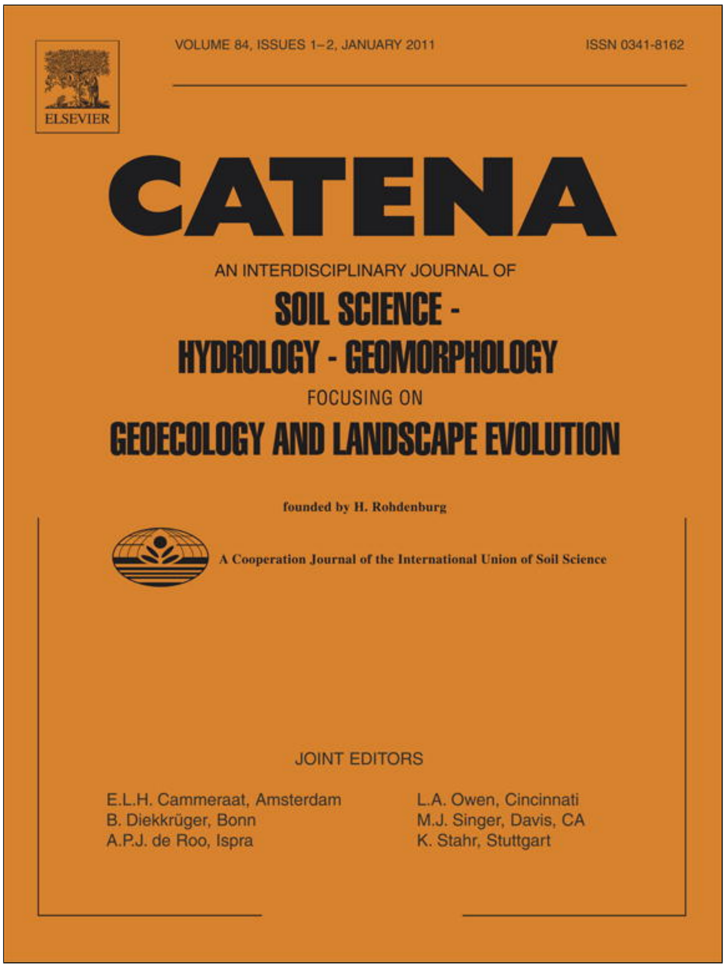

VOLUME 84, ISSUES 1–2, JANUARY 2011 ISSN 0341-8162

ELSEVIER

CATENA

AN INTERDISCIPLINARY JOURNAL OF
SOIL SCIENCE -
HYDROLOGY - GEOMORPHOLOGY
FOCUSING ON
GEOECOLOGY AND LANDSCAPE EVOLUTION

founded by H. Rohdenburg

A Cooperation Journal of the International Union of Soil Science

Catena 84 (2011) 79–88

Contents lists available at ScienceDirect

Catena

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

Environmental impact of introducing plant covers in the taluses of terraces: Implications for mitigating agricultural soil erosion and runoff

V.H. Durán Zuazo [a,*], C.R. Rodríguez Pleguezuelo [b], F.J. Martin Peinado [c], J. de Graaff [d], J.R. Francia Martínez [b], D.C. Flanagan [e]

[a] IFAPA Centro Las Torres-Tomejil, Crtra Sevilla-Cazalla, km 12.2, 41200 Alcalá del Rio (Sevilla) Spain
[b] IFAPA Centro Camino de Purchil, Aptdo. 2027, 18080 Granada, Spain
[c] Dpto. de Edafología y Química Agrícola, Universidad de Granada, C/Severo Ochoa s/n, 18071Granada, Spain
[d] Land Degradation and Development Group, Wageningen University, Droevendaalsesteeg 4, 6708 PB, Wageningen, The Netherlands
[e] USDA-Agricultural Research Service, National Soil Erosion Research Laboratory, 275 S. Russell Street, West Lafayette, IN 47907-2077, USA

ARTICLE INFO

Article history:
Received 16 August 2010
Received in revised form 14 October 2010
Accepted 21 October 2010

Keywords:
Terraces
Erosion
Agricultural runoff
Soil organic carbon
Heavy metals
Water quality

ABSTRACT

South-eastern Spain, and in particular the coastal areas of Granada and Malaga, feature a large area under subtropical crops, with orchards established on terraces built along the slopes of the mountainous areas. The climate, characterized by periodically heavy rainfall, variable in space and time, and with the common agricultural practice of leaving the taluses with bare soil, are the main factors encouraging soil erosion, runoff, and subsequent transport of pollutants. Over a two-year period, six plant covers were applied [(*Thymus mastichina* (Th), *Lavandula dentata* (La), native spontaneous vegetation (Sv), *Anthyllis cytisoides* (An), *Satureja obovata* (Sa), *Rosmarinus officinalis* (Ro)] in comparison to a control of bare soil (Bs) to determine the effectiveness of the covers in reducing soil erosion, runoff, and potential pollution risk by agricultural nutrients (N, P, and K) and heavy metals. Also, carbon losses were monitored in the transported sediments by runoff and in eroded soils. For this purpose, 16 m^2 erosion plots (4 m × 4 m) were laid out in the taluses of the terraces. When the yearly data were compared, the control plot (Bs) showed significantly higher soil erosion and runoff rates (26.4 t ha^{-1} yr^{-1} and 55.7 mm yr^{-1}, respectively) than the treatments with plant covers. The plant covers studied registered the following results in runoff: Ro > Sa > An > Th ≈ La > Sv (41.7, 38.2, 35.5, 16.9, 16.1, and 12.4 mm yr^{-1}, respectively) while annual soil erosion gave the following results: Sa > An > Ro > Th > Sv > La (18.0, 13.5, 13.4, 5.5, 4.4, and 3.2 Mg ha^{-1} yr^{-1}, respectively). This means that Sv reduced runoff and soil-erosion rates compared to Bs by not less than 78 and 83%, respectively. Nevertheless, La and Th plots were also very effective plant covers in reducing runoff and soil erosion (71.2 and 87.8; 69.5 and 79.2%, respectively) in comparison with the Bs plot. The heaviest nutrient losses in runoff and eroded soils were found in Bs and the lowest in the La, Th, and Sv plots. Bs and Ro plots registered the highest carbon losses (829.9 and 652.1 kg ha^{-1}, respectively), the lowest carbon-loss rates being measured in La, Sv, and Th plots (145.2, 140.3, and 109.3 kg ha^{-1}, respectively). The results indicate that heavy metals (Mn, Cr, Co, Ni, Cu, Zn, Mo, Cd, and Pb) in these types of agroecosystems may also be a potential pollutant due to transport by agricultural runoff. There was a major reduction of heavy-metal transport by plant covers in relation to the control of bare soil. The results of this research support the recommendation of using plant covers with multiple purposes (aromatic–medicinal–culinary) on the taluses of subtropical-crop terraces in order to reduce erosion and pollution risk.

1. Introduction

Soil has been termed by the International Soil Science Society as a "limited and irreplaceable resource". Without this resource, the biosphere would collapse, with devastating effects on humanity. In this sense, soil erosion by water is the detachment of soil particles by the direct action of raindrops and runoff water, and the transport of these particles by splash and very shallow flowing water to small channels or rills (Durán et al., 2010). This environmental problem ranks as one of the most serious problems worldwide and its effects are long lasting (Pimentel et al., 1995), exerting both physical and chemical effects. Physical effects involve soil erosion from cultivated fields and deposition in streams and water bodies, while chemical effects involve the loss of plant nutrients and other chemicals (Stroosnijder, 1995). The removal of these nutrients by erosion leads to negative nutrient balances and reduces land productivity

* Corresponding author.
E-mail address: victorh.duran@juntadeandalucia.es (V.H.D. Zuazo).

0341-8162/$ – see front matter
doi:10.1016/j.catena.2010.10.004

(Van den Bosch et al., 1998). In addition, most of the organic matter is close to the soil surface in the form of decaying leaves and stems, and therefore topsoil erosion also depletes soil organic matter. To date, all carbon-budget calculations have relied on the assumption that there are additions to the soil-carbon pool in solid forms, the only losses are gaseous. Recently, this has been recognized as erroneous, since soils and landscapes are dynamic (Lobb et al., 2002). Transported soil material by erosion contains carbon and therefore influences the cycling of this element in soils (Lobb et al., 2002).

The Mediterranean climate is characterized by unpredictable rainfall fluctuations from year to year with high-intensity rainfall events, increasing the vulnerability to erosion. Soil erosion is one of the major environmental problems in several areas of Spain, which have been described as the most threatened in Europe (Vallejo et al., 2005). This fact can be considered to be the result of various factors: fragile natural ecosystems (irregular terrain with steep slopes), long-period of human exploitation, land misuse, and land abandonment (Kosmas and Danalatos, 1993; Thornes, 1996; Kosmas et al., 2000). These processes have been varying in space and time at least for the last 4000 years within the Mediterranean basin (Brandt and Thornes, 1996). Concretely in Spain, more than 22 million ha (43.8% of the land) are affected by erosion rates higher than 12 Mg ha^{-1} yr^{-1}, exceeding the tolerable limit of soil formation (Rojo, 1990). In 2006, 12.6% of the land was affected by erosion rates higher than 50 Mg ha^{-1} yr^{-1}, and 34.1% of the land had erosion rates from 10 to 50 Mg ha^{-1} yr^{-1} (DGB-MMA, 2008).

Many mountainous areas in Spain have been terraced during the last few decades, and especially since the admission to the European Union in 1986, which has been one of the main driving forces for agriculture development. According to Durán et al. (2003), along the coast of Granada (SE Spain), intensive irrigated agriculture has been established on these terraces on steep slopes with subtropical crops [avocado (*Persea americana* Mill.), mango (*Mangifera indica* L.), cherimoya (*Annona cherimola* Mill.), litchi (*Litchi chinensis* Sonn.), and others]. The detached soil from the taluses of orchard terraces accumulates on the platform of the terrace below, hindering manual fruit harvesting and orchard maintenance. In this sense, talus erosion, making terrace reconstruction necessary, poses a serious economic challenge for farmers. Local farmers usually eliminate vegetation from the taluses of the terraces because most of plants are weeds. In addition, the importance of vegetation in controlling erosion and runoff is widely accepted. The relation between erosion and vegetation is the result of various complex processes that act at different time scales (Coppin and Richards, 1990; Morgan et al., 1986). In the short term, vegetation influences erosion mainly by intercepting rainfall and protecting the soil surface against the impact of rainfall drops, and by intercepting runoff. In the long term, vegetation influences the fluxes of water and sediments by increasing the soil-aggregate stability and cohesion and by improving water infiltration (Bochet et al., 2006; Durán et al., 2008; Durán and Rodríguez, 2008).

When soil is eroded, plant nutrients such as nitrogen (N), phosphorus (P), and potassium (K) are lost. Since topsoil is usually relatively rich in nutrients, eroded soil typically contains about three times more nutrients than the soil left on the eroded land. Therefore, to offset the damages that erosion inflicts on crops, large quantities of fertilizers are intensively used. These extra inputs can harm human health and pollute the environment (Pimentel et al., 1995).

High P concentrations in surface waters are a major cause of eutrophication, with detrimental impact on water quality, since P is usually the nutrient that limits algae growth in freshwater bodies. In agricultural terms, P losses represent a decline in nutrients for the system, to which the farmers usually attach little importance due to the low prices of fertilizers. However, from an environmental perspective, these losses can mean a serious deterioration in water quality. On the other hand, nitrate (NO_3) is a common chemical pollutant in agricultural areas. In contrast to P, the NO_3 is highly soluble and generally does not adsorb to soils. Rather, NO_3 tends to move with water into the soil profile. In general, nutrient losses are expected to be reduced in soil-management systems that preserve plant residues. The third major nutrient, potassium (K) is an important nutrient in fruit production, and therefore local farmers tend to apply heavy amounts of this element to encourage good-quality fruit.

Apart from theses three major nutrients, the increased inputs of heavy metals in soil have also received attention, since transport of these elements may result in increased contents of heavy metals in groundwater or surface water (Alloway, 1995; Moore et al., 1998). Heavy metals can be included in commercial fertilizers and other agrochemicals. Soils receiving repeated applications of these products could show increases in heavy-metal concentration in runoff (Moore et al., 1998).

An understanding of how vegetation disturbance and the construction of terraces for subtropical-crop cultivation on the coast of Granada affects runoff and soil erosion is urgently required in order to focus soil-management on mitigating soil erosion and thereby move towards sustainable agriculture. The aim of this study was to test, under field conditions, the response of runoff, soil erosion, nutrient, carbon losses, and heavy-metal transport to different plant covers, including aromatic and medicinal plants and native vegetation during two hydrological years.

2. Materials and methods

2.1. Description of the study area

The study area is located in the south-eastern part of the Iberian Peninsula (Lat 36°48′00″N, Long 3°38′0″W) (Fig. 1), some 7 km north of the Mediterranean coast at Almuñécar (Granada, SE Spain) at 183 m a.s.l. The relief is rough and steep, and most of the area presents slopes steeper than 30%, as reported at the plot and watershed scale by Rodríguez et al. (2009a). The study terrace, representative of those commonly found in the study area, is a reverse-sloped bench-terrace type with a toe drain measuring 160–170 m long. The platform was 2–3 m wide and the talus 3–5 m high. The platform had a single row of bearing mango trees (*Mangifera indica* L. cv. Keitt) spaced 3 m apart. Local temperatures are subtropical to semi-hot within the Mediterranean subtropical climatic category. The average annual rainfall in the study zone is 449.0 mm. The soils, formed from weathered slates, vary in depth, and some are rocky, providing generally very good drainage, especially in the fill used to construct the platforms. The soils of the zone are Typical Xerorthent (Soil Survey Staff, 1999). The main characteristics of these soils are presented in Table 1.

2.2. Experimental field design

Fourteen closed plots of 4 m × 4 m (16 m^2) each were established on the taluses of the terraces. They were sufficiently wide to minimize edge or border effects. Each plot consisted of a galvanized enclosure, drawer collector, sediment and runoff collector, and tanks for storing runoff. The boundaries of each plot were defined by 50 cm galvanized steel sheets and inserted up to 20 cm below the soil surface to prevent soil from leaving or entering the plot. To avoid the effects of position, all fourteen plots were established in one line, oriented parallel to the slope and adjacent to each other (Fig. 2).

Five types of aromatic–medicinal–mellipherous plants were used as covers: *Thymus mastichina* L. (Th) *Lavandula dentata* L. (La), *Satureja obovata* Lag. (Sa), *Anthyllis cytisoides* L. (An), and *Rosmarinus officinalis* L. (Ro) each replicated twice. Two years before beginning this study the covers on were planted the taluses of orchard terraces. The planting grid was 40 × 40 cm, with approximately 81 introduced plants per plot. Also, two of the erosion plots were left with native spontaneous vegetation growing in the study area (a spontaneous mixture of annual herbaceous weeds: *Papaver rhoeas*, *Convolvulus* sp.,

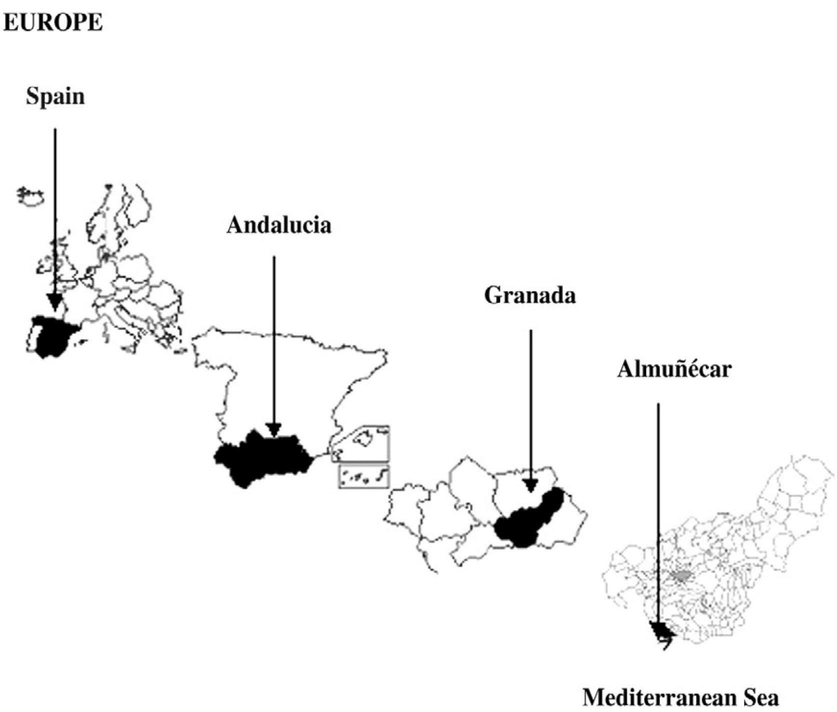

Fig. 1. Location of the study area in south-eastern Spain (Almuñécar, Granada).

Malva sylvestris, *Reseda phyteuma*, *Anacyclus* sp., *Sinapis arvensis*, *Medicago* sp., *Chrozophora* sp., *Taraxacum officinale*, *Chenopodium* sp., *Poa annua*, *Bromus* sp., etc.). Finally, two erosion plots were left with bare soil as a control.

The climatic data were taken from a local weather station (<20 m from the plots). For each of the events, maximum intensity at 30 min (I_{30}), and kinetic energy were calculated ($KE = 210 + 89\ log_{10}\ I$) (Wischmeier and Smith, 1978; Brandt, 1990). The erosion index of a particular event was calculated by multiplying the kinetic energy of the rain by its maximum intensity (Wischmeier, 1976).

2.3. Field work and laboratory analysis

Runoff, sediments, and eroded soil were collected at the base of each plot. The runoff in each tank was measured and sampled after each rainfall event. Sediment concentration in runoff was determined in aliquots, which were decanted and dried at 105 °C. Sediment yield was calculated by multiplying the runoff volume (total water in the tanks) by the average sediment concentration. Eroded soil was collected directly by sampling from drawer collector of closed erosion plot.

Table 1
Physico-chemical parameters of soil from the taluses of orchard terraces at 15 cm depth (n = 28).

Soil characteristics	
Slope	214%
Boulders	Slight
Textural class	Loamy sand
Sand (g kg^{-1})	684 ± 79
Silt (g kg^{-1})	228 ± 39
Clay (g kg^{-1})	88 ± 19
pH (H_2O)	7.7 ± 0.4
Organic matter (g kg^{-1})	7.9 ± 2.1
Nitrogen (g kg^{-1})	0.4 ± 0.2
Available P (mg kg^{-1})	9.0 ± 2.1
Available K (mg kg^{-1})	175 ± 21

Nutrient loss in runoff was expressed by the following equation:

$$\text{Total load} = \sum \text{nutrient conc.}\left(\text{mg L}^{-1}\right) \times \text{Total runoff depth(mm)} \tag{1}$$

Nutrient loss in sediment was expressed by the following equation:

$$\text{Total load} = \sum \text{nutrient conc.}\left(\text{mg kg}^{-1}\right) \times \text{Weight of sediments}\left(\text{kg m}^{-2}\right) \tag{2}$$

Each runoff sample was analysed for NO_3^-, NH_4^+, PO_4^{3-}, and K in accordance with standard methods for the examination of waters (APHA, AWWA, WPCF, 1995) and each sediment sample was analysed for N, P, and K plant-available content following standard methods for soil analysis (MAPA, 1994).

In addition, the heavy-metal concentration was also determined in each runoff sample by inductively coupled plasma mass spectrometry (ICP-MS) with a Perkin Elmer SCIEX ELAN-5000A spectrometer.

A representative sub-sample of the sediment was air-dried and analysed for organic carbon by weight differences after combustion at 550 °C for 2.5 h (Head, 1984).

In each field plot, soil-surface samples (0–15 cm) were taken at the beginning of the study, and after 12 and 36 months in all the plant covers in order to study the evolution of soil organic matter, using standard soil-examination methods (MAPA, 1994). All soil samples were previously passed through a 2-mm sieve to remove litter and stones, mixing the three-samples of each plot, obtaining a homogeneous sample.

Plant-cover percentage was estimated following the method of Agrela et al. (2003), using a 1 m^2 grid with 100 squares. This consists of evaluating the different cover percentages estimated in each of the squares on a scale of 0 to 5, thus obtaining a value matrix, the mean of which indicated the plot cover percentage.

2.4. Statistical procedures

Analysis of variance (ANOVA) was performed in order to ascertain whether differences in runoff and sediment yield existed among the

Fig. 2. Closed erosion plots in the taluses of orchard terraces with the different plant covers.

different plant-cover types. The runoff, soil erosion, and nutrient losses were selected for the measured variables (dependent variables), and the plant-cover types were the controlled variables (independent variables). Differences between individual means were tested using the least significant difference test (LSD) at $p<0.05$.

Irrespective of this, data from rainfall, I_{30} and EI_{30} vs. runoff, eroded soil, and sediment concentration from the overall rainfall events and both assessed seasons are presented, assessing their relationship through the correlation coefficient (r) of each plant cover.

3. Results and discussion

3.1. Rainfall characteristics for the study period

Statistical characteristics of the rainfall depth, I_{30} and EI_{30} of the erosive rainfall events (with generation of runoff) during the study period are shown in Table 2. Total erosive rainfall for the first and the second hydrological year was 250.4 and 410.6 mm, respectively, of which only 15 and 18 caused soil erosion. The first year was relatively dry, with a lower cumulative annual rainfall than the mean over the last 30 years for the area (449.0 mm), but the second year had higher rainfall. This temporal variability with rainfall as stated by Quadrelli et al. (2001) could be related with significant trends that have been found over most European areas during the winters considered. The associated pattern is characterized by a general and substantial decrease in rainfall over southern Europe. Monthly rainfall amounts varied markedly, with very dry conditions in July and August of both years and wetter months, in November (63.7 mm) of the first year and September (117.1 mm) of the second year. These rainfall events were characterized by a low mean rainfall intensity of 87 and 61% for the first and the second year, respectively, with a mean intensity <5 mm h^{-1} while only 0 and 22% for the first and the second year, respectively, had a mean intensity >10 mm h^{-1}. The high variability of monthly rainfall from one year to another is reflected in the recorded data. The inter-seasonal, as well as the interannual rainfall variability is also clearly displayed (Fig. 3).

Table 2
Statistical characteristics of rainfall for both years.

	Rainfall (mm)	I_{30} (mm h^{-1})	EI_{30} (MJ mm ha^{-1} h^{-1})
Year 1			
Average	16.7 ± 9.5	2.8 ± 2.6	7.3 ± 7.9
Max.	42.8	9.1	26.9
Min.	5.0	0.3	0.5
Total	250.4	41.8	110.0
Events	15	15	15
Year 2			
Average	22.8 ± 24.9	8.4 ± 13.5	26.9 ± 49.9
Max.	107.9	58.7	215.6
Min.	4.4	0.2	0.30
Total	410.6	151.6	485.9
Events	18	18	18

I_{30}, Maximum intensity at 30 min; EI_{30}, erosivity index; ± standard deviation.

3.2. Runoff and soil-erosion response

According to a comparison of the results for the yearly data, the control plot (Bs) (the common practice by farmers to maintain the soil of the taluses by applying herbicides) had significantly higher rates of soil erosion and runoff than did the rest of the treatments with plant covers (26.4 Mg ha^{-1} and 55.7 mm yr^{-1}, respectively). The plant covers studied gave the following results in runoff: Ro>Sa>An>Th≈La>Sv (41.7, 38.2, 35.5, 16.9, 16.1, and 12.4 mm yr^{-1}, respectively) whereas annual soil-erosion rates gave the following trends: Sa>An>Ro>Th>Sv>La (18.0, 13.5, 13.4, 5.5, 4.4, and 3.2 Mg ha^{-1} yr^{-1}, respectively). This signifies that Sv reduced runoff and soil erosion with respect to Bs by 78 and 83%, respectively. On the other hand, La and Th were also very effective plant covers in reducing runoff (71 and 88%, respectively) and soil erosion with regard to Bs (70 and 79%, respectively). Our results for annual soil erosion on bare soil were much higher than those reported by Bautista (1999) in Alicante (SE, Spain) for closed erosion plots and natural rainfall (0–8 Mg ha^{-1} yr^{-1}), Castillo et al. (1997), and Romero et al. (1998, 2000) in Murcia (0.012–1.84 Mg ha^{-1} yr^{-1}) and by Durán et al. (2005) for the same area in bare soil (9.1 Mg ha^{-1} yr^{-1}). This was due to the extraordinarily aggressive erosive event (107.9 mm and 58.7 mm h^{-1} for rainfall depth and I_{30}, respectively) registered in the second year of study, which significantly boosted the annual rate of erosion and runoff. These high erosion rates are very common on steep sloping land with land-use changes from a natural landscape to agricultural systems (Lal, 1990). Fig. 4 presents the analysis of variance concerning the effect of the plant covers on the average runoff and soil erosion. The lowest soil-erosion rates were recorded under Th and Sv (0.14 and 0.17 Mg ha^{-1}, respectively), these values differing significantly with respect to the other plant covers tested. Bs was the treatment that showed the highest erosion rates (2.36 Mg ha^{-1}). In terms of runoff, significantly lower values for Sv and Th were recorded in comparison with Bs (0.7, 0.9, and 3.3 mm,

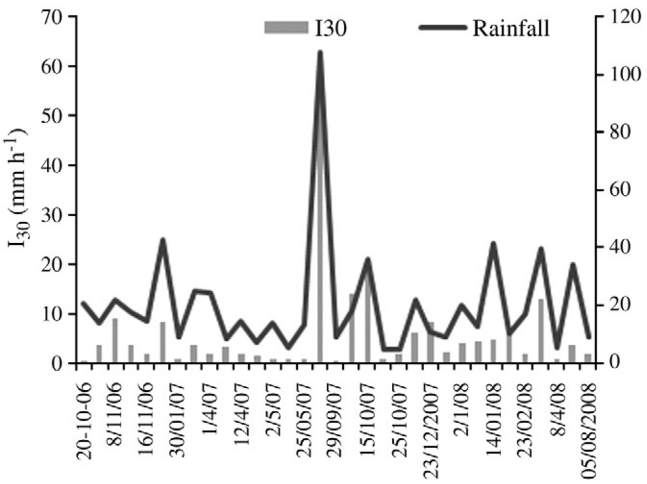

Fig. 3. Rainfall depth and maximum intensity at 30 min (I30) for the erosive events during the two hydrological years.

respectively). However, the rest of the plant covers (La, An, Sa, and Ro) did not significantly differ from each other (Fig. 4). The trend for runoff and for soil erosion was found to be higher during the second study year, when the highest rainfall was recorded. Compared to bare soil, Th and Sv reduced the runoff with 94 and 93%, and reduced erosion with 71 and 79%, respectively. The least effective treatment against soil erosion among the plant covers was Sa, which resulted in a reduction of only 39%, while the least effective to combat runoff was Ro, which gave a reduction of only by 26%, with respect to Bs (Fig. 4). In general, the plant covers softened the mechanical impact of the raindrops on the soil surface of the taluses, diminishing the surface runoff, thereby aiding soil conservation. The importance of vegetation in erosion control is attributed to two main effects: on the one hand, the direct protection of the soil surface by the canopy and litter covers that intercept rainfall, and on the other hand the indirect improvement of the soil physical and chemical properties, essentially through the incorporation of organic matter (García et al., 1995; Bochet et al., 1998).

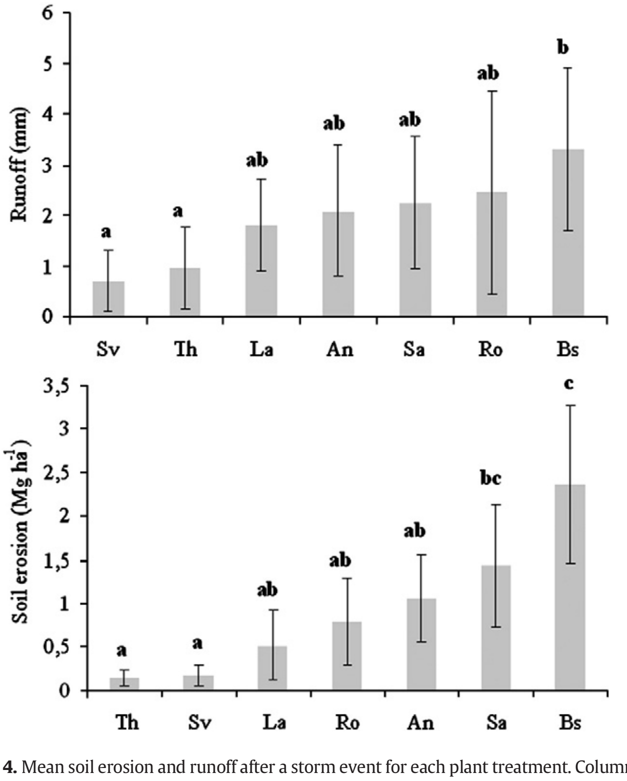

Fig. 4. Mean soil erosion and runoff after a storm event for each plant treatment. Columns with different letters are statistically different at the level 0.01 (LSD). Sv, Spontaneous vegetation; Th, *Thymus mastichina*; La, *Lavandula dentata*; An, *Antyllis cytisoides*; Sa, *Satureja obovata*; Ro, *Rosmarinus officinalis*; Bs, bare soil. Vertical bars represent standard deviation (n = 33).

The measurements made on the erosion plots showed that in all plant covers, runoff started to occur with rains of over 5–15 mm (Fig. 5). From these data, linear relationships were established between the amount of rainfall and the runoff. A more detailed summary of the relationships between soil erosion and runoff is shown in Table 3. The runoff correlated better with soil erosion for some plant covers (*Satureja obovata, Anthyllis cytisoides*, and *Rosmarinus officinalis*), and for bare soil. The remaining plant covers in general presented less relationship with the parameters studied. The highest percentage of soil covered correlated with the lowest runoff and soil erosion rates. Fig. 6a shows the evolution of the percentage of soil covered during the two-year monitoring period by each type of plant during the study period and the relationship between soil erosion (Fig. 6b) and runoff (Fig. 6c) with this percentage. Sa, An, and Ro plots were the plant covers with the lowest percentage of soil covered and therefore showed the highest soil erosion and runoff rates. On the other hand, Sv, Th, and La, covered the soil more efficiently, ameliorating the production of soil erosion. This agrees with Thurow et al. (1986) and Hofmann and Ries (1991), who reported that erosion rates increase with a decrease in the amount of plant cover. Therefore, native vegetation, with its greater cover, produces more biomass and thus augments the organic-matter content and structural stability of the soil. In this sense, Table 4 shows the average soil organic matter (SOM) percentage after 12 and 36 months of installing the plant covers. Plant covers increased SOM with time, this being higher in the Sv plot, followed by La and Th (Table 4) and lower in Sa and finally Bs plots. This low SOM content in Bs when compared with the rest of treatments was due to the easy breakdown of soil aggregates, being more exposed to soil erodibility (Fullen 1992; Fenton et al., 2005). All plant covers provided greater soil organic-matter content with respect to the initial situation, especially for the spontaneous vegetation. In contrast with Bs, which decreased with respect to the initial content, this is related to high rates of soil erosion in this plot. In this context, Rodríguez et al. (2009b) reported the significant effect of litter decomposition and nutrient release of plant covers used for erosion control in the taluses of orchard terraces, which enhanced soil fertility. This increase in SOM and therefore soil quality as result of plant covers agrees with many other authors (Andreu et al., 1998; Sánchez et al., 2002; Durán et al., 2006). Therefore, the benefits from plant covers are crucial for the improvement of soil quality.

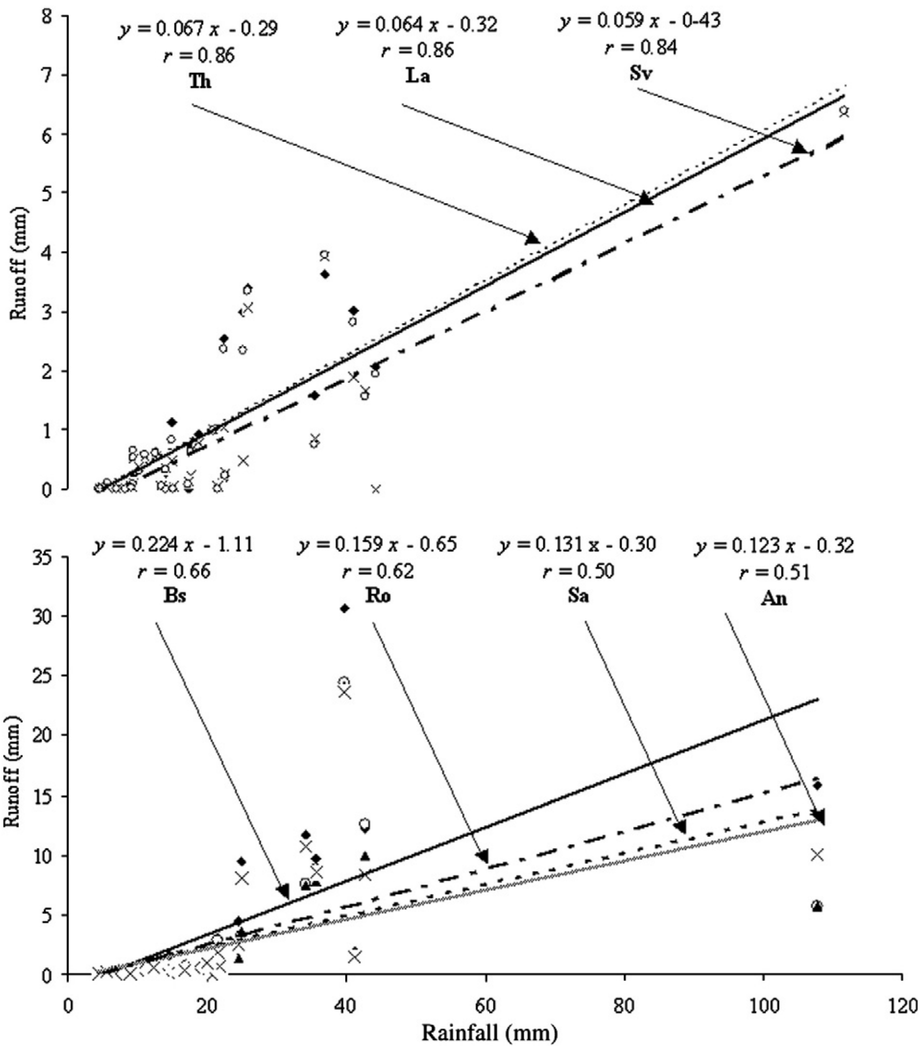

Fig. 5. Runoff (mm) vs. rainfall (mm) for the six plant cover treatments and the control plot (Bare soil). Sv, *Spontaneous vegetation*; Th, *Thymus mastichina*; La, *Lavandula dentata*; An, *Antyllis cytisoides*; Sa, *Satureja obovata*; Ro, *Rosmarinus officinalis*; Bs, bare soil.

3.3. Nutrient losses control

Table 5 lists the results for the N–NO_3, N–NH_4, P–PO_4 and K losses per area by runoff and N, P, and K losses per area in sediments and eroded soil. The greatest total N–NO_3 losses per area were recorded in Bs plot, while the lowest were measured in Th and La. The NO_3 losses showed the following order: Bs>Ro>An>Sa>Sv>Th>La. However, N–NH_4 followed the pattern: Bs>Sa>An>La>Ro>Sv>Th. For P–PO_4 and K, the highest losses were again recorded in Bs. The results of the present study indicate that the Bs plot had the highest rate of nutrient losses in terms of runoff per area and the lowest were recorded in Th, Sv, and La, with the exception of K, for which the lowest loss rates were found in the An and Sa plots.

In general, the transported amount of N–NH_4 per area was lower than for N–NO_3. The dominance of N–NO_3 in the three plots suggests that the dissolved nitrogen in the runoff came mainly from N fertilizers applied on the platform of the terraces for fruit cultivation, rather than from the soil, and the same applies to the dissolved NH_4^+ contained in the runoff. Our results for N–NO_3 annual losses from bare soil were much lower than those reported by Ramos and Martínez (2006) in vineyards. On the contrary, P–PO_4 losses recorded in this experiment (from 0.012 to 0.040 kg ha^{-1} yr^{-1} for Th and Sv plots, respectively) were similar to those found by Francia et al. (2006), who reported rates from 0.07 to 0.29 kg ha^{-1} yr^{-1} in olive orchards under different land management and similar to those of Ramos and Martínez (2006). K losses in runoff ranged from 106.3 to 289.2 mg m^{-2} yr^{-1} for An and Bs, respectively (Table 5), and from 21.1 to 100.5 mg m^{-2} yr^{-1} for La and Bs, respectively in sediments and eroded soil. These K losses rates were lower than those reported by Francia et al. (2006) in olive orchards (47.0–333.8 mg m^{-2} yr^{-1}). This appreciable amount of dissolved

Table 3
Relationship between runoff (mm) and soil erosion (Mg ha^{-1}) for all the treatments.

	Thymus mastichina	*Lavandula dentata*	Spontaneous vegetation	Bare soil	*Satureja obovata*	*Anthyllis cytisoides*	*Rosmarinus officinalis*	All treatments
r	0.51	0.66	0.68	0.79	0.95	0.62	0.80	0.70
R^2	0.26	0.44	0.46	0.62	0.92	0.39	0.63	0.48
p	**	***	***	***	***	***	***	***
Equation	$y=3.7+6.2x$	$y=-1.9+12x$	$y=1.4+10.3x$	$y=4.0+16.7x$	$y=1.4+8.5x$	$y=9.5+8.0x$	$y=4.3+3.2x$	$y=2.3+10.4x$

**$p<0.01$; ***$p<0.001$.

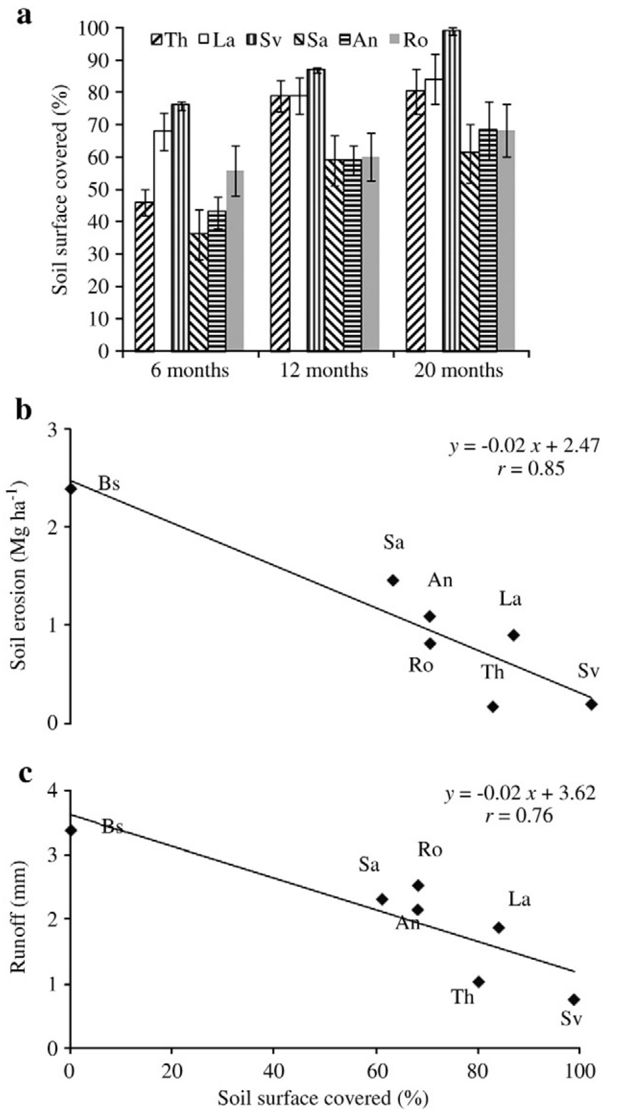

Fig. 6. Soil coverage by different plant cover and its influence on soil erosion and runoff. Soil surface covered by plant covers throughout the study period (a). Relationship between soil erosion and soil surface covered (b). Relationship between runoff and soil surface covered (c).

potassium resulted in K-rich runoff, and came presumably from K fertilizers (K_2SO_4, KH_2PO_4, KCl, and KNO_3) used for fruit production. Total nutrient losses in agricultural systems may be affected by various factors: soil-management, plant covers as well as rainfall characteristics. Bare-soil plots produced the highest nutrient losses, which are diminished by the use of plant covers, most effectively by La, Sv, and Th, except for K losses, which were greatest in the An, Sa, and Th plots.

3.4. Nutrient concentration in runoff

The average N–NO_3 concentration in the runoff ranged from 4.9 to 24.3 mg L^{-1} for La and Sv, respectively, and showed the following

Table 5
Annual nutrient losses with runoff, sediments and eroded soil under the different plant covers.

Plant cover	In runoff				In sediment and eroded soil		
	NO_3–N	NH_4–N	PO_4–P	K	N	P	K
	(mg m^{-2} yr^{-1})						
Th	148.2	7.4	1.2	179.4	218.8	7.8	44.1
La	80.6	19.2	1.9	214.8	78.9	3.8	21.1
Sv	302.5	11.7	1.9	216.2	187.9	12.1	99.3
Sa	312.8	40.0	2.5	114.3	796.0	19.9	63.7
An	334.5	24.5	3.6	106.3	778.0	17.0	34.9
Ro	366.2	17.6	3.1	227.6	528.0	15.8	52.9
Bs	367.8	44.1	4.0	289.2	1025.4	34.8	100.5

Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, Spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

order for the respective plant covers: Sv>Ro>An>Bs>Sa>Th>La (Table 6). The maximum concentration rates detected for a storm event exceeded 50 mg L^{-1} in Sv, which is the permissible limit for drinking water according to the WHO (2007). However, in most of the events, N–NO_3 concentrations in runoff exceeded the 10 mg L^{-1} upper limit recommended for drinking water by the U.S. EPA (1976) as well as for irrigation waters reported by Ayers and Westcot (1987). In addition, in most of the events recorded for Sv, the concentration was within the class 20–50 mg L^{-1}, which is a high enough concentration to indicate the influence of human activities, according to Spalding and Exner (1993). Average N–NH_4 concentrations in runoff ranged from 0.43 to 1.60 mg L^{-1} for Th and Sv, respectively, exceeding 0.5 mg L^{-1} in most of the erosive events and for all the plant covers, this concentration being standard for public water supplies (Huetter, 1992) and the 5 mg L^{-1} for irrigation (Ayers and Westcot, 1987). Average P–PO_4 concentrations in the runoff ranged from 0.05 to 0.20 mg L^{-1}, the highest average value being reached in La and An, and the lowest in Th (Table 6). In most of the events and for all the treatments, the concentration exceeded established limits usually associated with the eutrophication of surface waters: from 0.01 mg P L^{-1} (Vollenweider, 1968; Vollenweider and Kerekes, 1980) to 0.05 mg L^{-1} (U.S. EPA, 1976). However, the P concentration in runoff was well below the recommended level of 2 mg L^{-1} for agricultural use (Ayers and Westcot, 1987).

The highest average K concentrations were registered in La, Sv, and Ro and the lowest in Sa and An plots. The upper limit recommended for drinking water of 12 mg L^{-1} (Griffioen, 2001) was exceeded for all the plant covers studied. K concentrations were rather high because this element is relatively mobile and, although K does not directly result in eutrophication, the impact and risk as a potential pollutant when applied as fertilizer should be taken into account. The excessive use of K fertilizers (K_2SO_4 and KNO_3) for improving subtropical fruit quality is a potential source of pollution (Shinde et al., 2006).

In these steep areas (214% of slope in the present study), intensive irrigated agriculture incorporate substantial amounts of plant nutrients with high risks of transport by nutrient-enriched runoff during the rainfall period to lowlands.

Table 4
Average percentage of soil organic matter (0–5 cm) under each plant cover*.

Time (months)	Th	La	Sv	Bs (g kg^{-1})	Sa	An	Ro
12 (After planting)	7.6 (±2.3)	8.7 (±3.7)	11.4 (±3.1)	7.4 (±3.4)	7.7 (±2.7)	8.2 (±3.4)	8.6 (±2.2)
36	7.9 (±1.7)	9.1 (±2.8)	12.3 (±2.2)	6.3 (±1.7)	8.1 (±1.9)	8.6 (±2.1)	9.3 (±3.3)

*The plant covers were introduced in the taluses of terraces two years before beginning the present study; (±Standard deviation); Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, Spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

Table 6
Average nutrient concentration in the runoff for the different plant covers.

Plant cover	$N–NO_3$	$N–NH_4$	$P–PO_4$	K
	(mg L^{-1})			
Th	8.9 ± 10.0 (29.1)	0.4 ± 0.4 (1.4)	0.06 ± 0.05 (0.15)	10.9 ± 11.4 (43.3)
La	4.5 ± 6.1 (24.1)	1.8 ± 1.7 (4.7)	0.20 ± 0.21 (0.71)	28.7 ± 33.7 (130.0)
Sv	24.3 ± 19.3 (69.3)	1.6 ± 2.4 (7.8)	0.09 ± 0.05 (0.17)	24.1 ± 14.5 (56.0)
Sa	9.6 ± 6.6 (22.5)	0.5 ± 0.4 (1.7)	0.09 ± 0.03 (0.23)	5.8 ± 4.9 (21.3)
An	11.2 ± 8.4 (29.8)	0.9 ± 0.9 (3.4)	0.10 ± 0.06 (0.27)	5.2 ± 4.6 (22.0)
Ro	11.8 ± 7.8 (29.4)	0.8 ± 1.2 (5.2)	0.07 ± 0.03 (0.11)	12.6 ± 8.4 (34.0)
Bs	10.5 ± 12.9 (49.3)	1.5 ± 1.8 (6.3)	0.09 ± 0.05 (0.20)	12.7 ± 10.8 (35.0)

± Standard deviation; values between parenthesis are maximum concentration; n = 33. Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, Spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

3.5. Control of carbon export

Table 7 shows the total amount of organic carbon (C) lost by sediments and eroded soil per year. In general, our results for carbon losses are low due to the low content of organic matter in this soil. During the second study year, C losses were higher than in the first year. During the first year, total C losses by sediments and eroded soil followed the pattern: Bs > Sa > Th > Sv > Ro > La > An. Bs registered the greatest losses (202.5 kg ha^{-1}), and An the least (55.4 kg ha^{-1}) (Table 7). During the second year, this trend changed: Bs > Ro > Sa > An > La > Sv > Th. The Bs and Ro plots sustained the heaviest C losses (829.9 and 652.1 kg ha^{-1}, respectively), while the lightest C losses were measured in La, Sv, and Th plots (145.2, 140.3, and 109.3 kg ha^{-1}, respectively). Therefore, the An and La plots reduced C losses by 73 and 69%, respectively, with respect to Bs losses during the first year. Similarly, for the second year, Th, Sv, and La plots reduced C transport by 89, 85, and 85%, respectively, with respect to Bs plot. C losses in sediments transported by runoff ranged from 5.0 to 829.9 kg ha^{-1} for Sv and Bs plots (both values for the second year). These results are lower than those found by Bertol et al. (2005) for soybean and oats, who reported 16, 36, 152, and 1779.9 kg ha^{-1} for non-tillage, minimum tillage, conventional tillage and bare soil, respectively, but much higher than those recorded by Dos Santos et al. (2007), reporting from 0.2 to 55 kg ha^{-1} for different soil-management systems in semi-arid environments. It is well known that organic matter is among the first soil components to be removed by water erosion, not only because of its higher concentration on soil surface but also for its low density (Barrows and Kilmer, 1963). Thus, protection of the soil surface against organic-carbon losses by sediment is feasible by using plants covers, this being

Table 7
Annual organic-carbon losses by sediments and eroded soil during the study period.

Plant cover	In sediment*	In eroded soil (kg ha^{-1})	Total
Year 1			
Th	47.2 ± 18.4	51.0 ± 14.6	98.2
La	34.4 ± 15.4	28.4 ± 10.7	62.8
Sv	6.3 ± 2.2	70.3 ± 22.4	76.6
Sa	69.7 ± 21.1	51.2 ± 15.5	120.9
An	27.3 ± 11.8	28.7 ± 12.4	56.0
Ro	29.1 ± 17.4	33.9 ± 17.8	63.0
Bs	79.6 ± 11.9	123.1 ± 36.8	202.7
Year 2			
Th	45.2 ± 14.7	64.0 ± 42.1	109.2
La	74.1 ± 22.7	71.6 ± 25.4	145.7
Sv	5.0 ± 2.4	90.4 ± 19.5	95.4
Sa	445.9 ± 157.4	95.9 ± 30.8	541.8
An	332.7 ± 100.4	97.0 ± 17.4	429.7
Ro	652.1 ± 89.7	115.6 ± 24.1	767.7
Bs	829.9 ± 214.5	130.3 ± 41.9	960.2

± Standard deviation; *suspended sediment in runoff. Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

Table 8
Heavy-metal concentration in runoff from the different erosion plots.

	Mn							Cr							Co						
	(µg L^{-1})																				
Plant cover	Th	La	Sv	Bs	Sa	An	Ro	Th	La	Sv	Bs	Sa	An	Ro	Th	La	Sv	Bs	Sa	An	Ro
Average	305.9	224.1	384.1	179.3	307.19	170.9	192.8	1.5	2.2	3.0	2.4	9.0	1.8	0.8	4.5	5.1	4.5	3.8	5.5	3.3	0.8
SD	596.0	287.5	855.1	425.0	837.79	318.4	447.2	2.0	2.7	5.1	4.5	35.9	3.5	1.3	4.8	6.1	5.6	5.6	14.6	5.2	1.3
Max.	2454.8	1112.6	3723.1	1889.8	3634.82	1237.8	1908.9	6.57	10.0	14.8	14.8	175.7	13.8	5.3	15.8	22.0	17.3	17.3	70.7	19.8	5.3
Min.	0.4	2.2	0.2	1.06	0.5	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.0
	Ni							Cu							Zn						
Average	4.1	20.4	2.8	2.4	11.2	3.5	2.9	4.8	6.4	5.6	7.2	78.2	6.6	5.3	3694.6	3820.9	1533.9	1421.0	3009.7	2289.6	1300.8
SD	7.6	78.8	3.1	2.2	41.5	5.7	7.6	3.3	5.0	5.3	6.3	348.7	4.8	3.2	3954.5	3610.5	1784.1	1087.1	4597.9	2118.2	1090.3
Max.	31.8	355.3	9.4	9.4	203.5	25.6	38.2	15.1	22.2	17.2	24.0	1706.3	19.8	12.6	10,006.0	11,600.0	5768.2	3966.5	20,446.7	7979.9	4236.5
Min.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	1.3	0.0	0.5	0.2	0.8	1.2	6.7	27.2	0.0	2.7	2.2	6.3	1.9
	Mo							Cd							Pb						
Average	2.5	2.6	3.4	2.5	3.4	3.0	3.2	0.1	0.2	0.0	0.1	12.0	0.1	0.0	0.6	0.7	0.5	0.4	1.5	0.2	0.1
SD	3.2	1.6	2.6	2.2	3.8	3.0	2.6	0.2	0.9	0.0	0.3	57.6	0.1	0.0	0.7	1.2	1.1	0.7	6.0	0.7	0.4
Max.	12.8	6.3	8.0	7.8	18.8	14.2	10.2	0.9	4.1	0.1	1.5	282.5	0.2	0.1	2.0	3.5	4.0	2.7	28.8	2.7	1.8
Min.	0.6	0.6	0.5	0.1	0.6	0.7	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

SD, standard deviation; Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

Table 9
Average annual heavy-metal losses per unit area for the different plant cover.

	Plant cover						
	Th	La	Sv	Sa	An	Ro	Bs
	(mg m^{-2} yr^{-1})						
Mn	5.2 (±1.2)	3.6 (±2.9)	4.8 (±3.2)	11.7 (±9.1)	6.1 (±5.9)	8.0 (±5.8)	10.0 (±6.7)
Ni	0.07 (±0.01)	0.33 (±0.21)	0.04 (±0.03)	0.43 (±0.56)	0.12 (±0.08)	0.12 (±0.09)	0.13 (±0.22)
Mo	0.04 (±0.03)	0.04 (±0.03)	0.04 (±0.03)	0.13 (±0.09)	0.11 (±0.08)	0.13 (±0.08)	0.14 (±0.13)
Cr	0.02 (±0.03)	0.04 (±0.02)	0.04 (±0.03)	0.34 (±0.23)	0.06 (±0.05)	0.03 (±0.02)	0.13 (±0.17)
Cu	0.08 (±0.05)	0.10 (±0.08)	0.07 (±0.05)	2.98 (±1.9)	0.23 (±0.19)	0.22 (±0.19)	0.40 (±0.38)
Cd	0.00 (±0.00)	0.00 (±0.00)	0.01 (±0.01)	0.02 (±0.01)	0.00 (±0.00)	0.00 (±0.00)	0.01 (±0.01)
Co	0.08 (±0.06)	0.08 (±0.05)	0.06 (±0.07)	0.21 (±0.32)	0.12 (±0.09)	0.08 (±0.03)	0.21 (±0.26)
Zn	62.7 (±46.8)	61.4 (±39.8)	19.0 (±21.0)	80.6 (±67.1)	81.2 (±78.3)	54.2 (±45.7)	79.2 (±89.1)
Pb	0.01 (±0.0)	0.01 (±0.0)	0.01 (±0.02)	0.03 (±0.02)	0.01 (±0.02)	0.01 (±0.01)	0.02 (±0.03)

(±Standard deviation); Th, *Thymus mastichina*, La, *Lavandula dentata*, Sv, spontaneous vegetation; Bs, bare soil; Sa, *Satureja obovata*; An, *Anthyllis cytisoides*; Ro, *Rosmarinus officinalis*.

one of the most effective conservation practices in these subtropical agroecosystems.

3.6. Heavy-metal transport by runoff and its control

The heavy-metal concentrations in the runoff collected during the two agricultural seasons varied greatly (Table 8). The Cd and Pb concentrations were generally low while the Mn concentrations ranged from 0.1 to 3723.1 μg L^{-1}, and average An and Sv concentrations ranged from 170.9 to 384.1 μg L^{-1}, respectively. In this sense, concentrations of Mn greatly exceeded the 50 μg L^{-1} tolerance limit for drinking water (U.S. EPA, 1976). Average Cr concentrations ranged from 0.8 to 9.0 μg L^{-1} for Ro and Sa plots, respectively. The highest concentration values ranged from 5.3 to 175.5 μg L^{-1}. The Cr concentrations were lower than the drinking-water standard (100 μg L^{-1}), except for one event in Sa plot. Average Co concentrations ranged from 0.8 to 5.5 μg L^{-1} for Ro and Sa, respectively, with peaks of 5.3 to 70.7 μg L^{-1} for Ro and Sa, respectively, exceeding the 2.8 μg L^{-1}, which is the limit of Co for soil water (NMHPPE, 1998). Cd concentrations ranged from 0.0 to 12 μg L^{-1} for Ro and Sa, respectively, with a high peak of 282.5 μg L^{-1} for a storm event in the Sa plot. In most of the events, Cd concentrations were within the 5 μg L^{-1} standard for drinking water (Stewart et al., 2001), except for Sa plot. Average Ni concentrations were from 2.4 to 20.4 μg L^{-1} for Bs and La plots, respectively, with the highest peaks detected again for the Sa and La plots. The concentrations exceeded the 100 μg L^{-1} (drinking-water standard; Stewart et al., 2001) but not the 1400 μg L^{-1}, which is the established limit for surface waters according to the U.S. EPA (1976) for Ni. Average Cu concentrations ranged from 4.8 to 78.2 μg L^{-1} for Th and Sa plots, respectively, and the maximum concentrations were again recorded in Sa (1706.6 μg L^{-1}). For most events, concentrations did not surpass either 280 μg L^{-1}, the highest value of Cu found in a published assessment for natural surface waters of the USA (Manahan, 1991) or the limit value for Cu in drinking water (1000 μg L^{-1}; U.S. Public Health Service, 1962). Similarly, concentrations were lower than 100 μg L^{-1} (maximum permissible limit for drinking water, according to the Spanish Ministry of Health (BOE, 1990). The Cu concentrations were similar to those reported by He et al. (2004) for several runoff samples collected from agricultural lands (0.00–1475 μg L^{-1}). Average Zn concentrations ranged from 1300.8 to 3820.9 μg L^{-1}, and the highest peaks were found in the Sa and La plots (20,446.7 and 11,600.0 μg L^{-1}, respectively). Our results were much higher than those of He et al. (2004) for agricultural lands (0.0–1401.0 μg L^{-1}) and, after some events, concentrations exceeded 5000 μg L^{-1}, the maximum permitted for drinking water (Manahan, 1991). These high values for Zn concentrations may be due to the heavy Zn applications for foliar deficiencies in mango orchards and also probably from the material of galvanized sheets from the erosion plots themselves. Pb ranged from 0.1 to 1.5 μg L^{-1} and peaks were again recorded in Sa (28.8 μg L^{-1}) (Table 8). Pb concentrations in the Sa plot exceeded 15 μg L^{-1}, which is the standard for drinking water (Stewart et al., 2001) but were below 50 μg L^{-1}, the standard limit for drinking water according to Spanish Ministry of Health (BOE, 1990). Our Pb concentrations were very similar to those found by He et al. (2004) for different agricultural fields.

Heavy-metal losses per area are shown in Table 9. The greatest losses were recorded for Zn and the least for Ni, Mo, Cd, and Pb. For each element, the heaviest losses were recorded in the Sa and Bs plots, and the lowest in Th, La, and Sv. The Sv cover reduced Mn, Ni, Mo, Cu, and Zn losses by 52, 69, 71, 82, and 76%, respectively, compared to Bs. Among aromatic medicinal plant covers, Th had the lowest heavy-metal losses per area, except for Mn and Zn, for which the La plot was the lowest. Therefore, plant covers play an important role in controlling heavy-metal pollution risk, decreasing pollutant transport by runoff in comparison to the common soil-management practice in the study area with bare soil on the taluses of orchard terraces.

4. Conclusion

The present study highlights the impact of agricultural practices on soil erosion and runoff in taluses of orchard terraces, reflecting the need to protect these structures by using plant covers against soil loss and environmental pollution. The results of this research are in line with the findings of other studies, demonstrating the capacity of plant covers to reduce soil erosion and surface runoff on agricultural land. Thus, the implementation of aromatic plant covers in the taluses of subtropical orchard terraces substantially reduced soil erosion and runoff. Similarly, nutrient losses were reduced by using plant covers in comparison to the bare-soil treatment, especially in the Th, Sv, and La plots. In the same way, carbon losses by erosion were significantly reduced by the use of plant covers and at the same time, SOM was increased, due to the greater litter fall and nutrient cycling in this type of environment (Rodríguez et al., 2009b).

Under semi-arid conditions, where rainfall is responsible not only for the soil degradation but is also the main factor determining yields in subtropical agroecosystems, efforts need to be continued to develop sustainable systems for agriculture acceptable by the local farmers. In this context, the alternative cultivation of aromatic plant covers, such as thyme or lavender, could represent extra income for farmers and an environment-friendly measure that increases the stability of the taluses of the orchard terraces and helps minimize the risk of agricultural pollution.

Acknowledgement

The research work that led to this publication was sponsored by the following research project "Environmental Impact of Farming Subtropical Species on Steeply Sloping Lands. Integrated Measures for the Sustainable Agriculture" (RTA05-00008-00-00), granted by INIA, Spain and cofinanced by FEDER funds (European Union).

References

Agrela, F., Gil, J.A., Giráldez, J.V., Ordóñez, R., González, P., 2003. Obtention of reference value in the measurement of the cover fraction in conservation agriculture. In: Cury, B., Canalli, L.B. (Eds.), Proceedings II World Congress on Conservation Agriculture, Iguazu, Brazil, pp. 44–47.

Alloway, B.J., 1995. Heavy Metals in Soils. Blackie Academic & Professional, London, UK.

Andreu, V., Rubio, J.L., Gimeno, G.E., Linares, J.V., 1998. Testing three Mediterranean shrubs species in runoff reduction and sediment transport. Soil Tillage Res. 45, 441–454.

APHA, AWWA, WPCF, 1995. Standard Methods for the Examination of Water and Wastewater, 17th ed. American Public Health Association, Washington DC, USA.

Ayers, R.S., Westcot, D.W., 1987. Water Quality for Agriculture, Food and Agriculture Organization of the United Nations, Irrigation and Drainage Paper No. 29, Rev. 1, Rome, Italy.

Barrows, H.L., Kilmer, V.J., 1963. Plant nutrient losses from soil by water erosion. Adv. Agron. 15, 303–316.

Bautista, S., 1999. Regeneración post-incendio de un pinar (*Pinus halepensis* Miller) en ambiente semiárido. Erosión del suelo y medidas de conservación a corto plazo. Ph. D. University of Alicante, Alicante. Spain.

Bertol, I., Guadagnin, J.C., González, P.A., do Amaral, J.A., Brignoni, F.L., 2005. Soil tillage, water erosion, and calcium, magnesium and organic carbon losses. Sci. Agric. 62, 578–584.

Bochet, E., Rubio, J.L., Poesen, J., 1998. Relative efficiency of three representative matorral species in reducing water erosion at the microscale in a semi-arid climate (Valencia, Spain). Geomorphology 23, 139–150.

Bochet, E., Poessen, J., Rubio, J.L., 2006. Runoff and soil loss under individual plants of a semi-arid Mediterranean scrubland: influence of plant morphology and rainfall intensity. Earth Surf. Processes Landforms 31, 536–549.

BOE, 1990. Ministerio de Sanidad y Consumo 20/9/90. Real Decreto 1138/1990, de 14 de septiembre, por el que se aprueba la Reglamentación técnico-sanitaria para el abastecimiento y control de calidad de las aguas potables de consumo público.

Brandt, J.C., 1990. Simulation of the size distribution and erosivity of raindrops and through fall drops. Earth Surf. Processes Landforms 15, 687–698.

Brandt, J.C., Thornes, J.B., 1996. Mediterranean Desertification and Land Use. Wiley, Chichester, UK.

Castillo, V., Martínez, M.M., Albaladejo, J., 1997. Runoff and soil loss response to vegetation removal in a semiarid environment. Soil Sci. Am. J. 61, 1116–1121.

Coppin, N.J., Richards, I.G., 1990. Physical effects of vegetation. In: Coppin, N.J., Richards, I.G. (Eds.), Use of Vegetation in Civil Engineering. CIRIA, Butterworth, London, UK, pp. 49–86.

DGB-MMA, 2008. Ministerio de Medio Ambiente. . Available online at http://www.mma.es/portal/secciones/calidad_contaminacion/indicadores_ambientales/perfil_ambiental_2005/pdf/perfilambiental_2005_suelo.pdf (June 13, 2010).

Dos Santos, T.E.M., Montenegro, A.A.A., Silva, E.F.F., Neto, J.A.L., 2007. Perdas de carbono orgánico, potássio e solo em Neossolo Flúvico sob diferentes sistemas de manejo no semi-árido. Rev. Bras. Ciên. Agrár. 2, 143–149.

Durán, Z.V.H., Rodríguez, P.C.R., 2008. Soil-erosion and runoff prevention by plant covers. Review. Agron. Sustainable Dev. 28, 65–86.

Durán, Z.V.H., Martínez, R.A., Aguilar, R.J., Franco, T.D., 2003. El cultivo del mango (*Mangifera indica* L.) en la costa granadina, Granada, Spain.

Durán, Z.V.H., Aguilar, J.R., Martínez, R.A., Franco, T.D., 2005. Impact of erosion in the taluses of subtropical orchard terraces. Agric. Ecosyst. Environ. 107, 199–210.

Durán, Z.V.H., Francia, M.J.R., Rodríguez, P.C.R., Martínez, R.A., Cárceles, R.B., 2006. Soil erosion and runoff prevention by plant covers in a mountainous area (SE Spain): implications for sustainable agriculture. Environmentalist 26, 309–319.

Durán, Z.V.H., Rodríguez, P.C.R., Francia, M.J.R., Cárceles, R.B., Martínez, R.A., Pérez, G.P., 2008. Harvest intensity of aromatic shrubs vs. soil erosion: an equilibrium for sustainable agriculture (SE Spain). Catena 73, 107–116.

Durán, Z.V.H., García, T.I., Francia, M.J.R., Muriel, F.J.L., 2010. Soil erosion: causes, processes and effects. In: Fournier, A.J. (Ed.), Soil Erosion: Causes, Processes and Effects. Nova Science Publishers, Hauppauge, New York, USA, pp. 1–36.

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

Francia, M.J.R., Durán, Z.V.H., Martínez, R.A., 2006. Environmental impact from mountainous olive orchards under different soil-management systems (SE Spain). Sci. Total Environ. 358, 46–60.

Fullen, M.A., 1992. Erosion rates on bare loamy sand soils in east Shropshire, UK. Soil Use Manage. 8, 157–162.

García, R.J.M., Lasanta, T., Ortigosa, L., Ruíz, F.P., Mart, C., González, C., 1995. Sediment yield under different land uses in the Spanish Pyrenees. Mount. Res. Dev. 15, 229–240.

Griffioen, J., 2001. Potassium adsorption ratios as an indicator for the fate of agricultural potassium in groundwater. J. Hydrol. 254, 244–254.

He, Z.L., Zhang, M.K., Calvert, D.V., Stofella, P.J., Yang, X.E., Yu, S., 2004. Transport of heavy metals in surface runoff from vegetable and citrus fields. Soil Sci. Soc. Am. J. 68, 1662–1669.

Head, K.H., 1984. Manual of Soil Laboratory Testing. Soil Classification and Compaction Tests. ELE International Ltd. Fentech Press, London, UK.

Hofmann, L.K., Ries, R.E., 1991. Relationship of soil and plant to erosion and runoff on pasture and range. J. Soil Water Conserv. 41, 243–247.

Huetter, L.A., 1992. Wasser und wasseruntersuchung. Salle und Sauerlaender, Frankfurt, Germany.

Kosmas, C., Danalatos, N.G., 1993. Climate Change, Desertification and the Mediterranean Region. In: Rounsevel, M., Loveland, P. (Eds.), Soil responses to climate change: Implication for natural and managed ecosystems. NATO Advanced Research Workshop, Silsoe, UK, pp. 25–38.

Kosmas, C., Danalatos, N.G., Gerontidis, S., 2000. The effect of land parameters on vegetation performance and degree of erosion under Mediterranean conditions. Catena 40, 3–17.

Lal, R., 1990. Soil erosion and land degradation: the global risks. In: Lal, R., Stewart, B.A. (Eds.), Soil Degradation. Springer-Verlag. New York, USA, pp. 70–94.

Lobb, D.A., Burton, D.L., Lindstrom, M.J., Reicosky, D.C., 2002. Impacts of Soil Erosion on the Production and Emission of Greenhouse Gases and Carbon Sequestration in the Canadian Prairies. In: Smith, C.A.S. (Ed.), Soil organic carbon and agriculture: developing indicators for policy analyses. Proceedings of an OECD Expert Meeting, Agriculture and Agri-Food Canada, Ottawa and Organization for Economic Co-operation and Development, Paris/Ottawa, Canada, pp. 235–243.

Manahan, S.E., 1991. Environmental Chemistry, 5th ed. Lewis Publisher, Chelsea, MI, USA.

MAPA, 1994. Métodos Oficiales de Análisis. Tomo III Secretaría General Técnica del Ministerio de Agricultura, Pesca y Alimentación, Madrid, Spain.

Moore, P.A., Daniel, T.C., Gilmour, J.T., Shreve, B.R., Edwards, D.R., Wood, B.H., 1998. Decreasing metal runoff from poultry litter with aluminium sulphate. J. Environ. Qual. 27, 92–99.

Morgan, R.P.C., Finney, H.J., Lavee, H., Merritt, E., Noble, C.A., 1986. Plant cover effects on hillslope runoff and erosion: evidence from two laboratory experiments. In: Abrahams, A.D. (Ed.), Hillslope Processes, Allen & Unwin, London, UK, pp. 77–96.

NMHPPE, 1998. Netherlands Ministry of Housing. Physical Planning and Enviroment. Leidschendam, Netherlands.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267, 1117–1123.

Quadrelli, R., Pavan, V., Molteni, F., 2001. Wintertime variability of Mediterranean precipitation and its links with large-scale circulation anomalies. Climate Dyn. 17, 457–466.

Ramos, M.C., Martínez, C.J.A., 2006. Nutrient loses by runoff in vineyards of the Mediterranean Alt Penedès region (NE Spain). Agric. Ecosyst. Environ. 113, 356–363.

Rodríguez, P.C.R., Durán, Z.V.H., Muriel, F.J.L., Martín, P.F.J., Franco, T.D., 2009a. Litter decomposition and nitrogen release in a sloping Mediterranean subtropical agroecosystem on the coast of Granada (SE, Spain): effects of floristic and topographic alteration on the slope. Agric. Ecosyst. Environ. 134, 79–88.

Rodríguez, P.C.R., Durán, Z.V.H., Francia, M.J.R., de Graaff, J., Muriel, F.J.L., 2009b. Land-use changes in a small watershed in the Mediterranean landscape (Almuñécar, SE Spain): environmental implications of a shift towards subtropical crops. J. Land Use Sci. (In revision).

Rojo, L., 1990. Plan Nacional de Restauración Hidrológico-Forestal y Control de la Erosión. Tomo I. Memoria; Tomo II. Mapas, ICONA, Madrid.

Romero, D.A., López, B.F., Belmonte, S.F., 1998. Erosión y escorrentías en el campo experimental "El Ardal". Nueve años de experiencias. Papeles de Geografía 27, 115–130.

Romero, D.A., Belmonte, S.F., López, B.F., 2000. Erosión del suelo en ambiente semiárido extremo bajo diferentes tipos de litologías y suelos. V Reunión Nacional de Geomorfología, Sociedad Española de Geomorfología, Facultad de Ciencias Geológicas de Madrid, Madrid.

Sánchez, M.M., Soriano, M., Delgado, G., Delgado, R., 2002. Soil quality in Mediterranean mountain environments: effects of land use change. Soil Sci. Soc. Am. J. 66, 948–958.

Shinde, A.K., Dable, D.J., Jadhav, B.B., Kandalkar, M., Burondkar, M.M., 2006. Effect of dose and source of potassium on yield and quality of Alphonso mango (*Mangifera indica*). Indian J. Agric. Res. 76, 213–217.

Soil Survey Staff. Soil Taxonomy., 1999. A basic system of soil classification for making and interpreting soil surveys. Agric. Handbook N, vol. 436. USDA-USA.

Spalding, R.F., Exner, M.E., 1993. Occurrence of nitrate in groundwater, a review. J. Environ. Qual. 22, 392–402.

Stewart, J.C., Lemley, A.T., Hogan, S.I., Weismiller, R.A., Hornsby, A.G., 2001. Drinking Water Standards. Florida Coop. Ext. Service. Institute of Food and Agricultural Sciences, Univ. of Florida. Available online at http://edis.ifas.ufl.edu/SS297 (July 15, 2010).

Stroosnijder, L., 1995. Quantification of Nutrient Erosion. Proceedings Conference on Erosion and Land Degradation in the Mediterranean. University of Aveiro, Portugal, pp. 89–98.

Thornes, J., 1996. Desertification in the Mediterranean. In: Brandt, J., Thornes, J. (Eds.), Mediterranean Desertification and Land use. Wiley, Chichester, UK, pp. 1–12.

Thurow, T.L., Blackburn, W.H., Taylor, C.H., 1986. Hydrological characteristics of vegetation types as affected by livestock grazing systems. Edward Plateau Texas. J. Range Manage. 39, 505–509.

U.S. EPA, 1976. Quality Criteria for Water. US Environmental Protection Agency. United States Government Printing Office, Washington, D.C., USA.

U.S. Public Health Service, 1962. Public Health Service Drinking Water Standards. Public Health Service, Washington, D.C., USA.

Vallejo, V.R., Díaz, F.F., de la Rosa, D., 2005. Impactos sobre los recursos edáficos. In: Moreno, J.M. (Ed.), Evaluación preliminar general de los impactos en España por efecto del cambio climático. Spain, Ministerio de Medio Ambiente, Madrid.

Van den Bosch, H., Gitari, J.N., Ogaro, V.N., Maobe, S., Vlaming, J., 1998. Monitoring nutrient flows and economic performance in African farming systems (NUTMON). III. Monitoring nutrient flows and balances in three districts in Kenya. Agric. Ecosyst. Environ. 71, 63–80.

Vollenweider, R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus. OECD Report DAS/CSI6827, Paris, France.

Vollenweider, R.A., Kerekes, J., 1980. The loading concept as a basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. Prog. Water Tech. 12, 5–38.

WHO, 2007. Nitrate and Nitrite in Drinking Water. . Available on line at: http://www.who.int/water_sanitation_health/dwq/chemicals/nitratenitrite2ndadd.pdf (July 15, 2010).

Wischmeier, W.H., 1976. Use and misuse of the universal soil loss equation. J. Soil Water Conserv. 31, 5–9.

Wischmeier, W.H., Smith, D.D., 1978. Predicting Rainfall Erosion Losses: A Guide to Conservation Planning. USDA-ARS Agric. Handbook, 537, Washington, USA.