Catena 104 (2013) 153–160

Contents lists available at SciVerse ScienceDirect

Catena

journal homepage: www.elsevier.com/locate/catena

Vegetation cover reduces erosion and enhances soil organic carbon in a vineyard in the central Spain

M. Ruiz-Colmenero [a,*], R. Bienes [a,b], D.J. Eldridge [c], M.J. Marques [d]

[a] Applied Research Dept., IMIDRA, Finca El Encín. Ctra. A-2, km 38.2, 28800-Alcalá de Henares, Madrid, Spain
[b] Geology Dept., Universidad de Alcalá de Henares, 28800-Alcalá de Henares, Madrid, Spain
[c] Evolution and Ecology Research Centre, School of Biological, Earth and Environmental Sciences, University of New South Wales, Sydney, NSW 2052, Australia
[d] Geology and Geochemistry Dept., Universidad Autónoma de Madrid, Spain

ARTICLE INFO

Article history:
Received 14 August 2012
Received in revised form 3 November 2012
Accepted 6 November 2012

Keywords:
Aggregate stability
Soil organic carbon
Vineyard
Vegetative cover
Water erosion

ABSTRACT

Land degradation, and soil and nutrient loss, are significant environmental problems in semi-arid agricultural environments in the Mediterranean Basin. One land use that is particularly associated with the highest rates of erosion in Spain is extensive vineyards. We examined the effectiveness of two cover crops for improving soil physical properties and reducing erosion in a vineyard located in the Henares River basin southeast of Madrid, Spain. We assessed erosion from three replicate plots of 2 m^2 each with three treatments that comprised: traditional tillage, permanent cover of *Brachypodium distachyon* and spring-mown crop of *Secale cereale*. Erosion plots under traditional tillage yielded substantially more erosion (5.88 t ha^{-1} yr^{-1}) than *Brachypodium* (0.78 t ha^{-1} yr^{-1}) or *Secale* (1.27 t ha^{-1} yr^{-1}). While the concentration of SOC in sediments was greater for the cover crops, the mass-corrected loss of SOC was greater under tillage (0.06 t ha^{-1} yr^{-1}) than under *Brachypodium* or *Secale* (0.02 t ha^{-1} yr^{-1}). Root biomass was two- to four-times greater under the vegetation treatments. Our measure of aggregate stability for the tillage treatment remained between 7.9 and 5.4 drops over the four years of study and values for both cover crops exceeded that for Tillage by the end of the second year. The vegetation cover treatments increased SOC by 1.2% and intrapedal SOC by 10–60% compared with Tillage. By the end of the study, steady-state infiltration in the cover treatments was 45% greater than that under tillage, with the largest increase under *Brachypodium*. We attribute the greater infiltration on cover treatments to a greater abundance of larger pores on vegetated compared with tilled plots. Our study reinforces the notion that there are considerable benefits of using cover crops in rainfed vineyards, not only for prevention of soil erosion, but to enhance soil condition and potentially reduce the heavy reliance on industrial fertilisers.

1. Introduction

Land and soil degradation, particularly soil erosion and soil structure decline, contributes to substantial losses in productivity and economic livelihoods in semi-arid environments (de la Rosa et al., 2000). In the context of climate change and the increasing population, land degradation has become an issue of global concern. The specific nature of the geomorphology and the variable climate in the Mediterranean Basin mean that the soils are particularly susceptibility to human impact. Soil degradation is a cumulative process, and the time to onset of substantial degradation depends on factors such as crop management, vegetation cover, climate, and topography, and on soil physical properties such as texture, organic matter, structure and porosity. Of these, land use and surface cover are the principal determinants of erosion rates (Cerdan et al., 2002).

Numerous studies have attempted to determine tolerable or acceptable levels of soil erosion for dryland agricultural environments (Verheijen et al., 2009). Maximum rates of acceptable erosion in the literature are about 12 t ha^{-1} yr^{-1} (López-Bermúdez and García-Ruiz, 2007), but more recent studies have reported lower, long-term rates, particularly for Europe, in the vicinity of 1 t ha^{-1} yr^{-1} (Verheijen et al., 2009). Current estimates of erosion suggest that soil loss from agricultural land in Europe is 10- to 40-times greater than this value (Pimentel, 2006). About 44% of the land area of Spain is estimated to be affected by desertification, with erosion rates exceeding 12 t ha^{-1} yr^{-1} (Montanarella, 1999).

One land use associated with the substantially high rates of erosion in Spain is grape growing in extensive vineyards. In Europe, and particularly the Mediterranean Basin, rainfed vineyards are generally substantially degraded and highly eroded (Cerdan et al., 2002; García-Ruiz et al., 2010). Traditional vineyard management involves surface tillage, which creates bare soil most of the year and degrades the soil surface. It is not infrequent to find vineyards on hilly country showing substantial water erosion and with whitish soil surfaces, due to the white calcium

* Corresponding author. Tel.: +34 918879364; fax: +34 918879494.
E-mail address: marta.ruiz.colmenero@madrid.org (M. Ruiz-Colmenero).

0341-8162/$ – see front matter © 2012 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.catena.2012.11.007

carbonate-rich subsoil layers exposed in the surface. One method of moderating erosion in vineyards is to use inter-row cover crops, which not only control the vigour of the vines, but also help to reduce erosion and runoff.

A range of cover crops has been tested, with differing efficiencies, to control erosion in vineyards, and several studies have demonstrated the efficiency of herbaceous covers in woody crops such as olive groves, almonds, citrus and vineyards (e.g. Blavet et al., 2009). Site-specific differences in climate, soil and agronomic practices make it difficult to extrapolate the results to different areas (Romero, 1998), and specific conditions of the crop and vineyard need to be assessed. While cover crops are used widely to control erosion in more mesic areas, they are not used extensively in the Mediterranean Basin because of the belief that they will compete with the vines for soil water (Marques et al., 2011).

The extent to which cover crops compete for moisture and the extra benefits accruing in terms of soil physical amelioration is largely unknown for vineyards in the Mediterranean Basin. Vegetation cover can improve some physical–chemical parameters of the soil through both physical protection of the soil surface, the contribution of organic matter from the plant canopy and the root system (Bronick and Lal, 2005), and by the increased soil structural stability due to increasing soil organic carbon (SOC) and aggregation (Six et al., 1998). Indeed, the loss and redistribution of SOC is minimised due to the reduced tillage and the decrease of erosion (Eynard et al., 2005), though more research is needed to understand why some vegetation covers are more effective than others at enhancing aggregation (Wei et al., 2006). The SOC is a key factor in stability of aggregates (Martínez-Mena et al., 2012) because aggregate breakdown results in the oxidation of organic matter (Zotarelli et al., 2007). Moreover, an absence of plant cover and, consequently, of roots, prevents the buildup or maintenance of SOC.

Vegetation cover also affects the soil water balance by promoting the formation of biologically-produced macropores and by improving soil structural stability. Soil permeability is also directly related to root biomass (Bronick and Lal, 2005). Vegetation cover also increases plant-derived carbon, essential to restore degraded soils (Lal, 2009). Soil organic carbon stabilises macro-aggregates (Verchot et al., 2011) and increases infiltration by reducing the formation of physical crusts, thereby reducing surface sealing. At catchment scales, the lack of vegetation can lead to increased runoff (Cammeraat and Imeson, 1999) and sediment movement. Tillage can also increase erosion and result in declines in SOC associated with eroded sediments. While there is general agreement in the literature about the general links between erosion and nutrient loss, relatively little is known about how erosion affects the carbon balance in semi-arid agricultural environments (Berhe et al., 2007; Kuhn et al., 2009), and in particular, the effect of soil tillage, which is used widely in rainfed vineyards, on the SOC pool (Lal, 2005).

Plot-level studies have traditionally been used to examine the relationships between land management practices and processes of soil erosion and nutrient loss. Some studies have found comparable results between small (1 m^2) and large plots (10 m^2) in subtropical environments (Thomaz and Vestena, 2012). However, the complexity of the Mediterranean Basin makes the direct extrapolation of results from micro-plot to catchment scale problematic and imprecise (Boix-Fayos et al., 2006; Peeters et al., 2008). Also the scale dependency of geomorphic processes makes it difficult to extrapolate the results (García-Ruiz et al., 2010) without taking in account catchment-scale features (Le Bissonnais et al., 1998). However, micro-scale erosion plot data can provide valuable insights into the relative effects of different agricultural land management practices on runoff and soil loss, and can provide input data for more detailed process-based models (Peeters et al., 2008). Erosion plots have proved useful for estimating soil loss (Boix-Fayos et al., 2006) and the extent of SOC mobilization by interrill erosion. Although this fraction can be a significant component of the SOC movement, it is often overlooked in studies of the effects of management on the carbon cycle (Kuhn et al., 2009).

The objective of the work reported here was to examine the degree to which cover crops of the grass *Brachypodium distachyon* (L.) P. Beauv. and the cereal crop *Secale cereale* L. would reduce runoff, sediment movement and the loss of SOC in a rainfed vineyard compared with a standard tillage treatment. Specifically, we predicted that the mechanism responsible for soil erosion and especially for runoff control would be due to simultaneous processes involving changes in SOC, aggregate stability (Tisdall and Oades, 1982), total porosity (Reeves, 1997) and pore space connectivity (Schwen et al., 2011), resulting in an increase of plant available water capacity (Franzluebbers, 2002; Hudson, 1994) and infiltration (Ankeny et al., 1990), which in turn reduce runoff and sediment production. We were motivated by the need to demonstrate that cover crops can provide considerable environmental benefits in vineyards, and that any competition for water between vines and cover crops can be offset by the improvements of the above mentioned soil conditions, which would ultimately benefit the vines themselves. The lack of empirical data on the links between cover crops and soil condition in vineyards has hampered the implementation of the use of cover crops as a sustainable land management practice in semi-arid environments.

2. Materials and methods

2.1. The study area

Our study was conducted in the calcareous moor of Campo Real, in the Henares River basin, southeast of Madrid, in central Spain (40 21 27.7 N, 03 22 29.5 W). The climate is Mediterranean semiarid with an average temperature of 14 °C and an average annual rainfall of 386 mm (National Meteorological Agency, AEMET data from 1977 to 2000). Land use in the general area is predominantly rainfed crops such as vines, olives and almonds. Our study was conducted in a rainfed vineyard that is tilled yearly in the spring (two or three tillage passes depending on rainfalls) and the late autumn. The vineyard was located at 820 m.a.s.l in a highly eroded Miocene terrace dominated by shallow alluvial deposits derived from hard limestone (IGME, 1990). The general landscape is hilly (8° slope) and the soil classified as a Luvisol Calcic (FAO, 2006), equivalent to a Calcic Haploxeralf (SSS, 2010).

In September 2006, two treatments were applied, in a 2 m-wide strip, in the centre of the rows between the vines. The treatments consisted of two grass covers: *S. cereale* (rye) cut in spring (henceforth *Secale*) and a permanent cover of self-sown *B. distachyon* (henceforth *Brachypodium*). *Brachypodium* has been used widely for soil protection in olive groves (Saavedra and Pastor, 2002). These two grass treatments were compared with a conventional tillage control (henceforth Tillage) applied by farmers in the area. Both grasses were sown in September 2006 and the plots measured for four years. There were three replicate plots of each of the three treatments.

2.2. Soil erosion measurements

In each treatment we established three erosion plots (4 m long by 0.5 m wide, located along the slope) in the centre of the inter-rows between the vines (Fig. 1). Sediment was collected monthly, or after every significant rainfall event (>1 mm), in a Gerlach trough (Gerlach, 1967) situated at the bottom of each plot (sediment in Gerlach trough: G). Runoff water was directed to a box where suspended sediment in runoff water was collected separately (suspended sediments in runoff: SS).

2.3. Soil measurements

Prior to the imposition of treatments a 75 cm depth pit was excavated at the site and the following soil analyses undertaken: soil pH, total nitrogen by Kjeldahl digestion (Dewis and Freitas, 1970), available phosphorus (Olsen et al., 1954), extractable cations potassium and sodium by atomic emission spectrometry; and extractable calcium

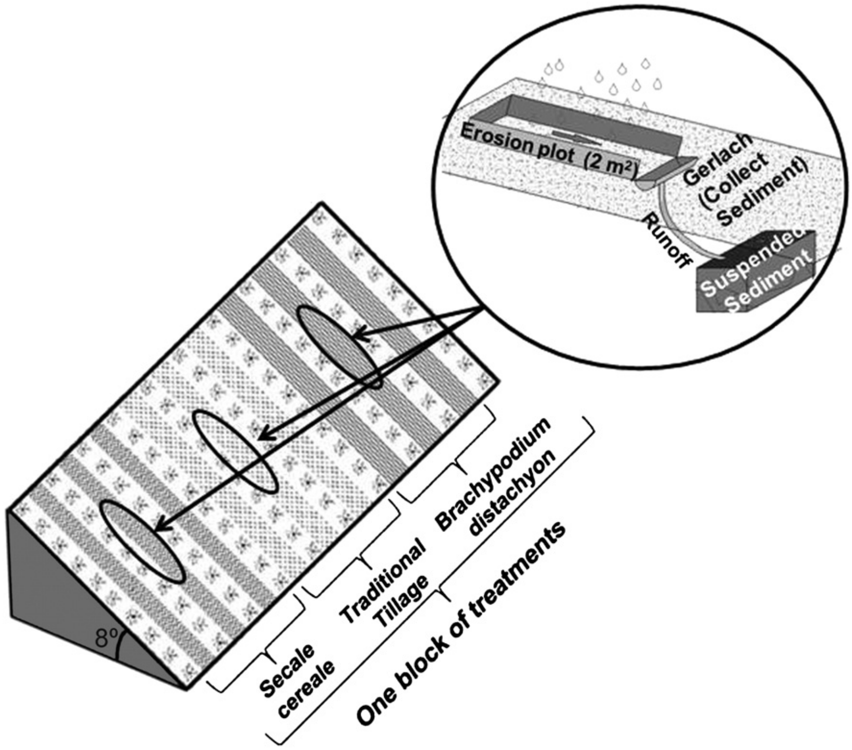

Fig. 1. Schematic representation of one of the three blocks of the experimental design. Every block comprises three consecutive strips of each of the three treatments. The nine erosion plots (three per treatment) were located in the middle of the central strip.

and magnesium by atomic absorption spectrometry (USDA-NRCS, 1996). Soil samples (0–10 cm; five subsamples) were collected each year from each treatment, air-dried in the laboratory, and sieved to obtain various sized fractions to undertake various soil tests as described below.

2.3.1. Soil organic carbon and aggregate stability

The soil organic carbon content (SOC) of the 2 mm fraction was determined using the Walkley and Black (1934) method, and soil organic matter (SOM) content determined using the relevant correction factor (1.72). A fraction of the soil samples was dry sieved (4.00 to 4.75 mm diameter) to extract macro-aggregates (Boix-Fayos et al., 2001) for assessment of aggregate stability. Fifty aggregates from each treatment were air-dried and weighted and subjected to the drop-test (Imeson and Vis, 1984) to evaluate resistance to drop impact. The number of drops required to fracture the aggregates (ND) was used as an index of aggregate stability.

2.3.2. Intrapedal porosity and organic carbon

The petroleum method was used to measure intrapedal porosity. Basically aggregates were weighed, submerged in petroleum oil for 24 h to allow pores to infill and the difference in mass used to calculate porosity according to the method of Busoni (1997). Intrapedal SOC was measured in two aggregate fractions micro-aggregates (<2 mm) and macro-aggregates (>2 mm) according to Porta et al. (2003) by the Loss On Ignition method after moderate temperature combustion (360 °C for 2 h; Konen et al., 2002).

2.3.3. Soil water relationships

We assessed volumetric soil moisture in the inter-rows of vineyards daily over 4 years with ECH_2O® moisture sensors. Two sensors were placed in each replicate of each treatment (three replicates per treatment), one at a depth of 10 cm and the other buried at 35 cm.

We collected nine 100 cm^3 undisturbed cores (depth = 5 cm) from the middle of the inter-rows of each treatment. The cores were saturated with water by capillarity in a sandbox. Twelve increasing suctions were applied until a balance was reached for each tension being then weighed again to determine the water content. First in the sandbox the cores slowly drained between pF 0 and pF 2. Then, the cores were changed to a plate extractor system (Richards, 1941) where tensions between pF 2.0 and 4.2 were applied. Thus, the soil moisture was measured from total saturation (pF ~0) to permanent wilting point (pF 4.2). Finally, the samples were completely oven dried (24 h at 105 °C).

Pores were defined as either macropores (>0.6 mm across; pF 0 to 1.8), mesopores (between 0.6 and 0.1 mm; pF 1.8 to 2.54) or micropores (<0.1 mm; pF > 2.54), consistent with current literature (e.g. Taboada et al., 2004). We use the pF data to derive water retention curves for each treatment (Hartge and Horn, 1992; Schofield, 1935). The curves allowed us to determine key information on measures of soil water storage such as field capacity (pF 2.54), permanent wilting point (pF 4.2) and therefore plant available water (henceforth PAW; between pF 2.54 and pF 4.2). We used our experimental data to derive the water retention curves using RETC software (van Genuchten et al., 1991) to obtain the parameters needed to use the empirical Eq. (1) modified by van Genuchten and Nielsen (1985); the corresponding curves were compared using Statistica 6.0 software (StatSoft, Inc., 2002).

$$\theta_h = \theta_r + (\theta_s - \theta_r)\left[1 + (\alpha h)^n\right]^{-m} \quad (1)$$

where θ_h = water content at tension h (m^3 water m^{-3}soil), θ_s = saturated water content (m^3 water m^{-3} soil), θ_r = the residual water content (m^3 water m^{-3} soil), α = inverse of the air entry suction (m^{-3}), n = a dimensionless value related to the pore size distribution, and $m = 1-(1/n)$, and h = the suction pressure (m^{-3}).

2.3.4. Infiltration rate

We assessed infiltration rate with a double-ring infiltrometer. Infiltration was measured twice in each treatment. Due to the extent of stones in the study site (20–25% stone cover), it was difficult to hammer the large rings into the soil. Consequently, we reassessed infiltration in the last year using a simplified infiltration test (USDA,

2001) using smaller single rings that were easier to push into the soil. Rings, 5 cm in diameter, were driven into the soil, the surface covered with plastic to avoid disturbance, and 50 ml of water (equivalent to 25 l m^{-2}) gently added after which the plastic was removed. The procedure was repeated 10 times for each ring, which received a total of 250 l m^{-2} of water.

In order to assess root biomass, three undisturbed soil cores (10 cm depth) per treatment were obtained using a drill (2.18 cm diameter). Soil samples were air-dried, soil aggregates were carefully crumbled and the roots were manually extracted with the help of sieves and tweezers and then weighted.

2.4. Statistical analyses

We examined differences in mean values using the non-parametric Kolmogorov–Smirnov test because transformations failed to normalise our data. Correlation analyses were undertaken using the Spearman test, a non-parametric equivalent of the Pearson test. The infiltration curves were fitted to the Kostiakov (1932) potential model ($K = cX^{-b}$). All analyses were undertaken in the Statistica 6.0 software package (StatSoft, Inc., 2002).

3. Results

3.1. Initial soil conditions

Pre-treatment soil analyses showed that the soils had a high pH (8.3 ± 0.6; n = 18) with a maximum $CaCO_3$ content of 25%. Electrical conductivity was low, indicating no salt accumulation. Total nitrogen (0.01 and 0.11%) and available phosphorus (36.5 mg kg^{-1}) were within the normal range for these soils. SOC was low (0.71 ± 0.14%), even at the topsoil (0–10 cm depth). The texture of this soil layer is sandy clay loam (58.6 ± 10.5% sand, 17.8 ± 5.8% silt, 23.6 ± 5.3% clay), which suggests medium to high rates of permeability (Hillel, 2004).

Prior to the establishment of the two grass treatments when the vineyard was under traditional tillage, the mean index of aggregate stability (ND) was 7.9 ± 1.2 drops, which is relatively low for cropping areas in semi-arid environments (Cerda, 2000).

3.2. Soil erosion

During the study we experienced 46 rain events, which generated a total sum of 47 t ha^{-1} of sediment. Average annual rates of erosion under tillage were about 6 ± 1.3 t ha^{-1} yr^{-1}; more than five-times greater than under *Brachypodium* or *Secale*, respectively (Table 1). Sediment entrained in both runoff water (SS) and in the Gerlach troughs (G) was also substantially greater under tillage and marginally greater under *Secale* than *Brachypodium*. Though there was less sediment generated under the cover crops, there was about 1.4-times more SOC in the sediment from the cover crop treatments than the plots under tillage (Table 2).

3.3. Aggregate stability

The increase in aggregate stability was slow, and it was not until two years after treatment establishment that there was a marked increase in stability. Surprisingly, our index of aggregate stability (ND) declined by about 30% under Tillage over the course of the three years. Intrapedal porosity failed to change significantly over time (Table 3). We detected a highly significant relationship between the average value of the aggregate stability index over the three years of the study and the average erosion yield (Spearman's R = −0.78; P = 0.01).

While the SOC% under the Tillage treatment was less than the cover treatments (Table 1) correcting these data on the basis of the mass of eroded sediment indicates that the total amount of SOC lost from tillage was 0.06 t ha^{-1} yr^{-1} ([3.02 t_{SS} ha^{-1} yr^{-1} (Table 1) · 1.3% SOC (Table 2)] + [2.86 t_G ha^{-1} yr^{-1} (Table 1) · 0.87% SOC (Table 2)]) compared with 0.02 t ha^{-1} yr^{-1} for both, *Brachypodium* ([0.58 t_{SS} ha^{-1} yr^{-1} (Table 1) · 2.14% SOC (Table 2)] + [0.20 t_G ha^{-1} yr^{-1} (Table 1) · 1.17% SOC (Table 2)]) and *Secale* ([1.02 t_{SS} ha^{-1} yr^{-1} (Table 1) · 1.68% SOC (Table 2)] + [0.25 t_G ha^{-1} yr^{-1} (Table 1) · 1.36% SOC (Table 2)]), respectively. For aggregates < 2 mm in diameter, intrapedal SOC ranged from 13.4 g kg^{-1} under Tillage to 18.3 g kg^{-1} under *Secale* and 20.6 g kg^{-1} under *Brachypodium*. For aggregates > 2 mm in diameter, SOC was 1.6-times greater under *Brachypodium* (26.4 g kg^{-1}) than under Tillage (17.0 g kg^{-1}), but unchanged for *Secale* (18.8 g kg^{-1}). Root biomass was greater under *Brachypodium* (0.2 mg cm^{-3}) and *Secale* (1 mg cm^{-3}) than under Tillage (0.05 mg cm^{-3}) at the end of the study.

Table 1
Mean and standard deviation (SD) of cumulative sediment yield (g m^{-2}) between September 2006 and September 2010 (46 rain events for three plots per treatment; n = 138 separate events). Different letters indicate significant differences at $P < 0.005$.

Treatment	Cumulative sediment yield 4 years (g m^{-2})		Average yield (t ha^{-1} yr^{-1})	% change compared with Tillage	
	SS in runoff water	Sediments in G trough		SS in runoff water	Sediments in G trough
	Mean ± SD	Mean ± SD			
Tillage	1208 ± 1254[a]	1142 ± 516[a]	5.88	–	–
Brachypodium	232 ± 340[b]	78 ± 34[b]	0.78	81	93
Secale	409 ± 595[c]	100 ± 25[c]	1.27	66	91

SS: suspended sediments. G: Gerlach.

3.4. Water holding capacity and infiltration

3.4.1. Water holding capacity

Total porosity was about 50% for all treatments, within the range for medium soil textures (Table 4). However, there was no consistent trend in porosity for different pore sizes among the three treatments. Microporosity was 1.2-times greater under the Tillage treatment than *Brachypodium*.

Based on empirical data, the soil water retention curves were consistent with predictions under the van Genuchten model. At high values of pF, the curve for Tillage was displaced to the right while at low pF values it was displaced to the left (Fig. 2) indicating a predominance of micropores and therefore few larger pores compared with the vegetated treatments. The curves also indicated that effective porosity, the contribution of mesopores and macropores (pF < 2.54), was greater in *Brachypodium* (37.5%) and *Secale* (33%) than under the Tillage treatment (20.5%;

Table 2
Mean and standard deviation (SD) values of soil organic carbon concentration (SOC, %) and soil organic carbon stock (SOC, t C ha^{-1}) in soil and in the eroded sediments in 2010 after 4 years of treatments. Different letters indicate significant differences at $P < 0.05$.

Treatment	Topsoil (0–10 cm depth) (n = 4)			SOC in sediments	
	Bulk density	SOC		In SS in runoff water (n = 3)	In sediments in G trough (n = 3)
	Mean ± SD	Mean ± SD	Mean ± SD	Mean ± SD	Mean ± SD
	(g cm^{-3})	(%)	(t C ha^{-1})	(%)	(%)
Tillage	1.35 ± 0.12[a]	0.59 ± 0.06[b]	8.0 ± 0.7[b]	1.30 ± 1.24[b]	0.87 ± 0.28[b]
Brachypodium	1.29 ± 0.18[a]	0.91 ± 0.20[a]	11.7 ± 1.6[a]	2.14 ± 0.49[a]	1.17 ± 0.13[a]
Secale	1.26 ± 0.13[a]	0.91 ± 0.05[a]	11.5 ± 1.2[a]	1.68 ± 1.20[a]	1.36 ± 0.27[a]

SS: suspended sediments. G: Gerlach.

Table 3
Mean and standard deviation (SD) of our index of aggregate stability (ND) and intrapedal porosity (%) of surface soil samples (0–10 cm) for each treatment at the beginning of the study and 1, 2 and 3 years later. Different letters indicate a significant difference at $P<0.001$ for each parameter measured.

Years	Treatment	Aggregate stability (ND)		Intrapedal porosity (%)	
		Mean ± SD	*n*	Mean ± SD	*n*
0	Tillage	7.9 ± 1.20^a	50	# #	#
1	Tillage	7.9 ± 4.16^a	200	29.5 ± 16.82^a	200
	Brachypodium	8.1 ± 4.70^a	200	26.8 ± 11.03^a	200
	Secale	8.3 ± 4.66^a	200	24.4 ± 16.23^a	200
2	Tillage	5.1 ± 2.08^c	100	28.2 ± 20.10^a	100
	Brachypodium	12.9 ± 11.48^b	100	28.5 ± 18.28^a	100
	Secale	9.4 ± 5.14^b	100	28.6 ± 13.88^a	100
3	Tillage	5.4 ± 1.76^c	75	31.7 ± 14.42^a	50
	Brachypodium	11.2 ± 6.93^b	75	30.0 ± 13.35^a	50
	Secale	9.0 ± 4.45^b	75	27.8 ± 13.24^a	50

= no data.

$P<0.05$). The fraction of moisture held between pF 2.54 and pF 4.2 did not differ among the three treatments (range: 10.6 to 13.3%; $P>0.05$).

Volumetric soil moisture varied from 18% during the autumn and spring rainfalls to 7% in summer. Applying these values to the water retention curves (Fig. 2), pF values would be lower under both *Brachypodium* (range: 2.2 to 3.5) and *Secale* (2.3 to >4.2) than under Tillage (2.4 to >4.2) due to the greater meso-macroporosity under the plants than under Tillage.

3.4.2. Infiltration

After three years of treatment, steady-state infiltration was greater under *Brachypodium* ($P<0.001$; 42 mm h^{-1}) than under *Secale* (18 mm h^{-1}) and under Tillage treatment (16 mm h^{-1}; Fig. 3A). Although values were considerably higher for the simplified test using a single ring than for the double ring test (Fig. 3B), the overall trends were similar. Thus, in the 4th year of treatments, significantly higher rates were obtained under *Brachypodium* (171 ± 63 mm h^{-1}) than either *Secale* (137 ± 85 mm h^{-1}) or Tillage (96 ± 43 mm h^{-1}; $P<0.025$).

4. Discussion

In the Mediterranean Basin, the management of vineyard soils has traditionally relied heavily on surface tillage to reduce weed competition and to break up surface crusts in the belief that this will increase the infiltration of water. Consequently, large areas of vineyards are now extensively degraded due to topsoil loss and physical and chemical deterioration of surface soils. Our study tested the effectiveness of two cover crops in preventing and reversing the effects of soil degradation in an eroded vineyard in central Spain. The use of vegetation to stop erosion and reverse land degradation is not new (Hooke and Sandercock, 2012) but it is not so often implemented in active woody crops in semi-arid areas where the competition for water can be an important handicap. Our results showed that, compared with conventional tillage, cover crops increased the mass of roots in the topsoil, and increased the concentration of intrapedal SOC and the relative proportion of meso- and macro-pores. These changes in soil physical conditions and aggregation resulted in greater infiltration, and consequently, a four- to six-fold reduction in erosion compared with conventional tillage. All these changes were relatively fast as significant differences were found after 3 or 4 years of treatment.

Table 4
Mean and standard deviation (SD) of pore size in surface soils (0–5 cm) in relation to treatment; $n=9$ for each treatment and sampling. Different letters indicate a significant difference at $P<0.05$ among treatments.

	Macropores	Mesopores	Micropores
pF	(%)	(%)	(%)
Pore size	0–1.8	1.8–2.54	>2.54
	>0.6 mm	0.6–0.1 mm	<0.1 mm
	Mean ± SD	Mean ± SD	Mean ± SD
Tillage	21.1 ± 2.73^a	8.8 ± 4.70^a	20.2 ± 2.52^a
Brachypodium	22.2 ± 2.77^a	12.1 ± 1.61^b	16.7 ± 2.24^b
Secale	23.7 ± 3.03^a	9.5 ± 3.97^a	19.4 ± 2.25^{ab}

4.1. Erosion and infiltration under different treatments

In our study, the average annual erosion rate under tillage was equivalent to 6 t ha^{-1} yr^{-1}, similar to rates reported for other areas in central Spain, which range from 5 to 12 t ha^{-1} yr^{-1} (Bienes et al., 2001). Our data suggest that changing to a management regime based on the use of cover crops would reduce this to about 0.8 t ha^{-1} yr^{-1} under *Brachypodium* or 1.3 t ha^{-1} yr^{-1} under *Secale*. Losses of 6 t ha^{-1} yr^{-1} are considered unsustainable for these semi-arid agricultural environments given that soil formation rates are an order of magnitude less (range: 0.3 to 1.4 t ha^{-1} yr^{-1}) than erosion rates for most European situations (Verheijen et al., 2009). Cover crops act to increase soil physical properties (described below), but also provide a physical barrier on the surface to increase the tortuosity of overland flow, thereby reducing the capacity of runoff to entrain sediment (Kosmas et al., 1997). Given that suspended sediment is typically high in the finest fractions such as clay to which are adsorbed nutrients such as nitrogen and phosphorus, reductions in erosion and runoff can reduce the loss of fertility of these soils (Martínez-Casasnovas and Ramos, 2006).

A number of studies have demonstrated a direct relationship between vegetation cover and soil water storage through reduced evaporation and increased infiltration (Chisci et al., 2001) resulting from increased

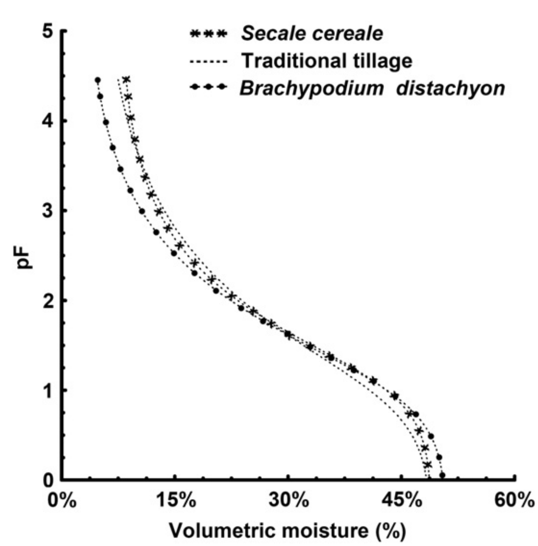

Fig. 2. Average Water Retention Curve for each treatment ($n=9$).

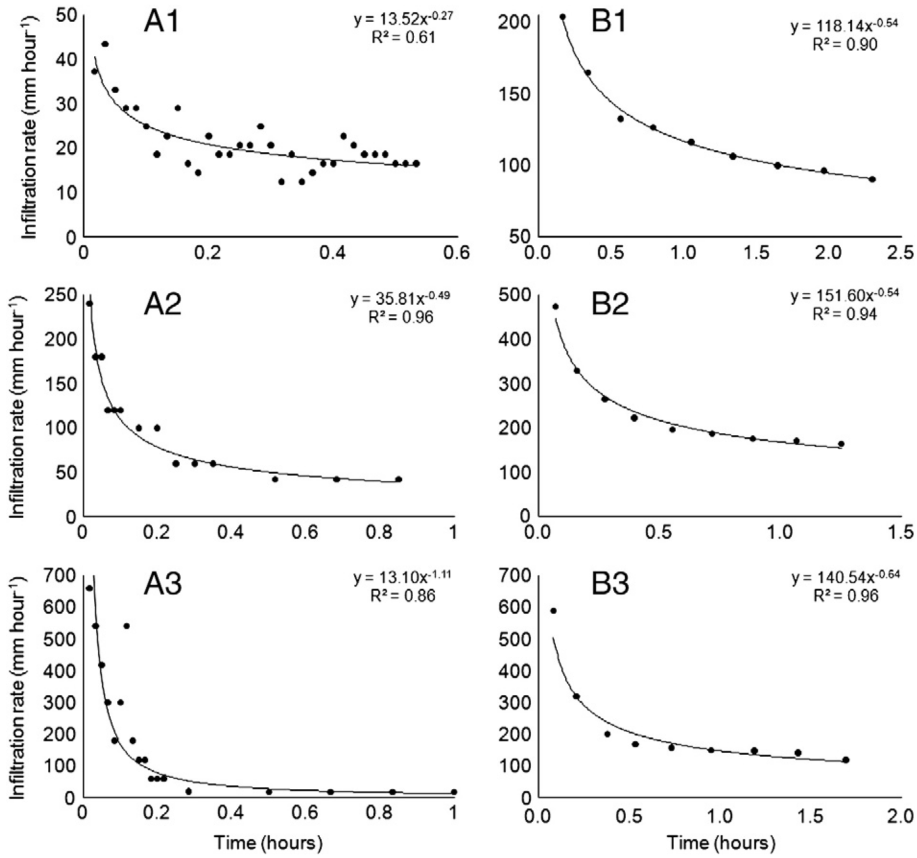

Fig. 3. Average infiltration curves for Year 3 using double ring test (A; $n=2$) and Year 4 using simplified ring test (B; $n=10$). Treatments: Tillage (1), *Brachypodium* (2) and *Secale* (3). Note the different scales on the both axes.

vertical porosity created by the root of perennial plants. In our study the relative contribution of micropores was greater under tillage than under *Brachypodium*, which we attribute to the collapse of larger pores under the impact of cultivation. Our results also indicate that a combination of reduced surface disturbance and increased vegetation cover and root biomass under *Brachypodium* and *Secale*, enhances the stability of surface soils, leading to greater soil water holding capacity and infiltration on an annual term. Differences in the relative contribution of large biologically-produced pores to total porosity among the three treatments explains, in a large part, the differences in infiltration rates that we observed across the sites.

Low frequency, high intensity rainstorms are a common feature of semi-arid Mediterranean systems (Ferrero et al., 2002). Consequently, the benefits of increased surface cover are likely to increase with increasing rainfall intensity. For example, Marques et al. (2010) showed that soil loss under traditional tillage treatments was five-times greater than under a cover crop under low intensity storms, but increased to 30-times greater under high intensity storms. While cover crops require soil moisture to survive in semi-arid Mediterranean environments, the benefits of a cover crop are likely to compensate some negative competitive effects with the crop by increasing the infiltration capacity of the soil. However, cover crops should be sown before maximum vine development in early spring to reduce the effects of competition for moisture (Ruiz-Colmenero et al., 2011).

4.2. Changes in soil organic carbon and porosity

Despite the initial low level of SOC in surface soils in our study (9.7 t C ha^{-1}), the value remained low (8.0 t C ha^{-1}) under tillage. The loss of SOC under tillage in this study may not appear overly high, but comparisons with other areas in the Mediterranean Basin (Quinton et al., 2010) indicate that our rate of SOC loss (6 t km^{-2} yr^{-1}) is substantially greater than comparable areas under tillage (~0.5 t km^{-2} yr^{-1}). We acknowledge that such a direct comparison is problematic, given differences in the erosivity of rainfall across different climatic zones. However, it provides a general view of the magnitude of SOC loss that can occur in vineyards, a crop that can remain active for several decades. Under inappropriate management, the crop can leave the soil in an irreversibly degradation state (COM, 2006). The loss of SOC also has implications for the maintenance of soil structure. This is particularly important when there is a limited carbon stock in topsoil. A review of SOC stock under the main types of land uses carried out by Rodríguez-Murillo (2001) showed that soils under vineyards accounted for 43 ± 29 t C ha^{-1} with very high variability. The results of this study are in the lower limit of this review as soils managed by tillage exhibited 8 t C ha^{-1}, but experienced a significant increase due to cover crops, achieving up to 11 t C ha^{-1} (Table 2). The index of aggregate stability was significantly higher under either *Brachypodium* or *Secale*, but only after two years of the study. The strong and significant negative relationship between soil aggregate stability and our annualised erosion rate clearly demonstrates the direct link between the recovery of aggregate stability in our soils and the susceptibility of soils to erosion, particularly in semi-arid Mediterranean environments (e.g. Barthes and Roose, 2002).

In general, losses of up to 60% of SOC are likely during the first few years of cropping following conversion of non-agricultural to agricultural land (Davidson and Ackerman, 1993). This decline in SOC content is likely to be more pronounced with increasing severity of erosion due to a loss of topsoil and therefore a reduced capacity of the residual surface to retain SOC (Fenton et al., 2005). Continual tillage will also reduce the potential for litter to contribute to increasing SOC or to protect the surface against raindrop impact (Le Bissonnais et al., 2007). The rate of increase in SOC with improved management is also slow. For example, Novara et al. (2011) found that about 50 years were required to increase SOC by 50% in abandoned vineyards in semi-arid environments. This suggests to us that the cost of erosion, both in terms of SOC and loss of soil fertility, is high (Martínez-Casasnovas and Ramos, 2006). Further, recovery of SOC and fertility in the absence of artificial fertilisers is likely to be a protracted process.

Soil structural stability can be enhanced by root exudates, polysaccharides and labile mucilages produced by living plant roots, as well as the detritus from decomposing dead roots, all of which are a source of labile carbon for microbes. Predictably, we found a greater root biomass under the vegetated treatments, with four- and two-times more biomass under *Brachypodium* and *Secale*, respectively, than under tillage. The increase in root biomass would be expected to result in increases in intrapedal porosity, as this is strongly influenced by fine plant roots. Contrary to expectation, however, intrapedal porosity did not differ among the three treatments.

Intrapedal porosity makes up only a small fraction of total soil porosity, which also includes voids between the aggregates and the pores remaining after root decomposition. We suggest, therefore, that increases in aggregate stability were due to increased SOC and to the greater biomass of plant roots. Both would lead to greater interparticle bonding and therefore greater aggregation without necessarily altering intrapedal porosity. Support for this comes from studies by Eynard et al. (2005) who found that macro-aggregate intrapedal SOC increased in the presence of vegetation and after cessation of tillage. In other studies, tillage has been linked to a reduction of C-rich macro-aggregates and the increase of C-depleted micro-aggregates (Six et al., 2000). Macro-aggregate stability is known to be positively correlated with SOC content (Blavet et al., 2009; Boix-Fayos et al., 2001) due to the strong linkage between the colloidal fractions of soils (Kirkby and Morgan, 1980). The level of SOC and the magnitude of aggregate stability in vineyards are both known to be correlated, and there may be a threshold, between 17 and 23 g SOC kg^{-1}, below which aggregate stability is low (Le Bissonnais et al., 2007). Comparing this apparent threshold with our results, it is clear that this is only achieved with macro-aggregates > 2 mm from the *Brachypodium* treatment. This reinforces our view that the sowing of cover crops, particularly *Brachypodium*, has the capacity to enhance soil structure over a period of four years.

5. Concluding remarks

Our study demonstrated that the planting of vegetative cover crops between the rows of vines in sloping vineyards can reduce losses from erosion and improve the infiltration of water. Soil organic carbon also increased, most likely because of an increased incorporation of vegetative residues and decomposition of roots. Aggregate stability also increased after two years, and this was linked to the higher intrapedal SOC. In contrast, traditional tillage had the opposite effect, with greater erosion, less infiltration, lower SOC and a preponderance of micropores, as indicated by the water retention curves. Changes in the parameters we measured occurred at different rates depending on the cover species involved. Overall, the greatest improvements in SOC and erosion prevention and infiltration occurred under *Brachypodium*, and four years was enough to bring about substantial improvements in soil quality.

Our results indicate that current management practices based on tillage of vineyard soils are unsustainable, leading to a loss of SOC and a reduction in the productive potential of soils that cannot be offset with the use of fertilisers alone. The use of a permanent cover of *Brachypodium* is one solution to the problem of soil degradation in vineyards. However, *Brachypodium* needs to be managed differently to prevent competition for water with the vines and a drop in production (Ruiz-Colmenero et al., 2011).

Acknowledgements

This experiment could be possible thanks to Projects FP06-DR3 IMIDRA, RTA2007-0086 INIA and Predoctoral grant FPI-INIA bec2008-0064. We thank the owner of the Carlos Gosálbez and Gosálbez-Ortí wineries for his assistance in the field and for allowing us to work on his land.

References

Ankeny, M.D., Kaspar, T.C., Horton, R., 1990. Characterization of tillage and traffic effects on unconfined infiltration measurements. Soil Science Society of America Journal 54, 837–840.

Barthes, B., Roose, E., 2002. Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. Catena 47, 133–149.

Berhe, A.A., Harte, J., Harden, J.W., Torn, M.S., 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience 57, 337–346.

Bienes, R., Pérez, R., Domínguez, M.A., 2001. Mapa de degradación de los suelos de la Comunidad de Madrid. Consejería de Medio Ambiente-Comunidad de Madrid, Madrid . (121 pp.).

Blavet, D., De Noni, G., Le Bissonnais, Y., Leonard, M., Maillo, L., Laurent, J.Y., Asseline, J., Leprun, J.C., Arshad, M.A., Roose, E., 2009. Effect of land use and management on the early stages of soil water erosion in French Mediterranean vineyards. Soil and Tillage Research 106, 124–136.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A.C., Soriano-Soto, M.D., 2001. Influence of soil properties on the aggregation of some Mediterrranean soils and the use of aggregate size and stability as land degradation indicators. Catena 44, 47–67.

Boix-Fayos, C., Martinez-Mena, M., Calvo-Cases, A., Arnau-Rosalén, E., Alvadalejo, J., Castillo, V., 2006. Causes and underlying processes of measurement variability in field erosion plots in Mediterranean conditions. Earth Surface Processes and Landforms 78, 267–285.

Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124, 3–22.

Busoni, E., 1997. Metodo della zolla ovvero dell'unità strutturale (ped). Metodi di analisi fisica del suolo. Ministerio per le Politiche Agricole, Milano, pp. 10–12.

Cammeraat, L.H., Imeson, A.C., 1999. The evolution and significance of soil-vegetation patterns following land abandonment and fire in Spain. Catena 37, 107–127.

Cerda, A., 2000. Aggregate stability against water forces under different climates on agriculture land and shrubland in southern Bolivia. Soil and Tillage Research 57, 159–166.

Cerdan, O., Le Bissonnais, Y., Couturier, A., Saby, N., 2002. Modelling interrill erosion in small cultivated catchments. Hydrological Processes 16, 3215–3226.

Chisci, G.C., Bazzoffi, P., Pagliai, M., Papini, R., Pellegrini, S., Vignozzi, N., 2001. Association of sulla and atriplex shrub for the physical improvement of clay soils and environmental protection in central Italy. Agriculture, Ecosystems and Environment 84, 45–53.

COM, 2006. 231-final. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions – Thematic Strategy for Soil Protection.

Davidson, E.A., Ackerman, I.L., 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20, 161–193.

De la Rosa, D., Moreno, J.A., Mayol, F., Bonsón, T., 2000. Assessment of soil erosion vulnerability in western Europe and potential impact on crop productivity due to loss of soil depth using the ImpelERO model. Agriculture, Ecosystems and Environment 81, 179–190.

Dewis, J., Freitas, F., 1970. Physical and chemical methods of soil and water analysis. FAO Soils Bulletin 10 (Rome).

Eynard, A., Schumacher, T.E., Lindstrom, M.J., Malo, D.D., 2005. Effect of agricultural management systems on soil organic C in aggregates of Ustoll and Usterts. Soil and Tillage Research 81, 253–263.

FAO, 2006. Guidelines for Soil Description, fourth ed. FAO, Rome.

Fenton, T.E., Kazemi, M., Lauterbach-Barrett, M.A., 2005. Erosional impact on organic matter content and productivity of selected Iowa soils. Soil and Tillage Research 81, 163–171.

Ferrero, A., Lisa, L., Parena, L.S., Sudiro, L., 2002. Run-off and soil erosion from tilled and controlled grass-covered vineyards in a hillside catchment. ERB and Northern European FRIEND Project 5 Conference, Demänovská dolina, Slovakia.

Franzluebbers, A.J., 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil and Tillage Research 66, 197–205.

García-Ruiz, J.M., Lana-Renault, N., Beguería, S., Lasanta, T., Regüés, D., Nadal-Romero, E., Serrano-Muela, P., López-Moreno, J.I., Alvera, B., Martí-Bono, C., Alatorre, L.C., 2010. From plot to regional scales: interactions of slope and catchment hydrological and geomorphic processes in the Spanish Pyrenees. Geomorphology 120, 248–257.

Gerlach, T., 1967. Hillslope for measuring sediment movement. Revue de Geomorphologie Dynamique 4, 173–175.

Hartge, K.H., Horn, R., 1992. Die physikalische Untersuchung von Bóden, 3 Auflage. Ferdinand Enke Verlag, Stuttgart . (177 pp.).

Hillel, D., 2004. Introduction to Environmental Soil Physics . (Academic Press, San Diego, CA. Elsevier Science, USA).

Hooke, J., Sandercock, P., 2012. Use of vegetation to combat desertification and land degradation: recommendations and guidelines for spatial strategies in Mediterranean lands. Landscape and Urban Planning 107, 389–400.

Hudson, B.D., 1994. Soil organic matter and available water capacity. Journal of Soil and Water Conservation 49, 189–194.

IGME, 1990. Mapa Geologico de Espana. Escala 1:50.000, Alcala de Henares. Instituto Technologico Geominero de Espana.

Imeson, A.C., Vis, M., 1984. Assessing soil aggregate stability by water-drop impact and ultrasonic dispersion. Geoderma 34, 185–200.

Kirkby, M.J., Morgan, R.P.C., 1980. Soil Erosion. John Wiley and Sons, London.

Konen, M.E., Jacobs, P.M., Burras, C.L., Talaga, B.T., Mason, J.A., 2002. Equations for predicting soil organic carbon using loss-on-ignition for North Central U.S. soils. Soil Science Society of America Journal 66, 1878–1881.

Kosmas, C., Danalatos, N., Cammeraat, L.H., Chabart, M., Diamantopoulos, J., Farand, R., Gutierrez, L., Jacob, A., Marques, H., Martinez-Fernandez, J., Mizara, A., Moustakas, N., Nicolau, J.M., Oliveros, C., Pinna, G., Puddu, R., Puigdefábregas, J., Roxo, M., Simao, A., Stamou, G., Tomasi, N., Usai, D., Vacca, A., 1997. The effect of land use on runoff and soil erosion rates under Mediterranean conditions. Catena 29, 45–59.

Kostiakov, A.N., 1932. On the dynamics of the coefficient of water-percolation in soils and on the necessity of studying it from a dynamic point of view for purpose of amelioration. Transactions of the Sixth Commission of the International Society of Soil Science, Russian Part A (17–21 pp.).

Kuhn, N.J., Hoffmann, T., Schwanghart, W., Dotterweich, M., 2009. Agricultural soil erosion and global carbon cycle: controversy over? Earth Surface Processes and Landforms 34, 1033–1038.

Lal, R., 2005. Soil erosion and carbon dynamics. Soil and Tillage Research 8, 137–142.

Lal, R., 2009. Challenges and opportunities in soil organic matter research. European Journal of Soil Science 60, 158–169.

Le Bissonnais, Y., Benkhadra, H., Chaplot, V., Fox, D., King, D., Daroussin, J., 1998. Crusting, runoff and sheet erosion on silty loamy soils at scales and upscaling from m^2 to small catchments. Soil and Tillage Research 46, 69–80.

Le Bissonnais, Y., Blavet, D., De Noni, G., Laurent, J.Y., Asseline, J., Chenu, C., 2007. Erodibility of Mediterranean vineyard soils: relevant aggregate stability methods and significant soil variables. European Journal of Soil Science 58, 188–195.

López-Bermúdez, F., García-Ruiz, J.M., 2007. La degradación del suelo por erosión hídrica en España. In: Cerdá (Ed.), Erosión y Degradación del suelo agrícola en España.

Marques, M.J., García-Muñoz, S., Muñoz-Organero, G., Bienes, R., 2010. Soil conservation beneath grass cover in hillside vineyards under Mediterranean climatic conditions (Madrid, Spain). Land Degradation and Development 21, 122–131.

Marques, M.J., Velasco, A., Ruiz-Colmenero, M., Cuadrado, J., Bienes, R., 2011. To what extent are farmers willing to collaborate to control land degradation? In: Donald, G., Vermang, J. (Eds.), Land Degradation Processes and Assessment: Wind Erosion, Interrill Erosion, Gully Erosion, Land Cover Features. Unesco Chair of Eremology, Ghent University, Belgium. ISBN: 978-90-5989-439-6 (173 pp.).

Martínez-Casasnovas, J.M., Ramos, C., 2006. The cost of soil erosion in vineyard fields in the Penedès–Anoia Region (NE Spain). Catena 68, 194–199.

Martínez-Mena, M., López, J., Almagro, M., Albaladejo, J., Castillo, V., Ortiz, R., Boix Fayos, C., 2012. Organic carbon enrichment in sediments: effects of rainfall characteristics under different land uses in a Mediterranean area. Catena 94, 36–42.

Montanarella, L., 1999. Soil at the interface between agriculture and environment. Agriculture, Environment, Rural Development: Facts and Figures — A Challenge for Agriculture. Report for the European Commission.

Novara, A., Gristina, L., La Mantia, T., Rühl, J., 2011. Soil carbon dynamics during secondary succession in a semi-arid Mediterranean environment. Biogeosciences Discussions 8, 11107–11138.

Olsen, S.R., Cole, C.V., Watanabe, F.S., Dean, J.F., 1954. Estimation of Available Phosphorus in Soils by Extraction with Sodium Bicarbonate. U.S. Department of Agriculture Circular . (939 pp.).

Peeters, I. Van, Oost, K., Govers, G., Verstraeten, G., Rommens, T., Poesen, J., 2008. The compatibility of erosion data at different temporal scales. Earth and Planetary Science Letters 265, 138–152.

Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environment, Development and Sustainability 8, 119–137.

Porta, J., Lopez-Acevedo, M., Roquero, C., 2003. Edafología para la agricultura y el medio ambiente. Ediciones Mundi-Prensa, Madrid . (929 pp.).

Quinton, J.N., Govers, G., van Oost, K., Bardgett, R., 2010. The impact of agricultural soil erosion on biogeochemical cycling. Nature Geoscience 3, 311–314.

Reeves, D.W., 1997. The role of soil organic matter in maintaining soil quality in continuous cropping systems. Soil and Tillage Research 43, 131–167.

Richards, L.A., 1941. A pressure–membrane extraction apparatus for soil solution. Soil Science 51, 377–386.

Rodríguez-Murillo, J.C., 2001. Organic carbon content under different types of land use and soil in peninsular Spain. Biology and Fertility of Soils 33, 53–61.

Romero, L.R., 1998. Olive farming in the age of science and innovation. Olivae 72, 42–51.

Ruiz-Colmenero, M., Bienes, R., Marques, M.J., 2011. Soil and water conservation dilemmas associated with the use of green cover in steep vineyards. Soil and Tillage Research 117, 211–223.

Saavedra, M., Pastor, M., 2002. Sistemas de cultivo en olivar: Manejo de malas hierbas y herbicidas. Editorial Agrícola Española S.A. (428 pp.).

Schofield, R.K., 1935. The pF of the water in soil. Transaction of the International Congress on Soil Science 3rd, II, pp. 38–48.

Schwen, A., Bodner, G., Scholl, P., Buchan, G.D., Loiskandl, W., 2011. Temporal dynamics of soil hydraulic properties and the water-conducting porosity under different tillage. Soil and Tillage Research 113, 89–98.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America Journal 62, 1367–1377.

Six, J., Paustian, K., Elliott, E.T., Combrink, C., 2000. Soil structure and organic matter: I. distribution of aggregate-size classes and aggregate-associated carbon. Soil Science Society of America Journal 64, 681–689.

SSS, 2010. Soil Survey Staff, Keys to Soil Taxonomy, 11th ed. USDA-Natural Resources Conservation Service, Washington, DC.

StatSoft, Inc., 2002. STATISTICA (data analysis software system), version 6. www.statsoft.com.

Taboada, M.A., Barbosa, O.A., Rodríguez, M.B., Cosentino, D.J., 2004. Mechanisms of aggregation in a silty loam under different simulated management regimes. Geoderma 123, 233–244.

Thomaz, E.L., Vestena, L.R., 2012. Measurement of runoff and soil loss from two differently sized plots in a subtropical environment (Brazil). Earth Surface Processes and Landforms 37, 363–373.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science 33, 141–163.

USDA, 2001. Soil Quality Test Kit Guide. Natural Resources Conservation Service, United States Department of Agriculture . (82 pp. http://soils.usda.gov/sqi/assessment/test_kit.html).

USDA-NRCS, 1996. Soil Survey Laboratory Methods Manual. The Soil Survey Analytical Continuum. Soil Survey Investigations Report, 42. US Dept. of Agriculture. Natural Resources Conservation Service, National Soil Survey Center, Washington, D.C.

van Genuchten, M.T., Nielsen, D.R., 1985. On describing and predicting the hidraulic properties of unsaturated soils. Annales Geophysicae 3, 615–628.

van Genuchten, M.Th, Leij, F.J., Yates, S.R., 1991. The RETC Code for Quantifying the Hydraulic Functions of Unsaturated Soils, Version 1.0. EPA Report 600/2-91/065. U.S. Salinity Laboratory, USDA, ARS, Riverside, California . (83 pp.).

Verchot, L.V., Dutaur, L., Shepherd, K.D., Albrecht, A., 2011. Organic matter stabilization in soil aggregates: understanding the biogeochemical mechanisms that determine the fate of carbon inputs in soils. Geoderma 161, 182–193.

Verheijen, F.G.A., Jones, R.J.A., Rickson, R.J., Smith, C.J., 2009. Tolerable versus actual soil erosion rates in Europe. Earth-Science Reviews 94, 23–38.

Walkley, A., Black, A.I., 1934. An examination of Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. Soil Science 37, 29–38.

Wei, C., Gao, M., Shao, J., Xie, D., Pan, G., 2006. Soil aggregate and its response to land management practices, review. China Particuology 4, 211–219.

Zotarelli, L., Alvesc, B.J.R., Urquiagac, S., Boddeyc, R.M., Six, J., 2007. Impact of tillage and crop rotation on light fraction and intra-aggregate soil organic matter in two Oxisols. Soil and Tillage Research 95, 196–206.