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Determining the potential impacts of fire and different land uses on splash erosion in the margins of drylands

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Abstract

This research aimed to estimate the splash erosion and its evolution during the first months in specific land uses after a forest fire. The study area was located in Congosto (North-West Spain), in the margins of Spanish drylands after a wildfire occurred in May 2012, which burned 15.56 hectares of scrubland and Pinus reforestation. Two different burned land uses were selected and compared to control areas: i) burned pine forest; and, scrublands. Rainfall intensity and the number, sizes and speed of raindrops were measured by an optical disdrometer and soil loss by funnels. Moreover, infiltration, soil moisture content, aggregate stability, water repellence, pH and organic matter were also measured. Results showed that the highest soil losses occurred in the burned areas, especially in the scrubland plots. The most influential factors were the presence of bare soil and the very low vegetation recovery rate. Changes in soil properties did not significantly influence splash erosion, although an increase in the presence of smaller classes of aggregates could promote erosion in the scrubland. We conclude that the vegetation ecosystem restoration is the key issue to be considered after a wildfire, especially, in those types of land uses which are severely affected by the fire in the margins of drylands.

KEYWORDS: ecosystem recovery; soils; splash erosion; wildfire.

1. INTRODUCTION

Fires are one of the most aggressive and powerful agents of land degradation in forests because it removes the vegetation cover that protects the soil and enhances runoff activation and, subsequently, soil erosion (Alcañiz et al., 2020; Chen, 2006; Pausas et al., 2009). To date, several challenges must be further studied considering the relationship between soils, humans and fire (e.g. Bento-Gonçalves et al., 2012; Rodrigo-Comino et al., 2020; Santín and Doerr, 2016). It is stated by several authors that splash erosion is the first stage of soil erosion activation, producing the collapse of unstable aggregates (Jomaa et al., 2012; Sadeghi et al., 2017). This fact is due to the raindrop impact and consequent detachment of soil particles that directly affects, especially, sites with lack of vegetation (Kavian et al., 2019; Liu et al., 2015). Splash erosion is determined by the characteristics of the soil surface it falls on, and intensity (Angulo-Martínez et al., 2012; Dunne et al., 2010). It is also determinant the size of drops and the speed, which can be modified by vegetation interception during their fall (Belmonte Serrato and Romero Diaz, 1998; Šraj et al., 2008). This can be very variable depending on the weather type conditions influenced by the wind origin, speed and intensity, and affecting the final soil erosion results (Nadal-Romero et al., 2015; Rodrigo-Comino et al., 2019). When the individual raindrops overcome the kinetic energy threshold, they can detach and transport soil particles (Salles et al., 2000, 2002). This threshold will also depend on the size and weight of the soil particles and organo-mineralogical composition of the soil aggregates (Furbish et al., 2007).

In Spain, the scientific community is paying more and more attention to this issue because the arson has significantly risen at the central plateau of the Iberian Peninsula after 2000 (Vecín-Arias et al., 2016). Since the last century, they are focusing on the relation between wildfires, ecosystem recovery, soil restoration and water resources (Cerdà, 1998). Immediately after the fire, the absorbing properties of the ash layer can reduce the potential activation of runoff to a negligible rate, decreasing the soil water repellency (Cerdà and Doerr, 2008). However, recent investigations also affirm that the ash layer soon loses this biological ability and is drastically exposed to immediate raindrop impact (Oliveira-Filho et al., 2018). As a result, after an initial stage, forest fires trigger higher soil erosion rates due to the absence of vegetation as other land uses such as agricultural fields, increasing in water repellency and reducing the water storage capacity (Bodí et al., 2012; Úbeda et al., 2006). Using an experimental set up rather exhaustive would permit to bring additional quantitative information on the effect of fire on soil erosion processes, which is scarce. Understanding the splash effect would give new insights to develop more specific soil erosion control measures in burned lands.

Fires change some key soil properties such as aggregate stability (Fox et al., 2007) or soil water repellency (Keizer et al., 2008), which also will deeply affect the splash effect (Zavala et al., 2009). Also, fire may produce changes in much stable physical and chemical characteristics of the soil such as compromising the mean weight diameter of aggregates, the distribution of aggregates, pH, organic matter content and aggregate stability (Fernández et al., 2016), which highly increase the potential splash erosion (Saedi et al., 2016). But the changes related to splash erosion that the fire produces in the soil also depends on the fire intensity and severity of fire (Jordán et al., 2011; 2016), which depends on several factors such as the previous ecosystem quality. Jordán et al. (2010) stated that high soil moisture minimizes the intensity of the fire, and, conversely, a fire produces a decrease of moisture content. For instance, phenomena such as soil compaction, building terraces in steep slopes (Hammad et al., 2006; Martínez-Hernández et al., 2017), tillage or engineering constructions (Abrantes et al., 2018; Awal et al., 2019) may influence the mean weight diameter of the soil aggregates and also decrease the infiltration capacity (Freebairn et al., 1991; Moldenhauer and Kemper, 1969). This would generate a surface water layer that may protect the soil from direct raindrop impacts, decreasing, subsequently, the splash erosion. Also, the released material can compact the soil surface and fill pores creating an impermeable seal that may leave soil particles exposed and ready to be washed away (Di Prima et al., 2018; Morin and Van Winkel, 1996).

The changes produced by the heating and combustion during the fire also affects the plant recovery (Bodí et al., 2012). High-intensity fires produce high seed mortality in the soil seed bank and the roots system decreasing the regeneration capacity of seeder species (Trabaud, 1998). On the other hand, under moderate intensity fires, the capacity of regeneration is higher, not showing symptoms of damage even after repeatedly burned (Schaffhauser et al., 2012). It is reported that some land uses need about 25 years to naturally return to their previous state, after a moderate fire, for example in *Pinus* spp areas or scrublands (Tang et al., 2013). The capacity of regeneration depends on the species of plants and the frequency of the time elapsing between consecutive wildfires because fires can make the soil nutrient status poorer, producing a transition from mature ecosystems to scrublands (Keesstra et al., 2017). Therefore, the main goal of this research is to determine the potential impacts of fire and different land uses on splash erosion, comparing two types of ecosystems: scrubland and Pinus reforestation. We investigated: i) how splash erosion behaves relating to the rainfall characteristics during the first months after fire; ii) which type of land uses depending on their capacity of recovery registers a clearer reduction of soil erosion; and, iii) which soil physical and chemical changes are produced by the fire in the margins of drylands.

2. MATERIALS AND METHODS

2.1. Study site

This study was carried out in a burnt area located in Congosto (León province, NW of Spain; 703205,55 X; 4722016,89 Y; 29T), in a hillslope exposed at SE (720-850 m above sea level), with an average inclination between 20 and 25°(Figure 1). This area is characterized by a Mediterranean climate, with annual precipitation of 580.5 mm and temperatures of 4.8ºC in January and 21.9ºC in July. Soils can be classified as Distric Cambisols (IUSS Working Group WRB, 2014), developed on shales and sandstones (Ramírez-Estévez and Reguera de Castro, 1995). In the land-use characterized by forests, the main tree species is *Pinus radiata* D. Don, cohabiting with other woody species such as *Cistus salvifolius* L., *Cistus ladanifer* L., *Erica arborea* L. meanwhile, the scrubland area presents several species such as *Cistus salvifolius* L., *Cistus ladanifer* L., *Erica arborea* L., *Erica australis* L. and *Erica cinerea* L, with scattered *Quercus ilex ssp. rotundifolia (Lamk.)* T. Morais *M.* (Table 1).

This area was affected by a wildfire on 15th May 2012. The fire burned 7.58 ha of Pinus reforestation and 15.5 ha of the scrubland area. There were different degrees of fire severity. In the scrubland, the litter and the vegetation were burnt, while in the pine area the underwood vegetation was burnt but the top of the trees was only partially burnt. Here, after that, it was noted that the needles of the trees fell on the soil creating new litter.

2.2. Experimental design and data collection

2.1. Plot design and splash erosion measures

Four different areas were selected (two burned land uses and two control plots): i) burned pine forest; ii) pine control forest; iii) burned scrubland; and, iv) control scrubland. All of them registered similar inclination, altitude, aspect, soil type, vegetation cover and type conditions (Figure 2). In each area, plots of 60 m^2 (20 m x 3 m) were installed and monitored. In each plot, there was a systematic sample design characterized by six splash funnel devices (Terry, 1998) set up in a horizontal line, each one separated from the next by of about four meters. A more detailed description of this double funnel device can be found in Fernández-Raga et al. (2019). One additional splash device was installed as a control one to check the soil transported by the air (Figure 3). There were 18 sampling periods from 29 June to 10 December 2012 (Figure 4) delimited by every rainfall event, after which the filter paper was changed, dried and weighed again in the laboratory. In suppl. Material, every rainfall event obtained by the disdrometer and the Spanish meteorological agency (AEMET; http://www.aemet.es/es/portada) were included. Also, daily sunlight (hours), maximum and average wind speed (km/h) were included.

The determination of the splash erosion load collected in the funnels was conducted by a gravimetric process, after heating the samples for 24 h at 105 ºC. To measure rainfall characteristics, rainfall intensity and number, size and speed of raindrops, an optical disdrometer Thies Laser Precipitation Monitor was installed at 9.3 km away at 42º33`0,3" N-6º34`51" W (Figure 1) at the roof of the University, because power supply requires a continuous connection to the electrical power grid and security reasons. This disdrometer is widely used for splash erosion studies (Angulo-Martínez et al., 2016; Fraile and Fernández-Raga, 2009). The highest rainfall amount registered was 22 l/m^2 (25/11/2012) by the disdrometer (Suppl. Material).

This device allows detecting the interruption of a laser beam (780 nm) by the raindrops from 0.13 mm to 8 mm of a diameter that crosses the sample area (30 mm width x 1 mm high x 160 mm long).

2.2. Soil properties and hydrological response

A secondary and parallel sampler line was also installed two meters downhill. Soil samples were collected to evaluate the effect of fire on some soil properties such as pH, organic matter content, mean weight diameter (MWD), distribution of aggregates and structural stability. These samples were collected in the first 5 cm depth to observe, the effects on the top layer, two months after the fire.

In the laboratory, pH was measured with an electrode (1:2.5 soil: water ratio) and the organic matter content with wet combustion with potassium dichromate (M.A.P.A., 1986). To obtain the mean weight diameter and the determination of the distribution of aggregates, dry sieving was used following the method proposed by Kemper and Rosenau, (1986). The structural stability was analysed with the drop impact test (Low, 1954).

Also, five measurements of soil water repellency, infiltration and moisture content were made to the splash erosion (Figure 3). The infiltration measurements were carried out by assessing the time of the water descent in a syringe in contact with the soil (Fernández-Raga et al., 2012). Soil water repellency was measured using a water drop repellency test (WDPT), and the results analyzed by comparing with the scale proposed by Doerr et al. (1998). Finally, volumetric soil moisture was measured using gravimetric analysis.

2.3. Vegetation recovery and fire severity

The resilience of the burned vegetation was assessed conducting two samplings in July (two months after fire) and November (six months after fire). In each sampling period, five experimental sampling units (1 m^2) were set around the funnels to estimate the visual percentage cover of woody vegetation and herbaceous species, litter and bare soil. In November, only burned areas were sampled to analyse the vegetation recovery because of the few and slow variations observed.

In the same sampling units, fire severity measuring the minimum diameter of remaining twigs as a mean of three twigs was evaluated. Fire severity was classified according to Cardillo et al., (2007): a) High severity: >10-15 mm; b) Moderate severity: 2-10 mm and c) Low severity: 1-2 mm (Table 1).

2.4. Statistical analysis

The Spearman rank coefficient was used to analyze the correlation between rainfall parameters, splash erosion and soil properties. The effect of wildfire on soil properties (pH, organic matter, mean weight diameter and structural stability and vegetation cover) was analysed using two-way analysis of variance (ANOVA). We consider as factors the type of vegetation (pine forest or scrub) and its status (burned or control). Previously, the normality was verified by the Kolmogorov-Smirnov test and the homogeneity of variances by the Levene test. Changes in the percentage of bare soil, woody and litter were analyzed using a repeated-measures ANOVA, with time as the repeated measure and the type of vegetation (pine forest or scrub) and the status of it (burned or control) as factors. When statistically significant differences ($P < 0.05$) were detected, the Tukey's test was applied. Finally, to confirm the influence of vegetation cover (bare soil, woody, litter and grasses), and soil properties (pH, organic matter, mean weight diameter and structural stability) on the total soil loss through splashing, a principal component analysis (PCA). Afterwards a correlation analysis (Spearman correlation rs) between splash erosion the two main components. All analyses were performed using STATISTICA 6.0 (Stafsoft 184-2001).

3. RESULTS

3.1. Splash erosion increased due to fire

Splash erosion increased after fire depending on the rainfall kinetic energy in both areas. In the scrubland, the increase in splash erosion produced during all the periods was nearly 15 times higher after the fire (from 1.2 g in the control plot to 18.7 g in the burned one) and 16 times higher for the pine area (from 0.4 g in the control plot to 6.3 g in the burned one) than the control plots (Figure 4). Table 2 shows the strong correlation between eroded soil and kinetic energy. A significative relationship ($P<0.05$) between splash erosion and four measured rainfall characteristics was detected (maximum kinetic energy, accumulated kinetic energy, accumulated precipitation and maximum intensity) in the burned area. On the other hand, no significant correlation ($P<0.05$) was found with any rainfall parameter and erosion in the control areas. Erosion did not register a significant correlation with the maximum kinetic energy, which obtained the highest correlation with the splash erosion.

3.2. Soil properties and hydrological response

Splash erosion decreased when an increase of water repellency was registered. The same pattern was also confirmed concerning soil moisture and infiltration in the burned areas (Table 2). The values of soil moisture content decreased in the unburned areas, but following a different temporal trend. While in the burnt Pinus areas, soil moisture values were higher during the three first weeks after the fire, decreasing later to less than 20%, the soil moisture content in the burnt scrubland was always much lower than in the control plot (Figure 5).

We also observed a decrease in eroded soil when the infiltration was higher, being significant in the pine control plot (Table 2). The infiltration rate was generally low during all the period, with values lower than 100 mm/h. Only three measurements conducted in the scrubland plot control and burned Pinus areas excessed 200 mm/h (Figure 5). All the studied areas showed slight water repellency. In addition, in the burned scrubland ($r = -0.69$; $P = 0.001$) and burned pine forest ($r = -0.47$; $P = 0.046$), a significant negative correlation was found; however, it coincided with a high variability in soil water repellency, both spatially and temporary. A significant negative correlation between eroded soil and water repellency was found in the control plot with pines ($r = -0.71$; $P = 0.001$), although with very low values of soil loss (Table 2).

Considering the effect of fire on soil properties, the mean weight diameter and organic matter content showed the same decreasing trend in both areas after the fire, meanwhile, aggregate stability and pH increased in the scrublands and decreased in the pine plot (Figure 6). The mean weight diameter of aggregates (MWD) of the unburned plots ranged from 0.94 mm to 0.98 mm in both land-uses. After the fire, it significantly decreased in the scrubland (nearly 30%) to 0.6 mm (F=37.23; P<0.001) but no changes were found in the pine plot. There were also significant differences between both types of vegetation (P = 0.002, F = 13.43) in the burned areas. The fire produced a decrease in the aggregate sizes (Figure 7), but the intensity of this change was significantly different. In the pine areas, aggregates bigger than 2 mm were the most stable and representative class before and after the fire, but with a light an increase between 1 and 0.25 mm.

We can deduce that this loss of aggregate sizes could be directly related to the changes in the organic matter, which is registered in Figure 6. This decrease in organic matter content was different in both areas (F = 10.42; P = 0.005), registering values of 4.1% in the Pinus plot and 21.7% in the scrublands; but they were not significantly different because of the high variability among samples. The pH showed differences between land uses but not after the fire. between control and burned scrublands (F = 5.776; P = 0.002), with values which vary between 5.3 and 6.2, respectively.

3.3. Vegetation recovery monitoring

The fire produced significant differences in both ecosystems related to the vegetation recovery (F=132,1; P <0.001). It was confirmed because of an increase in bare soil from 25-32% to 100%. Also, there was a significant decrease (F=286.6; P<0.001) in the coverage of woody species from 60% in the pine forest and 88% in the scrubland between 0.5-4% in both plots. Herbaceous species before and after the fire showed no significant differences (Table 3). The litter layer before the fire was close to 63% in both plots, remaining only of some pine litter (10%) and no scrubland litter (0%) after the fire. They obtained significant differences between vegetation types (F= 63.9; P <0.001).

In both sampling periods, lower soil losses in the unburned areas than in burned ones for Pinus areas and scrublands were found (Figure 4). Six months after the fire, a significant recovery for woody species took place (F=9.93; P=0.013); however, higher soil losses occurred in October coinciding with an elevate number of rainfall and bare soils that remained (Table 3).

3.4. Relationship between splash erosion, soil properties and vegetation variables.

The principal component analysis explained 53% of the variance (axis I), discriminating burned (negative side) and unburned (positive site) samples of pine forest areas and scrublands (Figure 8). The positive part of axis I is characterized by a higher weighted mean diameter of aggregates, indicating more stable soil aggregates. It also included a higher percentage of woody plants and herbaceous species. The negative part of axis I include burned scrubland samples, characterized by a higher cover percentage of bare soil. Axis II (explained a total variance of 19%) separated the soil samples collected in areas of pine forests (located on the negative side of the axis) of those belonging to the scrublands (located toward the positive side), with a higher content of organic matter content and aggregate stability. There is a negative significant correlation ($rs=-0.76$; $P<0.001$) between splash erosion and the complex environmental gradient represented by de axes I, indicating a strong connection between burned areas and soil loss due to splash erosion and the importance of vegetation to protect the soil. There was no significant correlation between splash erosion and the values located in axes II ($rs=0.22$; $P=0.33$).

4. DISCUSSION

We observed that the characteristics of the rainfall events and the lack of vegetation can determine the intensity of the splash erosion. This could be because the highest splash erosion impacts occurred during the first three months after the fire. In this research, during the monitoring period, no extreme rainfall events were registered, and the splash erosion was not as high as expected. However, the highest splash events were produced during the highest rainfall periods. Also, there are changes when the vegetation is recovered, as other authors also found in abandoned areas or deforested territories such as the Mediterranean mountains or the Loess Plateau (Cerdà et al., 2019; Chen et al., 2007). The vegetation recovery can play a key role in the interception process, enhancing the infiltration processes and developing new organic horizons (Martínez-Casasnovas et al., 2009). However, after the fire occurred, the vegetation was not successfully recovered in the scrub plot due of the short period since the fire, and land degradation processes were drastically noted (Fernández and Vega, 2014). Paying attention to the other forms of soil erosion, to put in context these findings in a broader perspective, some examples at larger scales can be cited. For example, Salesa et al., (2020) quantified soil erosion in a recently fire-affected territory to assess the soil loss on mountain trails obtaining an average soil loss from 1287 and 1404 Mg ha^{-1}. In northern Arizona, the Schultz Fire burned 6100 ha. Neary et al., (2012) monitored a series of flood events and due to erosion in bare soils, a substantial A horizon and much of the B horizon was eroded, generating gullies and rills. This can be considered relevant since, before the fire, no rills or gullies were developed because of the thick O horizon.

After the fire, there was immediate desiccation and loss of protection of the soil, because of the reduction of vegetation cover, which coincided with a decrease in the soil moisture content (Francos et al., 2016). The reduction of the soil moisture could be also related to an increase of the soil temperature by blackening or the presence of hydrophobic substances. Varela et al., (2007, 2005) found reductions up to 57% of moisture after fires, which coincide with our study where reached to 80% during some studied periods. However, there was an exception for the first sampling campaign carried out in the burned pine area, which conserved an elevated soil moisture content. Possibly, the main reasons were the high content of ashes on the surface soil, which can retain the water after raining, generating a new layer with organic material (Cerdà and Doerr, 2008). Ryżak et al., (2015) found that a decrease in soil moisture content may significantly favour the amount of splashed soil. This dynamic was confirmed in our study with a negative but not significant correlation.

This wildfire has modified some soil properties affecting the impacts of splash erosion on soils but highly variables in time and space. One of them was the water repellency. Fox et al. (2007) and Jordán et al. (2010) indicated that an increase in water repellency could induce an increase in surface runoff and erosion. On the other hand, other authors such as Bako et al. (2016) stated that this situation only occurs when the soil reaches a saturation point at which the splash can transport the material after separating the soil aggregates. This was also confirmed in early studies in Mediterranean areas characterized by arid and semi-arid climates, non-consolidated soils and low organic matter content (Cerdà, 1998; Imeson, 1983; Lavee et al., 1996; Poesen and Ingelmo-Sanchez, 1992). However, nowadays, there is no consensus about the value of this critical water depth and saturation point because of the influence of numerous factors such as porosity, rock fragments, root development or soil texture. Therefore, the high values of hydrophobicity or repellency found in these soils could be related not only to the fire but due to pedogenetic factors. These soils are developed under evergreen species such pine and heaths with resins, waxes and aromatic oils as other authors also found, coincident with other burned areas and water repellency responses (Doerr et al., 2000; Doerr and Moody, 2004). Some authors also highlighted the above-mentioned key role played by soil texture and aggregate stability (Bughici and Wallach, 2016; Moody et al., 2009). In this study, the sandy texture and acid condition could be a possible factor related to the greater susceptibility to hydrophobicity.

Another factor mentioned in the literature is the fire intensity. This fact could be also related to the changes in the aggregate distribution after the wildfire. There was a different response in both areas, which could be due to different fire severities. The large increase of smaller aggregate classes in the scrublands showed that the impact has been much higher there than in the pine area, coinciding with a reduction of the aggregate stability and, subsequently, increased susceptibility to splash erosion. The results found in the scrublands agree with Varela et al. (2007, 2005) who found that the reduction of the diameter after the fire, can especially affect the 2-5 mm fractions, increasing the values between 0.25-0.05-mm diameter.

There was a drastic decrease in the organic matter content, especially, in the scrubland. This fact can induce to think in a high intensity of the fire, decreasing the organic matter from the pyrolysed vegetation (Hernández et al., 2013; Jordán et al., 2011). The increase in pH, probably after solubilization of the ashes and block of the organic matter development, which also confirms the high temperatures reached in the scrublands (Bodí et al., 2012), because according to Giovannini and Lucchesi, (1997) and Marcos et al., (2007), it is necessary to reach temperatures higher than 450°C for a remarkable change in pH. Therefore, it can be inferred that in the scrubland plots, higher temperatures were reached.

The splash erosion potential is determined by the state and the percentage of cover of the soil surface. Therefore, regrowth of vegetation after a fire would influence the splash erosion potential. However, the relation between ecosystem recovery after fire and splash erosion has not been addressed in literature at all. Some laboratory studies (e.g. Shinohara et al., 2016) have shown that the aerial parts of herbaceous plants have a significant effect on the soil erosion rate, while roots have very little influence on the splash. But other scientists have found that the forest structure, especially the canopy cover and height may produce conjoining throughfall drops from branches and leaves that are responsible to produce 2.6 times higher erosion rates than open-field drops. This would increase the importance of understory vegetation such as shrubs, litter and herbs to protect soil surface against erosion (Geißler et al., 2012).

According to the recovery of the vegetation in both ecosystems, we also observed a lower cover during the study period, which is correlated to the increase in eroded soil. This could indicate that the intensity of the fire affects the speed of the recovery of vegetation, and this recuperation would be decisive in reducing the time of the bare soil exposed and, therefore, likely to be eroded (Pausas et al., 2009). Vegetation recovery time varies among ecosystems. For example, Rashid, (1987) in Algeria found that oak communities show smaller percentages of bare soil, and the regeneration of them after a wildfire is faster. Indeed, after the fire, the Pinus communities are the most susceptible ecosystems to be burned, although there is a decrease in the number of Pinus seedlings and diversity and species richness after the fire (Alvarez et al., 2007; Kim et al., 1999; Romeo et al., 2019).

The low recovery was probably due to the small number of rainfall events after the fire in May, enhancing the percentage of bare soils. Therefore, we can confirm the importance of water to enhance vegetation recovery after fire. Some authors indicated that the regrowth of these species should be visible from 3-6 months after the fire, but 6 months after fire very few signs were found (e.g. Calvo et al., 2003). Only in the pine plot, there was a slight but significant change. With a higher availability of water, the results could have changed very much. This may be another confirmation for the lower splash erosion rates in the burned pine area.

5. CONCLUSIONS

Fire influenced the soil characteristics but this was highly variable in time and space. The fire was able to decrease the soil moisture content in the two different studied land uses (scrubs and Pinus), and increase soil water repellency. These hydrological responses produced an increase in the intensity of splash erosion. Changes in pH, aggregate stability and organic matter allowed us to understand that, possibly, the intensity of the fire was higher in the scrub plots than in the Pinus ones. Moreover, the low amount of rain during the five monitored months possibly influenced vegetation recovery. Finally, the scrubland area appears to be more sensitive to splash erosion than the pine forest area. This is a clear reason for intervention and the development of sustainable control measures to reduce land degradation. Given the extent of scrubland in the world's drylands, this has wide relevance beyond the study area.

Data availability: The data that support the findings of this study are available from the corresponding author, upon reasonable request.

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FIGURES

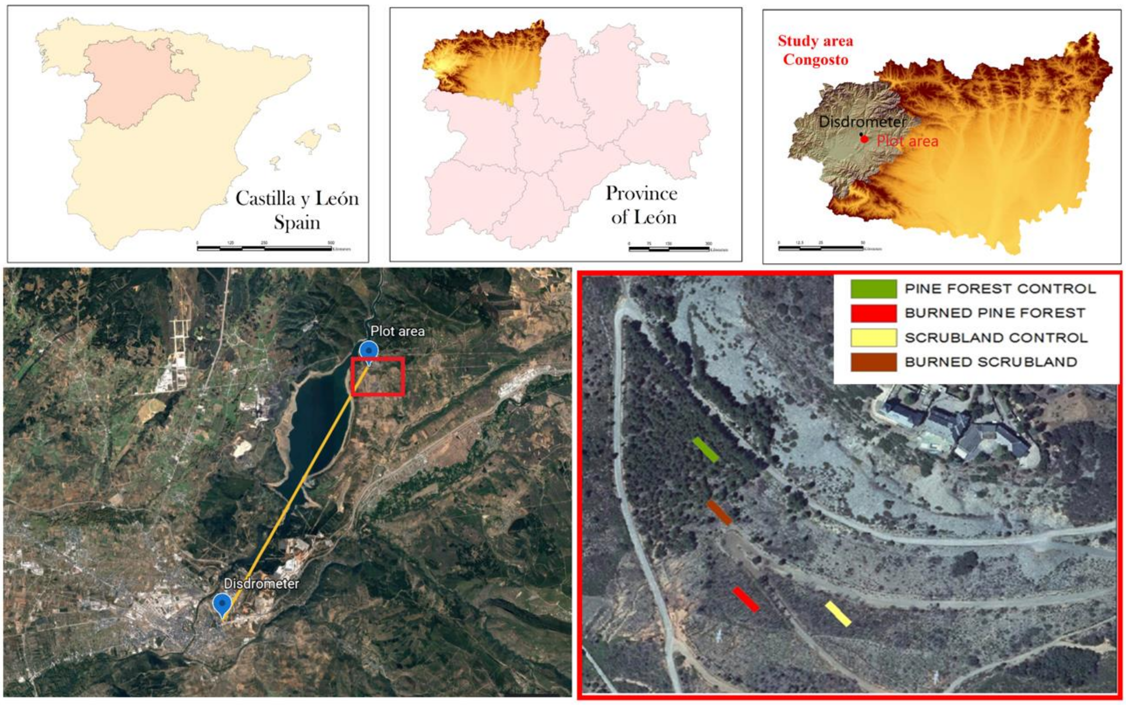

Figure 1. Study site with the four sampling areas.

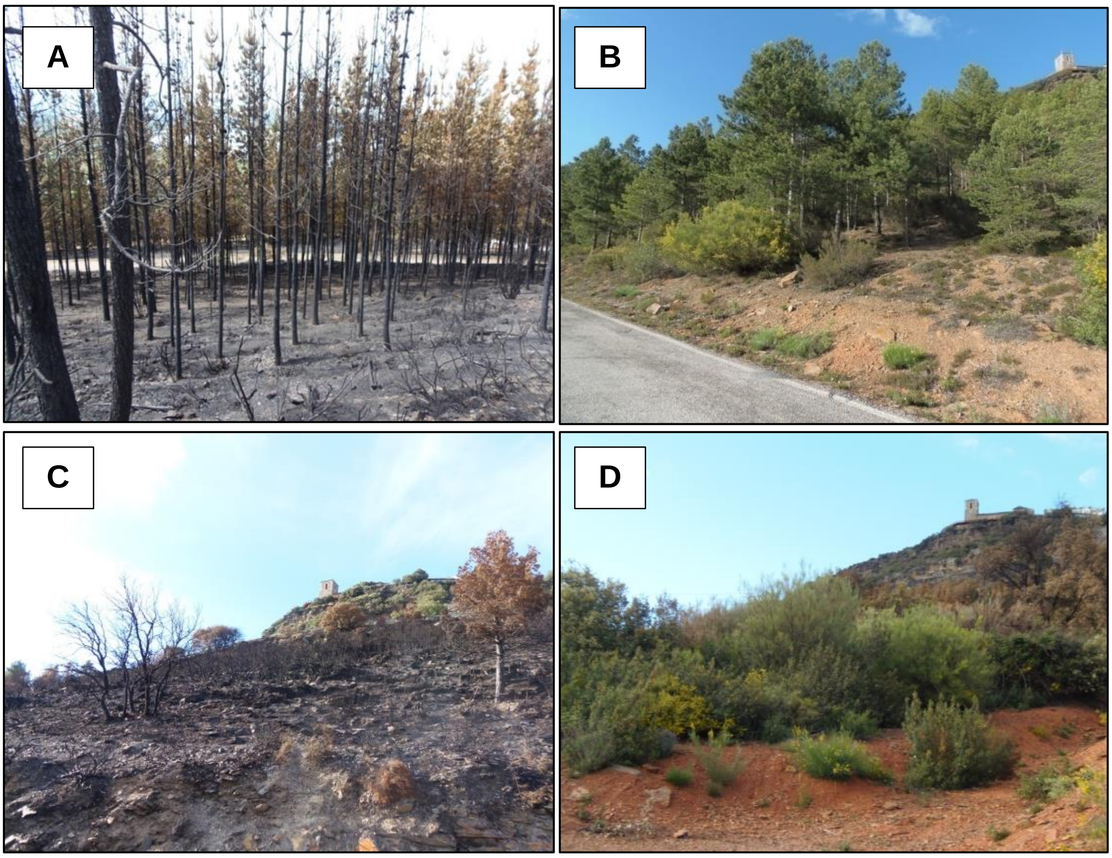

Figure 2. View of the four sampling areas: a) burned pine forest; b) control pine forest; c) burned scrubland and d) control scrubland.

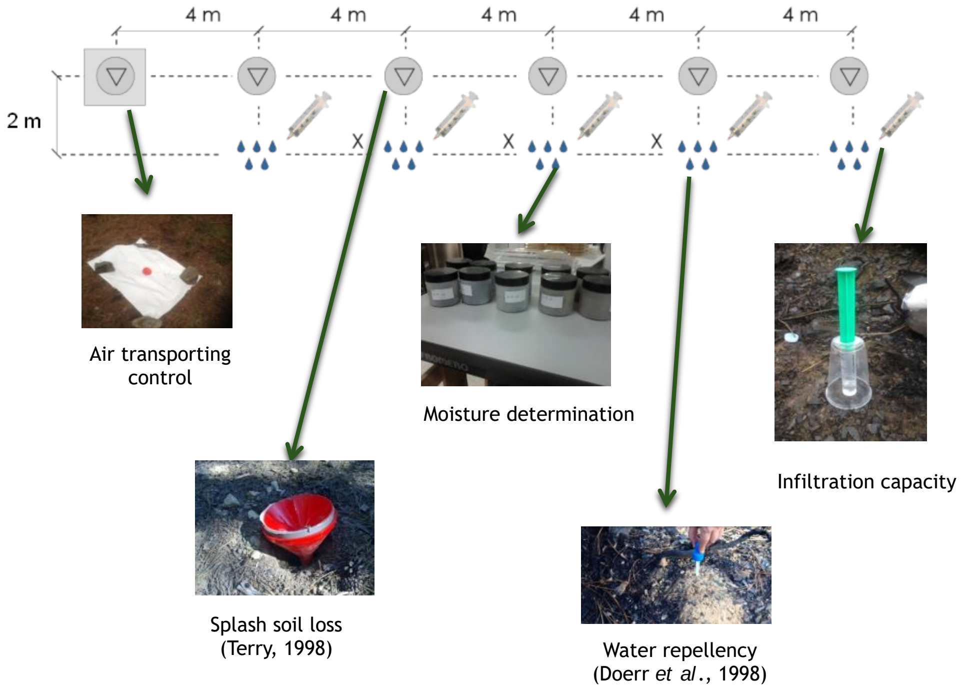

Figure 3. Design of sampling areas consisting on 6 funnels (one is the control), 5 points of measuring water repellency, 5 points measuring infiltration capacity and 3 points to take samples of soil (Fernandez-Raga, 2013).

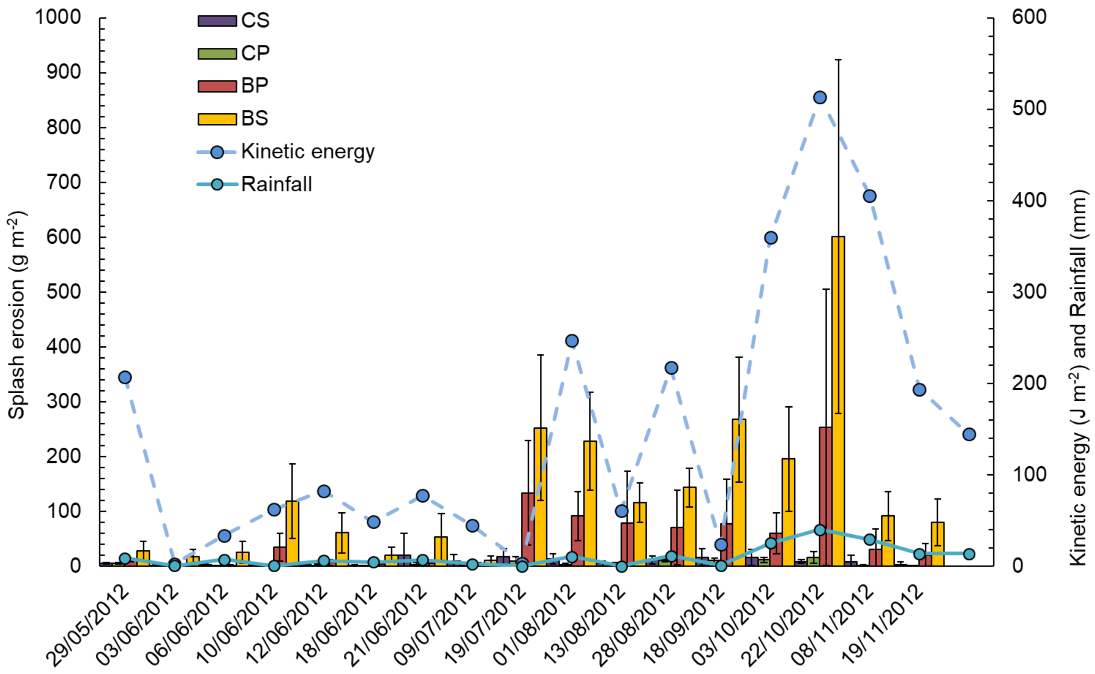

Figure 4. Interactions between kinetic energy, rainfall and mean values of splash soil detachment (g m^{-2}) and standard deviation corresponding to the four study areas (BP: burned pine forest BS: burned scrubland, CP: control pine forest, CS: control scrubland) during the sampling period included between 05/29/2012 and 12/11/2012.

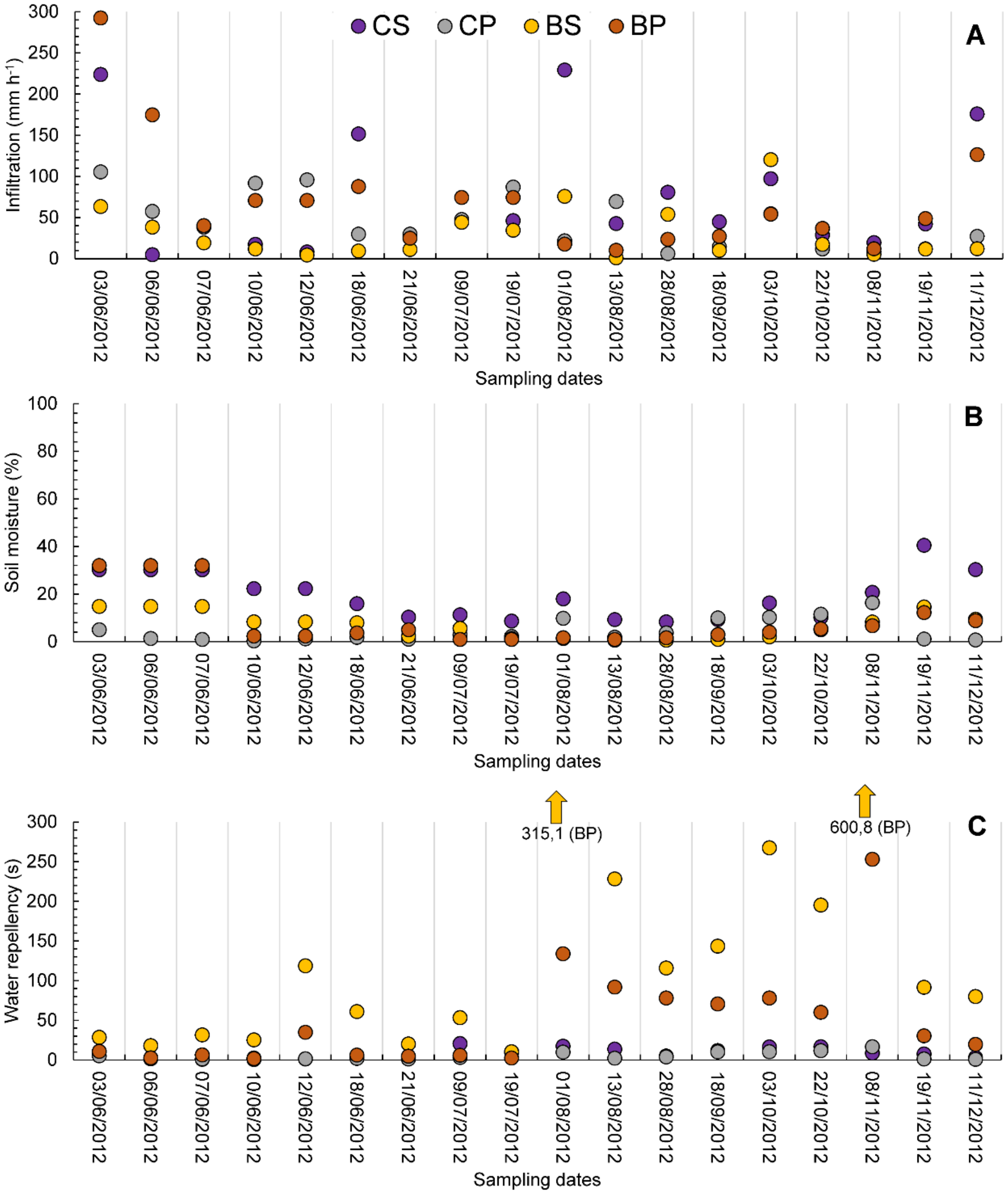

Figure 5. Infiltration (A), soil moisture (B) and water repellency (C) data of recovery ecosystems during the sampling periods. BP: burned pine-forest BS: burned scrubland, CP: control pine forest, CS: control scrubland.

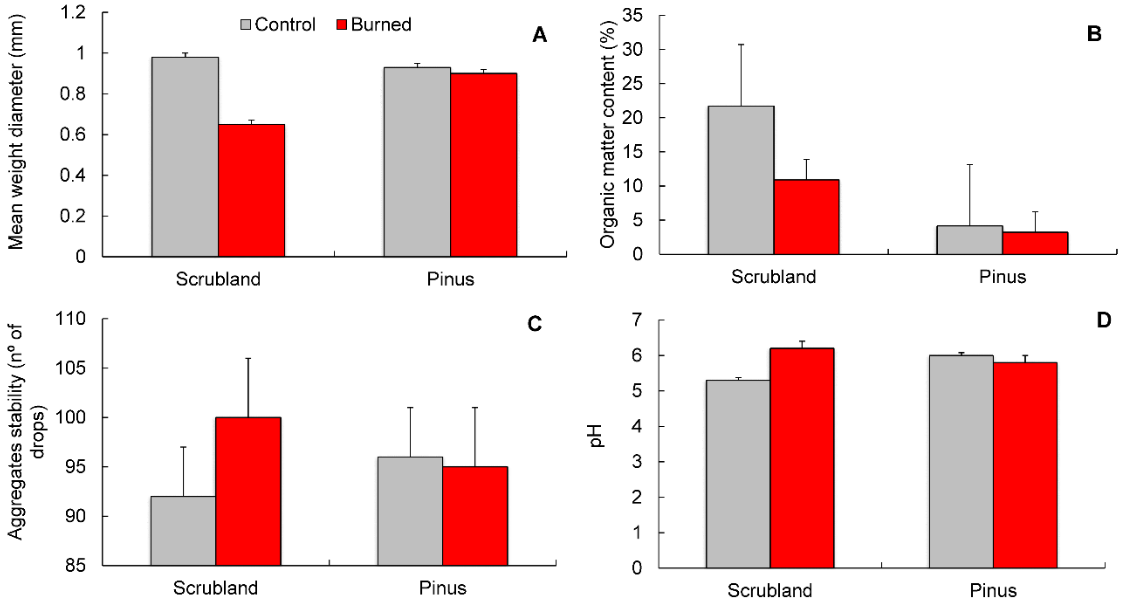

Figure 6. Mean values and standard deviation of a) mean weight diameter, b) organic matter content, c) aggregate stability and d) pH in the burnt and unburn plots for the scrubland (S) and the pine reforestation (P) after the wildfire. Different letters show significant differences (P<0,05).

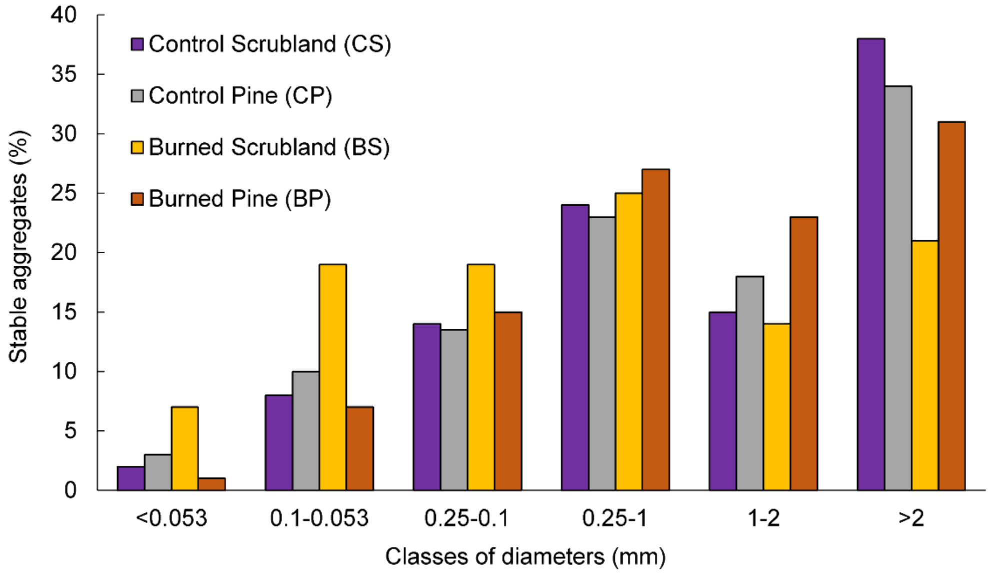

Figure 7. Percentage of soil aggregates by size classes.

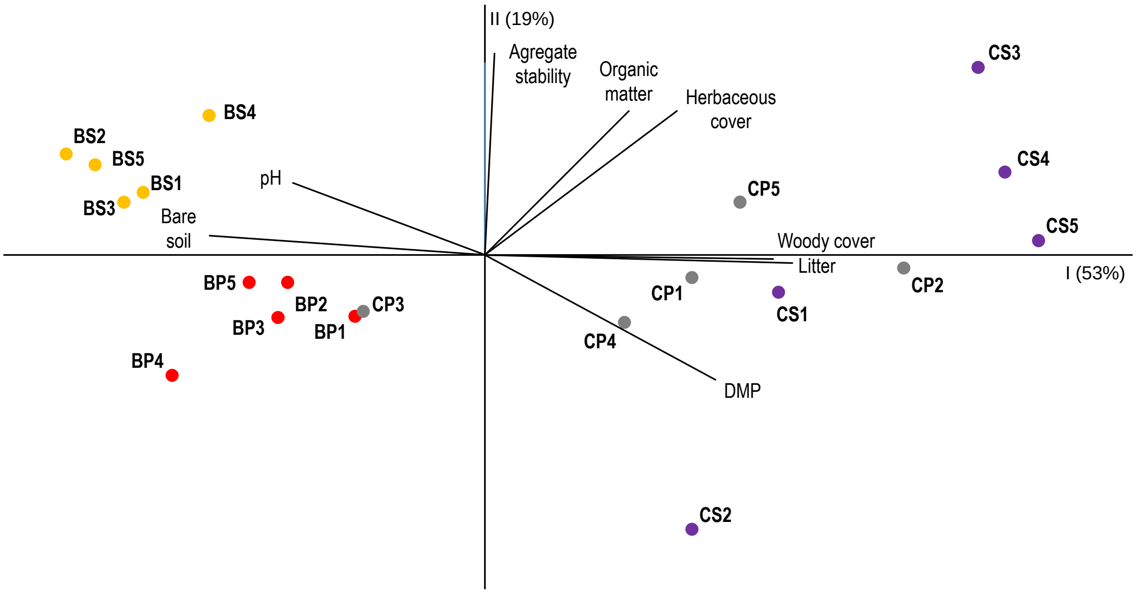

Figure 8. Situation in the plane defined by the first two axes of the principal component analysis of soil and vegetation properties in the four study sites. BP: burned pine-forest BS: burned scrubland, CP: control pine forest, CS: control scrubland.

TABLES

Table 1. Mean vegetation cover (%) corresponding to the four study sites two months after the fire took place.

Species	**CP**	**CS**	**BP**	**BS**
Pinus radiata	36	-	-	-
Quercus ilex	-	63	<1	<1
Erica arborea	3	-	-	-
Erica cinerea	20	2	-	-
Cistus ladanifer	1	-	-	-
Erica australis	-	-	3	-
Cistus salvifolius	-	14	-	-
Genista hystrix	-	2	-	-
Genista florida	-	7	-	-
Hebaceous species	2	3	-	1
Litter	63	63	10	-
Bare soil	32	25	95	100
Minimum diameter of remaining twigs (mm)	-	-	0.5	0.3
Fire severity	-	-	Moderate	Moderate

Note: CP: control pine forest; CS: control scrubland; BP: burned pine forest and BS: burned scrubland

Table 2. Relationships found among rainfall characteristics and soil properties and the amount of recovered soil after splash erosion in burned scrubland (BS), burned pine forest (BP), control scrubland (CS) and control pine forest (CP)

Rainfall characteristics	Units	BS	BP	CS	CP
Accumulated Kinetic energy	J m^{-2}	r=0.66; p=0.002	r=0.68; p=0.001	r=0.27; ns	r=0.44; ns
Maximum Kinetic energy	J m^{-2}	r=0.76; p=<0.001	r=0.79; p=<0.001	r=0.11; ns	r=0.36; ns
Accumulated Precipitation	mm	r=0.56; p=0.01	r=0.59; p=0.009	r=0.14; ns	r=0.40; ns
Mean Intensity	mm h^{-1}	r=0.44; ns	r=0.43; ns	r=0.12; ns	r=0.19; ns
Maximum intensity	mm h^{-1}	r=0.52; p=0.02	r=0.51; p=0.029	r=0.25; ns	r=0.33; ns
Number of drops	drops	r=0.44; ns	r=0.41; ns	r=0.20; ns	r=0.32; ns

Soil properties	Units	BS	BP	CS	CP
Repellency	s/drop	r=-0,69 p=0.001	r=-0.47; p=0.046	r=-0.01; ns	r=-0.71 p=0.001
Infiltration	mm h^{-1}	r=- 0.09; ns	r=- 0.38; ns	r=- 0.01; ns	r=- 0.55 p=0.018
Humidity	(%)	r=-0.34; ns	r=- 0.23; ns	r=- 0.43; ns	r=-0.268; ns

Table 3. Mean values of woody plants, herbaceous, litter and bare soil cover, number of woody species and soil loss per study site in both sampling periods (two and six months after fire) (n=5).

	Study sites							
	CP		CS		BP		BS	
	July	Oct	July	Oct.	July	Oct.	July	Oct.
Woody plants (%)	60	-	88	-	4	12	0.4	7
Herbaceous (%)	2	-	3	-	-	-	1	-
Litter (%)	63	-	63	-	10	10	-	-
Bare soil (%)	32	-	25	-	95	90	100	95
Woody species richness (nº)	4	-	5	-	2	2	1	2
Splash erosion (g)	0.04	0.05	0.12	0.09	0.45	1.36	1.00	3.01

Note: CP: control pine forest; CS: control scrubland; BP: burned pine forest and BS: burned scrubland