resprouters occupied the highest percentage area in all the plots (Fig. 3b, e). By 18 months after the wildfire, resprouters were dominant in terms of number of species and the area covered in all the plots (Fig. 3c, f).

Richness, evenness, density, diversity and maximum diversity

The results showed significant differences between time and treatment with regards to richness (S) and maximum diversity (HMax), while evenness (I_T) and diversity (H') only showed significant differences over time. H' showed significant differences between treatments. No differences were identified between time and treatment interactions (Table 2). Significant differences were identified between plots 10 months after the wildfire. S was significantly higher in CR and NT than in the control. In M15B and CL, S at10 months after the wildfire was significantly higher than after 2 months. In M15B, I_T and D at 10 months after the wildfire were significantly higher than after 2 months (Table 3). Diversity was significantly higher in CL than in M15B 18 months after the wildfire. Ten months after the wildfire, HMax was significantly lower in the control plot compared to values recorded in NT, M15B, CR and CL. Eighteen months after the wildfire HMax was significantly higher in NT, M15B, CR and CL than in the control plot and M15B. In M15B and CL, the HMax was significantly higher 10 months after the wildfire than after 2 months (Table 4).

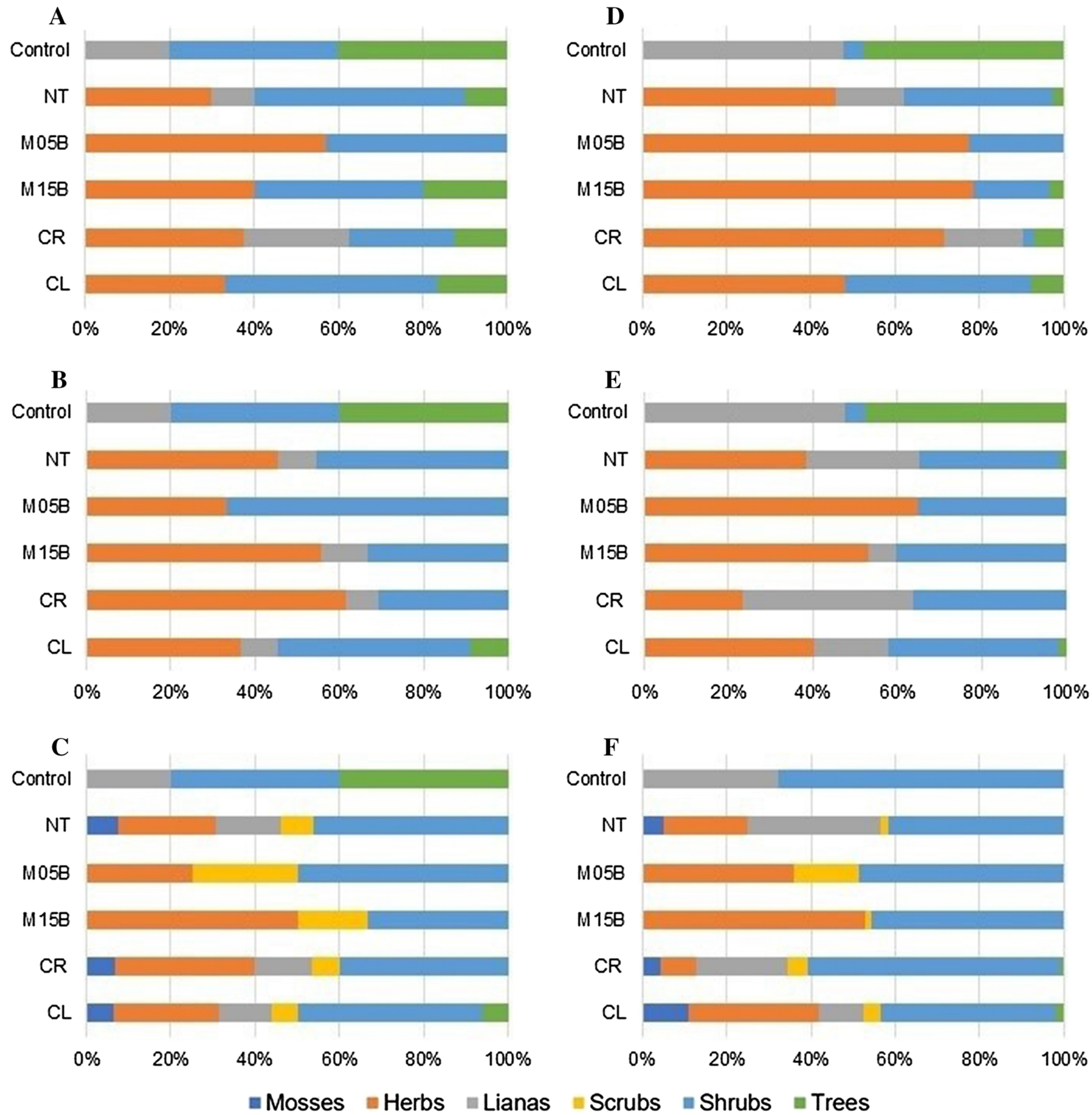


Fig. 2 Life form of the species identified in the studied plots. Percentage of number of species at 2 months (**a**), 10 months (**b**), and 18 months (**c**). Percentage of cover at 2 months (**d**), 10 months (**e**), and 18 months, **f**). Treatments: no treatment (NT), managed in 2005 (M05B), managed in 2015 (M15B), cut and leave (CL) and cut and remove (CR)

Discussion

Vegetation cover and distribution

Although all plots presented an increase in percentage cover, none were completely covered 18 months after the fire. The delayed recovery is probably attributable to (1) the high severity of the fire, (2) the management carried out, and (3) fire recurrence. Mediterranean vegetation is adapted to fire disturbance (Tessler et al. 2016); however, when exposed to a fire of high severity (as in our study), vegetation recovery was restricted because of the high temperatures reached, removal of high levels of nutrients by volatilization, and generation of large amounts of smoke and ash by the fire. Recovery is especially difficult on slopes because after high severity fires the soil is normally bare or has a reduced ash cover and is, therefore, vulnerable to water and wind erosion (Fernandez-Manso et al. 2016; Viana-Souto et al. 2017; Pereira et al. 2018). In the months immediately after the fire (August and September 2015), 88.1 mm of precipitation fell, when the soils were most exposed to the agents of erosion. However, precipitation events did not seem especially intense since no evidence of erosion was found in the plots 2 months after the fire. Pre-fire management reduces the biomass, but it can have a detrimental impact on soil properties (e.g., soil

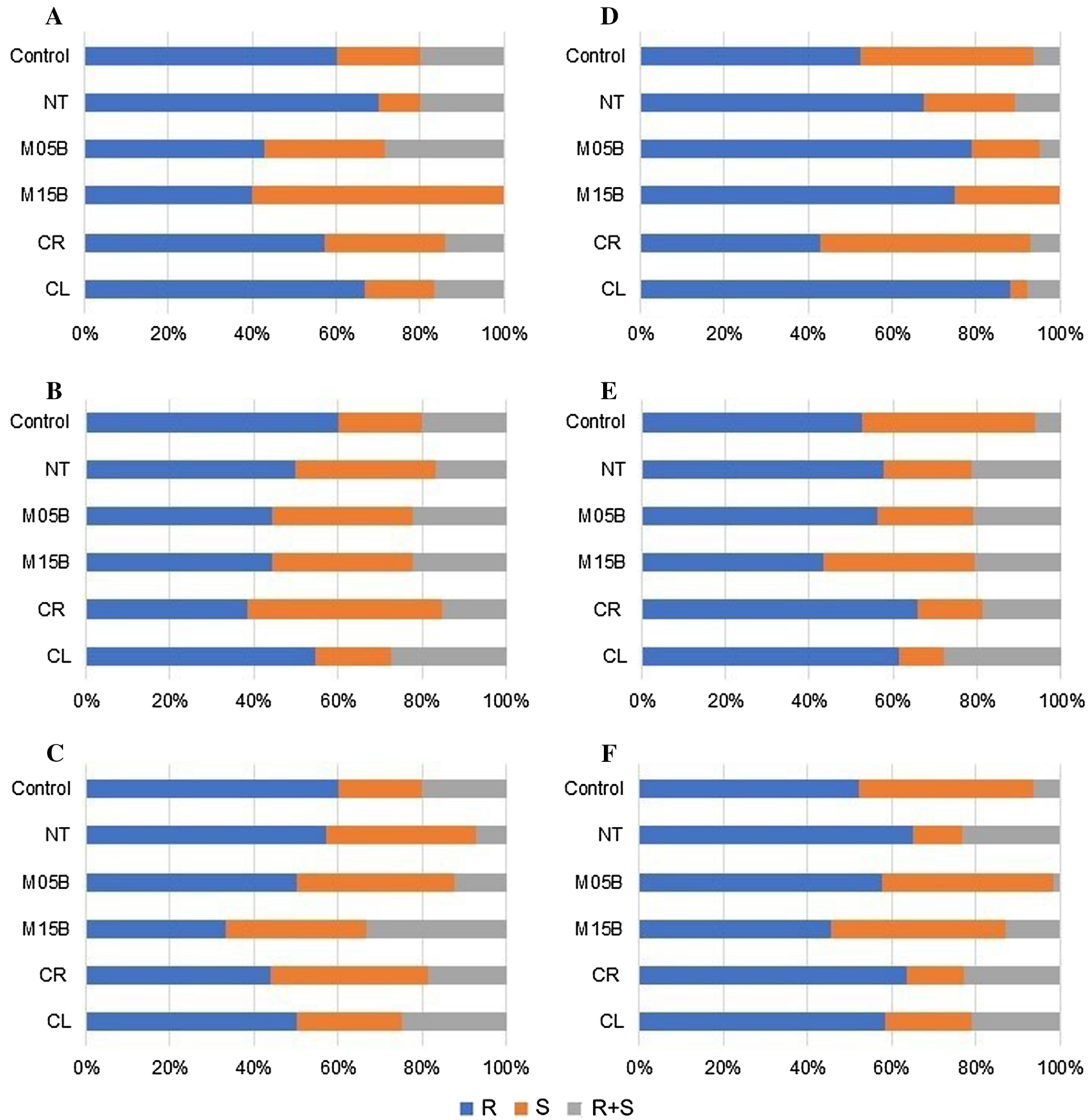


Fig. 3 Number of resprouters (R), seeders (S) and species with both regenerative capacity (R + S), percentage of number of species at 2 months (**a**), 10 months (**b**), 18 months (**c**) after the wildfire. Percentage of cover after 2 (**d**), 10, (**e**) and 18 months (**f**). Treatments: no treatment (NT), managed in 2005 (M05B), managed in 2015 (M15B,) cut and leave (CL) and cut and remove (CR)

compaction, moisture and microbiology) (Cambi et al. 2015; Toivio et al. 2017), thereby reducing the capacity of vegetation recovery post-wildfire and thus most likely responsible for the reduced rate of recovery in the M05B and M15B plots after 18 months (< 50%). Shive et al. (2013a, b) also observed that areas managed before the fire had lower vegetation cover than in untreated areas after a high severity wildfire. Among the plots studied, CL had the highest plant cover, probably reflecting its higher levels of soil organic matter. Post-wildfire management was carried out manually, thus avoiding negative soil effects (Francos et al. 2018b), in contrast to salvage logging (Garcia-Orenes et al. 2017). The recurrence of fire events can affect soils and vegetation recovery (Pereira et al. 2018; Tessler et al. 2016). In the present case, the last fire occurred in 1986 (29 years before), making it unlikely, in natural conditions, for vegetation cover to have fully recovered, as observed in the control plot. In the Mediterranean environment, shorter fire intervals (4–6 years) have the most negative impacts on vegetation cover (Tessler et al. 2016).

Two months after the fire, with the exception of CR (covered mainly with *Euphorbia serrata* L.), the most common species in all the plots was *Brachypodium retusum* a vigorous resprouter (Duguy and Vallejo 2008), as discussed in previous studies (Pausas 1999; Badia and Marti 2000). The destruction of vegetation by fire presents

Table 2 Two-way ANOVA results

Vegetal Index	Factors	p
Richness (S)	Time	***
	Treatment	**
	Time × Treatment	n.s.
Evenness (I_T)	Time	**
	Treatment	n.s.
	Time × Treatment	n.s.
Density (D)	Time	**
	Treatment	n.s.
	Time × Treatment	n.s.
Diversity (H')	Time	n.s.
	Treatment	*
	Time × Treatment	n.s.
Maximum diversity (Hmax)	Time	***
	Treatment	**
	Time × Treatment	n.s.

Significant differences in Tukey post hoc test: *$p < 0.05$. **$p < 0.01$ and ***$p < 0.001$. n.s., not significant

an opportunity for perennial graminoids to resprout due to the reduced level of competition and greater availability of light (Santana et al. 2014). In the period immediately following a fire, therefore, they tend to account for a high percentage of plant cover. In addition, this species has a well-developed root system that facilitates its expansion. The success of *B. retusum* in colonizing fire-affected areas during the first year after the fire is attributed to it capacity to resprout from the rhizomes and the positive effect of fire in increasing sexual reproduction (Vila-Cabrera et al. 2008). The regenerative capacity of *B. retusum* is not significantly affected by the high severity and frequency of fires (Caturla et al. 2000; De Luis et al. 2004). Yet, it reduces the germination capacity of other plants, including *Pinus halepensis* (Pausas et al. 2004a, b), as observed in this study. *E. serrata* is a species that tends to appear in the first year after a fire, albeit never as a dominant species, as observed elsewhere (Trabaud et al. 1985; Caturla et al. 2000; Prieto et al. 2009). However, this species can reach high coverage rates in a short time (Herrero and Gutierrez 2006). We hypothesized that here the availability of nutrients and light facilitated seed germination of this plant.

Ten months after the fire, *B. retusum* was still the most common species in M05B and CL, *Rubia peregrina* L. in NT and CR and *Rosmariunus officinalis* L. in M15B. In all the plots, the percentage cover of *B. retusum* had fallen as a result of the recovery of other plants, including *Dorycnium pentaphyllum* Scop. and *R. peregrina* (NT), *Fumana thymifolia* L. and *Psoralea bituminosa L.* (M05B), *Arbutus unedo* L. and *R. officinalis* (M15B) and *R. peregrina* and *Erica multiflora* L. (CL). These plant dynamics can be attributed to factors such as the germination season, effect of heat on plant germination; the capacity of each species to respond to the fire disturbance, plant recovery as a consequence of the incorporation of ash into the soil profile;, and the effects of the pre- and post-fire management practices. The seeds of *D. pentaphyllum*, a facultative resprouter, are stimulated by fire; the probability of germination increases when exposed to 120 °C for 10 min (Herránz et al. 1998). Indeed, the species is considered a "fire seeder" (Buhk and Hensen 2006). The same applies to the seeds of *Fumana thymifolia* L. (Moreira and Pausas 2012) and *P. bituminosa* (Herránz et al. 1998; Luna et al. 2007). The germination of these species was marked in NT and in M05B, where soil temperatures were very likely high due to the large amount of biomass (Francos et al. 2018b). The aforementioned species are well adapted to fire, as observed in previous studies (López-Soria and Castell 1992; Buhk and Hensen 2006; Santana et al. 2010). Ten months after the fire there was no evidence of ash cover. The high alkalinity of ash produced in high severity fires inhibits seed germination (Izhaki et al. 2000). Soil pH in the studied plots was alkaline for 2 months after the fire, thus reducing germination. Ten months after the wildfire, pH levels had fallen because of ash removal (no evidence of ash cover was found 10 months after the fire), erosion, overland flow and incorporation into soil profile (Pereira et al. 2014), thereby improving conditions for seed germination. It is widely known that ash contributes to an increase in soil nutrients and, therefore, to germination and plant growth (Kuzyakov et al. 2018). Pre-fire management reduces the amount of biomass, but depending on the management carried out, this reduction may have detrimental impacts on the soil, seed bank and plant restoration. In addition, pre-fire treatment can promote the establishment of non-native plants (Hunter et al. 2006). Ten months after the wildfire, the most abundant species in the plots studied here varied. *R. peregrina* was especially abundant in the NT, CL and CR plots, with a clear predominance over *B. retusum. R. peregrina*, a species that is not as well adapted to fire as *B. retusum*, can nevertheless resprout and germinate vigorously after wildfire (Papio 1988; Cabezudo et al. 1995), in areas without pre-fire management and in north-facing slopes (Leon et al. 2015), such as the one studied here. In the pre-fire treated plots, *R. officinalis* L. and *B. retusum* Pers. that *R. officinalis* tends not to germinate more vigorously after fire (Quintana et al. 2004). This low germination is probably due to low seed production, soil density, persistence and germination (Clemente et al. 2007). *R. officinalis* does not have any specific fire-adaptation mechanisms and behaves as a colonizer of open areas created by fire (Moya et al. 2015). Another

Table 3 Descriptive statistics for species richness (*S*), evenness (I_T) and density (*D*) from inventory at 2, 10 and 18 months after wildfire with six different management practices: CUn (control unburned), NT (no treatment), M05B (managed in 2005 and burned), M15B (managed in 2015 and burned), CR (cut and remove) and CL (cut and leave) ($N = 3$)

Vegetal index	Treatment	Months after fire	Mean	SD	Min	Max
S	Control	2	3.67	0.577	2.85	4.48
		10	3.67 B	0.577	2.85	4.48
		18	3.67	0.577	2.85	4.48
		ALL	3.67	0.500	2.85	4.48
	NT	2	5.33	4.04	1.42	9.24
		10	7.33 A	1.16	3.42	11.24
		18	6.67	2.31	2.75	10.58
		ALL	6.44	2.56	1.42	11.24
	M05B	2	4.33	1.53	3.00	5.66
		10	5.00 AB	0.00	3.67	6.33
		18	5.33	0.58	4.00	6.66
		ALL	4.89	0.93	3.00	6.66
	M15B	2	2.33 b	1.16	1.00	3.67
		10	6.00 aAB	1.00	4.67	7.33
		18	3.67 ab	0.58	2.34	5.00
		ALL	4.00	1.80	1.00	7.33
	CR	2	3.67	1.53	0.97	6.37
		10	6.67 A	1.53	3.96	9.37
		18	7.33	2.52	4.63	10.04
		ALL	5.89	2.37	0.97	10.04
	CL	2	3.33 b	1.53	1.70	4.96
		10	6.33 aAB	1.16	4.70	7.96
		18	7.33 a	0.58	5.70	8.96
		ALL	5.67	2.06	1.70	8.96
I_T	Control	2	21	12.29	3.64	38.36
		10	21	12.29	3.64	38.36
		18	21	12.29	3.64	38.36
		ALL	21	10.64	3.64	38.36
	NT	2	12.33	7.51	5.09	29.76
		10	27	15.88	9.58	44.24
		18	20	12.16	2.58	37.42
		ALL	19.78	12.43	2.58	29.76
	M05B	2	20.67	8.33	11.48	29.86
		10	23.67	7.51	14.48	32.86
		18	21.33	1.16	12.14	30.52
		ALL	21.89	5.80	12.14	32.86
	M15B	2	9.33 b	7.57	0.05	18.62
		10	34.00 a	5.29	24.71	43.29
		18	23.33 ab	6.66	14.05	32.62
		ALL	22.22	12.13	0.05	43.29
	CR	2	14	16.64	7.42	35.42
		10	34	2.00	42.59	55.42
		18	36.67	20.21	15.25	58.08
		ALL	28.22	16.95	7.42	58.08
	CL	2	8.33	5.51	0.58	16.08
		10	19	7.55	11.25	26.75
		18	16	1.73	8.25	23.75

Table 3 continued

Vegetal index	Treatment	Months after fire	Mean	SD	Min	Max
		ALL	14.44	6.73	0.58	26.75
D	Control	2	5.25	3.07	0.91	9.59
		10	5.25	3.07	0.91	9.59
		18	5.25	3.07	0.91	9.59
		ALL	5.25	3.07	0.91	9.59
	NT	2	3.08	1.88	1.27	7.44
		10	6.75	3.97	2.39	11.11
		18	5.00	3.04	0.64	9.36
		ALL	4.94	3.11	0.64	11.11
	M05B	2	5.17	2.08	2.87	7.46
		10	5.92	1.88	3.62	8.22
		18	5.33	0.29	3.04	7.63
		ALL	5.47	1.45	2.87	8.22
	M15B	2	2.33 b	1.89	0.01	4.66
		10	8.50 a	1.32	6.18	10.82
		18	5.83 ab	1.66	3.51	8.16
		ALL	5.56	3.03	0.01	10.82
	CR	2	3.50	4.16	1.85	8.85
		10	8.50	0.50	3.15	13.85
		18	9.17	5.05	3.81	14.52
		ALL	7.06	4.24	1.85	14.52
	CL	2	2.08	1.38	0.14	4.02
		10	4.75	1.89	2.81	6.69
		18	4.00	0.43	2.06	5.94
		ALL	3.61	1.68	0.14	6.69

Different letters represent significant differences with Tukey's HSD post hoc test at $p < 0.05$ between inventory dates (capital letters) and treatment (lower case letters)

factor that might have contributed to the increase in this species was the wet period post-fire. Normally, the development of *R. officinalis* in fire-affected areas increases with water availability (Parra and Moreno 2018). Pre-wildfire management negatively affected soil properties, a characteristic visible 2 months after the fire (Francos et al. 2018a), and in this context it seems likely that *R. officinalis* was able to germinate in poorer soils given that the species is adapted to harsh environments.

Eighteen months after the fire in NT and CL, the most commons species were *D. pentaphyllum* and *Smilax aspera* L., while the abundance of *R. peregrina* had fallen slightly. *D. pentaphyllum* propagates readily and has a good capacity to resprout after fire (Alegre et al. 1998), being favoured by post-wildfire management. Moya et al. (2009) observed that this species was abundant in post-wildfire thinned areas. As such, it is highly likely that CR management favoured the development of this species. *S. aspera* typically disappears following a fire (Úbeda et al. 2006) and the species only records its highest levels of abundance after long periods free of fire (Kazanis and Arianoutsou 2004). While our last inventory was made only 18 months after the fire, *S. aspera* still had a particularly slow response to fire disturbance. Tavsanoglu and Gurkan (2009) observed that *S. aspera* tends to increase in coverage with increasing time after the fire episode. In the study area the species tended to occupy the shallow, poorer soils, which were likely subject to strong post-wildfire erosion. *B. retusum* was the most common species in the plots treated pre-wildfire (even though *R. officinalis* covered a high percentage of M15B). The high presence of this species in the pre-wildfire managed plots 18 months after the fire may be evidence of soil degradation. The dominance of *B. retusum* tends to reduce the presence of grasses (Bonet 2004; De Luis et al. 2004). Here, this affirmation was very likely due to the impact of rainfall (especially high in months 15 and 16 after the fire) on soil degradation, especially because a high percentage of soil remained bare as observed Francos et al. (2019) medium term after severe wildfire and mechanical salvage logging. The slight increase in vegetation cover between 10 and 18 months after the wildfire in the pre-wildfire-treated plots was at the expense of *B. retusum*. In the post-wildfire managed plots, the most common species were *D. pentaphyllum* (CR), *Rubus ulmifolius* Schott (CR and CL), *S. aspera* (CR and CL). *R. ulmifolius* is resilient to fire (Tarrega et al. 1997); however, previous studies reported a decrease in the area covered by the species (Mitchell et al. 2009) and the flowering of the shrub only several years after the wildfire (Trabaud and de Chaterac 1985). Silva et al. (2014) observed an increase in the abundance of this species in a fenced-off plot (to protect it from grazing) with time post-fire, as in our study. *B. retusum* was one of the dominant species in CL, but with a much lower percentage compared to that in pre-treated plots. However, as described earlier, this species tends to cover large areas in recently burned forests.

Two months after the wildfire, herbaceous species covered the largest areas of the studied plots, especially in those subject to pre-wildfire treatment. This finding is in accordance with previous studies in the Mediterranean basin, which noted that herbs were the most common life form in the immediately after a fire because of the marked availability of nutrients, light and the capacity of these species to spread more rapidly (Papio 1988; Caturla et al. 2000; Kazanis and Arianoutsou 2004). The high abundance of herbs in the pre-fire-managed plot is, in all probability, an indicator that this specific treatment negatively affected soil properties and vegetation, which may in turn have undermined the resilience of this ecosystem to fire. This conclusion is further supported by the fact that these plots had the lowest vegetation cover. Ten and 18 months after the wildfire, the dominance of herbs decreased, while the presence of shrubs increased. Despite this decrease, herb cover in the plots managed pre-wildfire remained high, indicating that shrub species faced greater difficulties in colonizing these areas (Shive et al. 2013a). This result is further confirmation of the negative impact of this treatment on post-wildfire vegetation recovery, as previously noted by Shive et al. (2013b). The areas managed post-wildfire and the NT plots presented a similar degree of vegetation recovery, allowing us to conclude that manual treatment had no negative implications for post-fire recovery, presumably because the soil was not disturbed (Francos et al. 2018a, b). Here, we should stress the emergence of mosses 18 months after the fire in these plots. This finding can be attributed to the fact that moss species formed part of the pre-fire vegetation in these plots (Puche and Gimeno 2000), while the disturbance induced in the pre-fire-managed plots served to inhibit the emergence of mosses. The increase in moss species was small and gradual (cf. De las Heras et al. 1992, 1995), showing that NT, CR and CL are in an advanced plant succession stage compared to the pre-fire-managed plots. Resprouters were the most abundant species in all the plots studied (with the exception of CR 2 months after the fire; however, the differences were not substantial). This observation is in line with previous studies that found that the area covered by resprouters after the fire tends to be greater than that corresponding to seeders (Clemente et al. 1996; Calvo et al. 2005; González-De Vega et al. 2016). Resprouters have a high regenerative capacity attributable to their basal sprouters, a mechanism that seeders do not possess (Tavsanoglu and Gurkan 2009). Resprouters' capacity for regeneration is especially evident in the resource-poor conditions after high severity fires or post-fire erosive events, which reduce the seedbank and nutrient availability. During nutrient scarcity, seeders cannot compete with resprouters (Buhk et al. 2007). Here, the poor rate of recovery presented by seeders is attributable to the high severity of the fire and the post-fire erosion, especially 10–18 months after the fire.

Richness, evenness, density, diversity and maximum diversity

Species richness was significantly higher in CR and NT plots than in the control plot 10 months after the fire as found in other studies (e.g., Caturla et al. 2000; Mitchell et al. 2009; Santana et al. 2014; Tessler et al. 2016). This increase is probably attributable to the incorporation of ash into the soil profile and the consequent increase in nutrient availability. Temporal differences were observed between M15B and CL; thus, 10 months after the fire richness was significantly higher than after 2 months in M15B, a result that can be attributed to the impact of pre-fire management in M15B. Despite exposure to similar degrees of fire severity, we hypothesized that pre-fire activities had detrimental impacts on soil, indeed, 2 months after the fire, species richness was extremely low (2.33) although it increased to 6.00 species/plot 10 months after the wildfire. In the case of CL, richness was significantly higher both ten and 18 months after the wildfire than just two months and this seems to reflect the positive impact of post-fire management in this plot.

In line with species richness (*S*) and for the same reasons, I_T and *D* values were significantly higher in M15B 10 months after the wildfire than after just 2 months, but otherwise, they did not differ were found among the plots at 2, 10 and 18 months after the wildfire. Significant differences were only found between sampling dates for M15B, providing further evidence that pre-fire management was detrimental for post-wildfire *S*, I_T and *D*. Specifically, diversity was significantly lower in M15B than in CL 18 months after the wildfire, in support of the aforementioned argument. Eighteen months after the wildfire, the effects of pre-fire salvage logging remained visible. In contrast, no differences were observed between

Table 4 Descriptive statistics for diversity (H') and maximum diversity (HMax)

Vegetal index	Treatment	Months after fire	Mean	SD	Min	Max
H'	Control	2	0.21	0.11	0.05	0.37
		10	0.21	0.11	0.05	0.37
		18	0.21 AB	0.11	0.05	0.37
		ALL	0.21	0.10	0.05	0.37
	NT	2	0.41	0.06	0.18	0.64
		10	0.33	0.18	0.10	0.57
		18	0.42 AB	0.21	0.19	0.65
		ALL	0.39	0.15	0.10	0.65
	M05B	2	0.22	0.06	0.14	0.30
		10	0.23	0.08	0.14	0.31
		18	0.25 AB	0.02	0.17	0.33
		ALL	0.23	0.05	0.14	0.33
	M15B	2	0.31	0.17	0.16	0.45
		10	0.18	0.04	0.03	0.33
		18	0.16 B	0.04	0.02	0.31
		ALL	0.22	0.11	0.02	0.45
	CR	2	0.57	0.42	0.22	0.93
		10	0.20	0.05	0.15	0.55
		18	0.22 AB	0.06	0.14	0.57
		ALL	0.33	0.28	0.14	0.93
	CL	2	0.56	0.38	0.20	0.93
		10	0.38	0.22	0.02	0.75
		18	0.46A	0.04	0.10	0.23
		ALL	0.47	0.24	0.02	0.93
Hmax	Control	2	0.56	0.07	0.46	0.66
		10	0.56B	0.07	0.46	0.66
		18	0.56B	0.07	0.46	0.66
		ALL	0.56	0.06	0.46	0.66
	NT	2	0.65	0.30	0.36	0.94
		10	0.86A	0.07	0.57	1.15
		18	0.80AB	0.17	0.51	1.09
		ALL	0.77	0.10	0.36	1.15
	M05B	2	0.62	0.15	0.49	0.75
		10	0.70AB	0.00	0.57	0.83
		18	0.72AB	0.05	0.60	0.85
		ALL	0.68	0.09	0.49	0.85
	M15B	2	0.32b	0.28	0.08	0.56
		10	0.77aA	0.07	0.53	1.01
		18	0.56abB	0.07	0.32	0.80
		ALL	0.55	0.25	0.08	1.01
	CR	2	0.53	0.21	0.31	0.76
		10	0.82A	0.11	0.59	1.04
		18	0.85AB	0.15	0.62	1.07
		ALL	0.73	0.20	0.31	1.07
	CL	2	0.49b	0.20	0.31	0.67
		10	0.80abA	0.08	0.62	0.98
		18	0.86aA	0.03	0.69	1.04
		ALL	0.72	0.20	0.31	1.04

Different letters represent significant differences in Tukey'a post hoc test at $p < 0.05$ between inventory dates (2, 10 and 18 months after wildfire, capital letters) and six treatments (lower case letters): CUn (control unburned), NT (no treatment), M05B (managed in 2005 and burned), M15B (managed in 2015 and burned), CR (cut and remove) and CL (cut and leave) (N = 3) non-managed and managed areas, as reported elsewhere (De las Heras et al. 2004), indicating that the post-fire treatment adopted in this area was not detrimental to vegetation. Ten months after the fire, the burned plots had a higher maximum diversity compared to that of the control plot, similar to the findings of González-De Vega et al. (2018). In line with the H' values reported, no significant differences were identified between the plots treated before and after the fire as found by Garcia-Orenes et al. (2017). Eighteen months after the wildfire, maximum diversity was significantly lower in control and M15B than in the other plots, again indicating that the pre-wildfire treatment negatively impacted vegetation recovery.

Overall implications for forest management

Pre- and post-wildfire management typically influence vegetation recovery. Pre-fire management seeks to reduce the accumulation of fuel and the occurrence of high severity wildfires (Kalies and Kent 2016). Yet, these activities, typically performed with heavy machinery, can have a detrimental impact on soils (Francos et al. 2016b, 2019) and reduce the capacity for post-fire recovery. In our study, the pre-fire managed plots had the poorest vegetation cover, which seems to indicate their reduced resilience to fire. Eighteen months after the wildfire, the percentage vegetation cover continued to be lower than on the non-treated plots, showing that pre-fire treatment was detrimental for vegetation recovery. The relative lack of vegetation cover increases vulnerability to soil erosion and nutrient transport (Pereira et al. 2018). Here, the plot managed before the wildfire (M15B) suffered the greatest impact as a result of burning, with the lowest vegetation recovery, a high presence of *B. retusum* and germination of *R. officinalis* (a species adapted to poor soils), and very low S, I_T, H' and Hmax values, especially immediately after the wildfire. In contrast, plots that received the post-wildfire treatment increased their vegetation cover, and S, I_T, H' and Hmax values did not differ from those of the NT plot. These results seem to indicate that manual fuel management in specific areas affected by high severity fires is a good solution. However, heavy machinery is most often used for post-fire management and negatively impacts soil properties and vegetation recovery (McIver and Starr 2001; Wagenbrenner et al. 2016; Garcia-Orenes et al. 2017; Francos et al. 2019). All in all, more sustainable practices are needed to reduce the amount of fuel to avoid negative impacts on soil and vegetation recovery after wildfires. Here, the impact of the pre-wildfire treatment was more detrimental than that of the actual wildfire. Prescribed fires (Úbeda et al. 2018) and grazing (Davies et al. 2016; Bruegger et al. 2016), for example, might be used to reduce the amount of fuel and the impact of salvage logging on soil properties.

Conclusions

Pre-wildfire treatments negatively impacted post-wildfire vegetation recovery, especially when done shortly before (here, 2 months) the outbreak of the fire. Such disturbances, especially to the soil, seem to accentuate the impact of wildfires, reducing the resilience of the ecosystem. In contrast, manual operations seem to offer certain benefits in terms of vegetation cover, increasing soil protection against erosion agents and limiting potential post-wildfire degradation. Nonetheless, the use of manual management is very limited due to its characteristics should be confined to reduced areas. Further studies are needed to determine the impact of specific pre-wildfire management practices on the impact of wildfires so that strategies can be designed for maintaining ecosystem resilience.

Acknowledgements This study was supported by POST-FIRE_CARE Project (CGL2016-75178-C2-2-R) sponsored by the Spanish Ministry of Economy and Competitiveness and AEI/FEDER, UE. Support was also received from the FPU Program (FPU 014/00037) sponsored by the Ministry of Education, Culture and Sports, the EST16/00183 to a short stay in Mykolas Romeris University (Vilnius, Lithuania) to the Ministry of Education, Culture and Sports and project 2017SGR1344 of the Generalitat de Catalunya. We also thank the University of Barcelona's technical services for the English revision of the manuscript. We also thank the Diputación of Barcelona for facilitating our fieldwork in the study area.

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