



Temporal dynamics of carbon storage in a Mediterranean mountain scrubland managed by prescribed fire

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ABSTRACT

Farmland abandonment and reduction of grazing activity, mainly in mountain areas with remote access and ageing population, have been contributing to shrub encroachment in such territories and, consequently, to increase fuel load available for triggering wildfires. Accordingly, it is necessary to use vegetation management practices in order to reduce wildfire risk, prescribed fire being one of the most common techniques used in the Mediterranean region. This research focused in the effects of a prescribed fire (PF) applied in Montesinho Natural Park (PNM), NE Portugal, on the temporal dynamics of carbon storage in mineral soil, litter layer (organic horizon), and shrub biomass. Before PF and thirty-six months after PF, aboveground shrub biomass was collected in areas of 1 m² in 11 plots randomly distributed in the experimental shrub area. Also, in the same plots, litter thickness was measured and soil samples were collected before, two, six and thirty-six months after PF, in order to assess carbon concentration, bulk density and coarse elements content. Despite low to moderate fire intensity, carbon storage changes were observed in all compartments evaluated. Thirty-six months after PF, carbon storage in aboveground biomass of shrub species (7.4 Mg C ha⁻¹) was roughly two-thirds of that recorded prior to PF, and in litter layer (1.6 Mg C ha⁻¹) it was about half of that in the original situation (before PF). In contrast, the mineral soil showed a 10% carbon increase (6.4 Mg C ha⁻¹). Based on the balance between losses (shrub species and litter layer) and gains (mineral soil), at the end of the monitoring period (36 months), there was an annual positive rate of carbon storage, equivalent to 0.2 Mg C ha⁻¹ year⁻¹. Even after anthropogenic disturbances, such as prescribed fire, shrub communities constitute important terrestrial carbon pools; hence, these ecosystems might play an important role in mitigating climate change.

1. Introduction

Social changes in Europe have contributed to the abandonment of agrarian activity (agriculture and livestock), and landscapes have increased their susceptibility to secondary succession, mainly represented by shrub encroachment (Fernandes et al., 2013; Petersen and Drewa 2014; Santín and Doerr, 2016; Michielsen et al., 2017), which led to increasing fire-proneness in some environments and, consequent, the risk of uncontrollable wildfires. The Mediterranean region has favorable characteristics to fire occurrence (climatic, environmental and socio-economic), fire being an important ecological process on natural landscape evolution in this region, as well as in other parts of the world (Pausas et al., 2008; Penman et al., 2011; Bixby et al., 2015; Pereira

et al., 2021). Nevertheless, wildfires cause great impacts on the ecosystems including organic and mineral soil horizons, vegetation and water, culminating in a devastating change on landscapes (Certini, 2005; Bowman et al., 2009; Lourenço et al., 2011; Inbar et al., 2014; Pellegrini et al., 2018).

In order to avoid or mitigate the occurrence and impacts of wildfires, it is becoming increasingly common to use prescribed fire as a tool to manage fuel load, thus changing the severe fire conditions characterizing most wildfires by planned and less intense burns (Santín and Doerr, 2016; Hunter and Robles, 2020). In general, prescribed fires produce less pronounced effects than those caused by wildfires (Nave et al., 2011; Pereira et al., 2021), since they are applied on adequate weather conditions, reaching lower intensity, severity, and energy production.

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Additionally, this management practice is often effective in controlling shrub species encroachment, contributing to enhance biodiversity (Policelli et al., 2019) and maintenance valuable natural resources conservation, such as the soil and water (Scharenbroch et al., 2012; Fonseca et al., 2017). The impact of prescribed fires on soil organic matter and consequently in the carbon cycle is an essential topic to climate change and soil quality, as vast landscapes are managed annually across the globe by prescribed fire. Soil organic carbon content is one of the most important indicators of soil quality, and a crucial driver of ecosystems sustainability (Lal, 2015). Thus, it is essential to apply territorial management practices that result in positive soil organic matter balances, which in turn translates into increased soil productivity and additional storage of atmospheric carbon. In the course of a prescribed fire (low-to-moderate severity) litter layer and plant biomass are partially transformed into black carbon as a result of their incomplete combustion. This new carbon form is an important pool of the global carbon cycle since it is much more stable than other carbon forms, representing therefore a carbon sink and an increase on soil fertility (Major et al., 2010; Knicker, 2011; Krishnaraj et al., 2016).

Fire impacts have increased considerably from the second half of the 20th century onwards (Fernandes et al., 2013). The high frequency of fires in the Mediterranean region raises concerns about emissions and carbon sequestration. In addition, land degradation associated with ineffective land use management led to carbon depletion from soils and reduced carbon storage capacity, as soil organic carbon is typically the largest carbon pool in terrestrial biosphere (e.g. Rau et al., 2010; Armas-Herrera et al., 2016; Fonseca et al., 2019). This soil organic carbon depletion is a worldwide concern and a major reason of soil degradation, mainly in the semi-arid Mediterranean regions (Lal, 2015). In northeast of Portugal, where this study took place, around 70% of the burned area corresponds to shrublands, followed by forest (20%) and finally by cropland (10%) (Cavalli et al., 2019). This trend is often observed in many Mediterranean countries and highlights the need for new models for managing the territory and valuing the services provided by

shrublands, such as soil and biodiversity conservation, carbon storage and sequestration, and water resource protection (Smith et al. 2011; Fonseca et al., 2017; Harper et al., 2018). Fire is one of the most important disturbances affecting carbon stocks in shrub communities (soil, organic horizon, and biomass), yet the effects of fire on carbon storage in this type of ecosystems are poorly understood. Accordingly, it is essential to increase knowledge about management techniques of shrub areas, which can act as support for managers, stakeholders and policymakers.

The aim of this study was to evaluate over a 36 months-period the changes in carbon stocks in shrub biomass, litter layer and mineral soil in a Mediterranean shrubland after a prescribed fire used as a fire prevention forest management practice. Even though this tool may have benefits for the reduction of wildfire risk, it is essential to understand its environmental impacts such as those affecting carbon sequestration and the coupled mitigation of greenhouse gases emissions.

2. Methods

Montesinho Natural Park (PNM) is a mountain protected area in the Mediterranean region, northeast Portugal, covering of about 75,000 ha. It is a mountainous territory composed of a set of natural, semi-natural and humanized landscapes, where there is high biological diversity, recognized at regional, national and European level (ICNF, 2018). Although PNM includes 92 villages, social changes such as rural population decline and the aging led to farmland abandonment and livestock reduction, and currently, around one-third of PNM area is covered by shrub communities. Thus, according to the PNM management plan, prescribed fires to control shrub encroachment have been used as a useful tool to protect existing forests against wildfires.

The present study was carried out in a shrubland, subject to a prescribed fire, located in PNM (41°53'57"N, 6°40'55"W) (Fig. 1). It is a plateau zone, 800 m asl, cut by deep valleys. The bioclimatic is supra-Mediterranean with 12 °C mean annual temperature and 850 mm

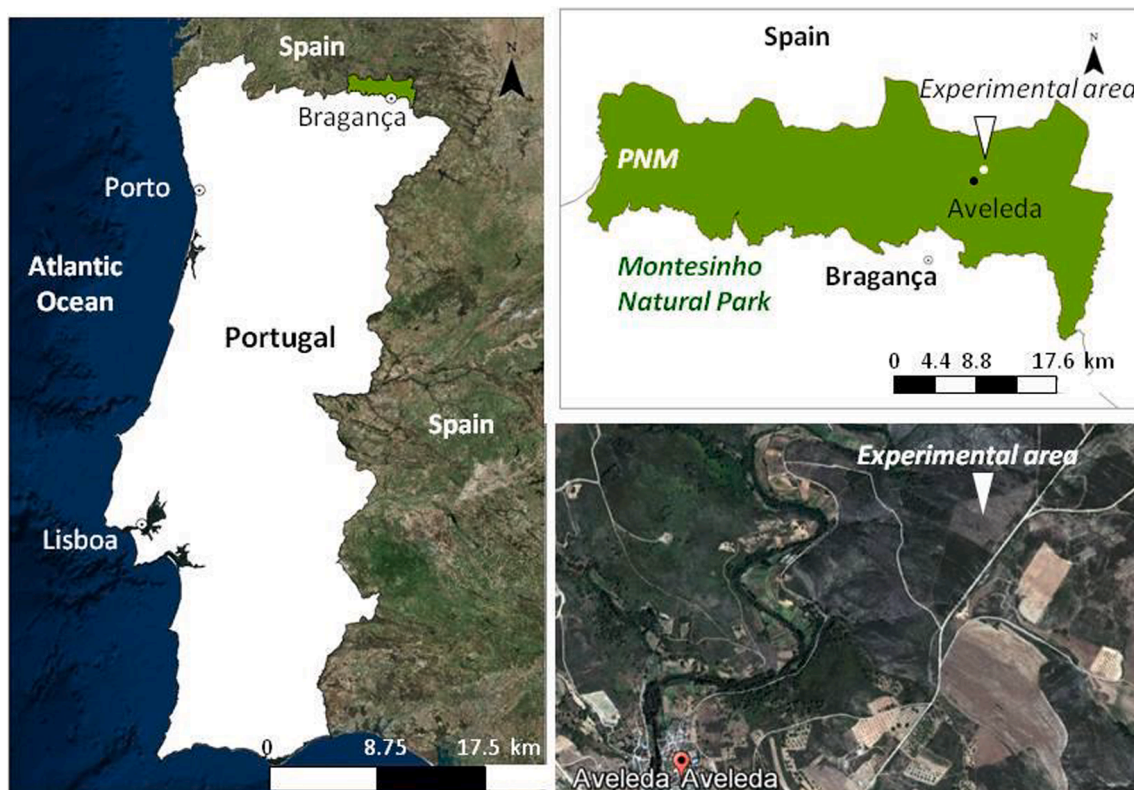


Fig. 1. Location of the study area in Montesinho Natural Park (PNM), NE Portugal.

mean annual rainfall, concentrated from October to April and a marked dry season among June and August (Agroconsultores and Coba, 1991). Soils are classified as Umbric Leptosols derived from schist (FAO/UNESCO, 1988; Agroconsultores and Coba, 1991) with high stoniness, medium-texture, low pH, moderate to high in organic matter content, low base saturation and cation exchangeable capacity, very low P and moderate K contents (Agroconsultores and Coba, 1991; Fonseca et al., 2017).

Prescribed fire was performed on a shrubby area, twelve years after a wildfire that partially destroyed a *Pinus pinaster* neighboring forest. This 5-ha area was subject to prescribed fire with the aim to protect the remaining forest that survived to the mentioned wildfire. Carbon storage in shrub biomass, litter and mineral soil was evaluated before prescribed fire (BF), and two (2 M), six (6 M) and thirty-six months (36 M) after fire.

The shrub vegetation, including the roots exposed to the surface, the litter layer and the soil surface conditions were observed immediately after the fire passage. According to the indicators established by Persons et al. (2010) the fire was considered of low to moderate severity. Thus, it was considered that about 10% of the superficial roots were destroyed by fire, which corroborates Persons et al. (2010), assuming that after 6 months 5% of the roots had already been recovered and at the end of the experimental period the root biomass was the same as that determined prior to fire.

Before prescribed fire (BF), 11 plots of 1 × 1 m were randomly established in the area to be burnt. In each plot, an inventory of shrub species was performed and all aboveground biomass was clipped, separated by species, their fresh weight being recorded in the field, and subsamples transported to the laboratory. Aboveground biomass subsamples were dried at 65 °C for 72 h to determine dry mass. Vegetation cover was assessed by visual observation of the sampling plots prior to clipping aboveground biomass. Before prescribed fire, 44% of the area was occupied by *Erica australis*, 30% by *Chamaespartium tridentatum* and 26% by *Cistus ladanifer*, meaning that the experimental site is representative of the shrub communities found in PNM (IPB/ICN, 2007; Fonseca et al., 2012). Shrub canopy cover reached 90–100 % before prescribed fire. The same procedure for aboveground biomass assessment was repeated 36 months (36 M) after prescribed fire. The shrub biomass remaining immediately after prescribed fire was estimated based on the combustion factors evaluated from field observations: 80% for *Erica australis* and *Chamaespartium tridentatum*; 50% for *Cistus ladanifer*. The evolution of aboveground shrub biomass two (2 M) and six (6 M) months after fire was estimated based on Eq. (1), established by Rambal (2001), for Mediterranean-type shrublands following disturbance, summed to the unburned biomass (the complementary to 100% of the above-mentioned combustion factors).

$$AGB = 2880(1 - e^{-0.0896t}) \quad (1)$$

where AGB is aboveground biomass of shrub species (g m^{-2} , dry mass); t is the time after disturbance in years.

Belowground shrub biomass (before and after fire) was estimated using the root-to-shoot ratio, 2.48 for *Erica australis* and 0.60 for *Chamaespartium tridentatum* and *Cistus ladanifer*, determined by Fonseca et al. (2012) in a similar shrubby area neighboring the present study site.

Litter thickness, needed to estimate carbon stocks in organic residues accumulated over the soil surface, was measured before fire (BF) and two (2 M), six (6 M), and thirty-six (36 M) months after fire. The average litter layer thickness (mm) for each one of the 11 sampling plots (1 m^2) was calculated from nine measurements taken at regularly spaced and fixed points within the plot (at the center of each one of the nine virtual squares equally dividing the plot area).

In the same above mentioned 11 plots, disturbed and undisturbed soil samples were collected before prescribed fire (BF) and 2 M, 6 M and 36 M after fire, at 0–5, 5–10 and 10–20 cm depths. Bulk density was determined in undisturbed samples, weighting oven dried soil (at 105 °C), collected in 100 cm^3 cylinders. Disturbed soil samples were air

dried and sieved with a 2 mm sieve, to determine the coarse elements content and total soil carbon organic concentration in the fine earth fraction (<2 mm) by the Walkley-Black method (Nelson and Sommers, 1996). Soil samples were tested with an acid-drop but no carbonates were detected, hence the total soil carbon was assumed to be comparable to soil organic carbon.

Aboveground and belowground shrub biomass values were converted to carbon (Mg C ha^{-1}), multiplying these values by 0.558 and 0.536, respectively, corresponding to the average carbon concentrations in dry matter obtained by Fonseca et al. (2012) in PNM shrub communities in the same area. The estimate of litter carbon storage was obtained using Eq. (2), proposed by Fonseca et al. (2012) for the PNM area.

$$LCS = 0.1229 LT + 0.0437(LT^2) = 0.81, P < 0.000 \quad (2)$$

where LCS is litter C storage (kg m^{-2}), LT is litter thickness (cm)

Soil organic carbon storage (C_{SOC} ; Mg C ha^{-1}) was evaluated by multiplying carbon concentration (Cc; %) by bulk density (BD; g cm^{-3}) and thickness (z; cm) of the mineral soil layer with a correction for coarse elements content (CE; v v^{-1}), using the Eq. (3) (Percival et al., 2000; Sil et al., 2017; Fonseca et al., 2019).

$$C_{\text{SOC}} = z Cc (BD - 2.65 CE/100) \quad (3)$$

The total carbon storage in the whole system (TC) per unit area (Mg C ha^{-1}) was estimated by summing the mean amount of carbon in different pools (Eq. (4)).

$$TC = C_{\text{SB}} + C_{\text{L}} + C_{\text{SOC}} \quad (4)$$

where C_{SB} is carbon content in above and belowground shrub biomass, C_{L} is carbon content in litter, C_{SOC} is carbon content in mineral soil.

The statistical treatment of the information collected included descriptive statistics, analysis of variance (one-way and two-way) and tests of means comparison (Tukey, $p < 0.05$). The variables under study passed the Kolmogorov-Smirnov normality test ($p > 0.05$). The Interquartile Range test was applied but no outlier was detected in the data sets.

3. Results

3.1. Carbon storage in shrub biomass and litter layer

The aboveground biomass consumed during the prescribed fire was 80% for *E. australis* and *C. tridentatum* and 50% for *C. ladanifer*. Fig. 2 shows the dynamics of carbon storage in aboveground biomass of the three species during the monitoring period (36 months). *C. tridentatum* showed significantly lower carbon storage in aboveground biomass than the other two species, which showed also quite different post-fire dynamics. *E. australis* had much higher carbon storage in aboveground biomass before fire but endured a much more pronounced fire impact as compared to *C. ladanifer*. Two months after fire (2 M), significant carbon losses in the aboveground biomass were observed in each one of the studied species. As compared to the BF condition, average losses amounted to 4.0 Mg C ha^{-1} (78% of the BF value), 2.0 Mg C ha^{-1} (77%) and 1.9 Mg C ha^{-1} (48%) for *E. australis*, *C. tridentatum* and *C. ladanifer*, respectively. At the end of the monitoring period (36 months) carbon storage in aboveground biomass was still significantly lower than that before prescribed fire (BF), representing absolute (and relative to BF) losses of 2.4 Mg C ha^{-1} for *E. australis* (47%), 1.0 for *C. ladanifer* (25%), and 0.9 (35%) for *C. tridentatum*.

The average increase rate of carbon storage associated to aboveground biomass recovery following prescribed fire can be estimated as $0.6 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ for *E. australis* (11% of BF carbon storage in aboveground biomass); 0.3 for *C. ladanifer* (8%); and 0.4 for *C. tridentatum*, (15%). At these average rates, recovery of the BF condition would be achieved in 7, 6 and 5 years respectively for *E. australis*, *C. ladanifer* and *C. tridentatum*.

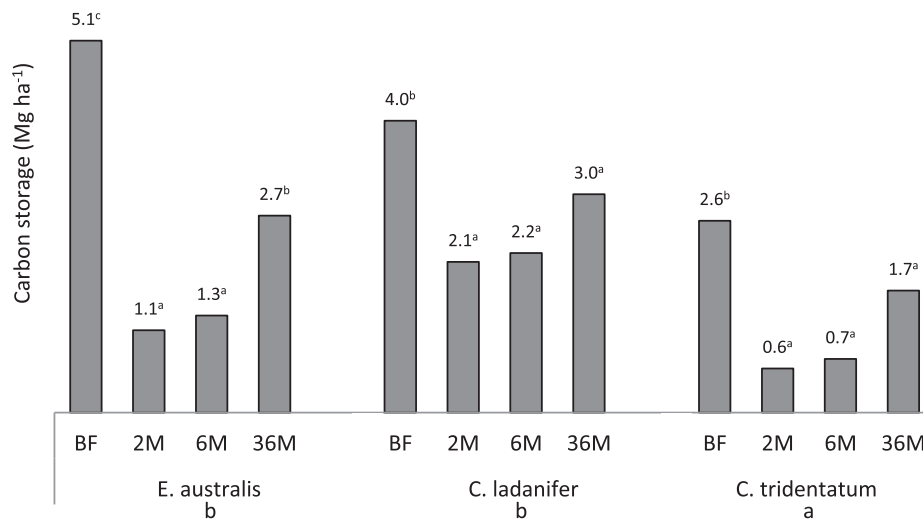


Fig. 2. Carbon storage in aboveground biomass of shrub species *Erica australis*, *Cystus ladanifer* and *Chamaespartium tridentatum*, before (BF), two (2 M), six (6 M) and thirty-six (36 M) months post-fire. The same letter indicates differences not statistically significant between compared categories of the tested effects: species and time since fire for each species (two-way ANOVA and Tukey test, $p > 0.05$).

Considering the three species together, thirty-six months after fire (36 M), aboveground biomass stored about two-thirds of the pre-fire carbon (7.4 Mg C ha⁻¹). The overall post-fire carbon loss (at 2 M) reached 7.9 Mg C ha⁻¹ (68% of BF carbon storage). The estimated carbon increase rate after fire is 1.3 Mg C ha⁻¹ year⁻¹ (11% of BF carbon storage).

The effects produced by the prescribed fire on the superficial roots were not significant, with the summed carbon storage in root biomass of the three species undergoing small variations over time (10%, 5% and 0% after 2 M, 6 M and 36 M, respectively) (Fig. 3). *C. tridentatum* and *C. ladanifer* showed not significantly different carbon storage values in belowground biomass, varying from 1.3 to 2.3 Mg C ha⁻¹, which represent around 36% of the carbon stored in the total biomass (above + belowground). In contrast, *E. australis* stored from 10.9 to 12.1 Mg C ha⁻¹ (roughly 70% of the total biomass carbon), absolute and proportional values those are significantly higher than those of the other two species.

Carbon storage in litter layer was significantly higher before the

prescribed fire (3.06 Mg C ha⁻¹) than two (0.97 Mg C ha⁻¹), six (1.02 Mg C ha⁻¹) and thirty-six months (1.60 Mg C ha⁻¹) after fire (Fig. 4). Two months after the fire (2 M), litter layer recorded a carbon loss of about 68% (2.09 Mg C ha⁻¹) and at the end of the monitoring period (36 M) litter carbon storage was roughly half of that existing prior to burning (52% of the original carbon).

3.2. Carbon storage in mineral soil

Fire did not significantly affect carbon concentration, coarse elements content and soil bulk density (Table 1), parameters that integrate Eq. (7) used in the determination of soil carbon per unit area (Mg C ha⁻¹). However, as compared to before fire soil conditions, at the end of the experimental period (36 M) there is a trend towards an increase in the carbon concentration and a reduction of coarse elements content and bulk density. Carbon concentration declined with soil depth while bulk density and coarse fragments increased (Table 1), following their typical variation patterns. The high stoniness found constitutes a common

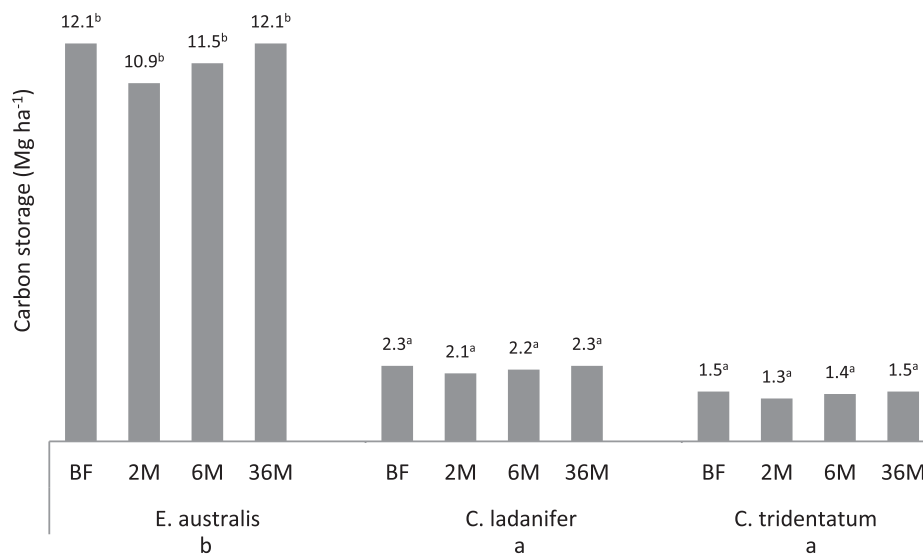


Fig. 3. Carbon storage in belowground biomass of shrub species *Erica australis*, *Cystus ladanifer* and *Chamaespartium tridentatum*, before (BF), two (2 M), six (6 M) and thirty-six (36 M) months post-fire. The same letter indicates differences not statistically significant between compared categories of the tested effects: species and time since fire for each species (two-way ANOVA and Tukey test, $p > 0.05$).

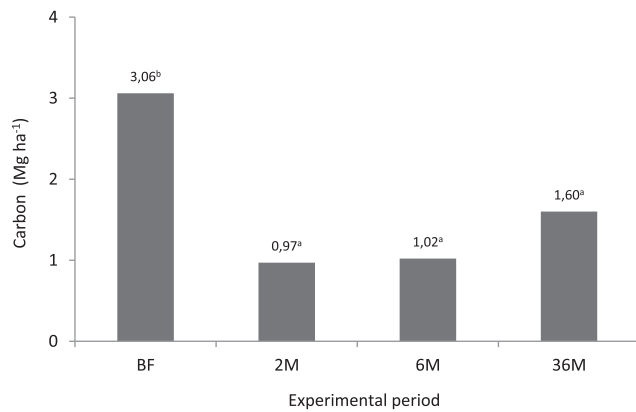


Fig. 4. Carbon storage in litter before (BF), two (2 M), six (6 M) and thirty-six (36 M) months post-fire. Average values with the same letter are not significantly different ($p > 0.05$).

Table 1

Soil carbon concentration (g C kg⁻¹), coarse elements content (%) and soil bulk density (g cm⁻³) in the soil layers before (BF), two (2M), six (6M) and thirty-six (36M) months post-fire, expressed as mean and standard deviation.

Depth (cm)	Experimental period			
	BF	2 M	6 M	36 M
SOC concentration (g kg⁻¹)				
0–5	46.5 ± 5.1	47.0 ± 7.8	43.7 ± 7.2	50.1 ± 8.1
5–10	38.5 ± 6.7	37.6 ± 9.6	34.5 ± 5.5	41.2 ± 7.0
10–20	30.8 ± 6.6	33.4 ± 10.5	28.0 ± 7.4	37.3 ± 6.3
Coarse elements content (%)				
0–5	22.1 ± 7.4	21.3 ± 6.4	20.9 ± 6.2	20.1 ± 6.4
5–10	29.1 ± 16.7	25.8 ± 14.7	25.1 ± 14.5	24.4 ± 14.8
10–20	37.4 ± 17.2	33.0 ± 15.6	32.1 ± 13.9	29.9 ± 16.3
Bulk density (g cm⁻³)				
0–5	1.13 ± 0.04	1.12 ± 0.06	1.15 ± 0.06	1.10 ± 0.09
5–10	1.19 ± 0.06	1.20 ± 0.08	1.23 ± 0.05	1.18 ± 0.12
10–20	1.26 ± 0.06	1.24 ± 0.10	1.29 ± 0.07	1.21 ± 0.14

feature of the Mediterranean basin soils.

Fig. 5 presents carbon stocks in the mineral soil layers. Two months after the fire (2 M), at all depths, there was a slight increase in soil carbon, yet not statistically significant as compared to BF. Six months after fire (6 M), carbon storage decreased in a way barely noticeable at all depths, increasing again thirty-six months after fire (36 M). In all layers, these were minor and not statistically significant changes in soil carbon storage over time; anyway, a positive balance was reached after three years. Carbon stored in the whole mineral soil compartment (0–20 cm) was 64.1 Mg C ha⁻¹ (BF), 67.3 Mg C ha⁻¹ (2 M), 63.5 Mg C ha⁻¹ (6 M) and 70.5 Mg C ha⁻¹ (36 M), meaning carbon gains of 5% (2 M) and 10% (36 M) and losses of 1% (6 M) in the post-fire period as compared to the original situation (BF). Between two (2 M) and six (6 M) months there was a carbon loss of 3.8 Mg ha⁻¹ while from six (6 M) to thirty-six (36 M) months there was an increase of 7.0 Mg ha⁻¹.

Soil layers sampled showed significantly different carbon storage values along the study period, the lowest occurring in the 5–10 cm layer and the highest at 10–20 cm soil depth. Comparing carbon storage before fire (BF) and thirty-six months (36 M) post-fire, gains are found in all the three layers, which are statistically significant at 10–20 cm depth. These results indicate a good resilience for soil organic carbon.

3.3. Carbon storage in the whole system

After thirty-six months (36 M) the largest carbon stock reduction was observed in the litter compartment (48% corresponding to 1.5 Mg C ha⁻¹), followed by that in the total shrub biomass (above and below-ground biomass), which decreased by 16% equivalent to 4.3 Mg C ha⁻¹ (considering BF as reference). In contrast, in the mineral soil, carbon storage increased by 10% (6.4 Mg C ha⁻¹). As compared to the pre-fire situation (94.7 Mg C ha⁻¹), total carbon stocks in the whole system (vegetation, litter layer and mineral soil) suffered a significant reduction at 2 M (86.4 Mg C ha⁻¹) and 6 M (83.8 Mg C ha⁻¹), recovering to a statistically similar status at 36 M (95.3 Mg C ha⁻¹) (Fig. 6). Actually, at the end of the experimental period (36 M), total carbon accumulated in the whole system exceeded in 1% the original amount (0.6 Mg C ha⁻¹ equivalent to an average increase rate of 0.2 Mg C ha⁻¹ year⁻¹).

The soil compartment was the largest carbon pool, representing more than 65% of the carbon stored by the scrubland, followed by vegetation (less than 30%) and finally by litter, which does not exceed 3% of the total carbon stored in the system.

Basically, fire had different impacts on shrub biomass, litter and mineral soil. Responses of vegetation and litter to disturbances caused by fire were more pronounced than that of mineral soil (Fig. 7). Carbon losses associated to shrub biomass and litter 36 months after fire were balanced by carbon gains in mineral soil in the same period, meaning that the whole system is a net carbon sink.

4. Discussion

Carbon storage dynamics in aboveground biomass of shrub species depends widely on the biomass combustion factor, and on the species' resprout ability after the prescribed fire. In the period between two (2 M) to thirty-six (36 M) months post-fire, the carbon storage capacity in aboveground biomass ranged from 1.1 to 2.7 Mg C ha⁻¹, 2.1 to 3.0 Mg C ha⁻¹ and 0.6 to 1.7 Mg C ha⁻¹ for *E. australis*, *C. ladanifer* and *C. tridentatum*, respectively. *E. australis* and *C. tridentatum* had a combustion factor of 80%, while *C. ladanifer* showed higher fire resistance, with a combustion factor of 50%. Available information on the impacts of fire on shrub communities is scarce, nonetheless Fernandes et al. (2000) in a work developed in Northern Portugal with *Erica umbellata* and *Chamaespartium tridentatum*, corroborate combustion factors presented in this study, for similar species. Also, with the purpose to add value to *C. ladanifer* as a renewable energy source, research is being conducted to obtain data on the combustion of *Cistus ladanifer* pellets, produced and tested under the European standards for solid biofuels (Pereira et al., 2013; Curió-Prieto et al., 2017). Two months after fire (2 M), carbon storage is closely related with biomass not consumed by fire, showing losses of 78, 77 and 48% for *E. australis*, *C. tridentatum*, and *C. ladanifer*, respectively. Six months after fire (6 M), the vegetation regrowth contribution to storage in aboveground biomass is still negligible. At the end of the monitoring period (36 M), the amount of carbon loss as compared to the pre-fire condition (BF) is similar for *C. ladanifer* and *C. tridentatum*, around 1 Mg ha⁻¹ (representing 25% of that at BF for *C. ladanifer* and 35% for *C. tridentatum*), while *E. australis* records losses of 2.4 Mg ha⁻¹ (equivalent to 47%). These results seem to be strictly related to specific regrowth patterns, as the *E. australis* presented some stumps without regrowth, and also some reprints did not survive to the dry summer period (June to August). After a disturbance, the response of species that resprout can vary considerably in number of resprouts and in individuals that resprout (Michielsen et al., 2017). Sprouting ability of shrub species depends on the natural conditions of the biome (Vesk and Westoby, 2004) and of species-specific development patterns (Gedroc et al. 1996). On the other hand, fire can alter the dominance of shrub species. Céspedes et al. (2014) observed a change in the balance of dominant shrub species in a Mediterranean shrubland during post-fire regeneration. Also, O'Connor et al. (2020) reported dominance reduction of resprouting shrub in woody encroached grassland after a

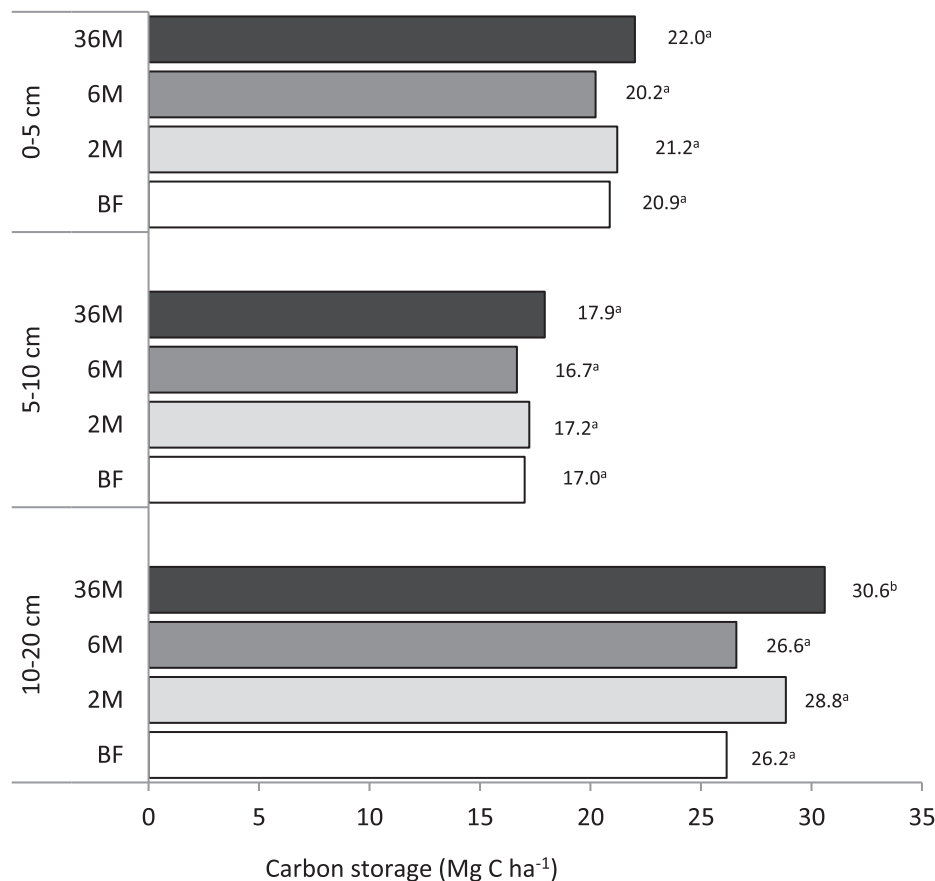


Fig. 5. Carbon storage in mineral soil layers before (BF), two (2 M), six (6 M) and thirty-six (36 M) months post-fire. Average values of each soil layer with the same letter are not significantly different ($p > 0.05$).

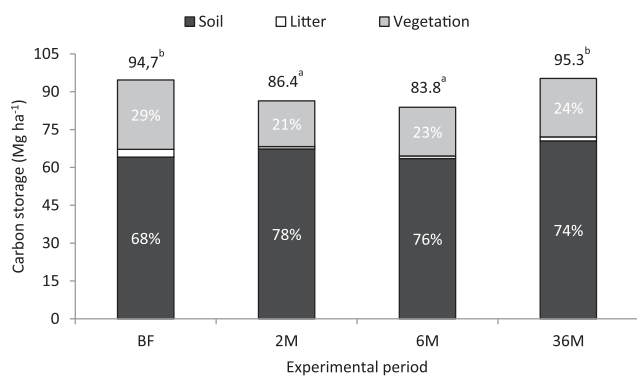


Fig. 6. Total carbon storage in entire system (soil, litter and vegetation compartments) before (BF), two (2M), six (6M) and thirty-six (36M) months post-fire. Average values with the same letter are not significantly different ($p > 0.05$).

prescribed fire.

The root-to-shoot ratio can be used as an adaptation indicator to severe drought conditions during the summer period in the Mediterranean region (Hilbert and Canadell, 1995). To manage the summer water deficit, some species allocate a larger amount of resources to the root than to the shoot part (Guilherme et al. 1998; Lloret et al. 1999). This adaptation mechanism is clearly observed in *E. australis* (showing a root-to-shoot ratio of 2.48 against 0.60 for *C. tridentatum* and *C. ladanifer*), which might mean a less resilient response to water shortage, reflected in a lower post-fire regeneration capacity and consequently higher carbon losses in aboveground biomass, as compared to the other species

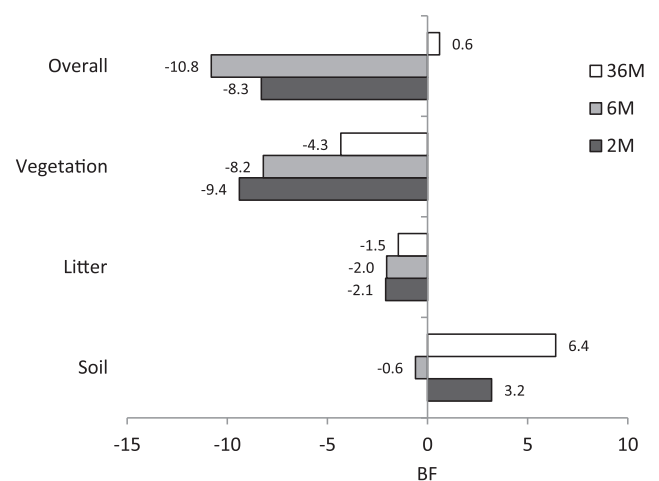


Fig. 7. Variation of carbon stored (Mg ha⁻¹) in the compartments soil, litter and vegetation, throughout the experimental period (2 M, 6 M and 36 M) compared to the pre-fire situation (BF).

(*C. tridentatum* and *C. ladanifer*). Reducing plant biomass production after fire can be seen as a strategy to deal with water stress during the dry period (Martí-Roura et al., 2011). The response of plant species to fire affects the biomass carbon amount released to the atmosphere, the black carbon formation during incomplete combustion of vegetation, and carbon dynamics over time in the system. Aboveground biomass is usually a short-term carbon pool in fire-prone ecosystems (Rau et al., 2010).

The carbon storage capacity in belowground biomass, in the period between two (2 M) to thirty-six (36 M) months post-fire, ranged from 10.9 to 12.1 Mg C ha⁻¹ for *E. australis*, 2.1 to 2.3 Mg C ha⁻¹ for *C. ladanifer* and 1.3 to 1.5 Mg C ha⁻¹ for *C. tridentatum*. The root systems of shrub species were little affected, as the fire had a medium to low severity according to Persons et al. (2010) criteria. Results obtained in the present study showed that a period of thirty-six months was enough for the complete recovery of the shrub roots. In this sense, carbon storage in belowground biomass slightly changed over time. For *E. australis* the belowground biomass is about 2.5 times higher than the aboveground biomass, roots playing an important role in carbon stocks (representing about 70% of the carbon stored in the total biomass of this species). Due to the characteristics of this species, in some PNM areas, the stumps and coarse roots of *E. australis* are collected by the local villagers for use as domestic fuel (fireplaces and ovens) (IPB/ICN, 2007), a traditional practice that has implications for the ecosystem's carbon cycle. In contrast, carbon stored in roots of *C. tridentatum* and *C. ladanifer*, show much lower proportions, representing only 36% of the total carbon stocks in shrub biomass.

Litter layer (organic residues accumulated on the soil surface) is a transient pool essential to maintain or increase soil organic matter content in the top soil. Litter amount consumed by fire can have undesirable consequences on soil properties and nutrient availability (Shakesby et al., 2015; Fonseca et al., 2017; Pellegrini et al., 2018). Litter suffered losses, on the one hand, due to direct fire combustion and on the other due to erosive processes (wind and water erosion), being part of the organic residues transported out of the system or accumulated in small depressions, leaving the soil surface partially bare.

Litter layer showed a significant post-fire decrease in carbon storage, yet smoothed overtime as burnt area recovery progressed. As compared to the before fire status, that decrease amounted 2.1 Mg C ha⁻¹ (68% of the pre-fire condition, corresponding to an average decline rate of 12.5 Mg C ha⁻¹ year⁻¹), 2.0 Mg C ha⁻¹ (67%, or 4.1 Mg C ha⁻¹ year⁻¹) and 1.5 Mg C ha⁻¹ (48%, or 0.5 Mg C ha⁻¹ year⁻¹) for the periods two (2 M), six (6 M) and thirty-six (36 M) months after fire, respectively. Nave et al. (2011), using meta-analysis to synthesize the results of 57 studies carried out in temperate forests, showed that prescribed fire had a significant negative effect on litter carbon storage, the average carbon loss reaching 46% of that before fire. Litter combustion is probably the major process influencing litter carbon decline, but other causes can also play a relevant role in litter carbon dynamics (Certini, 2005; Forbes et al., 2006; Nave et al., 2011; Krishnaraj et al., 2016; Mayer et al., 2020). As stressed before, in the present study wind and water erosion seemingly had a large contribution to the loss of organic compounds and ash resulting from the prescribed fire episode and deposited on the soil surface. In the first three days post-fire, strong winds occurred in the experimental area and field observations and measurements showed that more than 80% of the ash had disappeared in that short period. In addition, erosion microplots were installed to measure soil loss and runoff. The largest soil loss events by water erosion (washing together mineral and organic material) were recorded following the first post-fire rainfalls. Soil losses were measured during fourteen months, becoming negligible approximately twelve months after fire. Detailed information about water erosion in experimental area can be found in Fonseca et al. (2017). Similar results are also reported by other authors (Murphy et al., 2006; Moody et al., 2013; Santín and Doerr, 2016). Conversely, fire modifies the composition and structure of the litter not consumed. Part of litter might be transformed into small particles (<2 mm) (Mayer et al., 2020) and in pyrogenic carbon, which is very resistant to microbial decomposition and chemical transformations, as it is the case of black carbon (Certini, 2005; Foereid et al., 2011; Nave et al., 2011; Krishnaraj et al., 2016). These new forms of carbon compounds may be transferred from the litter layer downwards by several processes such as biological activity, soil water percolation, alongside with the formation of organometallic complexes (Gonzalez-Perez et al., 2008; Foereid et al., 2011; Patrício et al., 2018), therefore

contributing to increase carbon storage in mineral soil.

Fire effects on soil carbon storage are particularly important, since the soil compartment is the major terrestrial carbon pool that mitigates increasing atmospheric carbon dioxide and climate change (e.g. Santín and Doerr, 2016; Fonseca et al., 2019). Overall, soil carbon storage may exhibit insignificant changes following fire, as the increase in soil temperature during fire can be limited to a very thin surface layer, even in situations where fire consumed most of the litter layer and aboveground biomass (Stoof et al., 2013). Changes in soil characteristics are often an indirect consequence of fire that gradually become salient during the post-fire period (Santín and Doerr, 2016; Fonseca et al., 2017). In this study, an average increase of 3.2 Mg C ha⁻¹ was observed two months (2 M) after fire. This increase may be due to ash deposition, decomposition of organic material that did not undergo complete combustion, such as remains of branches and stems (González-Pérez et al., 2004; Pardini et al., 2004; Santín and Doerr, 2016), and new carbon compounds, transported from the litter layer into the mineral soil layers underneath, essentially in charred forms (Johnson and Curtis, 2001; Certini, 2005; Knicker, 2011). From two (2 M) to six (6 M) months after fire there was a decline in soil carbon storage of around 3.8 Mg C ha⁻¹, which may be associated to a gradual re-establishment of biological activity. This time span corresponds to the late-spring to early-autumn period, when the prevailing higher temperatures increase ecosystem respiration and the decomposition rate of organic compounds. Even the recalcitrant carbon forms (e.g. black carbon) experience important losses through respiration, water percolation and surface runoff (Major et al., 2010). The carbon losses in mineral soil in this period roughly match to the gains occurred in the first two months following fire. After thirty-six months, the increase in carbon in the uppermost mineral soil layers (0–5 and 5–10 cm) can be related to the organic residues on the soil surface added by three growing seasons of the regenerating vegetation. The litter layer is an important pool in soil–plant relationships, with considerable contributions to nutrient cycling and to the biogeochemical carbon cycle (Zhang et al., 2016; Fonseca et al., 2019). In the 10–20 cm mineral soil layer, the significant increase in carbon storage is seemingly related to the also significant increase in soil exchangeable aluminum during the same period, confirmed by earlier studies in the same experimental site (Fonseca et al., 2017). This may have favored organometallic complexes formation and the consequent migration of organic compounds in depth. This tendency of organic matter movement in the soil profile in the PNM area has already been referenced in other studies (Fonseca et al., 2012; Patrício et al., 2018). Since soil organic matter is the key component for maintaining the sustainability of ecosystems, the present study results give an indication that the soil had the capacity to preserve its environmental and productive functions in the time period considered. Other studies also report no significant changes in soil carbon after prescribed fire (e.g. Rau et al., 2010; Chen et al., 2019).

In all, carbon losses to the atmosphere from shrub biomass and litter burned during prescribed fire are not balanced by vegetation regrowth and residues input, and thus those compartments are a net carbon source after thirty-six months. In spite of the large magnitude of carbon losses from the vegetation and litter compartments, these are transient negative changes in carbon storage decreasing over time as post-fire recovery progresses. However, this recovery is closely dependent on fire recurrence and its intensity and severity (Alcañiz et al. 2018; Pellegrini et al., 2018). Because the mineral soil is the largest carbon pool in the system, the increase observed in the present research in this pool was sufficient to compensate the losses recorded in those other carbon pools. After thirty-six months (95.1 Mg C ha⁻¹), the system stores an amount of carbon slightly higher than that observed in the pre-fire situation (94.3 Mg C ha⁻¹) and, globally, behaved as a net carbon sink (0.2 Mg C ha⁻¹ year⁻¹). Clay et al. (2010) suggest that the reduction of carbon losses is related, for a certain period, with primary productivity increase and ecosystems respiration decrease. According to the results of the present study, mountain Mediterranean shrubland areas play an important role

in carbon storage and more attention should be given to management techniques used to maintain or improve ecosystems functions and processes in these areas (Hanes et al., 2018; Fonseca et al., 2019).

5. Conclusions

Based on carbon dynamics over time, this work gives a contribution to the validation of prescribed fire use as a vegetation management tool and consequent wildfire prevention. It is of great importance to obtain knowledge about this topic because the Mediterranean region suffers heavily from wildfires and, consequently, carbon emissions awaken to other problems such as climate change and soil quality. Mineral soil is the most important carbon pool of the shrubland system and it has balanced carbon losses from vegetation biomass and litter layer. However, soil carbon is also dependent on inputs from vegetation cycles. If the use of prescribed fire, as a vegetation management technique, is applied strategically under adequate weather conditions (not too hot, dry or windy), its action results in minor environmental impacts. It is important to emphasize that the soil preserved its environmental and productive functions by increasing the organic matter content, which constitutes the key factor of the sustainability of the ecosystems.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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