



## Soil and organic carbon redistribution in a recently burned Mediterranean hillslope affected by water erosion processes

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### ARTICLE INFO

Handling Editor: Ingrid Kögel-Knabner

#### Keywords:

Forest fires  
Coupled hillslopes  
Connectivity  
Soil organic carbon sink  
Density fractionation

### ABSTRACT

Forest fires cause many changes in the physical, chemical and biological soil properties such as aggregation and soil organic carbon contents (SOC) as well as on soil hydrology and erosion processes. Most studies on post-fire soil erosion in Mediterranean environments have been plot-based and research at hillslope or broader scale is scarce. Understanding SOC nature, distribution and modifications, as produced by forest fires and erosion, has become crucial to model and define the role of soil erosion as source or sink of C, and to sustainably manage ecosystem services related to the soil resource. This research provides data about the loss and redistribution of soil and SOC in a Mediterranean forest hillslope burned with high severity, at the Natural Park of Sierra de Espadán, Spain. Soil was sampled in coupled hillslopes (ca. 0.25 ha) (BU: burned, CO: control) at bottom (depositional), middle (transport) and top positions (eroding) at two depths (0–2 cm, 2–5 cm), and under two environments (UC: under canopy soil, BS: bare soil). Sediments were collected after each erosive event along one year, and yields were calculated. Samples were analysed to assess aggregate stability (AS), size and density fractionations, SOC contents and stocks. The main hypothesis is that fire affects soil characteristics related to aggregation and SOC stabilization and, together with erosion processes, may modify SOC distribution within aggregates and the burned hillslope.

Soils were in general very stable, but some differences in the results of the methods used were observed. Significant differences were found for the environment (under canopy vs bare) and soil depth but not for slope position. SOC content was high both at BU and CO with no significant differences. In the BU hillslope, a homogenization of SOC contents was observed along the hillslope, while in the CO, a higher SOC content was measured in the depositional and transport sites than in the eroding one. Similar trends were observed for SOC stocks. Only four erosive rain events were registered in this study, which generated no sediment yields in CO hillslope. In the BU one, sediment yields were measured (0.05–0.58 Mg ha<sup>-1</sup>, total 0.925 Mg ha y<sup>-1</sup>), which mobilised OC amounts ranging between 0.005 and 0.04 MgC ha<sup>-1</sup>. When samples were fractionated, changes were observed in the mass distribution of soil and sediment aggregates by size and density, and in the OC content between density fractions of BU soils with regard to sediment and CO soils.

According to the results, effective post-fire management should be oriented to control and reduce the erosion of aggregates < 2 mm, which present the highest SOC content and are very prone to be transported off-site. This fraction should include all the partially burned biomass (free light material), which acts as a first mulching and contains high amounts of OC that should be kept within the burned hillslope to increase soil fertility, promote vegetation recovery and act as a C sink. In the BU hillslope, eroded free light material might be buried at the de-

*Abbreviations:* AS, aggregate stability; BU, burned soil; BS, bare soil; CO, control soil; FLF, free light fraction; HF, heavy fraction; OLF, occluded light fraction; SOC, soil organic carbon; SOM, soil organic matter; UC, under canopy soil

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<https://doi.org/10.1016/j.geoderma.2021.115539>

Received 30 April 2021; Received in revised form 21 September 2021; Accepted 11 October 2021

0016-7061/© 2021

positional site and if the conditions are favourable for its conservation, SOC accumulation would be promoted, which may have implications for its stabilization, and the role of soil erosion as a C sink.

## 1. Introduction

Organic matter (OM) in soil is the largest and most dynamic reservoir of carbon (C) on Earth and, thus, a key factor in global carbon cycling (Cerli et al., 2012; Kirkels et al., 2014). Organic C is stabilized in soils through key mechanisms as physical isolation (occlusion), chemical interaction of OM with reactive soil minerals, and preservation of recalcitrant compounds (Berhe, 2012; von Lützow et al., 2006; Wang et al., 2014). These mechanisms relate to soil aggregation by providing physical and chemical protection of soil organic carbon (SOC) against decomposition, so that OM becomes non-accessible for soil microorganisms and fauna (Doetterl et al., 2016; Six et al., 2002), and by aggregate formation through SOC sorption to pedogenic metal oxides, clay minerals or by co-precipitation with polyvalent cations (Mikutta et al., 2006; Schmidt et al., 2011). It has been found that aggregate formation appears to be closely linked with SOC storage and stability (Hu and Kuhn, 2016; Wiesmeier et al., 2019). On the other hand, soil erosion can promote breakdown of aggregates at the eroding landform positions leading to exposure of previously protected SOC (Doetterl et al., 2012; Lal, 2003), which potentially increases its mineralization rate (Berhe, 2012; Zhang et al., 2006) and releases soluble compounds (Badía et al., 2014; Caon et al., 2014). However, transported and deposited SOC can be protected from decomposition if efficiently buried in slow turn-over environments, leading to large C sinks in colluvial and alluvial sediments depending on the rate of burial, the time since burial, the nature and amount of mobilized C, and the post-depositional conditions (Doetterl et al., 2016; Kirkels et al., 2014).

There are a number of disturbances affecting soil aggregation and SOC as changes in land use (Martínez-Mena et al., 2012; Wiesmeier et al., 2019) or forest fires (Shakesby, 2011; Shakesby et al., 2015), among others. Forest fires are considered one of the main causes of soil degradation in the European Mediterranean region (Caon et al., 2014; Garcia-Ruiz et al., 2013; Shakesby, 2011), affecting their physical, chemical and biological soil properties (Badía et al., 2014; Bento-Gonçalves et al., 2012; Campo et al., 2008), as well as an increase in water runoff and sediment loss (Campo et al., 2006; Cawson et al., 2012). Particularly important are the effects of forest fires in the first few centimetres of topsoil, in relation to changes in SOC quantity and quality (González-Pérez et al., 2004), aggregate stability and erosion processes (Mataix-Solera et al., 2011; Shakesby and Doerr, 2006).

Soil erosion disturbs topsoils and, preferentially, removes SOC from upslope places, resulting in the redistribution and burial of SOC in depositional environments (Martínez-Mena et al., 2012; Wang et al., 2014). There is a number of factors that can influence the net effect of transport and deposition on eroded SOC (and consequently on its fate) as the rate and nature of soil erosion, the amount and nature of the eroded C, soil texture, soil aggregation, the transport distance, and terrain attributes such as slope gradient and surface roughness (Doetterl et al., 2016). Slow but long-range transport may lead to a higher degree of decomposition of mobilized SOC, whereas fast but short-range transport might lead to the burial of mobilized SOC, with a lower degree of decomposition, at the depositional site (Berhe and Kleber, 2013; Kuhn et al., 2009; Quinton et al., 2010). At this zone, the rate of decomposition of eroded SOC can be reduced by a combination of processes. These are biochemical (recalcitrance of organic constituents mainly those associated to pyrogenic carbon), physical (protection with burial, aggregation, and changing water, air, and temperature conditions) or chemical (mineral-OM associations). As stated by Berhe et al. (2007), in this scenario, eroded SOC remaining near the surface of foothills could contribute to enhanced decomposition, higher mineralisation rates (Quinton et al., 2010; Van Hemelryck et al., 2010; Van Hemelryck et

al., 2011), whereas the decomposition rate of buried C stocks is likely to be reduced. In general, initial sediment C-enriched will be buried by the sediments transported during subsequent rains (depending on their magnitude and on the frequency of episodic heavy rainfall events or floods) (Nadeu et al., 2012).

Redistribution of SOC can be affected by forest fires, which can increase or reduce SOC stocks depending on several factors (González-Pérez et al., 2004; Shakesby et al., 2015). The role of soil aggregation and stabilization in SOC dynamics during erosion and deposition has attracted scientific attention in recent decades (Nadeu et al., 2011; de Nijis and Cammeraat, 2020). However, the effects of such processes on SOC stabilization have not been studied in Mediterranean forest soils, where bare-soil areas act as sources of runoff water, sediment, seeds and nutrients (including C) that move downslope and are captured by, and concentrated in, vegetation patches (Urgeghe and Bautista, 2015). Even less is known about SOC stabilization when disturbances, such as forest fires, lead to a decrease in vegetation cover and litter, therefore increasing the connectivity of runoff-source areas and the transport capacity of the flow, reducing the hillslope storage potential for water and sediments (Cammeraat, 2004; Mayor et al., 2011), and exposing the already fire-affected SOC.

Understanding SOC nature and reactivity upon changes, as those produced by forest fires, has become crucial to model and define the role of soil as source or sink of C, and to sustainably manage ecosystem services related to the soil resource (Faria et al., 2014). This knowledge starts by the identification of organic fractions with distinct chemical and biological functions and turnover times, characteristics strongly related to the form and/or the type of interactions with minerals (e.g. Rasmussen et al., 2005). Therefore, isolating fractions of OM occurring either inside or outside of aggregates or being part of organic-mineral associations, all of them different in terms of biochemical properties and functional relevance, has become a major research topic during the last two decades (Berhe, 2012; Cerli et al., 2012; Doetterl et al., 2012; Grünwald et al., 2006; Wang et al., 2014).

Physical fractionation by density has been proven useful to separate SOM and to identify meaningful soil fractions, which can be related to different stability and stabilization processes (von Lützow et al., 2007; Wagai et al., 2009, 2015; Nadal-Romero et al., 2016; Yeasmin et al., 2017). In contrast to chemical extractions, density fractionation allows for isolation of unmodified C fractions, and is theoretically related to the spatial arrangement and interactions of organic compounds and minerals (Cerli et al., 2012). The method separates light and heavy fractions (Christensen, 1992), taking advantage of the difference in density between minerals and organic material, and often by additional physical dispersion (e.g. sonication), aiming an aggregate disruption and subsequent release of the OM occluded therein (Golchin et al., 1994).

The free light fraction (FLF), which floats in a solution of given density without additional dispersion, comprises undecomposed, easily accessible OM, i.e. large organic fragments that underwent little physical and/or chemical transformation. The occluded light fraction (OLF) comprises much finer organic material with similar composition as the FLF but slightly more altered, stabilized by aggregation, and protected within aggregates, i.e., OM floating in the solution after aggregate disruption. The remaining OM fraction in sediments represents the heavy fraction (HF), in which C is strongly bound to minerals and cannot be completely separated from them, i.e. organic-mineral associations (e.g. Cerli et al., 2012; Kaiser and Guggenberger, 2007). Both floating (light) fractions are supposed to comprise mainly plant-derived debris (leaves, branches, and roots) plus some animal residues, charcoal, seeds, pollen, and microorganisms (Golchin et al., 1994; Wagai et al., 2009). The main differences between the two light fractions should be their size

and location within the soil matrix. Together with size fractionation, this technique would be helpful to study how erosion, transport and deposition can induce transitions in SOC from one fraction to another (i.e. active to passive or vice versa) for example by aggregate disruption or deep burial, which may change C mineralisation rates (Wang et al., 2014).

The present study intends to obtain a better understanding of SOC accumulation and stabilization in a post-fire Spanish Mediterranean hillslope under soil erosion and deposition processes. The main hypothesis is that fire affects soil characteristics related to SOC stabilization and, together with erosion processes, can modify the SOC distribution within aggregates and in the burned hillslope. Accordingly, the main objectives are: (a) to determine differences caused by fire in soil aggregation, SOC content and stock, at hillslope scale; (b) to evaluate the influence of different variables as hillslope position, environment (under canopy and bare soils) and depth on the changes of soil aggregation and SOC distribution; (c) to use density and size fractionations in burned and unburned soils, as well as in sediments, to assess the effects of fire and erosion on SOC distribution within aggregates; and, (d) to estimate the role of fire and erosion in the possible changes of SOC accumulation and stabilization (i.e. SOC stock) in a Mediterranean hillslope, in order to shed some light in the discussion about the role of soil erosion as source or sink of C.

## 2. Material and methods

### 2.1. Study site and sampling

This work was carried out in the municipality of Azuébar, Natural Park of Sierra de Espadán, in the Province of Castellón, Spain (Fig. 1). Coupled hillslopes (burned: BU, and control: CO, ca. 0.25 ha each one) belonging to the coastal foothills of the Iberian Mountain System were selected (BU: 39°50'45.11"N, 0°22'20.52"W; CO: 39°51'08.7"N, 0°22'17.6"W). Both slopes are located on forested concave hillsides, with ENE aspect, 25 – 28° of slope and an altitude around 370 m a.s.l. (more information is available in Supplementary information, SI, Fig. S1).

The climate of the area is *meso*-Mediterranean (mean annual precipitation of 450 mm), with a maximum precipitation in autumn (70 mm in October) and a second, but less rainy period in spring (45 mm in May). According to a nearby climatological station (Sot de Ferrer, 4.5 km from the hillslopes), the dry period (June - September), in which usually forest fires occur, presents a maximum temperature of 28 °C.

The mean annual temperature is 16 °C with a minimum of 5 °C approx. (BSk climate according to the Köppen-Geiger classification). Vegetation cover is characterized by a Mediterranean shrubland. The most abundant species are *Rosmarinus officinalis*, *Ulex parviflorus*, *Quercus coccifera*, *Rhamnus lycioides*, *Stipa tenacissima*, *Globularia alypum*, *Thymus vulgaris* and some disperse exemplars of *Pinus halepensis*. Soils are classified as Endodolomitic Skeletic Chromic Leptic Luvisol type according to the IUSS Working Group WRB (2015), which show variable depths between 20 and 50 cm thick, and clay-loam texture, developed on Triassic dolomite (Fig. S2).

The wildfire on Azuébar municipality occurred the 28/08/2014 and burned 10.59 ha of forested area, according to the Emergency Coordination Centre of the Generalitat Valenciana (no previous fires have been registered at least since 1994). About 70% of the affected area was in high- or moderate - high burn severity classes according to field burn indicators (Bento-Gonçalves et al., 2012; Tanase et al., 2010; Úbeda et al., 2006). This municipality has been affected by several wildfires during last years, registering up to five intentioned fires between 2012 and 2014.

Coupled Mediterranean hillslopes were sampled (19/09/2014) following a connectivity design according to which the top of the hillslope is the eroding zone, the middle part is the transport site and the foot's slope is the depositional zone (Bracken et al., 2015). Soil samples were gathered under two environments (under canopy soil: UC; inter-plants or bare soil: BS), and at two depths (0–2 cm and 2–5 cm). Shovel samples from each location, environment and depth (BU: n = 44; CO: n = 24, Fig. S3) were transported in sealed plastic containers, and dried at room temperature. Sediments were collected from four sediment fences constructed at the foot of the burned slope (Figs. 1 and S4). Only in the first erosive event (29–11-2014), sediment was collected in three out of the four fences. Sediment was negligible in the control hillslope, and samples could not be collected. Sub-samples of soil and sediment were ground for chemical analysis.

### 2.2. Soil characterization

Soil aggregate stability (AS) was measured by: (1) the water drop test (WDT), based on the number of drops (up to 200 drops in every test) required for the destruction of individual dry aggregates > 5 mm (Low, 1954), and (2) by measuring the percentage of aggregates < 2 mm that remain intact after the soil sample is subjected to the action of mechanical forces by wet-sieving (Kemper and Rosenau, 1986). The C and N contents of bulk soils, aggregate size fractions and

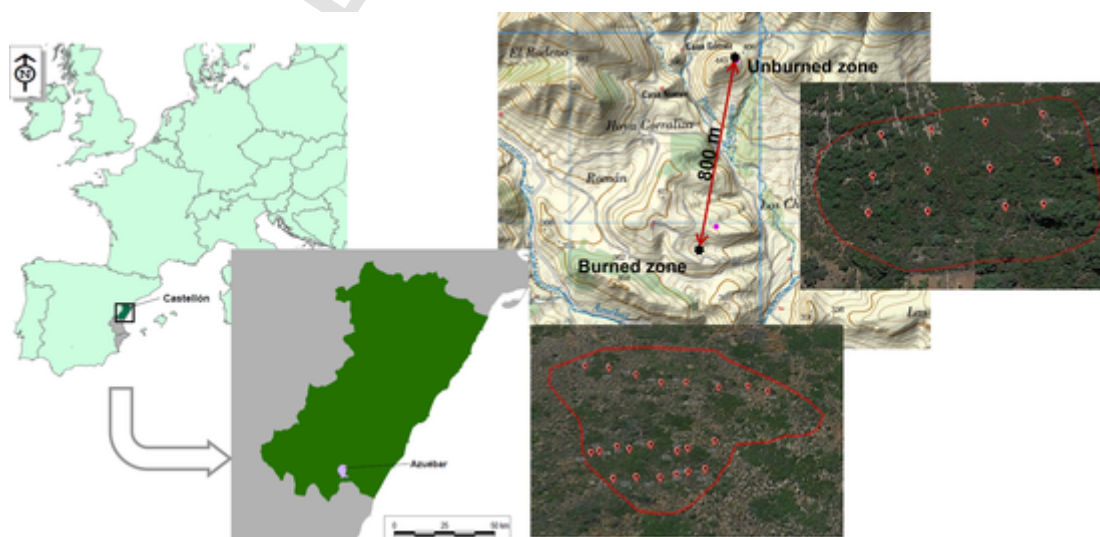


Fig. 1. Location of the study area and schemes of the burned and control hillslopes showing the distribution of samples. Coupled hillslopes in Azuébar, Natural Park of Sierra de Espadán, Spain, were used (Burned: 39°50'45.11"N, 0°22'20.52"W; Control: 39°51'08.7"N, 0°22'17.6"W).

density fractions were determined using a C and N analyser (Vario EL, Elementar GmbH, Hanau, Germany; Standard deviation (SD < 0.1% abs.). Calcium carbonate content ( $\text{CaCO}_3$ ) was determined using the Wesemael method ((van Wesemael, 1995)van Wesemael et al., 1955), from which the total inorganic carbon was also calculated (IC). The SOC was calculated subtracting the IC from the total carbon content (TC). The SOC and total N content (TN) contents were expressed in  $\text{g kg}^{-1}$  soil, while the SOC stock was expressed in  $\text{Mg ha}^{-1}$  (using the respective depth and bulk density). The C/N ratio was calculated using SOC and TN (SOC/TN). The bulk density (BD) was determined gravimetrically by drying a known volume of sample at  $105^\circ\text{C}$ .

### 2.3. Isolation of soil fractions

#### 2.3.1. Aggregate size fractionation

Samples were subjected to aggregate size fractionation according to the dry sieving method (Kemper and Rosenau, 1986). Briefly, 100 g of air-dried soil (and sediment) were manually passed through two consecutive sieves (5 and 2 mm mesh, respectively) to obtain three size fractions (<2, 2–5, >5 mm  $\varnothing$ ). Each fraction was weighed to determine its proportion and the SOC and TN content were determined in each fraction. Then, the relative distribution of SOC by aggregate size was calculated from the SOC content of each size fraction multiplied by its corresponding mass percentage.

#### 2.3.2. Soil density fractionation

Density fractionation was applied to bulk soil and sediment samples, by duplicate, following the methods of Golchin et al. (1994) and Cerli et al. (2012). Summarizing, 10 g air-dried material was weighed in a centrifugation tube and 50 ml of sodium polytungstate (NaPT, Sometu, Berlin, Germany) solution with a density of  $1.6\text{ g cm}^{-3}$  was added. This density was chosen according to Cerli et al. (2012), because it allows the separation of maximum amounts of almost pure organic material with minimum contamination by mineral material. After 1 h the suspension was centrifuged at 6800 g for 20 min at room temperature. The supernatant was vacuum filtered (GF/F filter, 0.7  $\mu\text{m}$ , Whatman GmbH, Dassel, Germany) and washed with deionized water until the conductivity of washed water was < 50  $\mu\text{S cm}^{-1}$ . The material collected from the filter represents the free light fraction, FLF <  $1.6\text{ g cm}^{-3}$ .

The soil remaining in the centrifugation tube was re-suspended into 50 ml of fresh NaPT solution, and sonicated at 300 J  $\text{mL}^{-1}$  (Sonopuls HD 3200 with VS70 probe, Bandelin electronic, Berlin, Germany). The ultrasound generator was calibrated according to Schmidt et al. (1999) in an ice-bath to keep the temperature <  $40^\circ\text{C}$ , and the sonication intensity was chosen after several pre-tests (determined by mass yields and SOC content of the obtained occluded fractions) on a subset of samples according to Cerli et al. (2012). The intensity used is a key factor to ensure the appropriate dispersion of pure OM enclosed within the aggregates, without contamination with the mineral part. Samples were

again centrifuged at 6800g for 20 min at room temperature, filtered and rinsed with deionized water as described above. This material is the occluded light fraction, OLF <  $1.6\text{ g cm}^{-3}$ .

The final residue in the tube was re-suspended in deionized water and centrifuged several times at 10,000g, in cycles of 1 h, until the conductivity of the supernatants was < 200  $\mu\text{S cm}^{-1}$ . The resulting solid material contains the heavy fraction (HF >  $1.6\text{ g cm}^{-3}$ ). All fractions were freeze-dried, homogenized and used for the determination of SOC and TN contents. Each fraction was weighed to determine its proportion and the SOC and TN content were determined in it. Then, the relative distribution of SOC by density fraction was calculated from the SOC content of each fraction multiplied by its corresponding mass percentage.

### 2.4. Statistical analyses

Normality of the data was tested using Shapiro-Wilk's test ( $n < 50$ ) and homogeneity of variances by means of Levene's test. For parametric analysis, analyses of variance with a General Linear Model and one-way ANOVA were carried out, and confirmed *a posteriori* with Tukey's or Student's t tests. Variables analysed were soil characteristics and SOC contents considering as fixed factors: treatment (T, with two levels, BU and CO), position (P, with three levels, eroding, transport and deposition), environment (E, with two levels, UC and BS) and depth (D, with two levels, 0–2 cm and 2–5 cm) having as random factor the sampling points. SOC contents obtained by size and density fractionations were also evaluated including the factor fraction (F, with size and density fractions as levels). Interactions between factors were also explored. For non-parametric analysis, differences were determined by the tests of Mann-Whitney U (M-W) or Kruskal-Wallis (K-W). In all the cases, results were considered to be statistically significant at  $p < 0.05$ . All statistical analyses were carried out using IBM SPSS Statistics 26®.

## 3. Results

### 3.1. Soil characteristics

Table 1 summarizes the soil properties determined in this study. In relation to the WDT, significant differences were only found between BU and CO, and between UC and BS (M-W,  $p < 0.05$ ). In CO soils, generally, >200 drops were needed to break aggregates, and no differences could be observed for position, environment or depth (Tables 1 and 2). In BU soils, only UC (200 drops) and BS (120 drops) were significantly different (M-W,  $p < 0.05$ , Fig. S5). Results from the wet-sieving test also showed significant differences between BU and CO, and between UC and BS (Student's t,  $p < 0.05$ ). Aggregates of BU soils were more stable ( $74.05 \pm 5.61\%$ ) than those of CO soils ( $57.51 \pm 16.45\%$ ; Table 1). However, differences in the AS of CO soils

**Table 1**

Mean values of soil characteristics organized by treatment, position, environment and depth. WDT: water drop test (number of drops), AS: aggregate stability by wet sieving (%), SOC: soil organic carbon ( $\text{gC kg}^{-1}$ ), OC stock: soil organic carbon stock ( $\text{Mg ha}^{-1}$ ), TN: total nitrogen ( $\text{gN kg}^{-1}$ ), SOC/TN: ratio soil organic carbon and total nitrogen. Different letters indicate significant differences at  $p < 0.05$  for each variable.

	Treatment		Position			Environment		Depth	
	Burned	Control	Eroding	Transport	Deposition	Under Canopy	Bare Soil	Topsoil	Subsoil
WDT (drops)	156 $\pm$ 68 <sup>b</sup>	198 $\pm$ 10 <sup>a</sup>	164 $\pm$ 65 <sup>a</sup>	182 $\pm$ 51 <sup>a</sup>	167 $\pm$ 60 <sup>a</sup>	195 $\pm$ 27 <sup>a</sup>	147 $\pm$ 71 <sup>b</sup>	166 $\pm$ 64 <sup>a</sup>	176 $\pm$ 53 <sup>a</sup>
AS (%)	74.05 $\pm$ 5.61 <sup>a</sup>	57.51 $\pm$ 16.45 <sup>b</sup>	67.63 $\pm$ 14.72 <sup>a</sup>	68.61 $\pm$ 14.11 <sup>a</sup>	68.43 $\pm$ 10.87 <sup>a</sup>	72.05 $\pm$ 6.91 <sup>a</sup>	64.38 $\pm$ 16.74 <sup>b</sup>	66.33 $\pm$ 14.60 <sup>a</sup>	70.09 $\pm$ 11.74 <sup>a</sup>
SOC ( $\text{gC kg}^{-1}$ )	56.88 $\pm$ 18.64 <sup>a</sup>	55.00 $\pm$ 25.64 <sup>a</sup>	49.55 $\pm$ 14.63 <sup>b</sup>	60.92 $\pm$ 26.34 <sup>a</sup>	58.56 $\pm$ 19.91 <sup>a</sup>	65.17 $\pm$ 21.56 <sup>a</sup>	47.26 $\pm$ 16.81 <sup>b</sup>	66.01 $\pm$ 23.42 <sup>a</sup>	46.42 $\pm$ 12.89 <sup>b</sup>
OC stock ( $\text{Mg ha}^{-1}$ )	28.94 $\pm$ 7.85 <sup>a</sup>	28.28 $\pm$ 12.17 <sup>a</sup>	24.21 $\pm$ 5.94 <sup>b</sup>	31.93 $\pm$ 11.72 <sup>a</sup>	30.24 $\pm$ 8.29 <sup>a</sup>	32.65 $\pm$ 9.19 <sup>a</sup>	24.76 $\pm$ 8.19 <sup>b</sup>	27.95 $\pm$ 10.43 <sup>a</sup>	29.46 $\pm$ 8.59 <sup>a</sup>
TN ( $\text{gN kg}^{-1}$ )	4.10 $\pm$ 1.02 <sup>a</sup>	4.46 $\pm$ 1.58 <sup>a</sup>	3.93 $\pm$ 0.98 <sup>b</sup>	4.35 $\pm$ 1.42 <sup>a</sup>	4.45 $\pm$ 1.30 <sup>a</sup>	4.73 $\pm$ 1.26 <sup>a</sup>	3.73 $\pm$ 1.02 <sup>b</sup>	4.83 $\pm$ 1.29 <sup>a</sup>	3.63 $\pm$ 0.87 <sup>b</sup>
SOC/TN	13.70 $\pm$ 1.74 <sup>a</sup>	11.93 $\pm$ 1.46 <sup>b</sup>	12.90 $\pm$ 0.69 <sup>a</sup>	14.32 $\pm$ 2.28 <sup>a</sup>	13.92 $\pm$ 1.60 <sup>a</sup>	14.03 $\pm$ 2.10 <sup>a</sup>	13.36 $\pm$ 1.24 <sup>b</sup>	14.00 $\pm$ 2.06 <sup>a</sup>	13.39 $\pm$ 1.33 <sup>a</sup>

**Table 2**

Significant differences grouped according to the different factors and their interactions. T: treatment, D: depth, E: environment, P: position, WDT: water drop test (number of drops), AS: aggregate stability by wet sieving (%), SOC: soil organic carbon (gC kg<sup>-1</sup>), Stock: soil organic carbon stock (Mg ha<sup>-1</sup>), TN: total nitrogen (gN kg<sup>-1</sup>), SOC/TN: ratio soil organic carbon and total nitrogen, ns: not significant, \*: significant at p < 0.05.

	T	D	E	P	T*D	T*E	T*P	D*E	D*P	E*P	T*D*E	T*D*P	T*E*P	D*E*P	T*D*E*P
WDT	*	ns	*	ns											
AS (%)	*	ns	*	ns	ns	*	*	ns	ns	ns	*	ns	ns	ns	ns
SOC (gC kg <sup>-1</sup> )	ns	*	*	*	ns	*	*	*	ns	ns	ns	ns	ns	ns	ns
Stock (Mg ha <sup>-1</sup> )	ns	ns	*	*	ns	*	*	ns	ns	*	ns	ns	ns	ns	ns
TN (gN kg <sup>-1</sup> )	ns	*	*	*	ns	*	*	*	ns	ns	ns	ns	ns	ns	ns
SOC/TN	*	ns	*	ns	ns	*	ns	ns	ns	ns	ns	ns	ns	ns	ns

depended on the environment (M–W, p < 0.05, Fig. S6), while in BU soils on the depth (M–W, p < 0.05, Fig. S6).

SOC content was relatively high in both BU and CO without significant differences between their means (Table 1), only the eroding position showed differences between treatments (Fig. S7). In BU soil, no significant differences were observed among eroding (54.08 ± 13.10 gC kg<sup>-1</sup>), transport (61.54 ± 23.53 gC kg<sup>-1</sup>), and depositional zones (54.39 ± 17.83 gC kg<sup>-1</sup>); while in CO, higher SOC contents were measured in the depositional (64.82 ± 22.39 gC kg<sup>-1</sup>) and transport (59.68 ± 33.01 gC kg<sup>-1</sup>) sites of the hillslope than in the eroding one (40.50 ± 13.89 gC kg<sup>-1</sup>). In this CO soils, SOC content was higher in UC than in BS (Student's t, p < 0.05), whereas in BU soil, it was higher in topsoil than in subsoil (Student's t, p < 0.05) (Fig. S7).

Similar results were obtained when analysing the SOC stock (Table 1). The values in CO soils were higher in transport (31.67 ± 15.82 MgC ha<sup>-1</sup>) and depositional sites (33.54 ± 8.96 MgC ha<sup>-1</sup>) than in the eroding one (19.65 ± 5.06 MgC ha<sup>-1</sup>) (ANOVA, p < 0.05), whereas in BU soils it was similar between positions (eroding: 26.49 ± 5.04 MgC ha<sup>-1</sup>, transport 32.05 ± 9.68 MgC ha<sup>-1</sup>, deposition 28.05 ± 7.39 MgC ha<sup>-1</sup>). SOC stock in CO soils was also higher in UC than in BS (Student's t, p < 0.05), and no differences were found between depths in any treatment (Table 2, Fig. S8).

TN content presented the same pattern than SOC with no significant differences between BU and CO soils (Table 1). Regarding the position effect, contents were similar in BU soils (eroding: 4.18 ± 0.91 gN kg<sup>-1</sup>, transport 4.22 ± 1.18 gN kg<sup>-1</sup>, and depositional 3.86 ± 0.97 gN kg<sup>-1</sup>); while in CO soils, the eroding site (3.43 ± 0.98 gN kg<sup>-1</sup>) was smaller than transport (4.62 ± 1.87 gN kg<sup>-1</sup>) and deposition (5.33 ± 1.28 gN kg<sup>-1</sup>) sites (ANOVA, p < 0.05, Fig. S9). In CO soils, lower content was found in BS than in UC, whereas in BU, it was higher in 0–2 depth than in 2–5 (Student's t, p < 0.05, Fig. S9). When the SOC/TN ratio was analysed different trends were observed. The fire affected this ratio, which was significantly higher in BU (13.70 ± 1.74) than in CO soils (11.93 ± 1.46) (M–W, p < 0.05, Table 1). Only in CO, the SOC/TN was higher in UC than in BS (M–W, p < 0.05), and no significant differences were found based on depth for any treatment (Table 2, Fig. S10).

### 3.2. Soil and organic carbon erosion

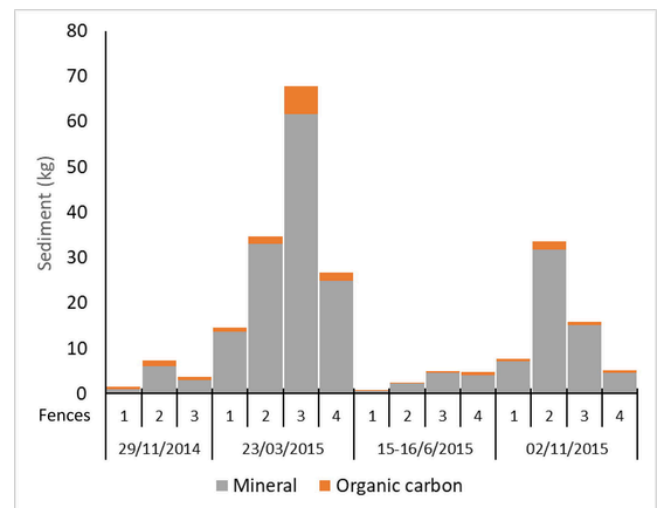
In the first year post-fire, only four rain events were erosive. Based on the data of the closest pluviometric station (Sot de Ferrer: 4.5 km from the study area), these were registered in 29/11/2014, 23/3/2015, 15–16/6/2015 and 2/11/2015 with total volumes of 64.2, 103.2, 41.2 and 131.6 L m<sup>-2</sup>, respectively (Table 3). These events did not produce any sediment in CO hillslope. Conversely, in the BU, sediment collected after each rainfall was 12.66, 143.63, 12.63 and 62.21 kg corresponding to sediment yields of 0.05, 0.58, 0.05 and 0.24 Mg ha<sup>-1</sup>, respectively. Considering the whole year, total yield was 0.925 Mg ha<sup>-1</sup> y<sup>-1</sup>.

Very high OC values were measured in the sediments of the first erosive event (181.71–366.10 gC kg<sup>-1</sup>), which were reduced in the following rainfalls (Table 3, Fig. 2). The lowest values were observed after the rain of 23/3/2015, the one that mobilised the highest amount of sedi-

**Table 3**

Sediment collected in the different fences after the four erosive events registered in the pluviometric station of Sot de Ferrer. Organic carbon content (OC) of the sediment, enrichment ratios (related to burned SOC content), sediment and OC yield in each event.

	Fence	Sediment (kg)	OC (gC kg <sup>-1</sup> )	Enrichment ratio	Sediment yield (Mg ha <sup>-1</sup> )	OC yield (MgC ha <sup>-1</sup> )
29/11/2014 (64.2 L m <sup>-2</sup> )	1	1.58	366.10	6.44	0.05	0.01
	2	7.36	181.71	3.19		
	3	3.72	183.98	3.23		
	4	–	–	–		
23/3/2015 (103.2 L m <sup>-2</sup> )	1	14.49	59.87	1.05	0.58	0.04
	2	34.73	49.13	0.86		
	3	67.73	89.75	1.58		
	4	26.68	64.42	1.13		
15-16/6/2015 (41.2 L m <sup>-2</sup> )	1	0.61	121.43	2.13	0.05	0.005
	2	2.40	81.52	1.43		
	3	4.91	75.49	1.33		
	4	4.71	126.63	2.23		
2-11-2015 (131.6 L m <sup>-2</sup> )	1	7.65	76.09	1.34	0.24	0.01
	2	33.57	52.79	0.93		
	3	15.80	48.80	0.86		
	4	5.19	99.81	1.75		



**Fig. 2.** Soil and SOC losses (kg) collected in the different fences after the four erosive events registered in the study area.

ment. Nevertheless, in general, the trend is decreasing during the year studied. Enrichment ratios, calculated as the ratio between OC content in sediments and in BU soil, also confirmed these trends (Table 3). In total, OC losses account for yields of 0.01, 0.04, 0.005 and 0.01 MgC ha<sup>-1</sup>, for each rainfall event registered. Accordingly, it is possible to state that in the fire of Azuébar a total of 0.925 MgC ha<sup>-1</sup> y<sup>-1</sup> were lost considering only the hillslope of this research.

### 3.3. Isolation of soil and sediment fractions

On one hand, significant differences were found for the mass of aggregates based on the size fractions and some interactions between size and other factors as treatment and environment (double interactions), and between them and position (quadruple interaction) (Table 4). For SOC content, differences were also found based on aggregate size and its interaction with the other factors (treatment, environment and depth) in double and triple combinations (Table 4). In CO soils, the percentage of aggregates < 2 mm ( $45.08 \pm 5.79\%$ ) was higher than those 2–5 mm ( $37.33 \pm 3.02\%$ ), which is also higher than those > 5 mm ( $17.58 \pm 4.11\%$ ) (K-W,  $p < 0.05$ ). SOC content was also higher in the fraction < 2 mm ( $6.83 \pm 3.70 \text{ gC kg}^{-1}$ ) than in the others, which were similar, 2–5 mm ( $4.82 \pm 2.33 \text{ gC kg}^{-1}$ ) and > 5 mm ( $4.61 \pm 1.75 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ ). In BU soils, the percentage of aggregates in the fraction < 2 mm ( $36.42 \pm 4.89\%$ ) was similar to the fraction 2–5 mm ( $35.57 \pm 3.19\%$ ), and both were higher than in the > 5 mm ( $28.00 \pm 6.78\%$ ) (K-W,  $p < 0.05$ ). However, SOC content of aggregates < 2 mm ( $8.47 \pm 3.47 \text{ gC kg}^{-1}$ ) was higher than that of aggregates 2–5 mm ( $4.91 \pm 1.25 \text{ gC kg}^{-1}$ ) and > 5 mm ( $4.51 \pm 1.21 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ ).

On the other hand, significant differences were observed for the mass of density fractions and their double interaction with the other factors, as well as in most of the triple interactions and in some of the quadruple ones (Table 4). For SOC content, differences were also found based on the density fractions and their interaction with treatment, position and environment factors (double interactions). Only one triple interaction was found to be significant (Table 4). In CO soils, the mass percentage in FLF ( $3.21 \pm 2.26\%$ ) was higher than in OLF ( $1.17 \pm 1.61\%$ ), and both were smaller than in HF ( $95.63 \pm 3.61\%$ ) (K-W,  $p < 0.05$ ). SOC content was similar in FLF ( $28.42 \pm 5.53 \text{ gC kg}^{-1}$ ) and in OLF ( $29.99 \pm 8.91 \text{ gC kg}^{-1}$ ), being both higher than in HF ( $4.35 \pm 1.30 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ ). In BU soils, the mass percentage in FLF ( $1.74 \pm 1.55\%$ ) was similar to OLF ( $1.79 \pm 0.96\%$ ) and both were smaller than in HF ( $96.48 \pm 2.18\%$ ) (K-W,  $p < 0.05$ ). But, SOC content of OLF ( $38.01 \pm 2.41 \text{ gC kg}^{-1}$ ) was higher than FLF

( $34.76 \pm 3.78 \text{ gC kg}^{-1}$ ), and both were higher than HF ( $3.26 \pm 0.79 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ ).

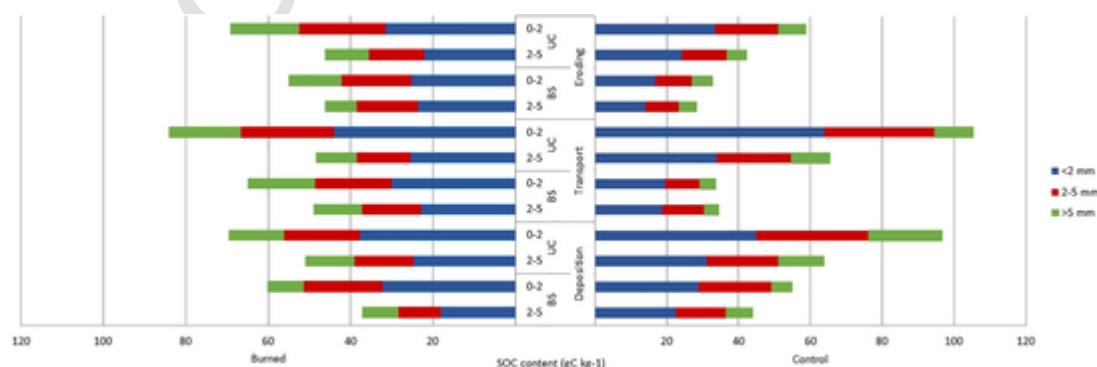
With these data (mass percentage and SOC content per size and density fractions), the relative distribution of SOC by fractions was calculated. In the size fractionation, significant differences were only detected for the SOC content weighed by fractions and their double interactions with other factors except position. None of the other interactions was significant (Table 4). In CO soils, the weighed SOC content of aggregates < 2 mm ( $29.32 \pm 14.96 \text{ gC kg}^{-1}$ ) was higher than those of 2–5 mm ( $17.36 \pm 7.80 \text{ gC kg}^{-1}$ ), which was also higher than those of > 5 mm ( $8.32 \pm 4.93 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ , Fig. 3). In BU soils, the weighed SOC content of aggregates < 2 mm ( $28.23 \pm 10.32 \text{ gC kg}^{-1}$ ) was higher than those of 2–5 mm ( $16.51 \pm 4.87 \text{ gC kg}^{-1}$ ), which was also higher than those of > 5 mm ( $12.14 \pm 4.97 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ , Fig. 3). In the density fractionation, significant differences were only assessed for the SOC content weighed by fractions and their double interaction with the other factors except depth. Triple and quadruple interactions were significant except when the depth factor was included (Table 4). In CO soils, the weighed SOC content of FLF ( $9.29 \pm 7.80 \text{ gC kg}^{-1}$ ) was higher than of OLF ( $3.56 \pm 5.44 \text{ gC kg}^{-1}$ ), and both were lower than of HF ( $42.15 \pm 16.84 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ , Fig. 4). In BU soils, the weighed SOC content of FLF ( $7.83 \pm 9.26 \text{ gC kg}^{-1}$ ) was similar to OLF ( $8.77 \pm 5.30 \text{ gC kg}^{-1}$ ), and both were lower than of HF ( $40.28 \pm 11.48 \text{ gC kg}^{-1}$ ) (K-W,  $p < 0.05$ , Fig. 4). Analysing significant differences between hillslope positions for each treatment and fraction, in CO soils, the weighed SOC content of HF was lower in eroding than in deposition site contrary to BU soils, in which these contents were similar.

In relation to sediments, significant differences were found for the mass of aggregates based on the size fractions but on the different sediment fences and dates. OC content only showed differences between dates, but when weighed by fractions presented the same results as the mass of aggregates. Double interactions were not significant for any combination of factors (Table 5). In sediment samples, the percentage of aggregates < 2 mm ( $68.73 \pm 9.06\%$ ) was higher than 2–5 mm ( $31.27 \pm 9.06\%$ ) (Student's  $t$ ,  $p < 0.05$ ), but the OC content was similar in both fractions (< 2 mm:  $13.42 \pm 7.15 \text{ gC kg}^{-1}$ , 2–5 mm:

**Table 4**

Significant differences for both soil fractionations grouped according to the different factors and their interactions. F: fraction, T: treatment, D: depth, E: environment, P: position, W (%) mass percentage of the different fractions, SOC ( $\text{gC kg}^{-1}$ ): soil organic carbon per fraction, SOC weighed ( $\text{gC kg}^{-1}$ ): soil organic carbon weighed per fraction, ns: not significant, \*: significant at  $p < 0.05$ .

		F	F*T	F*D	F*E	F*P	F*T*D	F*T*E	F*T*P	F*D*E	F*D*P	F*E*P	F*T*D*E	F*T*D*P	F*T*E*P	F*D*E*P	F*T*D*E*P
Size	W (%)	*	*	ns	*	ns	ns	ns	ns	ns	ns	ns	ns	ns	*	ns	ns
	SOC ( $\text{gC kg}^{-1}$ )	*	*	*	*	ns	ns	ns	ns	*	ns	ns	ns	ns	ns	ns	ns
	SOC weighed( $\text{gC kg}^{-1}$ )	*	*	*	*	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Density	W (%)	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
	SOC ( $\text{gC kg}^{-1}$ )	*	*	ns	*	*	ns	ns	*	ns	ns	ns	ns	ns	ns	ns	ns
	SOC weighed( $\text{gC kg}^{-1}$ )	*	*	ns	*	*	ns	*	*	ns	ns	ns	ns	ns	*	ns	ns



**Fig. 3.** SOC content ( $\text{gC kg}^{-1}$ ) in the different aggregate size fractions of burned and control soils (UC: Under canopy soil, BS: Bare soil).

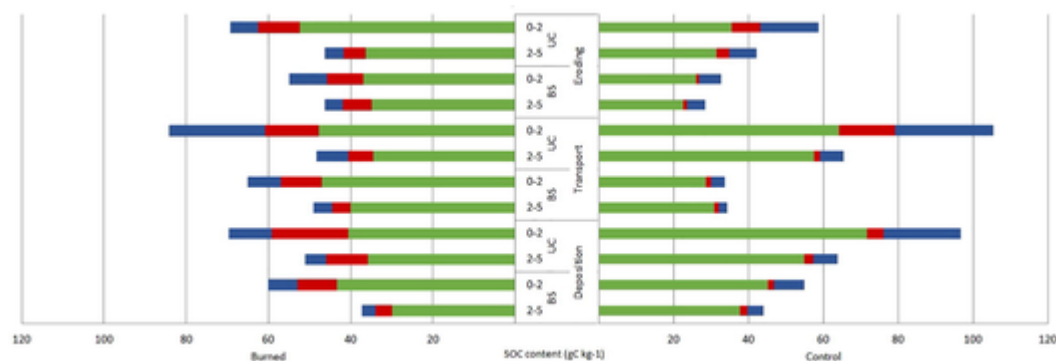


Fig. 4. SOC content (gC kg<sup>-1</sup>) in the different density fractions of burned and control soils (FLF: free light fraction, OLF: occluded light fraction, HF: heavy fraction, UC: Under canopy soil, BS: Bare soil).

Table 5

Significant differences for both sediment fractionations grouped according to the different factors and their interactions. F: fraction, Fe: sediment fence, Da: date, W (%) mass percentage of the different fractions, OC (gC kg<sup>-1</sup>): organic carbon per fraction, OC pond (gC kg<sup>-1</sup>): organic carbon weighed in each fraction, ns: not significant, \*: significant at p < 0.05.

		F	Fe	Da	F*Fe	F*Da	Fe*Da
Size	W (%)	*	ns	ns	ns	ns	ns
	SOC (gC kg <sup>-1</sup> )	ns	ns	*	ns	ns	ns
Density	SOC pond (gC kg <sup>-1</sup> )	*	ns	ns	ns	ns	ns
	W (%)	*	ns	ns	ns	ns	ns
	SOC (gC kg <sup>-1</sup> )	*	ns	ns	ns	ns	ns
		*	ns	*	ns	ns	ns

9.68 ± 4.73 gC kg<sup>-1</sup>, M-W, p > 0.05). When weighed, the OC content of aggregates < 2 mm (87.51 + 73.01 gC kg<sup>-1</sup>) was higher than those of 2-5 mm (24.33 ± 12.38 gC kg<sup>-1</sup>) (M-W, p < 0.05, Fig. 5). Aggregates > 5 mm were not found in any of the sediment samples collected. Significant differences were only based on the density fractions for their mass and OC content. Other factors as date and sediment fence did not present any significant influence. OC content weighed by fractions showed significant differences based on fractions and date. For the density fractionation, double interactions were also not significant for any combination of factors (Table 5). The mass percentage in FLF (12.33 ± 9.18%) was similar than in OLF (6.40 ± 4.12%), and both

were smaller than in HF (81.53 ± 12.49%) (K-W, p < 0.05). OC content was similar in FLF (36.33 ± 4.27 gC kg<sup>-1</sup>) and in OLF (36.44 ± 7.18 gC kg<sup>-1</sup>), being both higher than in HF (6.15 ± 2.16 gC kg<sup>-1</sup>) (K-W, p < 0.05). The weighed OC content of FLF (45.85 ± 47.55 gC kg<sup>-1</sup>) was similar to HF (42.98 ± 13.33 gC kg<sup>-1</sup>), and both were higher than for OLF (23.00 ± 24.63 gC kg<sup>-1</sup>) (K-W, p < 0.05, Fig. 6).

In relation to the recoveries of the size fractionations, averaged recovery of total soil mass and SOC was 95.12 ± 2.91% and 100.01 ± 18.44% (n = 48), respectively, for CO soils. In BU soils, average recoveries were 95.67 ± 2.34% and 107.28 ± 13.71% (n = 88) for the total soil mass and SOC, respectively. After density fractionation, the average recovery of CO soil mass was > 94%, ranging from 94.09 to 98.35%, which indicates little loss of material, averaging 2.18 ± 2.08% (n = 48). The recovery of OC was > 90.17%. Averaged recoveries of BU soils were 91.67 ± 2.13% and 89.23 ± 10.71% (n = 88) for the total soil mass and SOC, respectively. For the sediment, the average mass recoveries were in the range 97.63 to 99.32%, and from 103.29% to 124.60% for the OC (n = 30). In general, it was not possible to determine from which fraction was derived the C lost during the fractionation steps. Consequently, we assumed that recovered mass is equivalent to total mass when calculating the distribution of recovered C in the different fractions obtained by size and density.

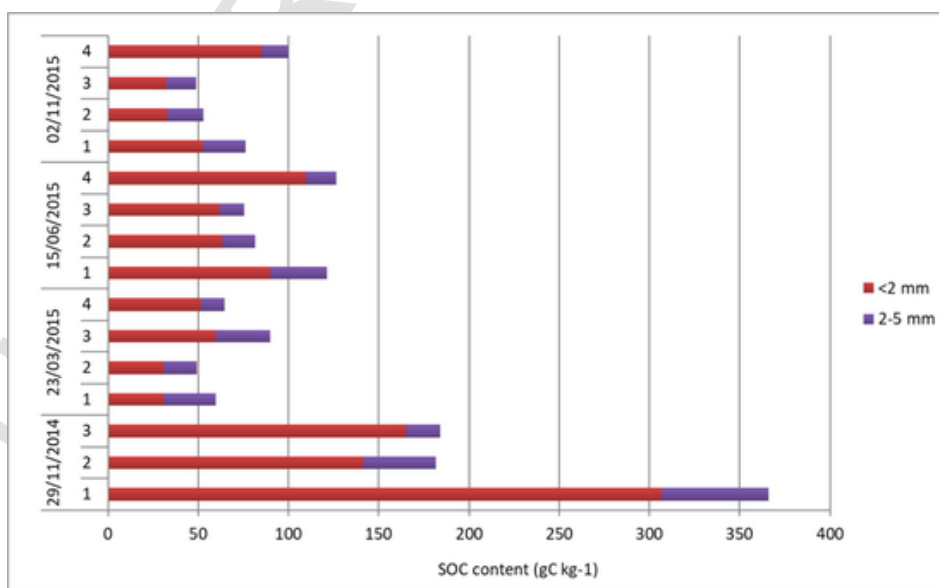


Fig. 5. OC content (gC kg<sup>-1</sup>) in the different size fractions of the sediment collected from the BU hillslope in every event and fence.

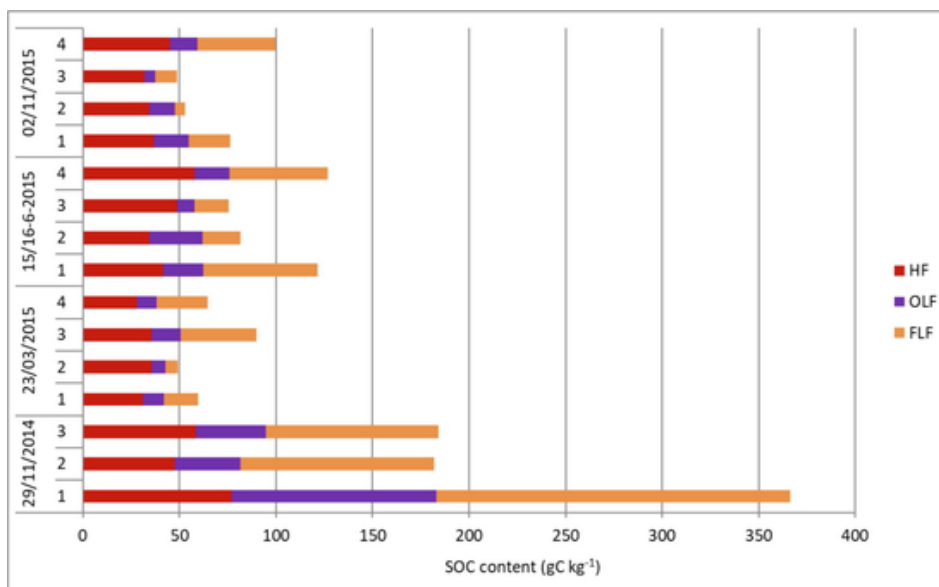


Fig. 6. OC content ( $\text{gC kg}^{-1}$ ) in the different density fractions of the sediment collected from the BU hillslope in every event and fence (FLF: free light fraction, OLF: occluded light fraction, HF: heavy fraction).

## 4. Discussion

### 4.1. Soil characteristics

Soils were in general very stable, but opposite results were observed (Table 1), which can be attributed to the slower wetting rate of the WDT as compared to the wet sieving. Imeson and Vis (1984) stated that WDT test is very suitable for soils of low aggregate stability, and the method would not be appropriate for the stable aggregates of this study, and other Mediterranean soils burned at high and moderate severity fires (Campo et al., 2008).

The response to AS to forest fires is complex since it depends on how fire has affected other related properties such as OM or calcium carbonate content, water repellency, etc. (Campo et al., 2008; Mataix-Solera et al., 2011). Stability of aggregates  $>5$  mm decreased after fire, while stability of those  $<2$  mm increased, with regard to CO soils. (Cerdá, 1993)Cerdá (1993) and Úbeda et al. (1990) also found a decrease in the stability of large aggregates, measured with the WDT, after fires in Pedralba (Valencia, E Spain) and in Catalonia (NE, Spain), respectively. Mataix-Solera et al. (2011) stated that stability of aggregates  $<2$  mm affected by fires can increase because of recrystallization of some minerals such as Fe and Al (hydr)oxides. Jimenez-Pinilla et al. (2016) confirmed that in Mediterranean soils heated at  $500\text{--}700$  °C, Fe (hydr)oxide minerals dehydroxylated to form maghemite, which may lead to a stabilization of aggregate structure and, hence, an increase in the AS. Arcenegui et al. (2008) and Terefe et al. (2008) explained such increases by the possible accumulation of hydrophobic substances on soil particles. It is also possible that fire's impact could have produced a homogenisation of aggregate stability between environments (in CO soils  $UC > BS$  but in BU soils  $UC = BS$ ), since the role of the vegetation cover for keeping a good soil structure and for protecting the soil against rain erosivity is lost (Campo et al., 2008). In general, it is possible to state that position on the hillslope did not have any effect on the stability of aggregates of different sizes affected by fire or not.

As explained for AS, researchers often state contradictory effects of fire on SOC content, related to variables as soil composition, plant community, fire severity, fire history, etc. (Certini, 2005; Terefe et al., 2008). In this study, significant differences were not found between SOC contents of CO and BU soils (Table 1). However, analysing the different hillslope positions, no significant differences were observed in BU soils, but significantly higher SOC contents were measured in the

depositional and transport sites than in the eroding one of CO soils. Similar results were reported by Berhe et al. (2012), who also found a higher content of C in depositional than in the eroding landform positions. These authors also concluded that depositional positions were able to store more C than the eroding ones. As stated by Kirkels et al. (2014) on a landscape scale, typically, the lowest levels of SOC can be found at high/mid-slope positions (eroding sites) and the highest at low positions (depositional sites). Accordingly, we state that fire caused a homogenization of this content between burned hillslope sites. Besides this, the higher SOC content observed in the eroding site of BU than of CO confirmed the addition of partially burned organic material to BU soils (Fig. S7).

As stated by Badía et al. (2014), many studies are flawed because soil samples analysed are too thick, which results in a “dilution” of the true effects of fires. This is not the case in this research, since topsoil (0–2 cm) was analysed separately from the subsoil (2–5 cm). In fact, in BU, the SOC content was higher in topsoil than in subsoil (Fig. S7). This together with the lack of difference between SOC content of BU and CO can be related to other factors such as the incorporation of ashes and charred litter into the soil (Badía et al., 2014; Certini et al., 2011).

In the studied hillslopes, the SOC stock presented similar trends to the SOC content, with higher values at transport and depositional sites than at the eroding one of CO soils, but similar values between the positions of BU soils. This can be explained because at eroding positions SOC stocks are reduced by erosion, with their rate and extent related to slope gradient and convexity (Doetterl et al., 2016). Physical removal of SOC causes direct depletion of the active C pool that can be exacerbated by increased on-site mineralisation, due to enlarged exposure, and by decreased productivity related to lower C return to soil (Lal, 2003). In depositional areas, the C budget is affected by competing processes such as changed (lower) mineralisation, biomass production and deep burial of SOC, which can be related to its accumulation and consequently, to the increase of the SOC stock, although relatively high decomposition of SOC in buried sediments can also happen (Kirkels et al., 2014). Consequently, it is still not clear what happens during the transport and deposition of eroded SOC, which has created a controversy on the role of soil erosion in the C cycle (de Nijs and Cammeraat, 2020). According to Doetterl et al. (2016), recent studies have highlighted two apparently contradictory processes taking place during transport and deposition. On the one hand, there can be increased mineralization and/or chemical transformation, because of both microbially facilitated



oxidative degradation and photo-degradation of C due to the detachment and breakdown of aggregates. This results in large quantities of labile SOC at depositional sites, where microbial and biological activity could be promoted, enlarging mineralisation and the net C flux to the atmosphere (Lal, 2003). On the other hand, selective transport and deposition of C-enriched sediments could favour aggregate formation by concentration of OM, clay and nutrients at depositional sites, and then enhancing physical and chemical protection of SOC (Berhe et al., 2012, 2013). In the case of our hillslope, the second process might be responsible of the accumulation of SOC in the transport and depositional positions of the CO hillslope (Fig. S7).

TN was also similar in BU and CO soils (Table 1) related to a possible incorporation of partially burned vegetation and ashes (rich in mineral N) to BU soils. Marañón-Jiménez et al., (2013) found in a burned Mediterranean pine forest that partially charred wood is a considerable pool of nutrients and nitrogen. However, Wan et al. (2001), stated that fire tends to decrease the soil TN content in the A horizon (0–2.6 cm and 0–5 cm) under different vegetation covers. Castro et al. (2006) reported similar results in the A horizon (0–15 cm) of a forest soil under *Pinus pinaster*, and Rau et al. (2010) in the 0–8 cm soil depth of a Pinyon Juniper Woodland burned under prescribed fires. Regarding the position effect, fire seems to have also caused a homogenization of the TN content between BU positions contrary to CO ones (Fig. S9).

The SOC/TN ratio was higher in BU than in CO soils (Table 1), despite that both SOC and TN did not show significant differences between treatments. This could be related to different incorporation ratios (probably higher for SOC than for TN in BU sites). Significant decreases in the C/N ratio have been traditionally reported, for the upper 3 cm of post-fire soils, generally attributed to preferential immobilization of N over C into recalcitrant heterocyclic N structures (Armas-Herrera et al., 2016; Certini et al., 2011; Knicker, 2007). No significant differences were found based on position or depth for any treatment (Table 2), probably also related to the different trends of SOC and TN incorporation.

#### 4.2. Soil and organic carbon erosion

Significant differences were observed in the sediment yields of the studied sites. While no sediments were registered at the CO hillslope (soil cover was ca. 80–90%), the four erosive rainfall events produced sediment yields in the BU site ranging 12.63–143.63 kg, which correspond to 0.05–0.58 Mg ha<sup>-1</sup> (Table 3, Fig. 2). These values can be considered low compared to other reported yields, although it is important to bear in mind that most Mediterranean soil erosion research after prescribed and experimental fires has been plot-based (Shakesby et al., 2015). When considering the time between the first and the last erosive rain, the sediment yield in our burned hillslope was 0.92 Mg ha<sup>-1</sup> y<sup>-1</sup>, which is lower than the rate of formation of soils in Mediterranean areas (probably < 1 Mg ha<sup>-1</sup> y<sup>-1</sup>) (e.g. García-Ruiz et al., 2013). It is important to highlight that no gullies or rill formation was observed during the studied period and that surface stoniness was visually estimated about 30–40%.

Direct comparison between erosion yields at hillslope scale is not straightforward. There is a number of factors that can affect the process of post-fire water erosion besides climate (especially rainfall volumes and intensities), as size, slope and aspect of the hillslope, soil characteristics, and its spatial variation, vegetation recovery as well as fire severity and connectivity (Cammeraat, 2004). Shakesby and Doerr (2006) summarised post-fire rates of hillslope water erosion in USA, NE Spain and Israel with values for unbounded plots (26 m<sup>2</sup> – 0.97 km<sup>2</sup>) draining into sediment traps (0.1–70 Mg ha<sup>-1</sup>). Despite not working in Mediterranean environments, worth mentioning are the studies published recently by Prats and colleagues. These researchers have evaluated erosion yields at hillslope and small catchment scale in different burned sites of Portugal. Thus, in a burned eucalypt plantation, and us-

ing sediment fences in hillslopes ranging from 83 to 131 m<sup>2</sup>, they measured sediment yields up to 4.62 Mg ha<sup>-1</sup> (Prats et al., 2016). Later, in convergent hillslopes or swales of 500 to 800 m<sup>2</sup> they monitored soil losses, observing decreasing rates from 2.2 to 0.4 and 0.11 Mg ha<sup>-1</sup> y<sup>-1</sup> for the first, second and third post-fire years, respectively (Prats et al., 2019).

After a high severity forest fire, all parts of the burned landscape can be potentially involved in sediment production, transport and storage, and thus there is more of a continuum of these zones than a set of discrete units. For traps collecting sediment from unbounded areas, their usually much larger contributing areas mean that there is more spatial heterogeneity in terms of infiltration capacity, and surface roughness, and hence more opportunities for storage, and generally lower erosion rates. The degree of on-hillslope connectivity is the key control of the long-term net delivery of slope-derived sediments to valley floors (Bracken et al., 2015). At this point it is important to bear in mind that, in the first erosive event, sediment was collected in three out of four of the fences while in the following ones, sediment was collected in all the fences installed at the foot of our burned hillslope, indicating a lower connectivity in the first event than in the others (Table 3, Fig. 2).

Soil erosion is also responsible for off-site redistribution or lateral movement of SOC, putting multiple processes in motion that affect SOC persistence in soil (Berhe and Kleber, 2013). We postulate that the SOC that is not eliminated by fires can be mobilized with sediments from eroding to deposition zones, since most (70–90%) of the eroded topsoil material is ultimately deposited downhill within the same source or in adjacent watersheds (Doetterl et al., 2016). Typically, redistribution patterns at hillslope scale show SOC and sediment losses from convexities (crests and divergent shoulders, i.e. eroding zones) and accumulation in concavities (valleys and foot-slopes, i.e. transport and/or depositional zone). Most erosion after fire occurs in the surface layer, where heat has decreased aggregate stability and consequently, cohesion between soil particles (Campo et al., 2008; Mataix-Solera et al., 2011). Aggressive rain events, typical of the Mediterranean autumn (Peña-Angulo et al., 2019), in combination with wet-dry cycles could cause the breakage of aggregates (Campo et al., 2006; Shakesby, 2011), and aggregate-protected SOC is released and rendered more accessible and thus more easily decomposable by organisms and enzymes.

The fate of eroded SOC is controlled by multiple factors. First, the transport distance and medium control differentiation. Similar to other studies conducted in areas with steep topography and forest cover (Martínez-Mena et al., 2008; Stacy et al., 2015), a significant OC enrichment in eroded sediments was measured in the study area (Table 3). This is related to the fact that water erosion tends to typically transport the clay and silt sized particles (including ashes) as well as the free light fractions (burned and unburned), which are commonly OC enriched (Prats et al., 2021), compared to coarser soil particles. As this material has the size and the weight of fine sand or coarser silty particles, it will settle much earlier after detachment than smaller particles. In our hillslope, soil erosion is transport-limited as can be inferred from the yields transported by each erosive rain. However, OC contents in sediments were progressively reduced during the studied period (Table 3). Authors suggest that sediment become less enriched in OC as time passes during an event since the more carbon-rich fine aggregates are depleted early in the event (Martínez-Mena et al., 2012). In addition, removal of topsoil material from the eroded position can expose subsoil material with less C content, which can explain the decreased OC values registered in the rain of 23/3/2015 (the event that mobilised the highest amount of sediments, Fig. 2). Selective erosion, characterised by preferred detachment of SOC, occurs via rain splash and interrill erosion, which are of minor importance for large-scale sediment redistribution but might be locally important (Kuhn et al., 2009). In the study area, with the initial erosion of the organic rich topsoil, both sediment and OC erosion rates are high (i.e. Prats et al., 2021), but continuous erosion results in OC depletion (quantity change) and exposure of subsoil

OC, which forms a more passive, slowly degradable pool (quality change) (Kirkels et al., 2014). In general, it is expected that easily degradable SOC would be depleted and relatively less accessible SOC would be enriched, causing less mineralization.

In the first post-fire year at our hillslopes, 17.85 kgC (8% of total sediment corresponding to  $0.07 \text{ MgC ha}^{-1} \text{ y}^{-1}$ ) were mobilised from the BU hillslope (mainly from eroding position) and were stored at the depositional zone, although they could be transported further in the valley. This erosion of SOC is lower than the one reported by Prats et al. (2016) after a wildfire in a eucalypt plantation in central Portugal. In their first year post-fire, these researchers measured  $0.89 \text{ Mg ha}^{-1} \text{ y}^{-1}$  of OM lost from hillslopes of about  $100 \text{ m}^2$ . These losses were significantly reduced to 0.36, 0.10, 0.09 and  $0.03 \text{ Mg ha}^{-1} \text{ y}^{-1}$  in the following four years. In a similar study, they compared post-fire erosion rates for six convergent hillslopes or swales of 500 to  $800 \text{ m}^2$  and reported OM losses of 0.55, 0.06 and  $0.02 \text{ Mg ha}^{-1} \text{ y}^{-1}$  for the three years monitored, respectively (Prats et al., 2019).

There is a number of factors that can influence the net effect of transport and deposition on eroded SOC (and consequently on its fate) as soil characteristics, terrain attributes (e. g. slope gradient, surface roughness), the rate and nature of soil erosion, the amount and nature of the eroded OC, and the transport distance (Doetterl et al., 2016). Slow but long-range transport may lead to a higher degree of decomposition of mobilized SOC (Polyakov and Lal, 2008) (Polyakov and Lal, 2008); while fast but short-range transport, as the one observed in the study area, might lead to the burial of mobilized SOC with a lower degree of decomposition, at the depositional site (Kuhn et al., 2009; Quinton et al., 2010; Berhe and Kleber, 2013). At this zone, the rate of decomposition of eroded SOC can be reduced by a combination of processes. Some of these processes are biochemical (recalcitrance of organic constituents mainly those associated to pyrogenic carbon), others physical (protection with burial, aggregation, and changing water, air, and temperature conditions), and still others chemical (mineral-organic matter associations). As stated by Berhe et al. (2007), in this scenario, eroded SOC remaining near the surface of foothills could contribute to enhanced decomposition, higher mineralisation rates (Quinton et al., 2010; Van Hemelryck et al., 2010; Van Hemelryck et al., 2011; Polyakov and Lal, 2008), while the decomposition rate of the one that is buried is likely to be reduced. In general, first sediment OC-enriched will be buried by the sediments transported during the following rains (depending on their magnitude and on the frequency of episodic, large magnitude events such as storms or floods) (Nadeu et al., 2012).

Traditionally, soil erosion has been regarded as a process which leads to a loss of OM from soils, and consequently, has been considered as a C source (Lal, 2003, 2008). However, Berhe et al. (2007), Berhe et al. (2012) argued that soil erosion can constitute a C sink if post-erosion C stocks increase by production of new photosynthate (net primary production, NPP) at eroding positions, where exposure of C-depleted sub-soil can reduce mineralisation (Kirkels et al., 2014), and/or if reduced decomposition rate of eroded SOC at depositional sites occur. According to Berhe and colleagues, unless the soil is eroded beyond a critical level, NPP on eroding positions continues, and the newly assimilated SOC would replace, at least partially, SOC that was eroded. Based on this, it is expected that similar processes would happen at the BU hillslope and, as vegetation recovers, NPP at the eroding position would replace the SOC transported by erosion (helping then to delete the homogenization observed immediately after the fire between the different positions).

#### 4.3. Isolation of soil and sediment fractions

Fire could decrease the mass of aggregates  $< 2 \text{ mm}$  (CO: 45% BU: 36%) and increase it for those  $> 5 \text{ mm}$  (CO: 18% BU: 28%). As stated before, recrystallization of Fe and Al (hydr)oxides (Jimenez-Pinilla et al., 2016) or the possible accumulation of hydrophobic substances on

soil particles (Arcenegui et al., 2008; Terefe et al., 2008) can increase AS and consequently, the size of burned aggregates. Aggregates less stable could be broken by raindrop impact and erosion transport since no sediment  $> 5 \text{ mm}$  was collected after any of the erosive events, and the size of more than 60% of aggregates was  $< 2 \text{ mm}$  (Fig. 4).

No change was observed in the SOC distribution of the different size fractions between BU and CO hillslopes. In both sites, SOC content was higher in the fraction  $< 2 \text{ mm}$  than in the others, in which it was similar. Significant differences in the SOC content observed between environments, in CO, and between depths, in BU, were also observed in this fraction. When SOC content is weighed with the mass percentage of each fraction, soil in both sites followed the same trend: " $< 2 \text{ mm}$ " (CO: 29%, BU: 28%)  $>$  " $2\text{--}5 \text{ mm}$ " (CO: 17%, BU: 16%)  $>$  " $> 5 \text{ mm}$ " (CO: 8%, BU: 12%) (Figs. 2, 3). Differences observed in the mass distribution and/or the SOC content were also found here. In summary, this fraction  $< 2 \text{ mm}$  ( $> 35\%$ ) contributed to  $\geq 50\%$  of the SOC content in all treatments, positions, environments and depths studied.

No light material was observed before the first separation step of the soil density fractionation (i.e. before ultrasonic dispersion), in agreement with the high stability of aggregates (Table 1). All separations yielded small proportions of material in the two light fractions, while most of the soil mass was recovered with the HF ( $> 95\%$ ), in both CO and BU hillslopes, suggesting that mineral association would be the most effective mean of SOM stabilization (Six et al., 2002; von Lützow et al., 2006) von Lützow et al., 2006), particularly in poorly drained depositional landform positions with high content of SOM and oxides, and high rate of OM input from NPP (Berhe et al., 2012). However, in CO soils, the mass percentage in the FLF ( $\sim 3.5\%$ ) was higher than in the OLF ( $\sim 1.5\%$ ), but in BU soils, the former was similar to the latter ( $\sim 2\%$ ). Accordingly, relatively undecomposed, easily accessible OM of control soils (FLF) would be burned or mineralized according to the increase observed in the HF of BU soils (with regard to CO soils). New organic-mineral associations would stabilize this altered SOC (Kaiser and Guggenberger, 2007; Cerli et al., 2012) that otherwise would be removed and transported downslope as part of the sediment (Fig. 5). No changes were observed in the OLF mass of soils, in both sites, which would mean that the amount of OM already stabilized by aggregation, and protected within aggregates (Golchin et al., 1994; Wagai et al., 2009) was not modified by fire. All the SOC occluded within aggregates was quantified since its separation/dispersion was done by sonication, which causes stripping-off of basically all adhering mineral material, resulting in a general cleaning effect (Cerli et al., 2012).

Fire's impact cause also changes in the SOC content of the different density fractions. In CO hillslope, the SOC content was similar in both light fractions ( $\sim 29\%$ ), while for BU hillslope the content was lower in FLF ( $\sim 34\%$ ) than in OLF ( $\sim 38\%$ ). SOC content of light fractions was about 7 and 11 times higher than for HF, in CO and BU soils, respectively. Such differences are clearly associated to the SOC nature since both floating (light) fractions are supposed to comprise mainly plant-derived debris plus some animal residues, with very high SOC contents, in relation to already transformed SOC bound to minerals (HF). Apparently, there is no difference in the size and location of light SOC within the soil matrix of CO soils. However, in BU soils, the addition of partly burned organic material to soil and its stabilization within "new" formed aggregates would help to explain the transition in SOC from one fraction to another (i.e. active to passive), which may also change C mineralisation rates (Kirkels et al., 2014). All these differences (in mass percentages and SOC contents) were also found considering hillslope positions, environments, and depths (Table 4), which confirms the influence of these factors in the SOC distribution and stabilization in Mediterranean forest soils.

SOC content weighed by mass percentage was higher in FLF than in OLF at the CO hillslope, but similar at BU, and all were lower than in HF (Fig. 3). Accordingly, most of the SOC is concentrated in HF, not because its content is high, but because this fraction represents about 95%

of the total. SOC content of the HF was similar between hillslope positions in BU soils, contrary to CO soils, in which this content was lower at eroding than at depositional positions. All factors considered in the analyses influenced significantly the SOC distribution except the depth. Accordingly, soil erosion and deposition processes are important mechanisms of long-term OM stabilization confirming the findings of Berhe et al. (2012). These authors also showed that SOM in depositional positions is less transformed and better preserved than in eroding ones (older OM is not necessarily more decomposed, just effectively preserved).

In the sediment samples, most of the aggregates were <2 mm, and less in the range 2–5 mm. No sediment >5 mm was collected after any of the studied events (Fig. 4). The OC content was similar in both fractions contrary to BU and CO soils, in which most of the SOC was accumulated in the fraction <2 mm (Figs. 2 and 3). Erosion, deposition and mainly, the breakdown and redistribution of soil aggregates >5 mm could help to explain this redistribution of OC. This aggregate disruption and the consequent deep burial of sediments, through successive erosive events, could also be responsible of the change in C mineralisation rates (Kirkels et al., 2014). OC contents in the sediments progressively reduced over the studied period, and consequently, significant differences were only observed between dates as also found by (Martínez-Mena et al., 2012) Martínez-Mena et al. (2012).

The mass percentage and OC content of sediment's LFs were similar, but while the masses were smaller than in HF, the OC contents were higher. As observed for soils, most of the OC in sediments is present in the HF (highest mass) but higher contents were measured in the LFs. Consequently, the weighed OC contents of FLF and HF were similar and higher (almost twice) than that of OLF. This fact can be explained by the storage location of the erosion-mobilized OC. A gradient could be observed with most of the OC being transported as non-protected OC (FLF) in the first rainfall, followed by OLF and then by HF. In the next events, HF became the fraction transporting most of the OC followed by FLF and OLF. Although using different fractionation techniques, (Martínez-Mena et al., 2008; Martínez-Mena et al., 2012) Martínez-Mena et al. (2008, 2012) found that mineral-associated OC (MOC) represented the main C pool in degraded soils and their sediments in southern Spain. This transported and deposited OC would be protected from decomposition, and stabilized, when the environmental conditions were favourable, leading to an OC sink in these sediments (Doetterl et al., 2016).

According to Berhe et al. (2012), lateral redistribution of SOM due to soil erosion has important implications not just for storage (accumulation) but also for its stability in dynamic landscapes. In their study, the preservation of relatively simple, plant-derived molecules (based on NMR spectroscopy) was favoured in depositional over erosional landscapes. Similarly, in our site, the first erosive events transported high amounts of FLF that would be stored, and buried as easily assimilable SOM in the subsoil of depositional position, suggesting that this is an important mechanism of SOM stabilization. Doetterl et al. (2016) also supported this statement when concluded that, if burial is fast and conditions at depositional sites are favourable for C conservation, sediment deposition at foothills, valleys and floodplains can result in the burial of C enriched in labile compounds. Additionally, as in Berhe et al. (2012), significant differences in the content of FLF between hillslope positions were not observed in our study. These researchers stated that since the stock of active SOM is not higher in the eroding positions than in the depositional positions, it is not likely that there would be massive loss of active C during redistribution as was previously suggested by Lal (2003), Lal and Pimentel (2008), Polyakov and Lal (2008) especially when the distance travelled by OM from the eroding/source position to the deposition/destination site is short. Both Berhe and Doetterl and their colleagues suggested that the combined process of soil erosion and deposition lead to the formation of new aggregates, and may facilitate modification of aggregate size and distribution within the soil pro-

viding physical stabilization for eroded SOM in depositional positions. As they claimed, successive deposition of this eroded SOM after each erosive event in low-lying depositional positions is usually accompanied by reduced oxygen availability and increased soil moisture content, which render OM less accessible and reduce the rate of decomposition. Consequently, SOC content would be increased (effective stabilization) at the BU depositional position and would tend to be higher than in the BU eroding one as described for the CO hillslope.

It is suggested that burned depositional sites may have high fractions of their OC stocks in active forms (free light material). The newly liberated OM or the OM that exists free from any mineral association (i.e. FLF transported by sediments in the first erosive events) can form new physical or chemical associations with the freshly deposited reactive minerals/colloids or those that are transformed to become more readily reactive with anions in the depositional position. The new or transformed mineral-OM associations have potential to improve its chances of persistence in the soil system as OM (dissolved and/or previously unprotected particulate) finds itself in a more stabilizing matrix context in depositional positions, where an organic fragment that was previously surrounded by large sand grains may be deposited among silt and/or clay particles thus allowing the formation of tighter and more efficient physical barriers for decomposition (Berhe and Kleber, 2013).

## 5. Conclusions

Wildfire impact on Azuebar's hillslope caused changes in several of the soil properties, which confirms the research hypothesis. Burned and control hillslopes showed differences in the stability of soil aggregates but trends depended on the size analysed and the method used. Factors as hillslope position (related also to hillslope steepness, length, and curvature), soil depth and environment (under canopy soil vs bare soil) influenced significantly the movement, and stock of OC in the landscape. Along the burned hillslope a homogenization of SOC contents was observed between positions, contrary to the control hillslope in which transport and deposition positions presented higher values.

This study provides evidence that fire and erosion caused changes in the mass distribution of aggregates by size and density, and in the SOC distribution within aggregates and at the hillslope positions, which may have implications for SOC stabilization, and the role of soil erosion as a C sink. According to the obtained results, effective post-fire management actions should be oriented to control and reduce soil erosion of the fraction <2 mm, which presents high SOC content and is very prone to be transported off-site. This fraction should also include all the partially burned biomass, like leave fragments, branches and other free light material, which besides acts as a first mulching, contains very high amounts of OC that should be kept at the burned hillslope in order to increase soil fertility and promote vegetation recovery. This eroded free light material should be buried at the depositional site and if the conditions are favourable for its conservation, OC accumulation and stabilization would promote extension of its residence time.

Conclusions of this research are based on one-year study after a high severity fire in the Spanish Mediterranean. At the long term, the net effect of post-fire soil erosion and redistribution on OM stabilization/decomposition may depend on the duration of the exposure (erosion site) and transport times, amount of soil eroded (i.e. rate, depth and time since burial), and the conditions of the new soil matrix (nature and amount of mobilized C). Consequently, further studies are necessary in such Mediterranean areas to know the extent of OC mineralization during transport and deposition processes, which are paramount for the assessment of OC fate (long-term dynamics and persistence) and therefore, for the discussion about the role of soil erosion as source or sink of carbon.

## Uncited references

van Wesemael, 1995; Cerda, 1993.

## 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.

## Acknowledgements

This work has been supported by the VALI + d postdoctoral contract (APOSTD/2014/010) of the Generalitat Valenciana. J. Campo also wants to acknowledge C. Celi, J. Westerveld and J. Schoorl for the help with laboratory work; and A. Revynthi, B. Peñarroya, M.I. Montoya and P. Yousefi for their great support.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.geoderma.2021.115539>.

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