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Research article

Weed cover controls soil and water losses in rainfed olive groves in Sierra de Enguera, eastern Iberian Peninsula

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ABSTRACT

Soil erosion is a threat for the sustainability of agriculture and severely affects the Mediterranean crops. Olive groves are among the rainfed agriculture lands that exhibit soil and water losses due to the impact of unsustainable practices such as conventional tillage and herbicides abuse. To achieve a more sustainable olive oil production, alternative, greener crop management practices need to be tested in the field. Here, a weed cover (CW) treatment is tested at an olive tree plantation that has undergone conventional mechanical tillage for 20 years and results were compared against an adjacent control plantation that maintained tillage as a weed control strategy (CO). Both plantations were under the same tillage management for centuries and macroscopic analysis confirms they are otherwise comparable. Compared to the CO, where tilled soil cover was zero, 20 years of CW (weeds cover 64%; litter cover 5%) had led to significantly higher values of soil bulk density and soil organic matter. Results from rainfall simulation experiments at 55 mm h^{-1} on 0.25 m^2 plots under CO (N = 25) and CW (N = 25) show that as a result of the improved soil structure, CW (i) reduced soil losses by two orders of magnitude (140 times), (ii) decreased runoff yield by one order of magnitude (from 2.65 till 27.6% of the rainfall), (iii) significantly reduced runoff sediment concentration (from 18.6 till 1.43 g l^{-1}), and (iv) significantly delayed runoff generation (CO = 273 s; CW = 788 s). These results indicate that weed cover is a sustainable land management practice in Mediterranean olive groves and promotes sustainable agriculture production in mountainous areas under rainfed conditions, which are typically affected by high erosion rates such those found in the CO plots. Due to the spontaneous recovery of plant cover, we conclude that weed cover is an excellent nature-based solution to increase in the soil organic matter content and soil erosion reduction in rainfed olive orchards.

1. Introduction

Soil erosion is a threat to the sustainable production in agricultural land due to the loss of nutrients, soil particles, water, and seeds (Keesstra et al., 2021). Soil erosion, among other soil degradation processes (Núñez-Delgado et al., 2020), is a major threat to the soil system of the Mediterranean ecosystem as it depletes the already meagre organic matter of the upper soil layer. Agriculture can result in negative environmental impacts due to soil erosion acceleration as a consequence of aggressive managements such as herbicides (Liu et al., 2016), tillage (Zhang et al., 2013), or soil compaction due to the widespread use of heavy machinery (Lima et al., 2019; Yao et al., 2019). A review of the current soil erosion rates in agriculture, reveals values that are several orders of magnitude higher than the soil formation rate (Chalise et al., 2019; García-Ruiz et al., 2017; S. D. Keesstra et al., 2019a, 2019b; Novara et al., 2019b), thus inducing land degradation and desertification (Briassoulis, 2019; Jucker Riva et al., 2017). Soil erosion assessment is a key instrument when pursuing sustainable production within an integrated and holistic approach in agriculture development (Bastianoni et al., 2001; La Rosa et al., 2008). To achieve sustainable or tolerable soil erosion rates, proper management should be stablished both in agriculture (Mattsson et al., 2000) and livestock production (Cederberg and Mattsson, 2000).

Olive (*Olea Europaea*) is a Mediterranean native crop with 7000 years since domestication. Olive, along with wheat and grapes, is one of the Mediterranean triad products that were widespread along the

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Mediterranean basin where wild olive (oleaster) is indigenous (Culham and Greene, 1988). After its spread by the Phoenicians, the Greeks, and the Roman Empire, the use of olive oil has been identified as part of the Mediterranean culture. Within the landscape, olive plantations characterize the traditional rainfed agriculture production in the Mediterranean. Today, Spain is the largest producer of olive oil with 6.6 million Mg y^{-1} and with the largest cultivated area, 2.6 million ha. Sixty percent of the world production is cropped in the fields of the Mediterranean basin, where olive oil is a basic nutrition product (Vossen, 2007). Olive fields and olive oil are part of the Mediterranean culture and some authors identify the Mediterranean with the land where the olive can be cropped. Loumou and Giourga (2003) claim that the life and the identity of the Mediterranean is found in the olive groves. Nobel Prize laureate Odysseas Elytis wrote that “With an Olive Tree, a Vineyard and a Boat, You Can Rebuild Greece”. Olive and the Mediterranean are perceived as twins by the society.

The traditional rainfed olive production involves intense and repetitive tillage to suppress weeds and reduce or control their competition for nutrients and water against the olive. This traditional abuse of tillage results in bare soils and then in higher soil erosion, that leads to soil degradation and a gradual loss in crop production. Researchers have already warned that such soil erosion rates are not sustainable. The research carried out by José Alfonso Gómez in Andalusia insists on the high erosion rates induced by steep slopes, herbicide abuse, intense tillage, and lack of cover crops, and the need to find sustainable management practices that promote soil ecosystems services (Gómez et al, 2006a, 2006b, 2014). In the historical perspective study of Vanwalleghem et al. (2011), it is shown that soil erosion in olive orchards is not only a contemporary issue but has been a persistent historical problem in the Mediterranean, at least since the 18th century (Amate et al., 2013; Marathianou et al., 2000; Vanwalleghem et al., 2010). Moreover, a review of the State-of-the-Art of soil erosion in the Mediterranean olives shows that the problem affects all regions and staple crops across the Mediterranean. Most of the research on soil erosion in olive plantations is carried out in Andalusia (Calderon et al., 2016; Rodríguez-Lizana et al., 2008; Taguas et al., 2009). However, high soil erosion rates are also measured in olive groves in central Iberian Peninsula (Sastre et al., 2017), Crete (Kairis et al., 2013; Karydas et al., 2009), and Eastern Spain (Rodrigo-Comino et al., 2017), and we still know little about the impact of cropping olives in other regions of the Mediterranean belt or other global producers such as United States, Brazil, and Argentina.

While soil erosion research in olive plantations is mainly experimental, modelling approaches are also present and necessary for management planning at larger scales (e.g. Panagos et al., 2015). Nevertheless, experimental research carried out at plot scale allows measurement of water and soil losses with accuracy and therefore the comparison of the impact of different management practices (Taguas et al., 2010; Novara et al., 2021). The plot approach is connected to the impact of gullying in soil erosion, and this is an emerging topic as the highest erosion rates are found in gullied areas (Taguas et al., 2012; Amare et al., 2019). The use of rainfall simulation experiments is also a source of information about the peak of erosion delivered during low frequency high magnitude rainfall events as demonstrated by Rodrigo-Comino et al. (2018). However, soil erosion monitored on plots was the source of information that contributed with direct information from natural rainfall events (Espejo-Pérez et al., 2013; Francia Martínez et al., 2006) and enriched the knowledge about the management impact on soil and water losses. The research conducted on soil erosion plots in Andalusia has also been complemented with the use of magnetic iron oxide tracers (Guzmán et al., 2013).

The impact of tillage in Mediterranean crops is a threat to the sustainability of the agricultural production in rainfed Mediterranean crop production. The negative impact of this agriculture management is found in other crops, too. For example, Novara et al. (2011) found high erosion rates due to tillage in the Sicilian vineyards. Within the seven management practices assessed by Novara et al. (2011), soil erosion rates reached 85 Mg ha^{-1} y^{-1} with conventional tillage in 2005, meanwhile other managements always yielded lower than 60 Mg ha^{-1} y^{-1} and sometimes lower than 20 Mg ha^{-1} y^{-1} upon the different types of cover crops. Keesstra et al. (2016) concluded that soil erosion rates are non-sustainable under tillage and herbicide treatments in apricots plantations is Eastern Spain. Soil erosion was extremely high in herbicide-treated plots with 0.91 Mg ha^{-1} h^{-1} of soil lost; in the tilled fields erosion rates were slightly lower with 0.51 Mg ha^{-1} h^{-1}. On the other hand, covered soils under organic farming management on apricot orchards showed an erosion rate of 0.02 Mg ha^{-1} h^{-1}. This is 46- and 26-times lower soil losses in organic weed covered managed soils in comparison to herbicide and tillage, respectively. Soil erosion measurements with plots under natural rainfall or under rainfall simulation experiments provide information at short-term, and soil erosion needs also to be assessed at medium- and long-term periods. The use of methods such as Improved Stock Unearthing Method (ISUM) provides long-term data (Rodrigo-Comino and Cerdà, 2018). Barrena-González et al. (2020) confirmed that after 20 years, high erosion losses were measured in vineyards in Extremadura under intensive tillage. ISUM calculated 45.7 Mg ha^{-1} y^{-1} in average whereas the Universal Soil Loss Equation (USLE) estimated 17.4 Mg ha^{-1} y^{-1}. Other crops are affected by the mismanagement of removal of the plant cover. Bayat et al. (2019) also found high erosion rates in persimmon plantations in Spain, with as much as 50 Mg ha^{-1} y^{-1} measured using ISUM. Raya et al. (2006) found extremely high erosion rates in almonds in Andalusia due to the bare soils and intense tillage, but also found that using thyme as a cover crop could reduce soil losses by 97%. This is a general trend found in different crops and research sites along the Mediterranean: high erosion rates that can be controlled using soil conservation strategies (Battany and Grismer, 2000; Casalí et al., 2009). This is a positive information as high erosion rates are causing environmental and economic damages (Panagos et al., 2018) that need to be controlled.

Here we use a simulated rainfall approach to determine the impact of weeds as a cover crop to reduce soil losses in olive groves. Rainfall simulation experiments allow an accurate measurement of the runoff and sediment delivery and the repetition of measurements to achieve a dataset to quantify the impact of the management. Moreover, the measurements can be done under the same weather conditions (season) and under similar soil conditions to avoid temporal variability in soil moisture and vegetation cover. Furthermore, rainfall simulation experiments are carried out at high magnitude – low frequency rainfall events, which allows to research how the extreme events, the ones that induce most of the runoff and soil erosion, and determines the annual soil losses. Rainfall simulators reduce the cost, the experimental period, and increase the accuracy of the measurements.

The objective of this research is to quantify the impact of weeds to control the soil erosion rates in an olive grove. A long-term approach (20 years) will allow to calculate the mean annual benefit to use weeds in olive rainfed agriculture.

2. Materials and methods

2.1. Study area

The Sierra de Enguera (El Teularet) Soil Erosion and Degradation Research Station (Figure S1 and S2) was established in 2002 to monitor soil and water losses in different crops and rangelands by means of plots, soil sampling, and rainfall simulators (Cerdà et al., 2017, 2018; Cerdà and Rodrigo-Comino, 2021). El Teularet is located in Sierra de Enguera in the southwest Valencia province (Eastern Spain, 750 m a.s.l., 38° 55′ N, 00° 50′ W) and was selected as representative of the Mediterranean traditional agriculture crops and rangelands. The climate shows a mean annual temperature of 12.7 °C. January is the coldest month (9.8 °C) and August is the warmest (25.7 °C). Mean annual rainfall is 540 mm with a typical Mediterranean summer drought.

Two paired study sites were selected within the El Teularet study area (Figure S1). The control (CO) study site (38°55′55.24″N; 0°49′46.43″W) is under tillage, while the weed cover (CW) study site (38°55′21.55″N; 0°50′26.35″W) has not been tilled for 20 years. In both study sites, olive crops (Blanqueta variety) are cultivated under the Valencia Organic Farming Committee (Comité d'Agricultura Ecológica de la Comunitat Valenciana) rules. Soils at the study sites are Typic Xerorthents developed over Cretaceous marls (Cerdà et al., 2018, 2021a). Soil texture at both study sites and at two depths shows no significant differences, with the exception of silt content sampled at 4–6 cm (Figure S1; Table S1). Silt content at 4–6 cm in the control (CO) study site was 44.48% (IQR = 42.15–46.23), whereas in the weed cover (CW) it was slightly lower at 41.13% (IQR = 39.13–43.2). Most samples classified as loam, with only three (3% of all samples) classifying as clay loam from different points and depths of both study site (Fig. 1). The similarity of soil texture across the two treatments, in addition to all other environmental and historical management similarities, allows for further comparison.

2.2. Experimental layout

In both study sites, olive crops were planted in 1997 and the experiments were carried out in August 2017, during the dry season, to avoid spatial and temporal variability of soil moisture. In each study site, 25 plots were established along two representative inter-rows, a least 2 m apart from each other. One rainfall simulation experiment was carried out per plot. The sampling strategy along the two tested treatments is shown in Figure S1. The CO treatment was tilled four times per year (March, May, July, and November). Each April, June, and August, the CW treatment was treated with a flail mower attached to a tractor (Figure S2).

2.3. Soil and surface sampling and analysis

Soil sampling took place at each of the 50 research plots, 25 plots for control (CO) and 25 for weed cover (CW). Samples were taken at 0–2 and 4–6 cm depth with a ⌀6 cm ring. Plant, litter, rock fragment, and bare soil cover were measured prior to the rainfall simulation experiments and were determined by measuring 100 points regularly distributed at each 0.25 m^2 plot. Grain size, soil moisture, and organic matter and bulk density was determined from the samples collected in August 2017. Soil moisture and organic matter and bulk density were measured in the lab from the samples collected during the experimental period of August 2017. The pipette method was used to determine grain size (Deshpande and Telang, 1950). Bulk density was measured using the core ring method. Soil organic matter was measured by means of the Walkley-Black method (Amate et al., 2013). Soil moisture was determined by the desiccation and measured in all the 100 samples: 2 depths × 2 managements × 25 plots.

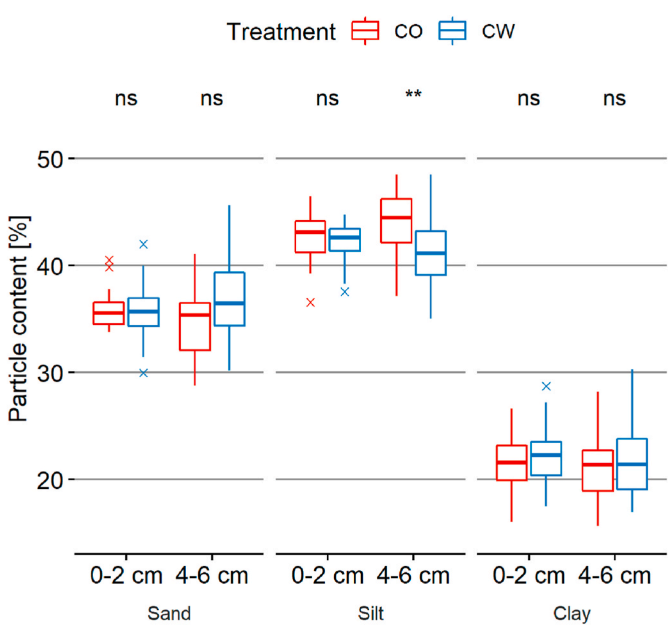

Fig. 1. Sand, silt, and clay percent content of soil samples taken from two sampling depths (0–2 and 4–6 cm) in the Control (CO) and Weed cover (CW) treatments. N.s. denotes no significance whereas ** denotes significance only at the $p < 0.01$ level. Outliers are denoted with ×.

2.4. Rainfall simulation experiments

The experimental setup for the rainfall simulation experiments was carried out during the sampling period of August 2017. It involved 50 rainfall simulation experiments (Cerdà et al., 2020) at rainfall intensity of 55 mm h^{-1} for 1 h over circular paired plots with an area of 0.25 m^2. Thus, on each plot a total of 13.75 l were precipitated in the course of 1 h. At each plot, runoff was collected at 1-min intervals and water volume was measured. The runoff coefficient was calculated as the percentage of rainfall water leaving the circular plot as overland flow. Runoff samples were desiccated (105 °C for 24 h) to determine the runoff sediment concentration. The sediment yield was calculated upon the runoff discharge and the runoff sediment concentration. The soil erosion rates were then converted to soil loss per area and time [Mg ha^{-1} h^{-1}].

During the rainfall simulation experiments, time to ponding (time required for 40% of the surface to be ponded) Tp [s], time to runoff initiation Tr [s], and time required by runoff to reach the outlet Tro [s] were recorded. These parameters show when the soil ponded, when the runoff was initiated, and when it reached the plot runoff collector. Tr-Tp [s] and Tro-Tr [s] were also calculated, and they indicate how quickly ponding is transformed into runoff and how much time is required for runoff on the soil surface to reach the plot outlet. Shorter periods of time imply lower infiltration rates, lower permeability, higher surface connectivity, and lower surface roughness. These parameters are indicators of the hydrological connectivity at the plot scale and have been successfully used to assess the runoff dynamics at pedon scale (Cerdà et al., 2020). Runoff coefficient Rc [%] was calculate following the equation:

$$Rc = \frac{Q}{V} 100\% \tag{1}$$

where Q [mm] is the total runoff from the plot and V [mm] is the total volume of water precipitated [mm] on the plot during the experiment. Sediment yield Sy [g] was calculated following the equation:

$$Sy = R \times Sc \tag{2}$$

where Sc [g l^{-1}] is the concentration of the sediment in the runoff and R [l].

Soil erosion rate was calculated in g m^{-2} h^{-1} and Mg ha^{-1} h^{-1} following the equation:

$$Se = \frac{Sy}{A} \tag{3}$$

where A [m^2] is the plot area, here equal to 0.25 m^2.

2.5. Statistical analysis

Measurement distributions did not always fit the normal distribution under the test Saphiro-Wilk, therefore medians together and interquartile ranges (IQR) are reported. In all cases, the non-parametric Wilcoxon test is used to assess statistical significance. Results are reported in boxplots with suspected outliers denoted with ×, significance is denoted in standard star code at different levels (i.e., *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$, ****$p < 0.0001$) and n.s. stands for no significance. Statistical analysis and plotting were conducted in R (R Development

Core Team, 2017) using packages *ggplot2* (Wickham, 2016) and *ggpubr* (Kassambara, 2020), and code is available on request.

3. Results

3.1. Soil cover

Regarding soil cover, in the control plot (CO) all plant, litter, and stone cover had a median value of 0. Stone cover was also negligible in the weed cover (CW) plot, with no significant difference from that of CO (Fig. 2). However, plant, and litter cover in CW were significantly higher, at 64% (IQR = 58–66) and 5% (IQR = 4–8), respectively.

3.2. Soil properties

Soil organic matter (SOM), bulk density (BD), and soil water content (SWC) were used to quantify the effect of the CW treatment on soil properties. Results are shown in Fig. 3 (also Table S2). SOM was significantly higher in the CW plot than in the CO plot, both at the upper (0–2 cm) and lower (4–6 cm) soil layer. Especially in the upper layer, median SOM of CW was over 100% higher than that of CO with respective values being 2.08% (IQR = 2.02–2.31) and 1.03% (IQR = 0.99–1.05), respectively (Fig. 3). BD was also significantly higher in CW, both in the upper (1.42 versus 1.23 g cm^{-3}) and lower (1.43 versus 1.26 g cm^{-3}) soil layer (Fig. 3b). Finally, SWC was significantly lower in both soil layers of the weed cover plot, especially in the upper layer where CW had 5.78% (IQR = 5.409–5.994) versus 3.48% (3.26–3.74) in the CO (Fig. 3c).

3.3. Runoff initiation

Rainfall simulation experiments allow assessment of the runoff formation and direct comparison between treatments due to the identical rainfall characteristics. Time to ponding (Tp), time to runoff (Tr), and time to runoff outlet (Tro) were determined in each plot (Fig. 4 and Table S3). In all cases differences were significant. In the control plot (CO), time to ponding was 86 s (IQR = 81–88), and 104 s (96–114) later runoff begun and after 81 s (IQR = 61–100) runoff had reached the outlet of the plot. The entire time required from the onset of rainfall to runoff at the outlet of the control plots was 266 s (IQR = 254–293). On the other hand, in the weed cover (CW) treatment, time to ponding was 158 s (IQR = 153–161), and 236 s (221–255) later runoff begun and

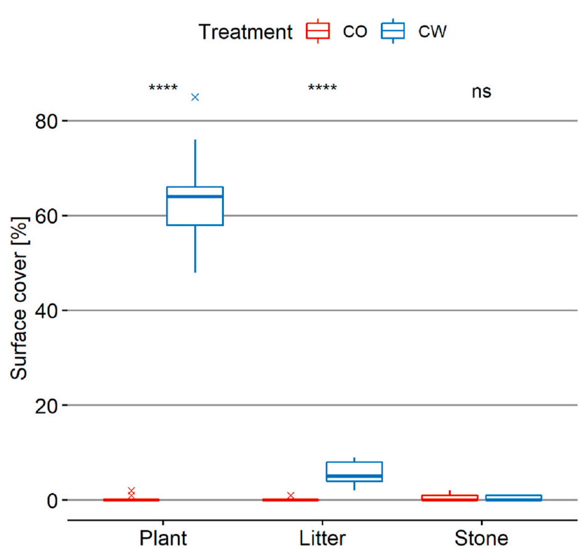

Fig. 2. Plant, litter, and stone cover [%] of plots in the Control (CO) and Weed cover (CW) treatments. **** denotes significance at the $p < 0.0001$ level (****) and n.s. denotes no significant difference. Outliers are denoted with × .

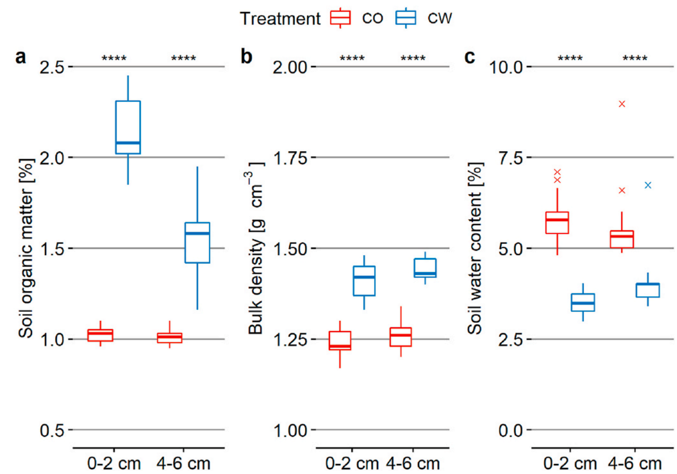

Fig. 3. Soil organic matter [%] (a), bulk density [g cm^{-3}] (b), and soil water content [%] (c) of soil samples taken from two depths (0–2 and 4–6 cm) in the Control (CO) and Weed cover (CW) treatments. All differences between treatments are significance at the $p < 0.0001$ level (****). Outliers are denoted with × .

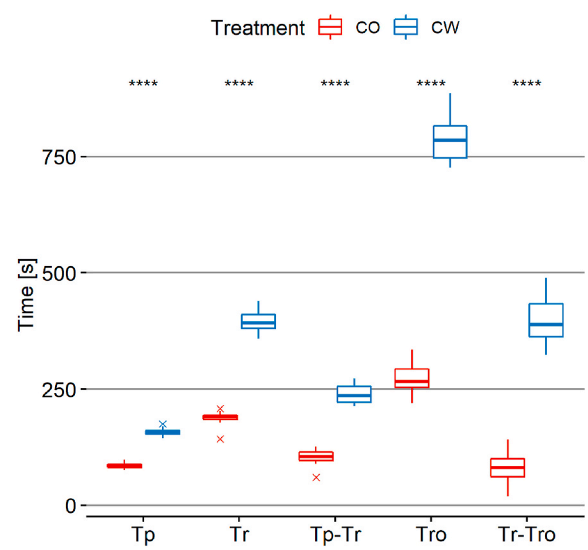

Fig. 4. Time to ponding (Tp), Time to runoff (Tr), and Time to runoff outlet (Tro), as well as differences Tp-Tr and Tr-Tro (all measured in s) in the Control (CO) and Weed cover (CW) treatments. All differences between treatments are significance at the $p < 0.0001$ level (****). Outliers are denoted with × .

after 389 s (IQR = 363–433) runoff had reached the outlet of the plot. The entire time required from the onset of rainfall to runoff at the outlet of the CW plots was 786 s (IQR = 747–816), almost 3 times as much as for CO.

3.4. Soil and water losses

Runoff coefficient (Rc), sediment concentration (Sc), and soil erosion rate (Se) are used to quantify the effect of the treatment on soil and water losses occurring during the rainfall simulation experiments (Fig. 5 and Table S4). In all cases differences were significant at the $p < 0.0001$ level (Fig. 5). Runoff coefficient in the CO plots was over an order of magnitude higher than that of the CW, with median values being 27.14% (IQR = 23.54–29.14) and 2.24% (IQR = 2.01–3.31), respectively (Fig. 5a). Sediment concentration [g l^{-1}] in the CO plots was also an order of magnitude higher than that of the CW, with median values being 17.69 [g l^{-1}] (IQR = 16.58–19.68) and 1.40 [g l^{-1}] (IQR =

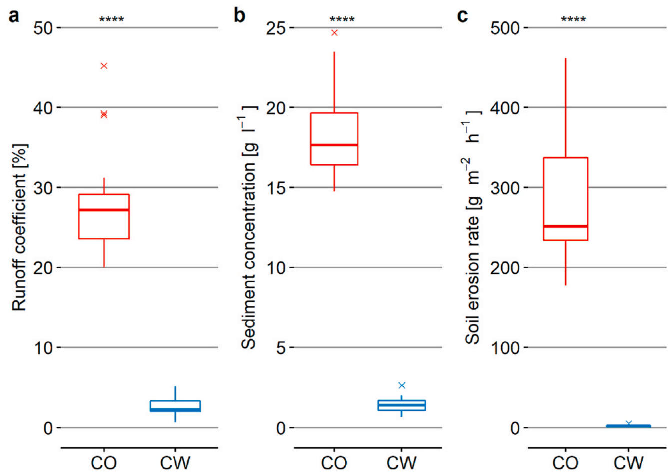

Fig. 5. Runoff coefficient [%] (a), sediment concentration [g l^{-1}] (b), and soil erosion rate [g m^{-2} h^{-1}] (c) in the Control (CO) and Weed cover (CW) treatments. All differences between treatments are significance at the $p < 0.0001$ level (****). Outliers are denoted with × .

1.07–1.67), respectively (Fig. 5b). Soil erosion rate in the CO plots was over two orders of magnitude higher than that of the CW, with median values being 250.97 g m^{-2} h^{-1} (IQR = 233.80–337.09) and 1.97 g m^{-2} h^{-1} (IQR = 1.24–2.36), respectively (Fig. 5c). According to the results, when upscaling soil erosion rates from plot to field scale CW treatment has negligible rates whereas CO produces a median of 2.51 Mg ha^{-1} h^{-1} (IQR = 2.34–3.37).

4. Discussion

The fifty rainfall simulation experiments conducted in August 2017 at the Sierra de Enguera (El Teularet) Soil Erosion and Degradation Research Station demonstrated that the long-term use of weeds as a cover crop transformed the marly soils from highly erodible to stable soils under the water erosion process due to changes in soil properties.

4.1. Soil organic matter

Confirming previous findings on the effect of cover crops and residue on SOM (Chalise et al., 2019; Martins et al., 2015), the CW treatment led to an increase of soil organic matter. As documented in the literature, organic matter benefits a wide range of soil functions and services such as arthropod fauna presence and diversity (Bàrberi et al., 2010), and development of better soil infiltration conditions which is fundamental to increase the soil water retention capacity and reduce the soil losses. Here, we found that CW led to an annual average increase of organic matter at 0–2 cm was 0.05% y^{-1} (from 1.03% to 2.08% in 20 years). Assuming uniform distribution of SOM in the topsoil, this increase is moderate compared to the change observed at a nearby irrigated citrus plantation by Novara et al. (2019a). In their study, Novara et al. (2019a) reported an increase of 0.26% y^{-1} (from 1.18% to 5.53% in 21 years). The relatively lower increase in SOM observed here can be attributed to the rainfed conditions that restrict the plant biomass production, while in citrus farms irrigation promotes the development of weed biomass and the subsequent accumulation of litter (Fig. 6a).

4.2. Bulk density

The non-linear relationship of SOM concentration on soil bulk density has been well documented and can be modelled with an exponential equation (Ruehlmann and Körschens, 2009). Here we confirm the general tendency of soils with higher SOM to have lower BD but also attribute this decrease in the absence of tillage in the CW treatment.

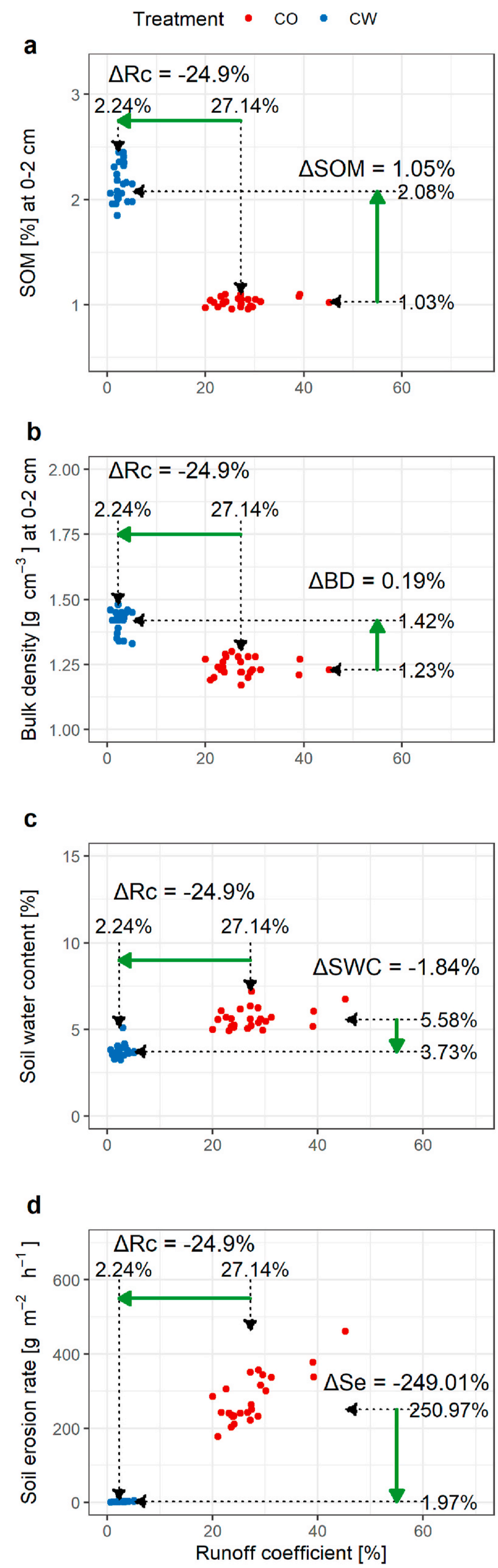

Fig. 6. Relation of runoff coefficient [%] at treatments CO and CW and differences of median values versus (a) soil organic matter content [%] sampled at 0–2 cm, (b) soil bulk density (0–2 cm depth), (c) mean soil moisture from 2 depths [%], and (d) soil erosion [g m^{-2} h^{-1}], over the 20 years of experimental data.

Indeed, similar results were also reported by Carter (1990) who found a bulk density of 1.34 ± 0.06 Mg m^{-3} for ploughed loam and 1.40 ± 0.03 Mg m^{-3} for direct drilling/seeding, as well as Lampurlanés and Cantero-Martínez (2003) who found a mean bulk density (0–120 cm depth) of 1.34 g cm^{-3} under no tillage and 1.22 g cm^{-3} under tillage.

Under such changes in bulk density, one would intuitively expect a reduction of soil water capacity and infiltration processes. Nevertheless, in agricultural soils, macropores rather that total porosity is the major determinant of soil saturated hydraulic conductivity. Weed cover and absence of tillage promotes the development of macropores, either due to soil fauna (Green and Askew, 1965), roots (Hatano et al., 1988; Mitchell et al., 1995), or cracks (Mitchell et al., 1995; Novák et al., 2000), and thus higher infiltration rates. Tillage disturbs macropores continuity and therefore, in terms of soil water conservation, it is not beneficial drying the wetting phase. Under no-tillage with cover crops, Osunbitan et al. (2005) has also observed higher bulk density and hydraulic conductivity of loamy sand soils compared to 3 different tillage treatments. Cerdà et al. (2021b) found that the use of herbicides increases in 20% the soil bulk density in Saturn-peaches crops in Eastern Spain, which result in high erosion rates. Due to this effect, the increase in soil bulk density did not affect runoff discharge as shown in Fig. 6b. Here we demonstrated that although the soils increased the bulk density by 0.0085 gr cm^{-3} y^{-1} in twenty years, the runoff reduced by 1.2465% y^{-1} as a consequence of moving from a matrix flow under tillage management to a preferential flow after 20 years with cover crops. Sander and Gerke (2007) found the importance of preferential flows in paddy fields were dye tracer penetrated vertically via preferential pathways to depths from 94 till 120 cm while most of the matrix remained unstained. Biopores and cracks were the main cause of macropores, and they caused high spatial variability of hydraulic conductivity. Clothier et al. (2008) reviewed the impact of preferential flow impact on water transport in the soil and found a generalized impact on the infiltration capacity of soils such as we found in our experiments in Sierra de Enguera experimental farm. Some authors, such as Janssen and Lennartz (2008) demonstrated that the preferential flow through the hard pan due to earthworm burrows, root channels and shrinkage cracks induce higher infiltration rates that can reach the groundwater level. This confirm that at the experimental site of Sierra de Enguera, weeds can contribute to increase the infiltration rates, reduce runoff discharge, and soil loss although the soil bulk density increased.

Another key issue here is that the low bulk density of the soils under tillage (CO) changes due to the raindrop impact that contribute to the formation of a soil crust that induce low infiltration rates and high runoff discharges (Robinson and Phillips, 2001). This is why some farmers apply a shallow tillage to avoid the soil compaction and take advantage of the cover crops, however this shallow and less intense ploughing strategy induces higher erosion rates (Novara et al., 2019b).

4.3. Surface cover

Plant and residue cover are key parameters on direct infiltration due to the development of macropore flow, but also due to the induced delay in the surface runoff transport. Recently, Bombino et al. (2019) found that in olive groves in the Mediterranean with a cover of chipped pruned branches reduces the runoff rate on average by 30%, mainly because of the increased soil infiltration rates, and they mention that the retention of vegetal residues may be advisable to reduce surface runoff generation rates. We report the runoff generation with the measurement of the time from the runoff generated in the plot (Tr) till the runoff measured at the exit of the plot (Tro). This time until the runoff reaches the plot outlet is relevant to understand that the plant cover reduces the surface runoff velocity, and this increase the ponding and the infiltration rate. Here, for each 1% of vegetation cover results in 4.74 s of delay of the runoff, and this is an opportunity for the ponded water to infiltrate and contribute to macropore flow (Fig. 7). Similar conclusions have been reached with the use of indigenous grasses (Zhang et al., 2012), crop residue mulches (Li et al., 2019; Telak and Bogunovic, 2021), and plant litter (Sun et al., 2016) found when they incorporated plant litter in the Loess Plateau soils to reduce the velocity of the water flowing on the soil surface.

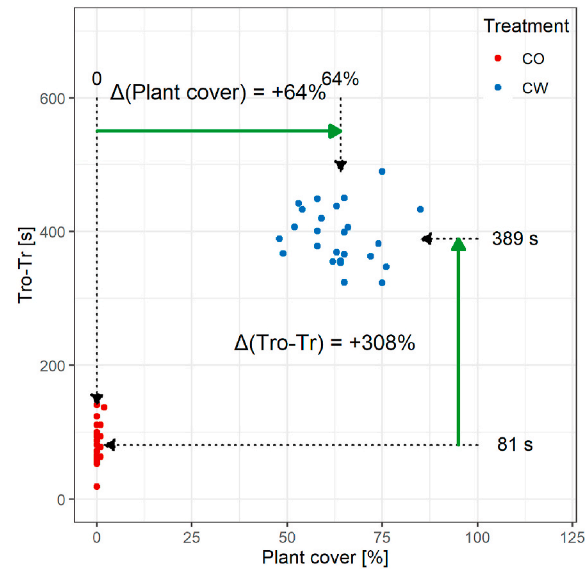

Fig. 7. Relation of Plant Cover [%] and Tro-Tr [s] per plot (Tillage and Weed cover) at treatments CO and CW and differences of median values.

4.4. Soil water

Tillage impacts on the soil moisture balance in two ways. As mentioned above, it disturbs macropores continuity and therefore, in terms of soil water conservation, it is beneficial during the drying phase since it hinders evaporation. Furthermore, under tillage management, top layer drying is controlled only by evaporation whereas, under weed cover, it also depends on plant transpiration which can be significant during the dry season (Schiller and Cohen, 1995). Here the measurements of soil moisture were conducted during a typically dry Mediterranean summer when soil moisture is at its minimum. The dataset informs about lower values in the CW treatment, both at 0–2 and 4–6 cm depth, than in the CO treatment. It is not clear whether the additional water stress due to weed cover can impact the development of the olive trees, but measurements show that the SWC difference is less pronounce in the deeper (4–6 cm) layer (Fig. 3c), therefore it is probably even less significant at the root zone. Regarding the effect of soil water on the runoff process, assuming that both CW and CO treatment have a similar soil water capacity, increased SWC present in CO contributes to shorter response times (Fig. 3 and Table S3) and a higher discharge (Fig. 3c).

4.5. Soil erosion

This strategy to reduce the tillage intensity (recurrence) has been used as an alternative to the traditional tillage. Turtola et al. (2007) applied in the Finnish clayey soils and they found that surface runoff increased with decrease in the tillage intensity due to the changes in soil roughness. Turtola et al. (2007) also found that the shallow tillage produced higher soil erosion rates (0.407–0.1700 Mg $ha^{-1}yr^{-1}$), still 48% higher than the no-tillage managed fields. This is why the authors concluded that in the flat clayey soils typical for southern Finland, tillage has a great influence on soil losses and conservation agriculture should be applied. Schillinger (2001) in the Pacific Northwest (USA) shows that long-term practice of minimum and delayed minimum tillage during fallow significantly increased surface residue and clod retention for erosion control with no adverse agronomic affects compared with conventional tillage. Both examples show that under cold and temperate climatic conditions shown a positive use of weed cover to reduce soil erosion. This is even more contrasted in the olive orchards in the Mediterranean where soils are bare when tillage is applied, and they show a permanent cover when they are under no-tillage management. Here the tested no tillage and weed cover treatment allowed the transformation of rainfed Mediterranean agriculture in a more ecosystem friendly land use. The treatment resulted in an average annual reduction of runoff by 1.24% and a reduction of soil losses by 0.20 Mg ha^{-1} y^{-1} during the 20 years covered by the measurements (Fig. 6d).

4.6. Rainfall simulation

Rainfall simulation experiments contributed with accurate information about the impact of weed cover on soil erosion and runoff. This method is typically applied under high rainfall intensities, to reproduce extreme rainfall event conditions, that are known to generate the highest runoff discharges and highest erosion rates. For example, from the rainfall events recorded in the universal soil loss equation (USLE) database, the top 10% in magnitude contribute 50% of all eroded soils, the 25 largest daily events deliver 46–63% of the load, and the 5 largest daily events delivered 23–39% of the sediments (González-Hidalgo et al., 2009). This observation is even more pronounced in semiarid ecosystems where extreme events are more frequent. For example, in the El Ardal research station, Murcia, López-Bermúdez et al. (1998) found that the extreme rainfall events were the most efficient to detach and transport soil material, and during a 9-year experiment Romero Díaz et al. (1998) found that the largest rainfalls events were the ones to determine the total soil losses. Since these determining extreme events take place at low frequency, rainfall simulation allows the collection of data that would otherwise require decades of observation to be collected within a few weeks.

4.7. Cover crops for sustainability

The high erosion rates in Mediterranean rainfed crops threaten the sustainability of their production and the ecosystem services they supply. At the global scale, the Mediterranean rainfed crops under conventional intensive tillage constitute a hurdle against the 2030 Sustainable Development Goals of the United Nations and the Land Degradation Neutrality challenge (Daliakopoulos and Keesstra, 2019; Keesstra et al., 2018a, 2018b; Visser et al., 2019) since erosion rates do not allow soil formation processes to restore soil losses. The intense tillage and bare soils result also in the loss of nutrients and induce a lost in the soil quality mainly due to the loss of organic matter and soil structural stability (Novara et al., 2019b; Ramos et al., 2010; Tejada and Benítez, 2020; Rodrigo-Comino et al., 2020a,b). Therefore, the need of innovative land management practices is required to reduce soil losses and contribute to the sustainability of Mediterranean agriculture (Daliakopoulos et al., 2019). Option such as the use of catch crops, mulches, and geotextiles are available to farmers, however they are expensive for low productivity and low turnover rainfed agriculture (López-Vicente et al., 2020). In fact, rainfed agriculture is threatened by land abandonment, which in turn affects soil and water yield, and degrades the traditional Mediterranean agricultural landscape (Nadal-Romero et al., 2019), therefore there is a need to find management practices that can reduce costs while increasing output value due to the ecosystem services sustainable agricultural soils can offer (Brady et al., 2019; Malherbe et al., 2019).

Here we test the effectiveness of weed cover as a sustainable land management practice for rainfed crops as it is the easiest to apply, has low cost, and is based on local resources. The combined advantages of the weed cover treatment, resulting to significant increase of soil organic matter and a simultaneous reduction of soil loss and plot runoff, have the potential to transform the rainfed agriculture in the Mediterranean in a more friendly ecosystem where soil and water are preserved as in a forest. We conclude that it is relevant to promote the use of weed cover as a sustainable management in Mediterranean olive plantations to achieve the Sustainable Development Goals of the United Nations. A recent state-of-the-art review shows that cover crops can also contribute to the conservation of the water resources when properly managed in semiarid ecosystems (Novara et al., 2021).

5. Conclusions

This work compares two study sites representative of tillage and weed cover by means of an experiment that was initiated in 1997 and finalized in 2017. Within twenty years, soil losses in the weed cover site reduced by two orders of magnitude due to the reduction in soil water losses and sediment concentration. Although tillage induced a lower soil bulk density, the increase in soil organic matter in the weed covered plots contributed to the reduction the runoff discharge and soil loss. We demonstrated that weed cover reduces soil and water losses in olive plantations and increased soil organic matter content, which is relevant to succeed with a sustainable management in Mediterranean olive plantations that will contribute to achieve the Sustainable Development Goals of the United Nations.

Author statement

Artemi Cerdà: Conceptualization, field work, laboratory work; Methodology, Formal analysis, Writing – original draft, writing – reviewing and editing, Project administration, Funding acquisition; Enric Terol: field work, laboratory work; Methodology, Formal analysis Ioannis N. Daliakopoulos Conceptualization, statistics, graphics, writing – reviewing and editing

Declaration of competing interest

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

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Appendix A. Supplementary data

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