



Benefits of adding forestry clearance residues for the soil and vegetation of a Mediterranean mountain forest

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ABSTRACT

Desertification is occurring throughout the mountainous areas of the Mediterranean. These processes lead to reduced soil fertility, increased soil loss, and reduced vegetation cover and species richness. To prevent further damage, it is recommendable to use low-cost approaches that are compatible with the European Strategy of Circular Economy guidelines. We investigated the systemic benefits from recycling of forest clearance residue by adding it to a dry Mediterranean mountainous area. More specifically, we performed afforestation without addition of residue in two control plots (C plots), and afforestation with addition of 10 Mg ha⁻¹ of clearance residue from a nearby region dominated by Aleppo pine (*Pinus halepensis* Mill.) in two other plots (PM plots). We conducted the experiments throughout 30 months after the afforestation process. Eighteen months after the intervention, the PM plots had significant increases in the soil organic carbon (SOC), and related increases in ecosystem productivity and stability. More generally, addition of clearance residues improved soil and vegetation recovery, and contributed to more successful afforestation. The improvements may be explained by an increase of infiltration process due to the physical changes in the soil following bio-waste addition. Addition of the forest residues increased the formation of soil macrochannels, and also increased the sink area, thereby improving the hydrodynamics of the ecosystem. Thus, soil loss was reduced by 98.2% in the PM plots relative to the C plots. Our study indicates that application of forest clearance residues to Mediterranean mountainous areas is an effective land management practice that produces very little waste, and it is in accordance with European policy.

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1. Introduction

Desertification in the Mediterranean region has increased due to climate change, its unique lithology and geomorphology, and human activities (Lavee et al., 1998; COM, 2002; Eaton et al., 2008; Martínez-Murillo et al., 2016). Desertification leads to reduced soil fertility, increased soil loss, and reduced vegetation cover and species richness (García-Orenes et al., 2009). This problem is particularly serious in mountainous forest areas, where soil organic carbon (SOC) dynamics are particularly sensitive to climate change (Olaya-Abril et al., 2017). Thus, when these regions pass certain thresholds of change, the ecological damage may be not be reversible, making active restoration activities necessary (Vallejo et al., 2000; Hueso-González et al., 2016; Muñoz-Rojas et al., 2016).

Forest managers commonly use revegetation programs to restore the function of mountainous forest ecosystems (Bretón et al., 2016; Nadal-Romero et al., 2016), because vegetation improves soil conservation by increasing SOC and reducing soil loss (Cerdà, 2001; Van Hall et al., 2017). Mountain forest soils with reduced natural vegetation experience decreases in organic matter and nutrients (Marqués et al., 2005; Hueso-González et al., 2015), which can make vegetation

recovery more difficult and lead to ecosystem degradation by a positive feedback loop (Lavee et al., 1998; Ruiz-Sinoga and Martínez Murillo, 2009). However, afforestation programs for dry Mediterranean mountain areas have only had limited success (Castro et al., 2002; Navarro and Palacios, 2004). This is because the dry Mediterranean environment has limited water during summer droughts (Maestre et al., 2003; Gouveia et al., 2017) and also has excessive radiation (Pumo et al., 2010), both of which impede seedling establishment and growth. Thus, alternative low-cost afforestation methods are needed that increase seedling survival and growth and do not have negative environmental impacts (Eldridge et al., 2012; Benigno et al., 2013).

The unique characteristics of the Mediterranean mountains contribute to the rapid spread of highly severe fires that are difficult to control (Francos et al., 2016; Tessler et al., 2016). Thus, before the summer drought, forest managers often clear vegetation to reduce the amount of combustibles, to counteract the losses from land management practices that consume biomass, and to reduce the overall risk of fire (Tedimm et al., 2016; Pereira et al., 2014). However, these vegetation residues are often subjected to prescribed burning or moved to a waste plant, activities with significant economic and environmental costs. Moreover, this land management practice is not in accordance with the Circular Economy Strategy of the European Union (COM, 2017a). This European guideline urges all member states to transition to a “circular economy”, in which resources are

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conserved and waste is reused. Within the framework of the Circular Economy Strategy, it is particularly important ensure the proper management of organic waste (bio-waste), due to the large volume generated and the environmental implications of mismanagement (COM, 2017b). According to the National Statistics Institute of Spain, waste management companies collected 21.3 million of metric tons of residues in 2015 (INE, 2016). The global emissions generated from the treatment and elimination of solid organic waste represent 4.0% of greenhouse gas emissions and 6.5% of emissions from diffuse sectors (CONAMA, 2016). Thus, it is necessary to find new approaches for clearance and recycling of mountainous forests that are compatible with the European Strategy of Circular Economy (Ghisellini et al., 2016).

We investigated the effect of reuse of forestry residues (bio-waste) as a restoration method in a dry Mediterranean mountainous forest. We hypothesized that use of forestry clearance residues in this region could enhance ecosystem productivity, stability, and hydrodynamics, and also improve vegetation recovery.

2. Materials and methods

2.1. Study area

The experimental plots are in a mountainous region of southern Spain, inside the Sierra Tejeda, Almirajara and Alhama Natural Park (Fig. 1; 36.7985173° N 3.8511693° W; WGS84). These plots are in the upper part of an alluvial fan that has calcareous conglomerates, and they are located at an elevation of 470 m.a.s.l. This region has a dry Mediterranean climate (mean annual temperature: 18 °C, mean annual rainfall: 589 mm). The plots are in an agriculture field that was abandoned and then recolonized by shrubs since the 1950s, and it was subjected to natural burns during the summers of 1975 and 1991. The surrounding region has an open forest of *Pinus halepensis* Mill. (Aleppo pine), and an understory of degraded Mediterranean scrub and tussocks. FAO (2006) describes these soils as lithic and eutric leptosols. > 50% of rock fragments are on the surface, with 56% gravel content in the soil profile. The soil has 60% sand, 32% silt, and 8% clay, and has a sandy-loam texture.

2.2. Experimental design

The four plots were 2 m × 12 m in size, had slope gradients of 7.5%, aspects of N170°, and were parallel to the line of maximum slope. We enclosed the plots with steel strips (50 cm × 1 m) that were 30 cm deep. The vegetation cover of each plot was cut uniformly in October 2010. A 250 L container was used to collect overland flow and sediment at the point of lowest elevation in each plot.

A HOBO weather station was installed in the experimental region in April 2011. We monitored rainfall with a tipping-bucket rain gauge (accuracy: 0.2 mm) every 15 min.

In May of 2011, we applied clearance residues (application rate: 10 Mg ha⁻¹; max. chipped diameter: 3 cm²; max. chipped length: 8.5 cm) from a nearby Aleppo pine forest to the surface of two plots (PM plots), and used the other two plots as controls (C plots). In November 2011, we afforested all four plots with 9 species (*Chamaerops humilis* L., *L. stoechas* Lam., *L. dentata* L., *L. multifida* L., *Pistacea lentiscus* L., *Rosmarinus officinalis* L., *Rhamnus alaternus* L., *Rhamnus oleoides* L., and *Thymus capitatus* L.) using the same number of plants and spatial pattern as previously used by managers of this park. The vegetation was planted in a grid, and the plants were 0.5 m apart. The soil was tilled to a depth of 25 cm during the afforestation procedure.

2.3. System productivity and stability

System productivity and stability were evaluated by measuring SOC (Hussain et al., 2016; Garau et al., 2017; Sarma et al., 2017). Soil samples from the afforested plots were sampled in: (i) spring 2012 (6 months after afforestation); (ii) fall 2012 (12 months after afforestation); (iii) spring 2013 (18 months after afforestation); (iv) fall 2013 (24 months after afforestation); and (v) spring 14 (30 months after afforestation) (Fig. 1). Four samples were collected from each plot (40 soil samples in total) from the soil surface, e.g., 0–10 cm depth, in which according to Pierce et al. (1994), most soil transformation occur. The SOC (%) was measured using the Walkley-Black method of oxidation with dichromate, and subsequent titration (FAO, 2006).

2.4. Soil vegetation recovery

The effect of the intervention on soil vegetation recovery was evaluated by measuring afforestation and different components of the soil surface. Measurements of the soil surface were taken at the end of the wet season (May 2013) and at the end of the dry season (September 13) (Fig. 2), according to procedures of Calvo-Cases et al. (2005) and Arnau-Rosalén et al. (2008). For analysis of soil surface components, a 12 m wide strip was marked within each plot; this strip extended from the highest elevation to the lowest elevation of each plot. This region was photographed using a camera placed 6 m above the center of each 4 × 2 m grid, along a transect. A Global Navigation Satellite System (Leica GPS1200) was used to geo-reference the four grid knots. Then, the ArcGis 10.2 Georeferencing Tool was used to rectify the images to the topographical grid, based on the four grid knots of each photograph. This analysis considered rainfall infiltration (sinks, i.e. bare soil with embedded rock fragments accounting for > 70% of cover, high density shrubs, low density shrubs, litter, dead plants, and annual plants) and overland-flow (sources, i.e. rock outcrops and bare soil areas with rock fragment accounting for > 70% of cover, and crusts), as described by Ruiz-Sinoga and Martínez Murillo (2009).

From 2011 to 2014, seedlings were measured: (i) 6 months following transplantation (May 2012); (ii) 12 months following transplantation (November 2012); (iii) 20 months following transplantation (June 2013); and (iv) 30 months following transplantation (May 2014) (Fig. 2). The number of surviving plants was also measured, as described by Castro et al. (2002) and Gómez-Aparicio et al. (2004). A plant was considered alive if it had living leaves, buds, or stems.

2.5. Hydrodynamics

The hydrodynamics of the study area was evaluated by determining rainfall, overland flow, and sediment yield (November 2011 to May 2014). The total overland flow from each event was determined by measuring accumulation in the 250 L containers. Overland flow samples (1 L) were collected after each event for laboratory analysis and measurement of sediment concentration. The sediment concentration was determined at three levels within each container, following mixing, and multiplied by the volume of water to calculate total sediment yield.

2.6. Statistical analysis

Normal distributions and homogeneous variance were verified using the Kolmogorov–Smirnov test and Levene's test, respectively.

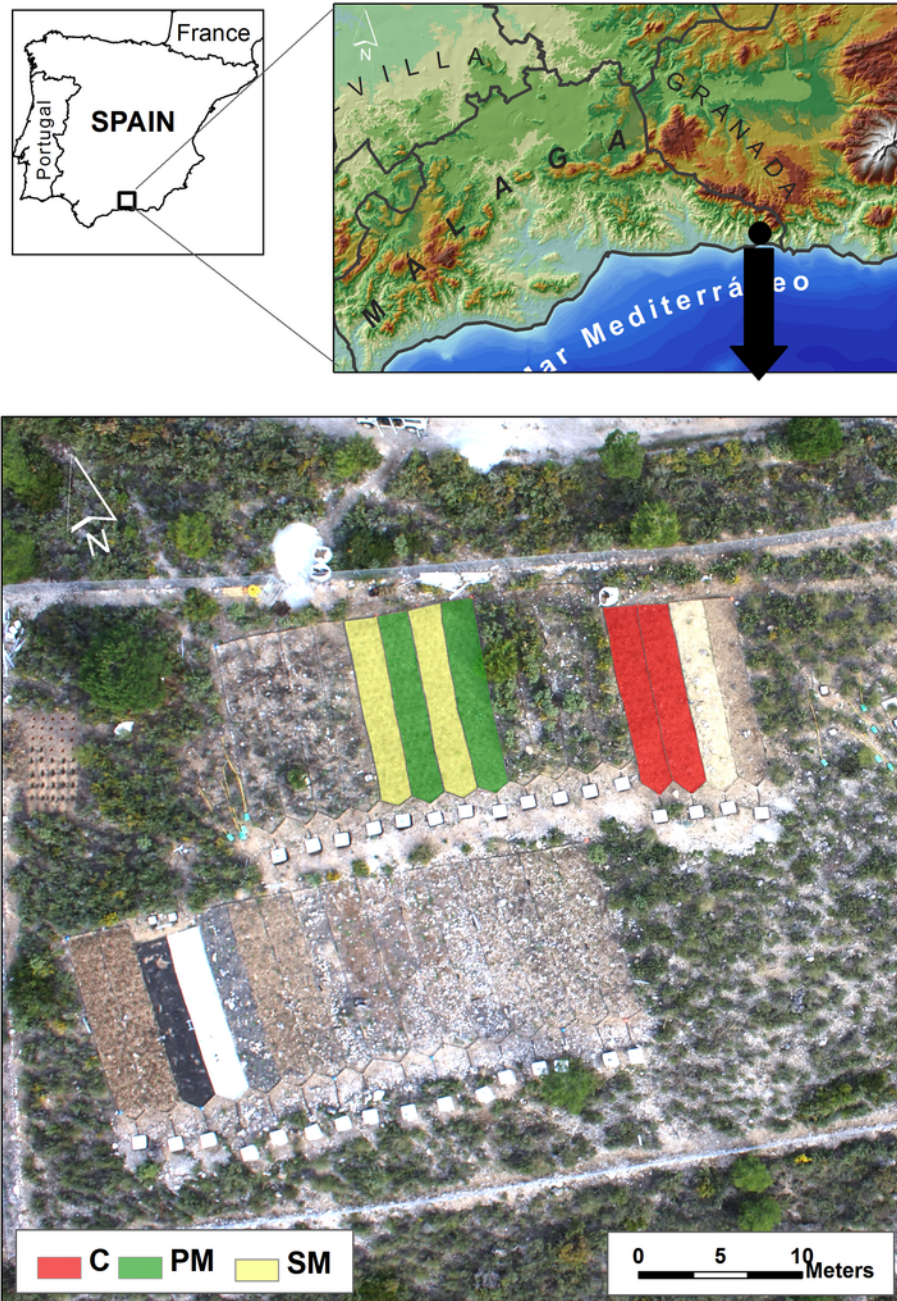


Fig. 1. Location of the experimental site. C: soil afforested, no amendment; PM: soil afforested and amended with clearance residues from a close Aleppo pine (*Pinus halepensis* Mill.) forest.

Differences in soil properties, eco-geomorphological patterns, and seedling survival were determined using a one-way analysis of variance (ANOVA). For data that was not homoscedastic (Levene's test: $p < 0.05$), the nonparametric Mann-Whitney U test was used. In all analyses, a p value < 0.05 was considered significant. All statistical analyses were performed using SPSS software (version 21) for Windows (IBM Corp., 2012).

3. Results and discussion

3.1. Effects on system productivity and stability

The SOC correlates with an ecosystem's capacity to provide nutrients to plants and to retain elements or compounds harmful to the environment or plants (Almendros et al., 2010; Puttaso et al., 2011; Ali and Yan, 2017). Thus, the SOC reflects important functional processes in the soil, such as storage of nutrients, water-holding capacity, stability of aggregates, and microbial activity (Casals et al., 2000; Eaton et al., 2008; Ghimire et al., 2017). Hence, the SOC is a

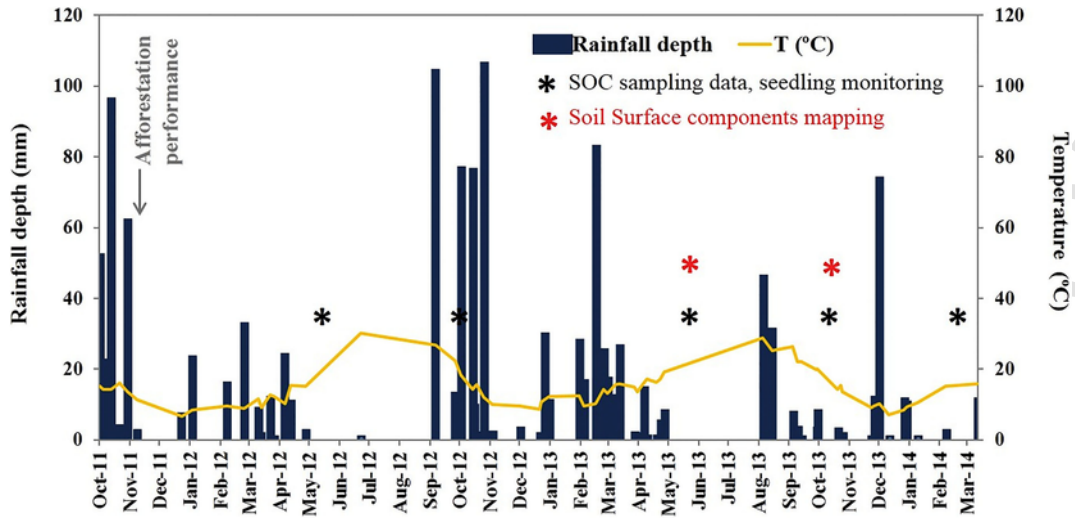


Fig. 2. Rainfall and temperature from October 2011 to March 2014.

key component of soil fertility that is related to an ecosystem's capacity for restoration (Hueso-González et al., 2014; Hussain et al., 2016; Garau et al., 2017; Sarma et al., 2017).

The PM and C plots all had increased SOC at the end of the wet season, due to the timing of the spring sampling (6, 18, and

30 months after afforestation) (Table 1). Previous researchers also described this intra-annual variability in Mediterranean areas (Casals et al., 2000; Ruiz-Sinoga and Romero Díaz, 2010; Gabarrón-Galeote et al., 2013). A low level of rainfall leads to homogenization of soil function and reduced SOC during the summer in dry mountainous areas (Ruiz-Sinoga and Martínez Murillo, 2009). Thus, we found that the low level of rainfall and high temperatures in the summers of 2012 and 2013 (Fig. 2) led to substantial mineralization of the organic matter fractions, and there were significant decreases in SOC during the autumns of 2012 and 2013 (12 and 24 months after afforestation; Table 1).

Analysis of SOC in the PM plots indicated there were only significant differences from the C plots at 24 and 30 months (Table 2; $p < 0.05$). There are three possible reasons why these plots did not differ in SOC during the first 18 months. First, we did not apply composting processes to the forest clearance residue, so this increased the time needed for decomposition (García-Gómez et al., 2005; Hueso-González et al., 2016). Second, the forest clearance residue had a high content of cellulose and lignin, impeding degradation (Duryea et

Table 1

Changes of soil organic carbon (%) over time in afforested control plots (C) and afforested plots treated with Aleppo pine debris (PM).

Time of exposure	n	Soil organic carbon			
		C plots		PM plots	
		Mean	SD	Mean	SD
6 months	8	3.21	0.30	4.60	0.70
12 months	8	1.83	0.00	1.71	0.40
18 months	8	2.97	0.42	2.52	0.20
24 months	8	1.41	0.31	2.14*	0.12
30 months	8	3.03	0.40	5.17*	0.20

* Indicates a significant difference relative to the control ($p < 0.05$).

Table 2

Spatio-temporal analysis of changes in percent cover by different classes in afforested control plots (C) and afforested plots treated with Aleppo pine debris (PM). Class 1, dead plants connected to soil; class 2, litter; class 3, bare soil; class 4, crusted soil; class 5, rock outcrops; class 6, shrubs; class 7, annual plants; class 8, bare soil with rock fragment cover of > 70%; class 9, bare soil with embedded rock fragment cover of > 70%.

		Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9	
C	Replicate 1	May-13	0.00	3.80	15.00	0.00	0.00	21.80	28.60	30.80	0.00
		Sep-13	22.90	15.30	1.50	11.40	0.00	41.70	0.00	0.00	7.30
	Replicate 2	May-13	0.10	5.40	2.60	5.30	0.00	18.70	23.40	44.60	0.00
		Sep-13	8.70	21.30	2.40	10.00	1.90	11.90	0.00	33.10	10.70
	Mean		7.93	11.45	5.38	6.68	0.48	23.53	13.00	27.13	4.50
	SD ±		9.34	7.19	5.57	4.47	0.82	11.09	13.13	16.51	4.66
PM	Replicate 1	May-13	0.00	14.50	0.90	0.00	0.00	38.20	22.00	24.40	0.00
		Sep-13	2.40	63.60	0.00	0.00	0.00	23.40	0.00	10.60	0.00
	Replicate 2	May-13	0.70	38.60	1.00	0.00	0.00	34.80	9.60	15.40	0.00
		Sep-13	0.00	58.10	0.60	0.00	0.00	27.10	0.00	13.20	1.10
	Mean		0.77*	43.77*	0.62*	0.00*	0.00*	30.88	7.90	15.90	0.27*
	SD ±		1.13	22.23	0.45	0.00	0.00	6.81	10.43	6.00	0.55

* Indicates a significant difference relative to the control ($p < 0.05$).

al., 1999; Jensen, 2009; Hueso-González et al., 2015). Third, the rate medium-high selected (10 Mg ha⁻¹) that became the mineralization process lower (Young et al., 2015). Other authors have demonstrated that addition of more organic waste to the soil increases the time needed to see an effect on SOC (Jordán et al., 2010, 2011).

We found that the bio-waste addition to the PM plots led to a significant increase of SOC at 24 months and an even greater increase at 30 months. This increase of SOC over time indicates a long-term improvement of soil function, a response that could lead to additional systemic responses (described below) (Bulluck et al., 2002; Adekalu et al., 2007; Mulumba and Lal, 2008).

3.2. Effects on soil recovery

Our spatio-temporal analysis of the eco-geomorphological pattern shows significant modifications in the PM plots (Table 2; $p \leq 0.05$;

Fig. 3). In particular, at 24 months, the PM plots had an 8-fold reduction in the percentage of bare soil (class 3) compared to the C plots (Table 2). Thus, in the dry and wet seasons, the percentage of bare soil in the PM plots was always < 1.0%, indicating a reduction of over-land-flow. Moreover, the PM plots had greater percentages of sink areas (class 2, litter: 43.8% ± 22.2). Analysis of annual plants indicated no significant differences due to seasonal effects (Table 2; $p > 0.05$). However, the PM plots had greater spontaneous generation of plants. More specifically, Fig. 4 shows the PM plots had 10 individuals of *Hedypnois rhagadioloides* L., 5 individuals of *Hyparrhenia hirta* L., 3 individuals of *Rosmarinus officinalis* L., 2 individuals of *Lobularia maritima* L., and 1 individual of *Limonium sinuatum* L. The PM plots also had significantly lower percentages of dead vegetation connected to soil comparing C plots (Table 2; $p < 0.05$). This is consistent with our observation of the increased survival rate of vegetation in the PM plots (Table 3).

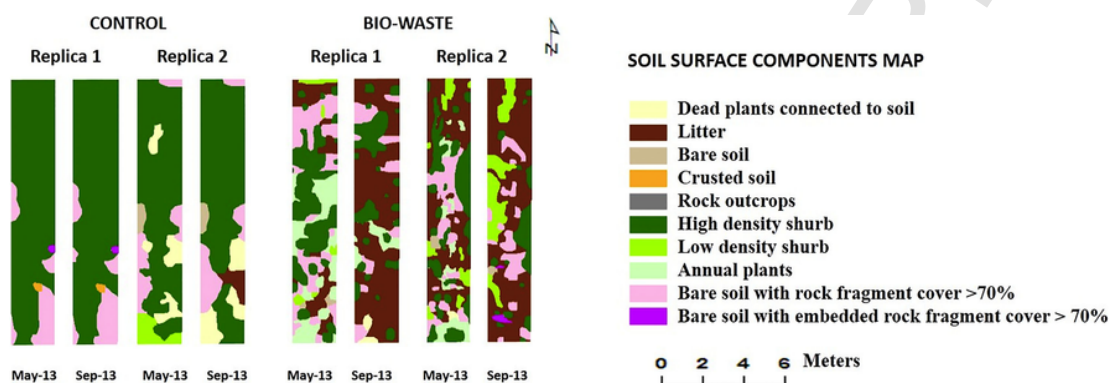


Fig. 3. Soil surface component maps from the two soil treatments in May 2013 and September 2013.



Fig. 4. Plots afforested and amendment with bio-waste (PM). From left to right: afforested plots treated with Aleppo pine debris (PM) (July 2013); *Hedypnois rhagadioloides* L. (May 2013); *Limonium sinuatum* L. (February 2013); *Lobularia maritima* L. (March 2013).

Table 3

Percentage of plants that survived in afforested control plots (C) and afforested plots treated with Aleppo pine debris (PM) during 2011–2014.

Month	<i>Lavandula stoechas</i> L.		<i>Lavandula dentata</i> L.		<i>Lavandula multifida</i> L.		<i>Rosmarinus officinalis</i> L.		<i>Thymus capitatus</i> L.		
	Mean	SD ±	Mean	SD ±	Mean	SD ±	Mean	SD ±	Mean	SD ±	
0	C (%)	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0
	PM (%)	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0	100.0	0.0
6	C (%)	58.1	11.7	62.5	17.7	91.6	11.8	100.0	0.0	56.5	8.8
	PM (%)	100.0*	0.0	100.0*	0.0	100.0	0.0	95.8*	5.9	100.0*	0.0
12	C (%)	58.1	11.7	50.0	0.0	83.3	0.0	75.0	0.0	56.5	8.8
	PM (%)	100.0*	0.0	100.0*	0.0	100.0*	0.0	95.8*	5.9	100.0*	0.0
20	C (%)	58.1	11.7	37.5	17.7	83.3	0.0	75.0	0.0	56.5	8.8
	PM (%)	100.0*	0.0	87.5*	17.7	100.0*	0.0	95.8*	5.9	100.0*	0.0
30	C (%)	58.1	11.7	37.5	17.7	74.95	11.8	75.0	0.0	56.5	8.8
	PM (%)	100.0*	0.0	87.5*	17.7	100.0*	0.0	95.8*	5.9	100.0*	0.0

* Significant difference relative to the control ($p < 0.05$).

Our examination of seedling survival indicated greater survival rates for all species in the PM plots (Table 3; $p \leq 0.05$). The highest survival rates were for *Lavandula dentata* and *Thymus capitatus*, which were $> 40\%$ greater in the PM plots than in the C plots (Table 3). Fig. 5 shows the soil profiles for the PM and C plots. A visual analysis of the profile shows that the PM plots had notable increases in macro-porosity. This is because during the initial process of afforestation of these plots, the soil was plowed to a depth of 25 cm so the bio-waste could be mixed into the soil. The survival of seedlings is strongly affected by the type and amount of porous soil (Cerdà, 1998; Hueso-González et al., 2016). The distribution and size of pores affect the air-water-soil ratios of the profile and the availability of water and nutrients (Ding et al., 2016; Otalvaro et al., 2016). Consequently, soils with greater porosity have greater infiltration and provide better support for seedling growth (Ellises et al., 2003). The addition of the forest residue to the PM plots increased the sink areas (Fig. 3) and also increased infiltration (Jordán et al., 2010). These two factors, along with the characteristically low mineralization rates for SOC (Table 1), which retard decomposition processes, increased the contact time of water with the soil so that macrochannels and macropores develop, water infiltration increases, and transplantation stress decreases. Thus, the use of forestry clearance debris in Mediterranean mountainous areas is an effective method to promote vegetation recovery, because it increases infiltration and the successful establishment of vegetation.

3.3. Effects on hydrodynamics

Many macrochannels formed in the soil of the PM plots ($\geq 50.0 \mu\text{m}$; Fig. 5), so the total sink area increased in these plots (C = 62.4%; PM = 83.59%; Fig. 3 and Table 2). This could have important implications for the hydrodynamics of this ecosystem. During the study period, there were 60 rainfall events and 1084.4 mm of total precipitation (Fig. 2). The rainfall threshold needed to activate overland flow is the same for the PM and C plots. Thus, the overland flow generation threshold is defined according two rainfall criteria (Fig. 6): a minimum rainfall depth of 10.6 to 15.0 mm and a minimum rainfall intensity (I_{15}) of 4.8 to 8.8 mm h^{-1} . Although this threshold

was in the same range for the PM and C plots, once the runoff started, the hydrological and erosive responses were very different in the PM and C plots, despite their similar exposures to rain (Fig. 6).

Fig. 2 shows that the climate in our study area is characterized by prolonged periods of drought, and intense and irregular rainfall events (Horton, 1933; Ferreira et al., 2000; Calvo-Cases et al., 2003, 2005). From a hydrological perspective, this precipitation pattern means that the rainfall intensity will usually exceed soil infiltration capacity, and there will be huge amounts of runoff and soil loss in the C plots, according to the Horton overland flow model (Horton, 1933; Ferreira et al., 2000; Calvo-Cases et al., 2003, 2005; Puech and Chabi-Gonni, 1984; Rao et al., 1998; Beven, 2002; Stomph et al., 2002; García-Ruiz et al., 2013). In fact, Fig. 6 shows that the C plots had a very rapid overland flow response, consistent with the Horton overland flow model. Fig. 6 shows that the trend of the C accumulated overland flow line is indicating that overland flow was always generated by rainfall excess (Lavee et al., 1998; Ward and Robinson, 2000; Beven, 2002; Calvo-Cases et al., 2003; Cammeraat, 2004; Latron et al., 2007). This was particularly significant in high intensity rain events, such as those on September 30, 2012 and November 18, 2012.

The PM plots had overland flow and sediment yield responses that were the opposite to those in the C plots (Fig. 6). In particular, the PM plots had smoother curves for accumulation of overland flow. This means that overland flow occurred only when the soil profile is saturated (soil saturation model) (Lavee et al., 1998). During erosive rain events, the increased sink area (litter, rock embedded rock fragment cover of $> 70\%$, and scrub surface) in our PM plots increased water infiltration (Ruiz-Sinoga and Martínez Murillo, 2009; Cerdà et al., 2016). Moreover, the formation of macro-pores or cracks in the soil increased infiltration rate and reduced overland flow and sediment yield (Hueso-González et al., 2015). This is consistent with our observation at the end of the study period, that the accumulated sediment yield was 179.8 kg in the C plots but was only 3.3 kg in the PM plots (Fig. 6). This implies that the addition of bio-waste to the PM plots reduced soil loss by 98.2%. Thus our results suggest that the use of vegetation forest residue (bio-waste) has positive effects in con-



Fig. 5. Soil profile for soil afforested and not amended (C) and soil afforested and amended with clearance residues from a close Aleppo pine (*Pinus halepensis* Mill.) forest in January 2014. In each image the checkered square is 1 cm on each site.

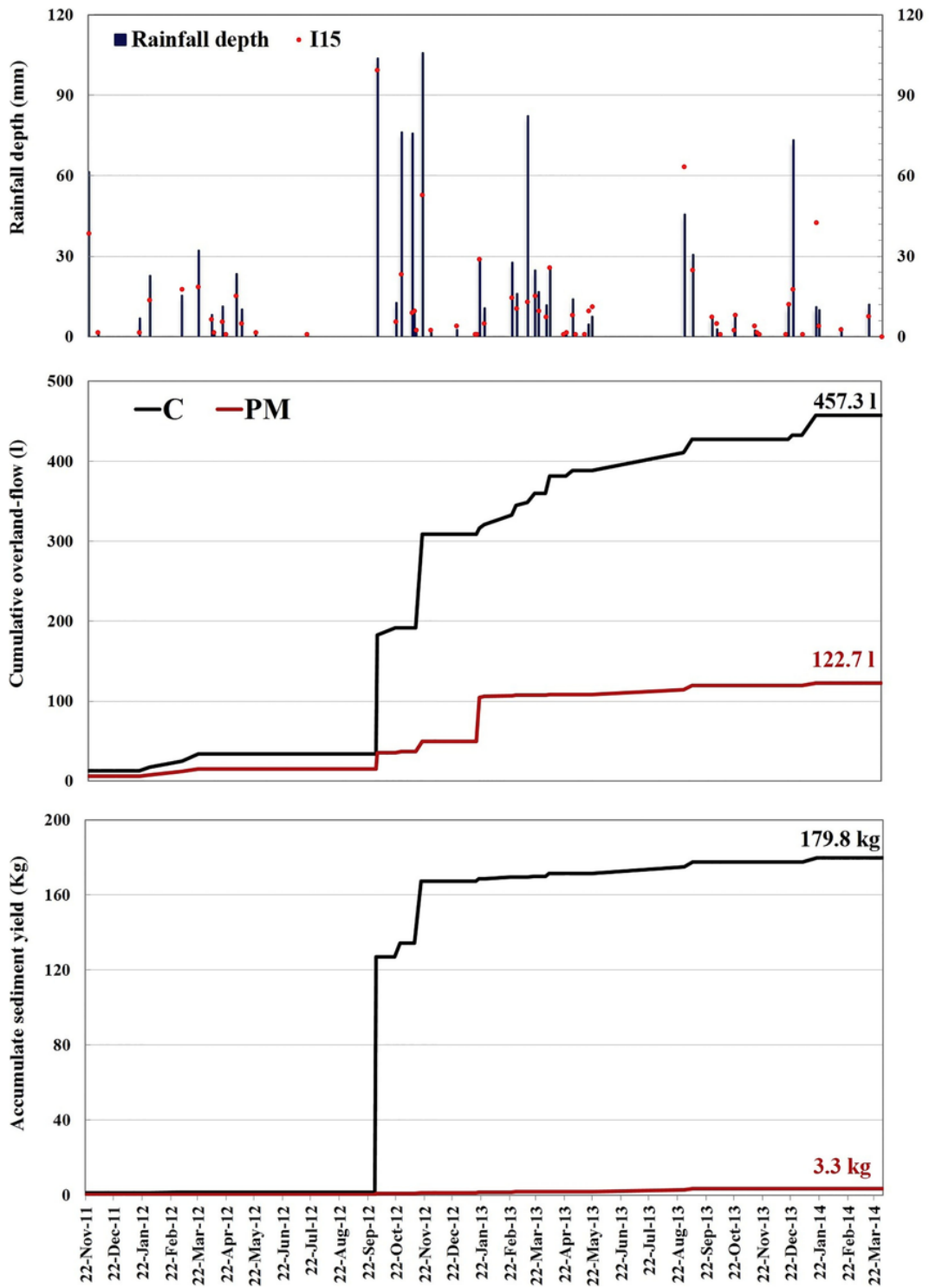


Fig. 6. Rainfall depth (mm), maximum intensity in 15 min (I_{15} , mm h⁻¹), cumulative overland-flow (l), accumulated sediment yield (kg). Where: C: soil afforested and not amended; PM, soil afforested and amended with clearance residues from a close Aleppo pine (*Pinus halepensis* Mill.) forest.

trolling overland flow and sediment yield production in this Mediterranean mountainous forest.

4. Conclusions

Forest managers commonly use vegetation clearance in most of the Mediterranean mountainous forests to prevent summer forest fires. Based on the European Circular Economy Strategy framework, the present study examined the effect of recycling of this forest residue as a soil amendment to enhance ecosystem productivity and vegetation cover, to control soil loss, and to reduce the economic and environmental costs required for bio-waste management. The recycling of this bio-waste promotes the opening of macro channels, and thereby increases water infiltration. This intervention provided important benefits to the hydrodynamics of this ecosystem. More specifically, there was a significant increase in SOC two years after the addition of 10 Mg ha⁻¹ of clearance residues from a *Pinus halepensis* Mill. forest. The low decomposition rate of the bio-waste led to maintenance of macrochannels in the soil during the study period. This alteration of soil structure led to greater seedling survival rates and greater growth of spontaneous vegetation, which in turn had positive roles in controlling overland flow and sediment yield production. Our results indicate that bio-waste application reduced soil loss by 98.16%. Apart from the systemic benefit to this Mediterranean forest, the reuse of vegetation debris also reduces waste production, as specified by the European Circular Strategy.

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